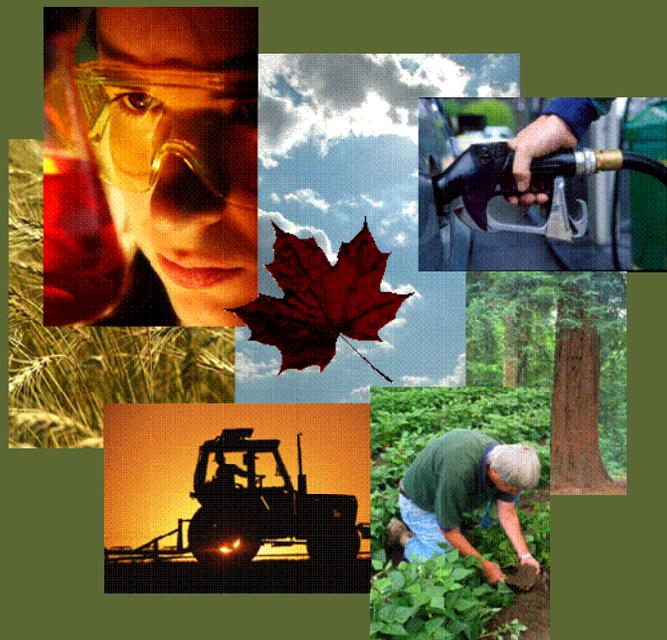


Threats and impacts of exotic pests under climate change: implications for Canada's forest ecosystems and carbon stocks

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Executive Summary

Insects and pathogens are important natural disturbance agents in Canada's forests, but while these organisms and their hosts have had a long time to co-exist and become adapted to each other, non-native species of insects and pathogens can flourish when they invade new territory that is enemy-free, potentially resulting in devastating effects to forest ecosystems. The emerald ash borer is a recent example of an invasive species that has been difficult to eradicate and will likely continue to decimate ash populations as it spreads. Historical examples such as chestnut blight, which all but extirpated its host tree species in North America, illustrate the profound effect that an exotic pest can have on Canada's forests.

As the climate changes, the ability of non-native forest pests to establish and spread in Canada may be improved, as the range of suitable environments expands and the door opens to insects and pathogens that previously posed less of a risk. Direct effects of climate change on forest pests will likely be realized in Canada as increased survival rates due to predicted warmer winter temperatures, and increased developmental rates due to warmer summer temperatures.

Canada's forests store vast amounts of carbon in living tree and plant biomass, as well as in the soil; these forest carbon stocks are influenced by forest disturbances, including outbreaks of insects and pathogens. In order to assess the future state of our forest ecosystems and forest C stocks, it is important to consider the threats and impacts of invasive, non-native species, under current and future climate projections.

Forest pests can impact ecosystem processes and C stocks by causing tree mortality over a large area, resulting in the regeneration of a new forest ('stand-replacing' disturbance), or by attacking host trees scattered throughout the forest and causing small gaps ('matrix' disturbance). In Canada's forests, it is likely that the largest effects of non-native insects and pathogens, particularly if they successfully invade the boreal forest region, will arise from increased frequency of stand-replacing disturbance, which decreases the overall amount of C stored on the landscape. The effects of 'matrix' disturbance are less clear, but the resulting shifts in forest species composition, as particular tree species are lost or reduced in abundance, can have lasting effects on forest ecosystem functioning and forest C stocks.

There is a long list of non-native pest species identified as posing a potential risk to Canada's forests; the case studies in this report examine the threats and impacts of several of these species (brown spruce longhorn beetle, European woodwasp, Asian gypsy moth, Asian long-horned beetle, emerald ash borer, hemlock woolly adelgid, and sudden oak death). For a few of these organisms, enough information on their climate tolerances or requirements was available for the creation of maps of their 'climate envelopes' (range over which the climate should be suitable for them) in Canada under different climate change scenarios. The results illustrate the degree to which anticipated changes in climate can expand a species' climate envelope, in particular showing the drastic changes expected to take place in the interior and northwestern parts of the country. The relative magnitude of changes projected for different emissions scenarios, and for different climate models, can also be observed. However, more information on the basic biology and ecology of these organisms is needed before meaningful risk assessments can be made; the maps created for this report are necessarily simplistic due to the small amount of data available for the mapping of each species' 'envelope', consisting only of temperature-related parameters. This report, therefore, represents a first step in assessing the potential ranges in Canada of a variety of non-native forest pests under different climate change scenarios.

Recommendations include the encouragement of research in several key areas, as well as the development of forest inventories that incorporate finer-scale distributions and relative abundances of tree species, and the provision of knowledge-sharing opportunities (e.g. symposia) for Canadian researchers and experts in the diverse fields (entomology, forest ecology, climate change biology, disturbance ecology, etc.) that relate to this issue.

1. Introduction

Global temperatures are expected to rise by 3 to 5°C over the next 100 years (Hengeveld 2000). Climate change will be most dramatic at high latitudes and in continental interiors. In Canada, increases in winter temperatures will likely be the most notable change, with increases of up to 20°C in the far north by 2100, according to some model predictions. In southern and central Canada, 5 to 10°C increases in annual temperature are predicted, with less warming occurring in coastal areas (Hengeveld 2000). Relatively smaller changes are expected in precipitation levels, meaning that more drought conditions will likely occur in parts of Canada as temperatures warm without a concomitant increase in moisture.

As temperatures rise, many possibilities for feedback (both positive and negative) exist between the biosphere and the atmosphere, and hence predictions about response of ecosystems are difficult to make. For example, if warming temperatures result in more C being released to the atmosphere, exacerbating the greenhouse effect, warming may be further accelerated.

Canada's forests represent a substantial amount of global terrestrial C stocks; the Canadian boreal forest, which comprises 75% of the country's forest land, holds 15% of global terrestrial C (Apps *et al.* 1993). It is therefore important to understand how Canada's forest ecosystems might be affected by climate change.

Climate change is predicted to affect Canada's forests in many different ways, including potentially facilitating the establishment and spread of non-native insects and pathogens. The "establishment and spread of alien pests under current and projected environmental conditions, including global change" has been identified as an emerging issue for Canada's forest sector (Canadian Forest Service 1999). Surprisingly little is known about the specific roles of insects and pathogens in Canada's forest ecosystems, and their influence on forest carbon dynamics; the bulk of research in this area has been aimed at minimizing economic losses due to disturbances by native forests pests. The purpose of this report is to synthesize the relevant information on the effects of climate change on forest insects and pathogens, the effects of insects and pathogen outbreaks on forest ecosystem processes and forest C, and the specific characteristics and tolerances of a selection of potentially invasive non-native forest pest, for the purpose of assessing the overall threats and impacts of non-native forest pests on Canada's forest ecosystems and forest C stocks.

2. Effects of climate change on forest ecosystems and forest C

Climate change is predicted to impact forest ecosystems in numerous ways, through effects on many different factors controlling forest processes (Aber *et al.* 2001). Increased atmospheric CO₂ may increase productivity by raising photosynthetic rates and/or increasing water use efficiency, given a sufficient supply of nutrients. Changes in growing season lengths and warmer temperatures may alter competition dynamics of vegetation at the edges of forest regions, shifting forest region boundaries (Thompson *et al.* 1998) and possibly extending the northern treeline northwards, while expanding the extent of grasslands (prairie region) and temperate forests (east/central Canada) at the expense of boreal forest at its southern boundary. Increased organic matter decomposition rates that may accompany climate warming could result in massive C loss from forest soils. Increasing drought frequency due to warmer conditions without a concomitant increase in precipitation may increase water stress, reducing productivity and predisposing forests to more frequent and severe wildfires and insect outbreaks (Kasischke *et al.* 1995). A changing climate will likely alter the disturbance dynamics of native forest insects and diseases, as well as facilitating the establishment and spread of non-indigenous species. While a growing body of research exists in the area of predicting the effects of climate and atmospheric change on forest productivity and fire regimes (though not without uncertainty) (e.g. Thompson *et al.* 1998), comparatively little has been done on the potential changes in insect and pathogen disturbance regimes (Woods *et al.* 2005), and even less on the threats that non-indigenous

insects and pathogens may pose. Interactions between insect disturbances and fire make predictions of changes in disturbance regimes under climate change difficult (Fleming 2000). This challenge is an important one to take on, however, as the nature and magnitude of potential changes in disturbance regimes, including disturbance from insects and pathogens, will play a major role in determining the fate of Canada's forest carbon stocks as the climate changes, through their influence on the state and distribution of forest ecosystems (Holling 1992; Fleming 2000).

3. Effects of climate change on forest insects and pathogens

The mechanisms by which climate change could alter disturbance patterns from insects and pathogens include: direct effects on the development and survival of herbivores and pathogens; physiological changes in tree defenses; and indirect effects from changes in abundances of natural enemies, mutualists and competitors (Ayres and Lombardero 2000). The first mechanism will presumably be most important in the case of invading, non-native species. Their short life cycles, physiological sensitivity to temperature, mobility, and reproductive potential will allow forest insects and diseases (native and non-native) to respond rapidly to climate change (Ayres and Lombardero 2000).

Direct effects of climate change on forest pests will likely be realized in Canada as increased survival rates due to predicted warmer winter temperatures, and increased developmental rates due to warmer summer temperatures. Ungerer *et al.* (1999) predicted that warmer winter temperatures as well as less variance in temperature would be more important than warmer summers for the northward expansion of the southern bark beetle (*Dendroctonus frontalis*) in the U.S. Increased climatic variability in the form of more extreme weather events could also possibly benefit forest pests by increasing the vulnerability of trees to infestation (e.g. more extreme weather events such as windstorms can produce damage on trees and allow entry to pathogens, while more lightning strikes can encourage bark beetle infestation) (Ayres and Lombardero 2000).

Williams and Liebhold (2002) modeled the response of mountain pine beetle to climate warming and found that the temperature is currently strongly limiting to the beetle, meaning that small changes in climate could remove significant limits on this species. Logan and Powell (2001) determined that a 2.5°C increase in annual temperature could result in a northward shift of 7° of latitude, potentially allowing this insect to invade the jack pine forests of the eastern boreal region. Logan *et al.* (2003) speculate that a warming climate may already be partly responsible for the apparent northward expansion of mountain pine beetle in B.C. However, once again demonstrating the complexities involved with predicting a species' response to climate change, Logan and Powell (2001) report that while a increase in temperature of just over 2°C would shift mountain pine beetles from semivoltine (requiring two years to complete a generation) into synchronous, univoltine (requiring one year to complete a generation) populations, a temperature range exists beyond which, with increased warming, populations would be shifted further into an unfavourable (for the insect) asynchronous voltinism due to disrupted seasonality requirements. Other insect species may be negatively affected by climate change if diapause requirements have a lower chance of being met thereby disrupting developmental cycles (Ayres and Lombardero 2000).

Physiological changes in host trees may include increases in the amounts of defensive chemicals produced, as a response to increased atmospheric CO₂ concentrations and/or moderate drought (Ayres and Lombardero 2000), increasing host resistance to pest attack. Raised CO₂ levels may also make conditions less favourable to insects by increasing the carbon to nitrogen ratio of tissues, which could result in nutritional deficiencies, but could also result in more feeding as the organisms attempt to gain enough nitrogen (Ayres and Lombardero 2000). Canada's boreal forest region in particular is expected to become warmer and drier, and while moderate drought may increase production of defense compounds in trees, drought stress can also make trees more vulnerable to mortality from insect attack (Fleming 2000; Hogg *et al.* 2002). Increased drought could be potentially advantageous to pests if sugar concentrations in foliage are increased as a result, making it more palatable to herbivores (Fleming and Candau 1998), or if stomatal conductance declines, raising microclimate temperatures on

tree foliage and thereby benefiting insects (Scarr 1998). Drought conditions can also indirectly influence insect populations through changes in resource allocation by host trees; for example, drought causes jack pine to produce more reproductive tissue, which results in greater survival of the jack pine budworm, a native forest pest (Fleming 2000).

The complex dynamics involved in the effects of climate change on community-level interactions make predictions particularly difficult to make (Williams *et al.* 2000). For example, pathogens dependent on insects for dispersal may experience negative effects if climate change disrupts the synchrony between the two types of organism (e.g. Williams and Liebhold 2002). Fleming and Candau (1998) point out that climate change could affect spruce budworm populations by altering any of the relationships among host species, the budworm, and its natural enemies.

While the bulk of the content of this report pertains to insect pests, as comparatively less relevant information was available on disease organisms, there is evidence that climate trends are affecting the severity of disease outbreaks in Canadian forests. For fungi and other disease-causing organisms, climate-change induced alteration of precipitation patterns may be a more important factor than temperature changes. Woods *et al.* (2005) reported that *Dothistroma* needle blight, a native fungus in temperate forests of the northern hemisphere, is causing unprecedented mortality of mature lodgepole pine in British Columbia. Normally, the blight is a relatively innocuous organism which can reduce growth rates, but rarely causes mortality in its natural range. A trend of increasing summer precipitation, coupled with the increase in host abundance due to the establishment of extensive plantations of its host tree, is blamed for the current epidemic.

According to MacLean *et al.* (2002), there are three relevant time frames in the consideration of pest outbreak patterns in a changing climate: the short-term, in which insect populations will likely respond to climate change more rapidly than their hosts; the medium-term, which may see widespread disruption to forest ecosystems, as trees experience difficulties regenerating, and shifts in tree species' ranges occur; and the long-term, where a new equilibrium will develop between insects and forest ecosystems.

4. Effects of insects and pathogens on forest ecosystems and forest C

It has long been recognized that insects and pathogens are important organisms in forest ecosystems, contributing to and affecting forest processes such as nutrient cycling and net primary productivity, and influencing forest structure, successional patterns, and species composition (Fleming 2000; Castello *et al.* 1995; Wilson 1987; Mattson and Addy 1975). However, the ecological roles of insects and diseases in forests are in general still poorly understood (Lewis and Lindgren 2000). In particular, the impact of changing disturbance regimes, including insect outbreaks, represents an important unanswered question with regards to understanding the effects of climate change on forest C (Gower 2003).

In Canada's forests, insects are an important disturbance agent, responsible for productivity losses estimated at almost 60 million cubic metres per year (Fleming 2000), and affecting more forest area annually than fire. The most important native pests (except mountain pine beetle, *Dendroctonus ponderosae*, in the west) are mainly defoliators; forest tent caterpillar (*Malacosoma disstria*) is the most important pest of hardwoods, spruce budworm (*Choristoneura fumiferana*) of conifers (Fleming 2000). The impacts of invading forest pests on forest ecosystems are not fundamentally different from that of native pests (Liebhold *et al.* 1995), as both reach outbreak proportions as a result of escaping their natural controls (either temporarily, as is the case with native species, or for an indefinite period of time, as with non-native invasive species), and both affect the same processes in forest ecosystems. However, many of the invasive, non-native insects that are at high risk of establishing in Canada, and that may enjoy an expanding potential range due to climate change, are wood boring insects that feed on cambial tissue or otherwise disrupt the flow of carbohydrates and water in the tree stem, and may have the potential to cause more rapid and widespread mortality than defoliators. Non-native pests may alter competitive dynamics among plants (Thompson *et al.* 1998); more frequent insect-induced

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stand-replacing disturbance in temperate hardwood forests (where, in contrast to the boreal forest, most tree species are not adapted to this pattern of disturbance) in particular may result in more profound changes in forest structure over the landscape (Ayres and Lombardero 2000).

Insect and pathogen-induced tree mortality results in forest disturbance ranging in magnitude from stand-replacement to small-gap formation, with effects seen on both short- and long-term time scales. Lewis and Lindgren (2000) describe the features that distinguish stand-replacing disturbance (stand-scale, dramatic change, initiates succession) from 'matrix' disturbance (individual tree-scale, modification, slow change). Insect or pathogen-induced disturbance at both scales affects forest ecosystem structure and function, and hence C stocks.

Stand-replacing disturbances such as some forest pest outbreaks and fire strongly influence forest carbon balance, particularly in Canada's boreal forest. The more frequent and severe the disturbance, the less carbon is stored on the forest landscape (MacLean *et al.* 2002; Fleming 2000; Kasischke *et al.* 1995). Tree mortality shifts forest structure towards younger age-classes that contain less living biomass and hence less carbon in the tree biomass pool (Fleming 2000; Kasischke *et al.* 1995). In the short-term, increased solar radiation at the soil surface due to canopy removal, as well as increased organic matter inputs and under some circumstances increased soil moisture, favour microbial growth and hence accelerate decomposition rates (e.g. Bhatti and Apps 2002). Hence, a significant amount of the C stored in the dead organic matter created by the disturbance can be released to the atmosphere in a relatively short time (Bhatti and Apps 2002; Fleming 2000). In the boreal forest, soil C levels are most strongly correlated with rates of litter inputs, which depend in part on age-class structure, which in turn depends on the type and frequency of disturbance.

Insect-induced stand mortality, compared to fire and harvesting, should result in the largest amount of biomass C being transferred to the forest floor (Figure 1), due to the absence of the large, direct losses that occur when biomass is consumed by fire or removed via harvest. So, insect disturbance may result in more potential for soil C accumulation compared to other stand-replacing disturbances. However, over the landscape, more stand-replacing disturbance in general can result in boreal forest soils being a net C source; when more area of forest represented by younger age classes, litter and CWD inputs to soil are reduced overall, and faster decomposition rates due to altered microclimate in young stands further tips the balance towards net C loss (Henrickson 2003; Bhatti and Apps 2002).

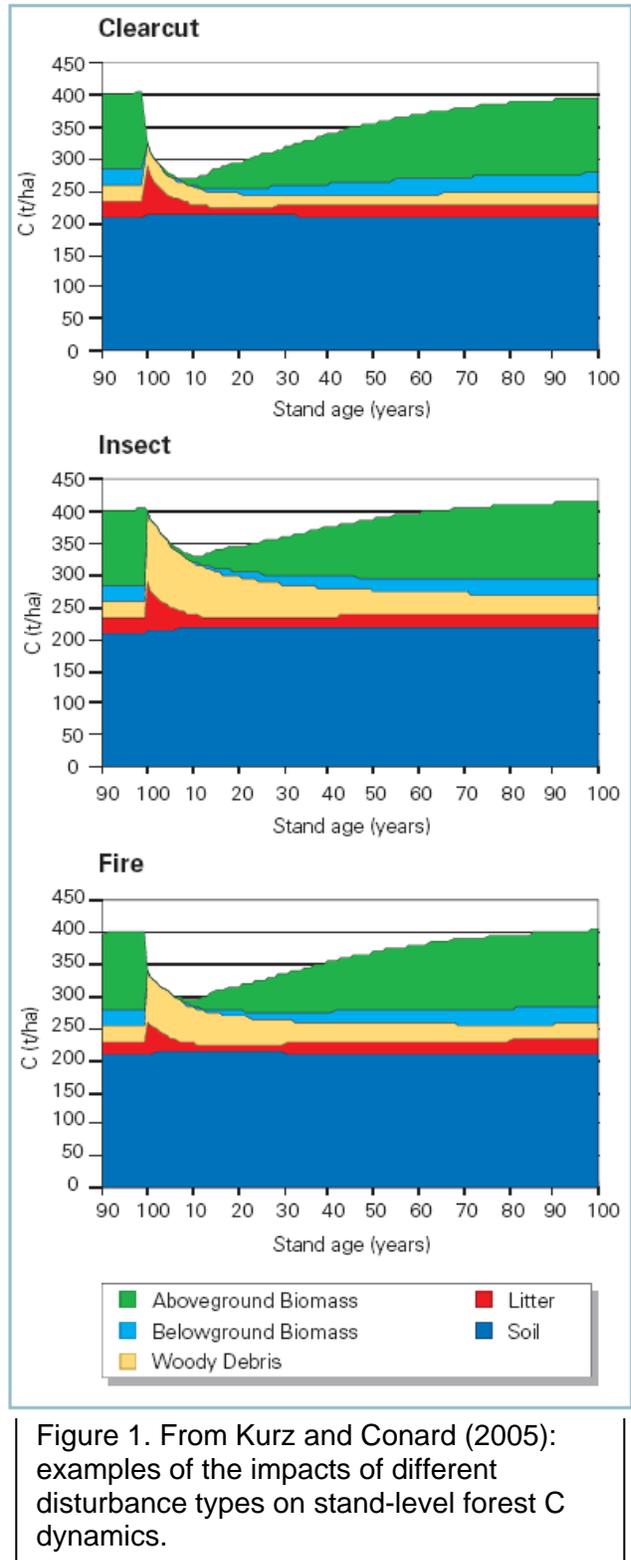


Figure 1. From Kurz and Conard (2005): examples of the impacts of different disturbance types on stand-level forest C dynamics.

Insect-induced tree mortality results in a transfer of biomass from live to dead organic matter pools, which can translate to high fuel loads and fire risk. If insect-damaged forests burn, then more C is lost immediately to the atmosphere as organic matter is consumed (Kasischke *et al.* 1995). The relationship between insect outbreak and fire is not always simple, however, and more research is needed to understand the interactions between these disturbance agents. For example, fire risk may be high immediately after a pest outbreak (Thompson *et al.* 1998), but if fire does not occur before leafy understory vegetation proliferates and shades the forest floor, the resulting high moisture conditions can reduce fire risk (Kulakowski *et al.* 2003; Thompson *et al.* 1998; Stocks 1987), at least until the volume of falling dead wood finally overwhelms the vegetation (Stocks 1987). In the long term, fire regimes could be affected by pest outbreaks if alterations of tree species composition over the landscape reduce or increase forest flammability. Hardwood foliage contains more moisture and is less flammable than conifer foliage; mixed forests of hardwoods and conifers in Canada are generally less at risk from fire than pure conifer stands. If outbreaks of conifer pests increase in frequency and/or severity, and successional dynamics result in increased hardwood abundance, fire risk could decline, having the general effects of conserving C on the forest landscape. However, the replacement of conifers by hardwoods can have the effect of altering nutrient cycling and organic matter decomposition rates through changes in microclimate and litter quality, and may result in smaller pools of C in the soil and forest floor. These potential counteractions in the effects of increased pest outbreaks, native or non-native, highlight the need for more data collection and modeling efforts.

Castello *et al.* (1995) note that biotic and abiotic forest disturbance agents are similar in effect in that they redistribute organic matter and nutrients, and influence succession and forest landscape patterns. However insect disturbance, even on a large scale, has important distinguishing features from forest fire, for example with regards to levels of tree mortality (Veblen *et al.* 1991; Kurz and Conard 2005). Pest outbreaks may be capable of causing 100% mortality in pure stands, and so forest plantations may be particularly at risk of complete stand mortality, however even the relatively monospecific boreal forest generally is composed of a mix of tree species at the stand level. Spruce budworm, a major defoliator in the Canadian boreal forest, has been found to attack balsam fir (*Abies balsamea*) more heavily than spruce (*Picea spp.*) in mixed spruce/fir stands (MacLean and MacKinnon 1997; Nealis and Regniere 2004), altering competition dynamics and the relative abundance of tree species. In the western U.S., spruce beetle (*Dendroctonus rufipennis*) outbreaks can cause high rates of tree mortality in Engelmann spruce/subalpine fir (*Picea engelmannii/Abies lasiocarpa*) stands, killing spruce and releasing fir, effectively accelerating succession towards a later seral species composition (Veblen *et al.* 1991), rather than 'resetting' succession back to an early stage as fire might do. Veblen *et al.* (1991) note the difference in maximum tree size between spruce and fir; the species composition change towards dominance by the smaller-statured fir translates into a reduction in the C stocks of the living vegetation. The productivity of surviving trees can be increased after an outbreak (Romme *et al.* 1986; Veblen *et al.* 1991). Romme *et al.* (1986) found that a tree mortality rate of 50% caused by a mountain pine beetle outbreak resulted in a loss of overall productivity of only 25%, and that pre-outbreak productivity levels were regained by 15 years after the disturbance.

Shifts in forest age-class structure and/or species composition will also have implications for wildlife habitat, especially if the abundance of old forest is reduced and homogeneity of forest patches over the landscape is increased.

Here it must be noted that inferences made about the effects of non-native pests on Canadian forests based on information about native pest impacts has its limitations; outbreaks of non-native forest pests may result in unforeseen differences in the resulting disturbance patterns. Romme *et al.* (1986) attribute the rapid return of pre-outbreak productivity levels after a mountain pine beetle outbreak to co-evolutionary relationships between the beetle and its host tree. The beetle tends to selectively attack larger, more mature trees, leaving the younger trees to regenerate the stand. The tendency of many of the non-native pests that threaten Canada's forests to attack a wider variety of host tree species, age-classes, and health levels may result in more complete stand mortality than outbreaks of more host-specific pests.

Relatively little information is available on the effects of wood-boring insects such as bark beetles on stand-level ecosystem processes (Veblen *et al.* 1991). The effects of wood boring insects differ from defoliators, for example in the lack of immediate return of N-rich tissues in frass (insect faeces) and damaged leaf pieces (Romme *et al.* 1986). At onset of tree mortality, resources are redistributed through leaf fall and woody debris inputs; the latter occurring over years or even decades as snags (standing dead trees) fall over. In contrast, a growing body of data exists on the stand-level effects of defoliators on forest processes.

Chronic, non-stand replacing disturbance due to defoliation by insects can reduce ecosystem productivity by slowing photosynthesis and increasing heterotrophic respiration (due to increased activity in the detrital and herbivorous trophic webs) (e.g. Fleming 2000), and hence reducing forest C sequestration. Hogg *et al.* (2002) found that forest tent caterpillar defoliation was the most important cause of variation in inter-annual tree growth in western Canada.

Defoliation can directly influence ecosystem processes such as nutrient cycling through the deposition of nutrient-rich frass, inputs of prematurely senesced leaf litter, increased nutrient leaching from damaged tissues, and enhanced decomposition rates due to thinning of the canopy and warming of the forest floor (Fleming 2000; Rinker 2001; Hunter 2001). Classen *et al.* (2005) called soil microclimate “a major driver of ecosystem processes”, and found that scale insect infestations altered microclimate at the soil surface by reducing tree leaf area leading to increased soil temperature and moisture. Jenkins *et al.* (1999) speculate that higher rates of net nitrogen mineralization and nitrification in stands affected by hemlock woolly adelgid (*Adelges tsugae*) were due to warmer soil temperatures from increased solar radiation at the soil surface. These authors provide a helpful conceptual model of the effects of tree mortality on forest soil processes (Figure 2).

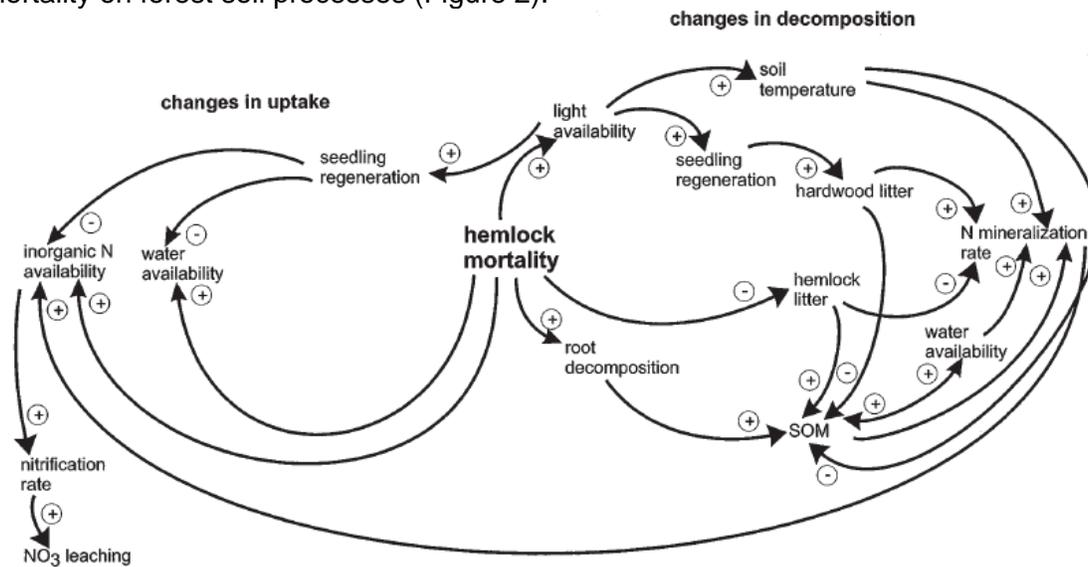


Figure 2. From Jenkins *et al.* (1999): a model of hypothesized relationships among forest structure, nutrient cycling, and forest insect-induced tree mortality.

Swank *et al.* (1981) found that outbreaks of a defoliating insect resulted in increased nitrate export to a forest stream, attributed to large increases in frass, litterfall, and mineralization rates, which, despite post-outbreak increases in leaf production and net primary productivity, resulted in a net loss of nitrate from the system. Others have found that the microbial biomass growth stimulated by insect frass deposition during an outbreak can result in immobilization of mineralized N, effectively retaining N in the system (Lovett and Ruesink 1995). The deposition of insect carcasses can also represent a pulse of nutrient-rich organic matter to the detrital trophic web, followed by a period of increased N availability to plants (Yang 2004), suggesting that the loss of primary productivity due to consumption of foliage by

herbivores may be at least partially compensated for by nutrient enrichment and faster growth following defoliation.

The long-term effects of non-stand replacing outbreaks (or 'matrix' disturbance, where host-specific attack results in scattered tree death and small forest gaps) can include altered tree species composition, affecting nutrient cycling indirectly through changes in litter quality. As eastern hemlock (*Tsuga canadensis*) is killed by the hemlock woolly adelgid (*Adelges tsugae*) and is replaced by black birch (*Betula lenta*) in forests of the northeastern U.S., Jenkins *et al.* (1999) predict that the increased inputs of hardwood litter will affect nitrogen mineralization rates. Lovett *et al.* (2002) found that nitrate losses to streamwater were related to the carbon to nitrogen ratio of adjacent forest soils, and that this in turn was influenced by tree species composition; high net nitrate production and nitrate loss is associated with high abundance of sugar maple and low conifer abundance (Lovett *et al.* 2002; Venterea *et al.* 2003).

Historical and current examples of the loss of tree species (including functional or local loss, not necessarily complete extirpation) from forest ecosystems in North America due to invasive pests include the ongoing decline of whitebark pine (*Pinus albicaulis*) in the western U.S. from white pine blister rust infection (Liebhold *et al.* 1995), the impact of chestnut blight on the American chestnut (*Castanea dentata*) in the northeastern U.S. and adjacent parts of Canada (Woods 1959; Castello 1995), and the loss of American elm (*Ulmus americana*) due to Dutch elm disease in the early to mid-1900s (Castello 1995).

Dutch elm disease has altered age-class structure and species composition in northeastern North American forests, as elms were replaced by other species. In some cases shade-tolerant trees such as sugar maple increased in abundance, altering the light environment at the soil surface by casting a deeper shade, and thereby influencing understory vegetation and tree regeneration. In wet lowlands, shrub density often increased in response to elm loss, shifting forest structure to a more open-canopied shrub rich ecosystem. Interestingly, the decline of elm likely led to an increased abundance of ash (*Fraxinus spp.*) species in many forests, a species that is now in danger from another non-native pest (Castello *et al.* 1995). The blight-induced loss of American chestnut likely increased tree species richness in some forests where chestnut had been the dominant species (Castello 1995). In other cases, the loss of chestnut from dry, pine-chestnut forests resulted in a shift from a closed forest structure to open pine heathlands; where eastern hemlock was present, a more conifer-dominant forest often resulted from chestnut loss (Woods 1959). Woods (1959) estimated that the filling in of even small gaps created by chestnut mortality took 25 years or more. Some types of wildlife will be affected negatively by tree species loss, as seen with the food shortages for grizzly bear caused by the decline of whitebark pine (grizzlies feed on the seeds), while others may be positively affected if pathogen outbreaks result in habitat enhancement through, for example, the formation of holes in trees (Castello 1995).

Ellison *et al.* (2005) reviewed the ecological effects of tree species loss due to insect or pathogen attack. Over time, the loss of tree species within the forest matrix will have many effects on forest ecosystems, including the seasonality and quality of litter inputs, the soil surface microclimate, the habitat suitability for wildlife species, and the hydrology and ecology of forest streams (Ellison *et al.* 2005). While all of these changes may also involve changes in forest C dynamics, little information is available on the specific effects of biodiversity loss on C stocks in forests. Recently, Bunker *et al.* (2005) modeled tree species loss in a tropical forest to investigate the effect of tree species removal on forest C. Effects depended on the characteristics of the species removed: for example, when species with dense wood or large maximum size were removed and replaced with a random mix of species from the surrounding forest, forest C stocks increased (Bunker *et al.* 2005). To our knowledge, no similar study exists for Canadian or North American forests, although this type of information would be valuable in assessing changes in biodiversity on forest C dynamics.

5. Case studies

5.1 Introduction

Many insects and pathogens not native to Canada have been identified as potential future invaders to Canada's forests, and there are certainly more that remain unknown. It is not the purpose of this report to speculate on all potential invasive forest pests, but to discuss the general threats and impacts posed by the interaction between non-native pest invasions and climate change. We have selected a set of non-native organisms, recognized as posing a risk to Canada's forest ecosystems, as case studies, and examined how climate change may increase their success. For some organisms, enough information was available to allow us to make preliminary assessments of the effects of projected climate change on their potential geographic range (their 'climate envelope'), and/or on the number of generations that can be produced annually within parts of their potential range in Canada. We then consider the implications for forest ecosystems and forest C stocks, by assessing the role of the host tree in the ecosystem and the potential consequences of the loss or reduction in abundance of the host, as well as by assessing the potential effects of the pest on the age-class structure over the forest landscape.

The list of 'case study' species includes the brown spruce longhorn beetle (*Tetropium fuscum*), European woodwasp (*Sirex noctilio*), Asian gypsy moth (*Lymantria dispar*), Asian long-horned beetle (*Anoplophora glabripennis*), emerald ash borer (*Agilus planipennis*), hemlock woolly adelgid (*Adelges tsugae*), and sudden oak death (*Phytophthora ramorum*). These organisms are either foreign to Canada but established in the U.S., previously intercepted in Canada or the Northern U.S., or newly established in Canada but not yet widespread. The list includes representatives of all the above categories, and is composed mainly of insects that cause tree mortality through consumption of phloem, xylem, and/or cambium tissue in tree stems, but also includes a defoliator (Asian gypsy moth) and a pathogen (sudden oak death).

For four of the organisms listed above (brown spruce longhorn beetle, European woodwasp, Asian gypsy moth, and Asian long-horned beetle), appropriate information was available on their climate tolerances or requirements; we have created maps of the potential ranges of these pests in Canada. We include discussions of the other listed species. Not enough data was available to create climate envelopes for the emerald ash borer; investigations into the phenology of this insect are still underway. The developmental rates of some life stages known but not others, and extent of its native range has not yet been confirmed (B. Lyons, pers. comm.). The hemlock woolly adelgid is thought to be adapting to colder temperatures as it progresses northwards from the northeastern U.S.; attempts to map its current and future range given the available cold tolerance data did not match with the occurrence of new infestations have recently been found on the southern edge of Lake Ontario. It is likely that HWA will move into Canada as it continues to adapt to colder temperatures at the northern edge of its range, and that climate change will facilitate this as winter temperatures warm. Finally, detailed mapping exercises have recently been undertaken by others to predict the potential spread of sudden oak death under climate change, and so we did not construct our own climate envelope for this pathogen.

We considered including the mountain pine beetle (*Dendroctonus ponderosae*) as a case study of a regionally native species with the potential to invade other regions, however due to a number of factors we have not included a discussion of this insect here, although the importance of this threat should not be overlooked. Other authors (particularly Logan *et al.*) have examined the mountain pine beetle's response to a warming climate and have made specific predictions about the amount of warming needed to push the MPB out of its current range and into the eastern boreal jack pine forests. That information is included earlier in this report as part of the section on the impacts of climate change on forest pests. Also, we address the potential impacts of widespread tree mortality in Canada's northern pine forests in the section below on the European woodwasp.

Climate mapping is a valuable way to visualize the risk posed by non-native invasive organisms (Baker *et al.* 2000). Although geographic potential is only one factor that determines an invading species'

success, modeling the potential ranges of invasive pests offers possibilities of proactive approach to management of these species (Peterson and Scachett-Pereira 2004), and modeling potential species distributions using species-specific developmental rate parameters has been deemed a useful approach (Logan and Powell 2001). McKenney *et al.* (2003) reviewed and analysed risk assessment methods applicable to exotic species in Canada, and argue for increased use of bioclimatic modeling of potential pests.

It is important to emphasize that the projected climate envelopes included in this report are only as robust as the information used to create them; for non-native species in general, data on climate requirements/tolerances is scarce. The maps that follow should be viewed with this in mind; they are a synthesis of the available information on how Canada's climate is expected to change, and the known climate requirements/tolerances of these insects. More research is needed on the biology and ecology of these and other potential non-native invaders if meaningful risk assessments are to be made. Nonetheless, our projections are useful for illustrating the degree to which anticipated changes in climate can expand a species' climate envelope, in particular providing a visual confirmation of the drastic changes expected to take place in the interior and northwestern parts of the country. Also, the relative magnitude of changes projected for different emissions scenarios, and for different climate models, can be observed. This report, therefore, represents a first step in assessing the potential ranges in Canada of a variety of non-native forest pests under different climate change scenarios.

5.2 Methods

Climate envelopes in Canada were determined for each chosen pest under different climate change scenarios. The climate envelope is the area of potential geographic distribution based on known environmental tolerances or requirements of the organism. For some of the 'case study' species, Canada's current climate is not limiting to their survival, but may limit the number of generations produced in a year; in these cases, this 'envelope' within which a species may be able to produce a certain number of generations per year was mapped.

Suitability indicator variables (c.f. Levia and Frost 2004) for each pest were obtained from information in the literature and from communication with researchers; the indicator variables included minimum mean annual temperature tolerances, degree day requirements above a threshold temperature, and monthly mean summer temperature requirements. Host tree ranges were mapped along with each pest species' potential range (if a pest has more than one host, the range of the most widespread host species was mapped). The available information on the climate requirements and tolerances of pest species was mainly temperature related (and the limiting factor in the Canadian climate, at least for most insects, is likely to be temperature rather than precipitation), and so the mapped potential ranges are based on temperature-related variables. Future work in this area should involve creating more detailed and fine-scale climate envelopes using a larger suite of variables derived from the species' native range.

Climate data were obtained from the Canadian Institute for Climate Studies (Canadian Climate Impact Scenarios (CCIS) <http://www.cics.uvic.ca/scenarios/index.cgi>). Degree days were calculated by sine-wave interpolation of mean monthly temperatures to arrive at mean daily temperatures (Brooks 1943), then by subtracting the threshold value from daily mean temperatures and summing over the year. Spatial referencing of the climate data was done using the mapping program iMap, and data manipulations in MSExcel.

We used climate projections from the Canadian Coupled Global Circulation Model (CGCM2), for SRES (IPCC Special Report on Emissions Scenarios) scenarios of medium-high (SRES A2) and medium-low (SRES B2) greenhouse gas emissions, and the Hadley Circulation Model (HadCM3), for these same scenarios as well as high (SRES A1) and low (SRES B1) emissions scenarios. Full descriptions of the different scenarios can be found in Nakicenovic *et al.* (2000)

5.3 Brown spruce longhorn beetle (*Tetropium fuscum*) (BSLB)

The brown spruce longhorn beetle (BSLB) is a wood boring beetle that feeds on the woody tissues of trees in the genera *Picea*. The insect breeds and develops in dead wood as well as live trees, and is tolerant of cold climates. Its natural range extends from Scandinavia to Turkey, Siberia, and Japan. Under current climatic conditions, it is capable of producing one generation every year (univoltine) over most of the range of its hosts in Canada.

BSLB was first collected in Halifax, Nova Scotia in 1990 (P. deGroot, pers. comm.), and became established in a forest park near to the likely port of entry (Dobesberger 2005). Red spruce (*Picea rubens*) appears to be its preferred host in Canada; it has also been found feeding on black spruce (*P. mariana*) and white spruce (*P. glauca*), and, unlike in its home range, it has been attacking healthy, vigorous trees. Efforts to eradicate have been only partially successful. Over 6000 trees have been removed, and the rate of tree infestation has declined significantly since 2000 (Henry *et al.* 2005), however it has been detected outside of the park in which eradication efforts have been taking place (Dobesberger 2005). Native *Tetropium* spp. are found in Canada, and there is some evidence that natural enemies of the native beetles (parasitoid wasps) may be attacking BSLB (Henry *et al.* 2005).

The beetle is designated as posing high risk to North American forests, due to its proven ability to successfully establish outside its host range, the presence of suitable hosts and climate, its ability to adapt to new hosts, and its high reproductive potential (Dobesberger 2005). The likelihood of the insect spreading throughout the range of its hosts is also very high: BSLB adults can fly several kilometres or more in distance, and the host spruce species are gregarious and widely distributed across northern North America (Dobesberger 2005).

While the climate in Canada is thought to be suitable for the establishment of BSLB over much of the range of its host trees (J. Sweeney, pers. comm.), climate warming scenarios predicted for Canada will likely increase the range over which the beetle is capable of bivoltinism, producing two generations each year (Figures 3 to 8). Bivoltinism has been observed in this insect in its native range, where average July and August temperatures are over 20°C (Schmitschek 1929). The occurrence of bivoltinism will not necessarily lead to increased population size and potential outbreak severity; success of the second generation produced late in the season may be limited as adults and eggs are not very tolerant of cold (J. Sweeney, pers. comm.).

The consequences of BSLB establishment across the range of spruce in Canada are potentially severe. Pure (or relatively monospecific) stands of spruce, particularly in the boreal forest, may be subjected to outbreaks that essentially lead to stand replacement. The resulting shift in age-class structure due to an increasing frequency of such disturbances would lead to a decline in forest C stocks over the landscape. Fuel build-up from tree mortality could contribute to a higher risk of fire, which would result in the immediate release of more C from dead biomass pools to the atmosphere. If hardwoods increased in abundance, for example if conifer mortality resulted in recruitment of aspen or birch, fire frequency may be reduced in the longer term by the shift in species composition over the landscape to less flammable species, and hence more C may be conserved. Conversely, the shift to hardwoods and the resulting change in microclimate and litter quality may result in C being lost from soil pools, and a reduced capacity for C storage in the forest floor.

In low, wet areas of the boreal forest, black spruce can be found growing on deep organic matter deposits, an environment unsuitable for most other tree species. Outbreaks of the beetle in this type of stand could potentially result in the conversion of forest to non-forest if the loss of spruce leads to dominance of shrubs such as alder, and less C being stored in the living vegetation. Closed spruce forests also provide important winter cover for a variety of wildlife species and the reduction of these habitats over the landscape would have potential consequences for biodiversity.

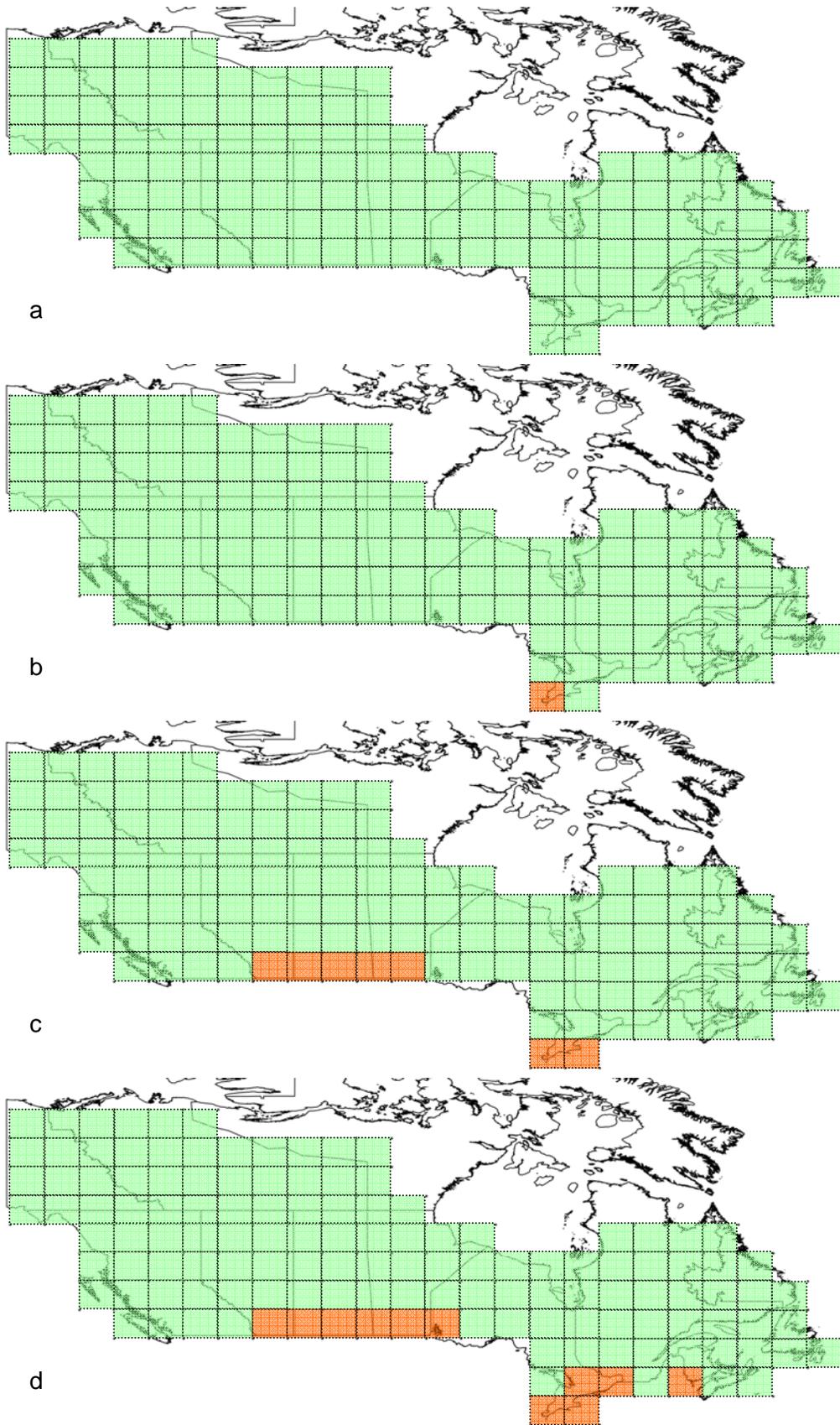


Figure 3.

Green shading:
host tree range and
potential range of
BSLB in Canada.

Orange shading:
Potential range of
BSLB bivoltinism
[mean July and
August temperatures
at least 20°C] in (a)
1990, (b) 2020, (c)
2050, and (d) 2080
under a **low
emissions
(SRES B1)** scenario,
based on **HadCM3**
projections.

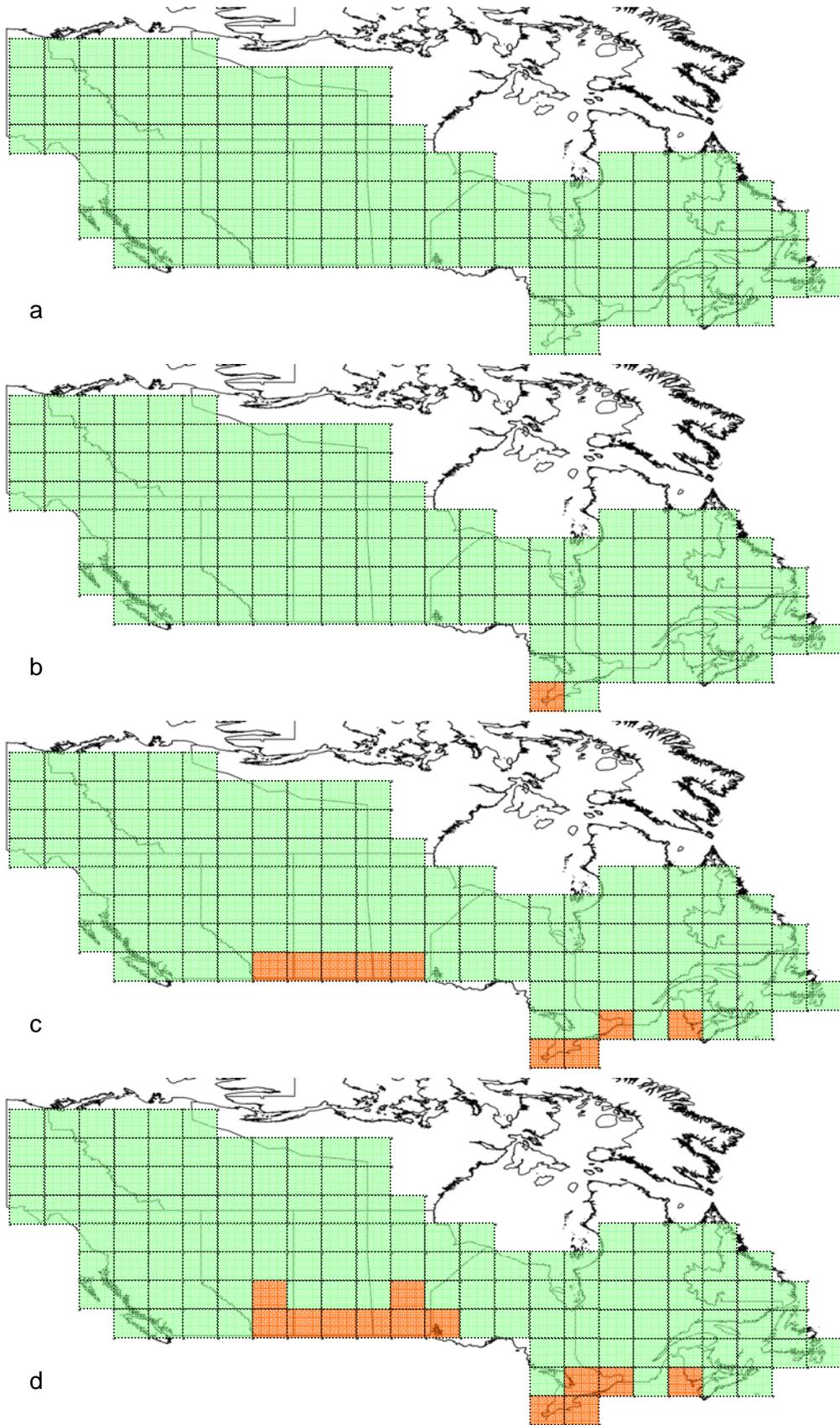


Figure 4.

Green shading:
host tree range and
potential range of
BSLB in Canada.

Orange shading:
Potential range of
BSLB bivoltinism
[mean July and
August temperatures
at least 20°C] in (a)
1990, (b) 2020, (c)
2050, and (d) 2080
under a **medium-
low emissions
(SRES B2)** scenario,
based on **HadCM3**
projections.

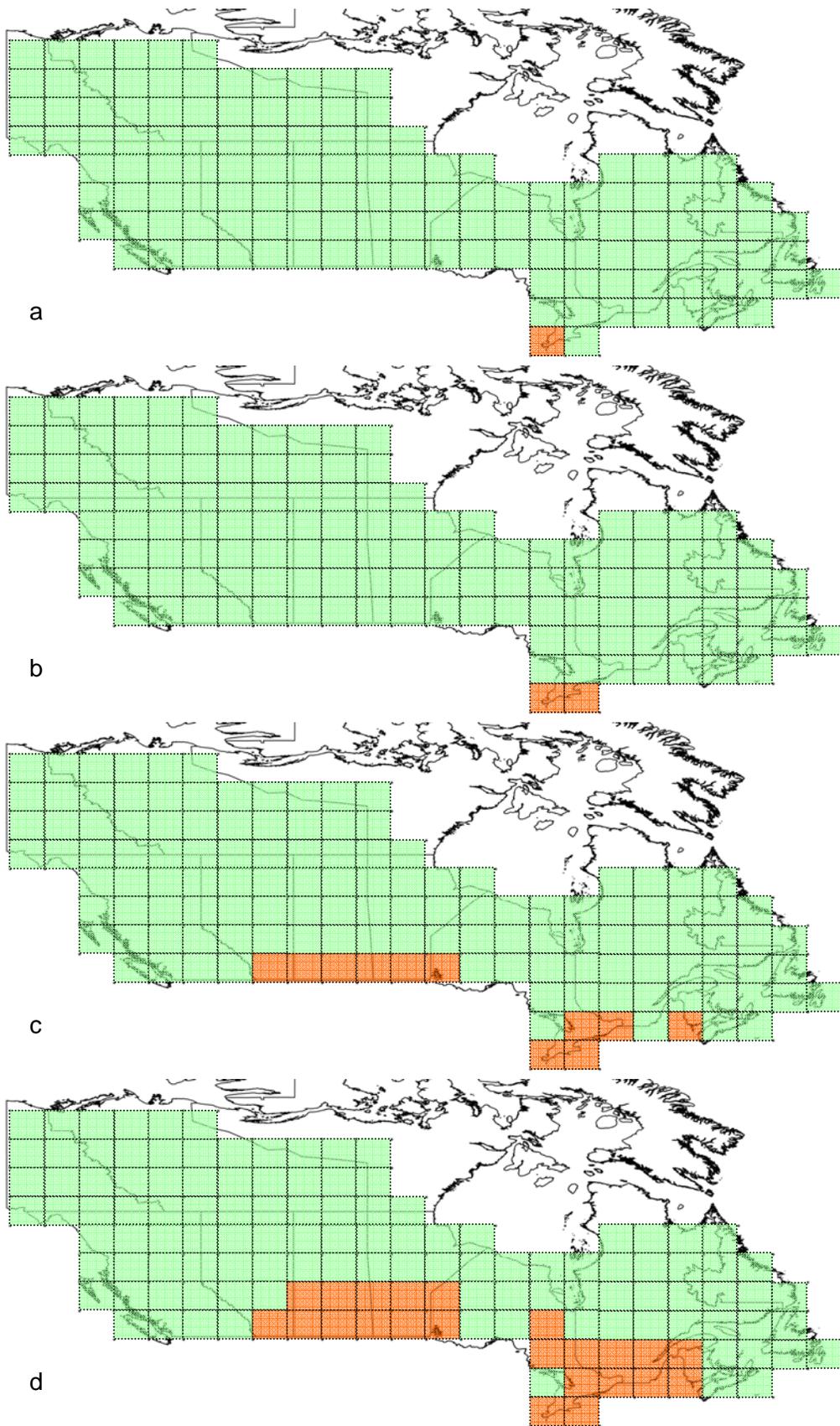


Figure 5.

Green shading:
host tree range and
potential range of
BSLB in Canada.

Orange shading:
Potential range of
BSLB bivoltinism
[mean July and
August temperatures
at least 20°C] in (a)
1990, (b) 2020, (c)
2050, and (d) 2080
under a **medium-
high emissions
(SRES A2)** scenario,
based on **HadCM3**
projections.

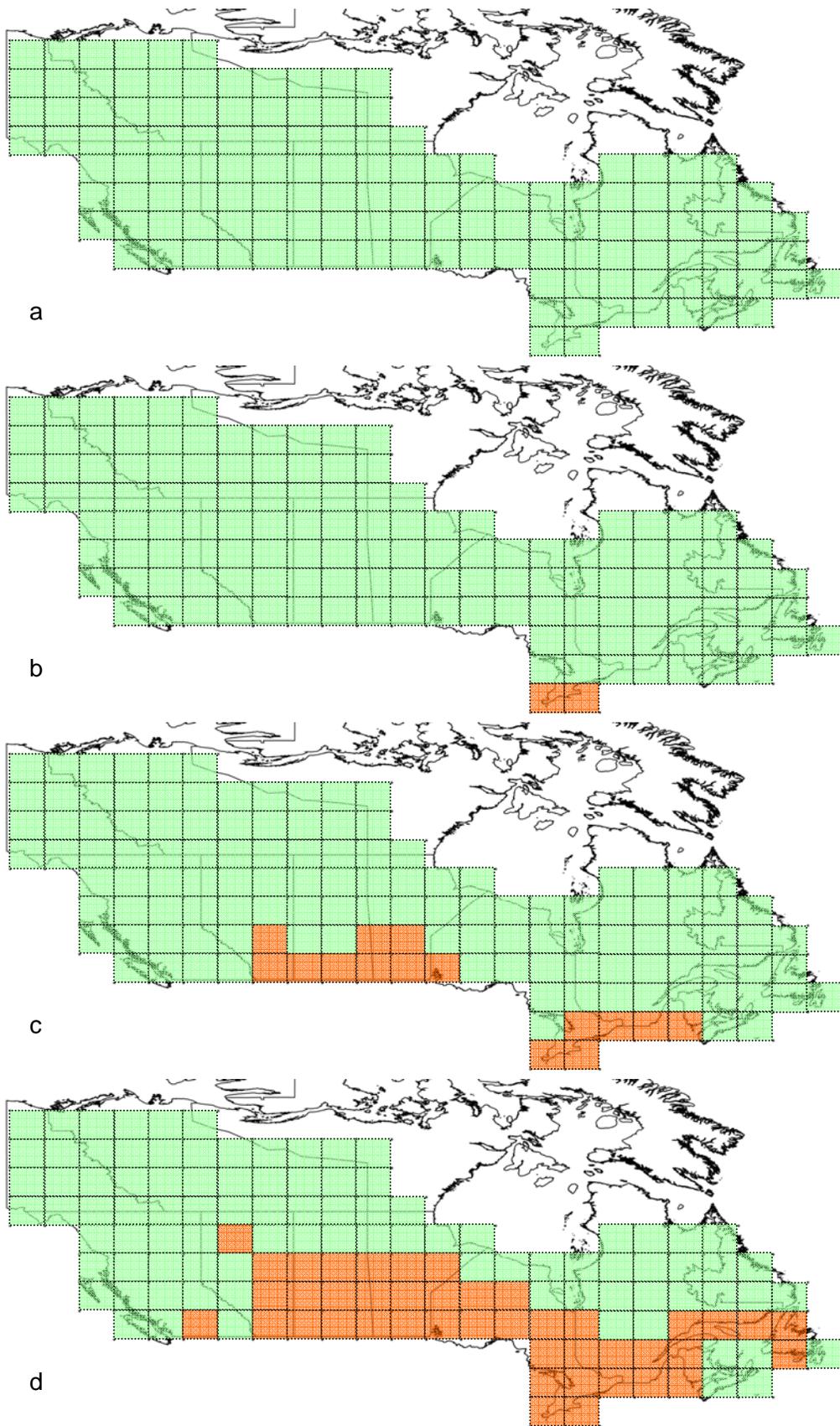


Figure 6.

Green shading:
host tree range and
potential range of
BSLB in Canada.

Orange shading:
Potential range of
BSLB bivoltinism
[mean July and
August temperatures
at least 20°C] in (a)
1990, (b) 2020, (c)
2050, and (d) 2080
under a **high
emissions
(SRES A1)** scenario,
based on **HadCM3**
projections.

Figure 7.

Green shading:
host tree range and
potential range of
BSLB in Canada.

Orange shading:
Potential range of
BSLB bivoltinism
[mean July and
August temperatures
at least 20°C] in (a)
1990, (b) 2020, (c)
2050, and (d) 2080
under a **medium-
low emissions
(SRES B2)**
scenario, based on **CGCM2**
projections.

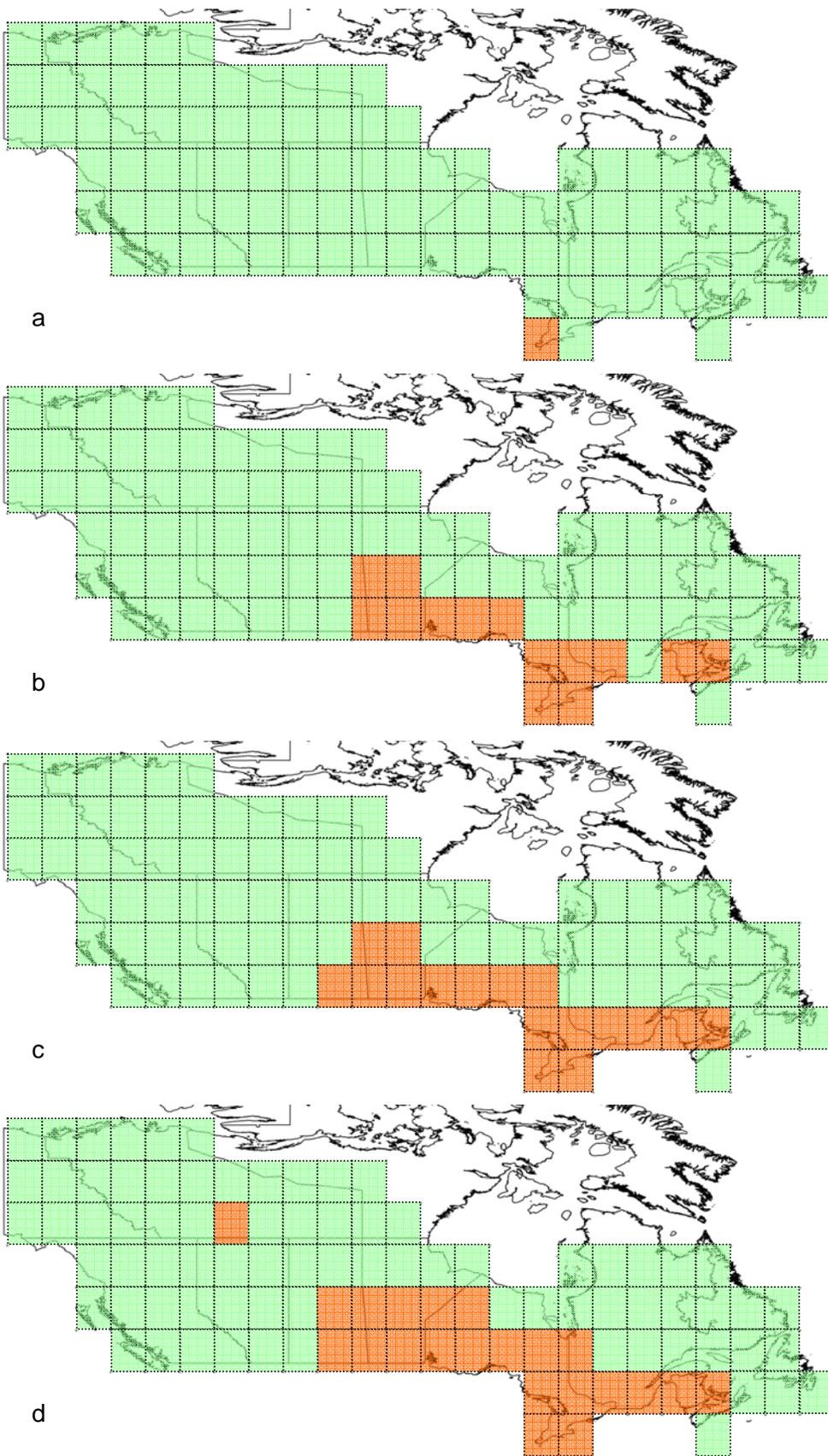
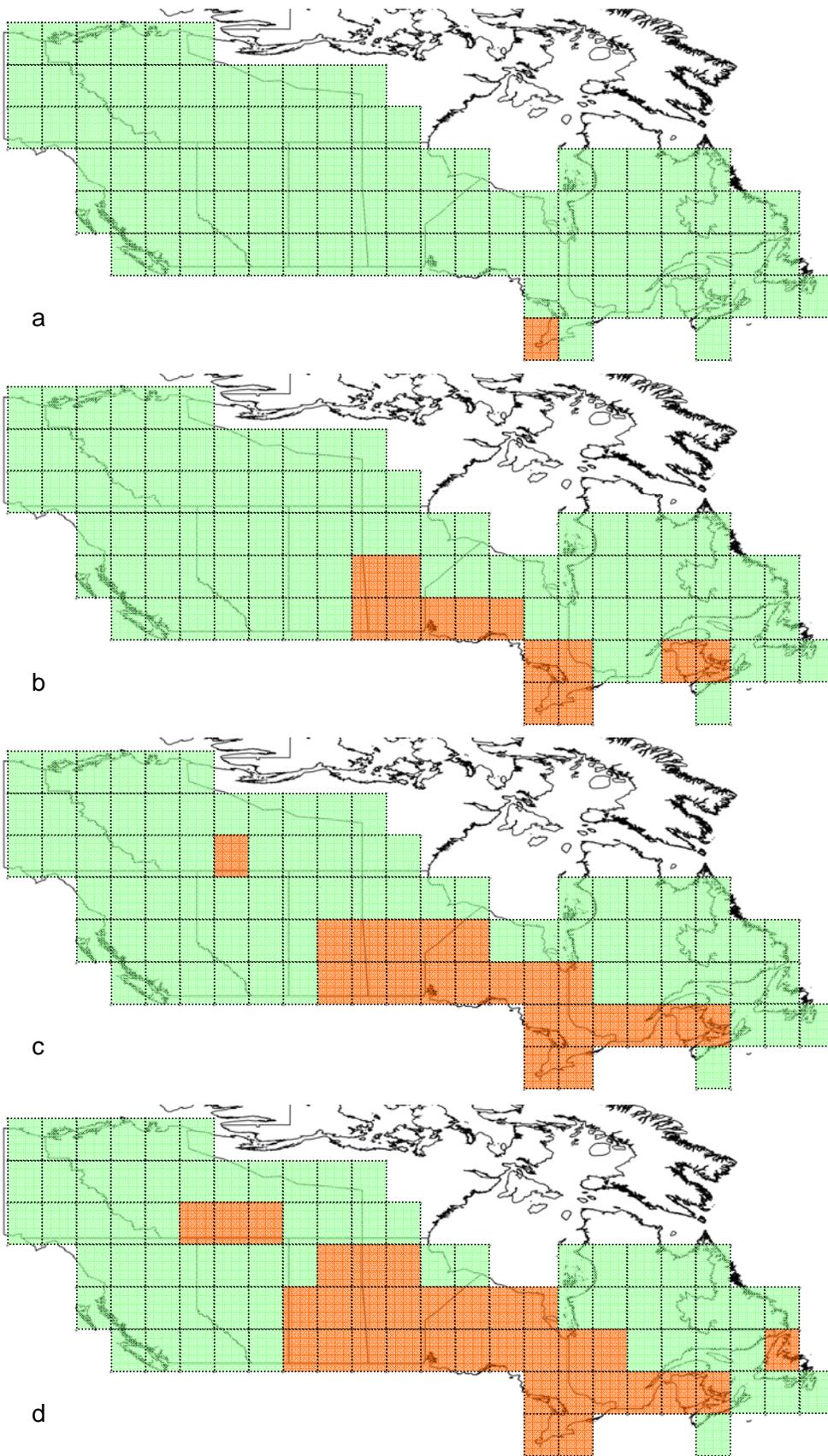


Figure 8.

Green shading:
host tree range and
potential range of
BSLB in Canada.

Orange shading:
Potential range of
BSLB bivoltinism
[mean July and
August temperatures
at least 20°C] in (a)
1990, (b) 2020, (c)
2050, and (d) 2080
under a **medium-
high emissions
(SRES A2)**
scenario, based on **CGCM2**
projections.



5.4 European woodwasp (*Sirex noctilio*) (EWW)

The European woodwasp (EWW) attacks pine trees (*Pinus* spp.), particularly those in the 2 or 3 needle group (P. deGroot, pers. comm.). Oviposition occurs in the trunks of live trees as well as dying and dead trees and logs, and during the process the woodwasp deposits a phytotoxic mucus along with propagules of a fungus into the wood (Bain 2005). The mucus kills the tree's sapwood, the fungus grows on the dead tissue, and the wasp larvae feed on the fungus. Trees are frequently, but not always, killed by the attack; vulnerability depends on tree vigour. Generally the first trees attacked are weakened or dying; as populations of the woodwasp grow, healthy trees are overcome as well (Ciesla 2003). Based on its native range in North Africa, Europe, Mongolia, and Siberia, the insect could likely establish across the geographic range of its hosts in North America.

In its native range, EWW is considered a minor forest pest; however, where it has invaded new territory, particularly in the Southern Hemisphere, it has become a major pest, particularly of pine plantations (Ciesla 2003). In New Zealand, outbreaks in the middle of the last century were associated with drought and overstocking. Biocontrol and silviculture to improve tree vigour have been used to control the wasp with some success (Bain 2005).

EWW is frequently intercepted at U.S. ports of entry, including in northeastern states (New York and New Jersey), and has recently (September 2005) been found in southern Ontario (K. Marchant and P. deGroot, pers. comm.). In Canada, the extensive jack pine and lodgepole pine forests of the north and west would be at risk if the insect were to become established (Ciesla 2003).

In cold climates, the woodwasp requires two years to complete a generation (Ciesla 2003; Madden 1981). In Canada, the current climate is thought to be suitable for the species over much of the range of pine. According to our predicted climate envelopes for this species, in a warming climate, EWW will still likely experience a semivoltine life cycle (two years per generation) over most of its host range in Canada; in limited areas it may be able to produce one generation per year (Figures 9 to 14). An accumulation of 2500 degree days over 6.8°C are required for the complete development of a generation (Madden 1981).

In Canada's temperate forest ecosystems, attacks on red pine (*Pinus resinosa*), a 2-needled pine, could lead to increased forest homogeneity and decreased habitat value for certain species. In these ecosystems, the replacement of pine with hardwoods could result in more C stored in tree biomass over the long-term, as pine wood is less dense than most hardwoods (e.g. pine wood holds about 200 kgC/m³, while sugar maple holds about 350 kgC/m³). However, the generally more rapid decomposition of hardwood litter could result in a smaller amount of C being stored in the forest floor. Red pine is a common plantation species in southern Ontario; extensive plantations of this species were established in the early to mid 1900s to combat erosion on sandy agricultural land. These plantations may be vulnerable to the woodwasp, with the risk of losing the important services, such as habitat and erosion control, that they provide on the agricultural landscape.

This insect would likely cause the most damage in the forests of boreal and western Canada, where large expanses of forests exist that are dominated by jack pine (*Pinus banksiana*) and lodgepole pine (*P. contorta*). Outbreaks of this species that cause high pine mortality rates could shift forest age-class structure leading to a decline in forest C stocks over the landscape. Replacement of jack pine or lodgepole pine with boreal hardwoods such as aspen (*Populus tremuloides*) or birch (*Betula papyrifera*) could mean less fire risk and more C storage on landscape due to lower disturbance frequency. Jack pine mortality could accelerate succession by releasing the understory black spruce cohort, if present. Boreal pine ecosystems provide important habitat for wildlife (e.g. jack pine/lichen forests are important for woodland caribou), and so outbreaks of EWW have the potential to affect biodiversity.

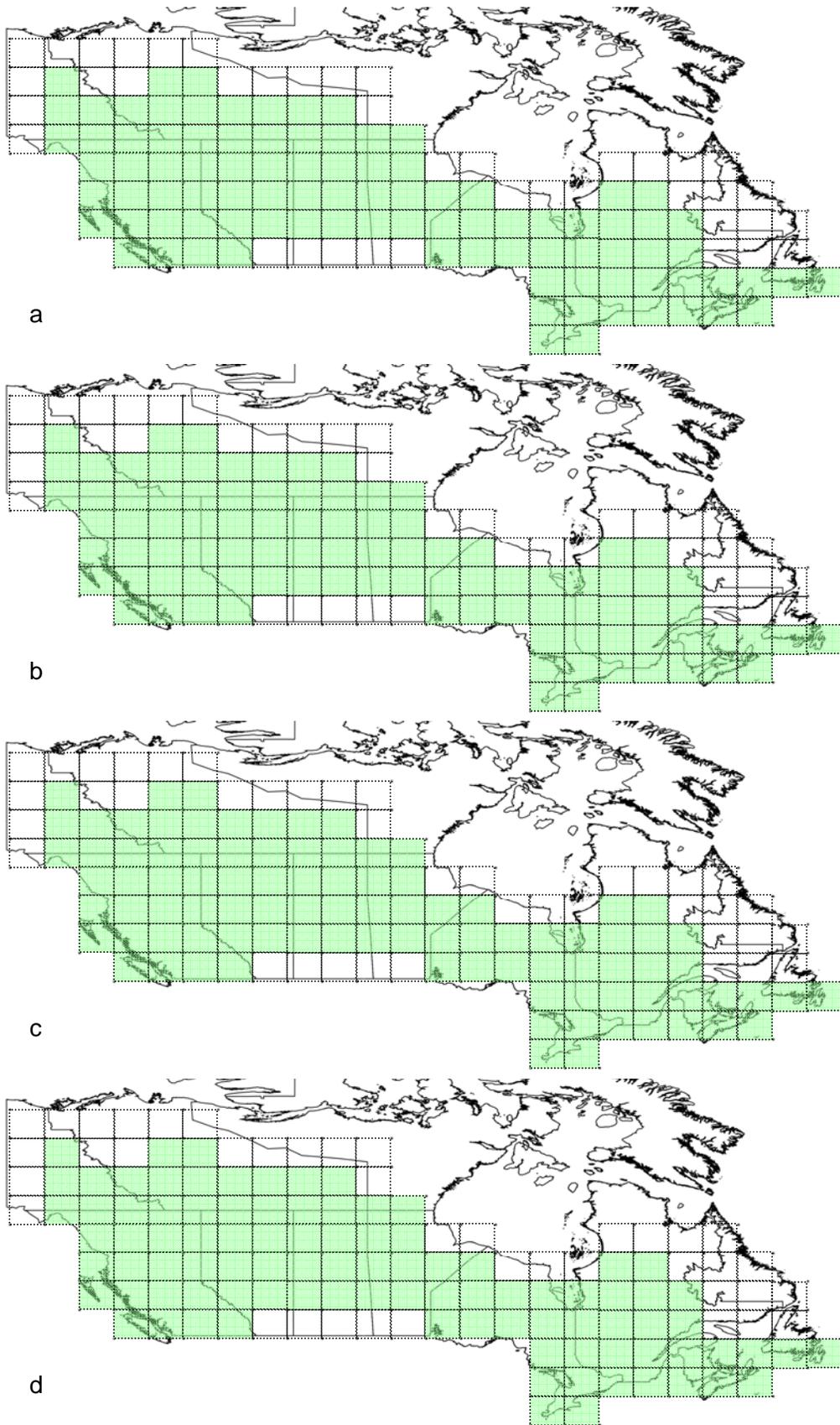


Figure 9.

Green shading:
host tree range and
potential range of
EWW in Canada.

Orange shading:
Potential range of
EWW univoltinism
[at least 2500
degree-days over
6.8°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **low emissions**
(SRES B1) scenario,
based on **HadCM3**
projections.

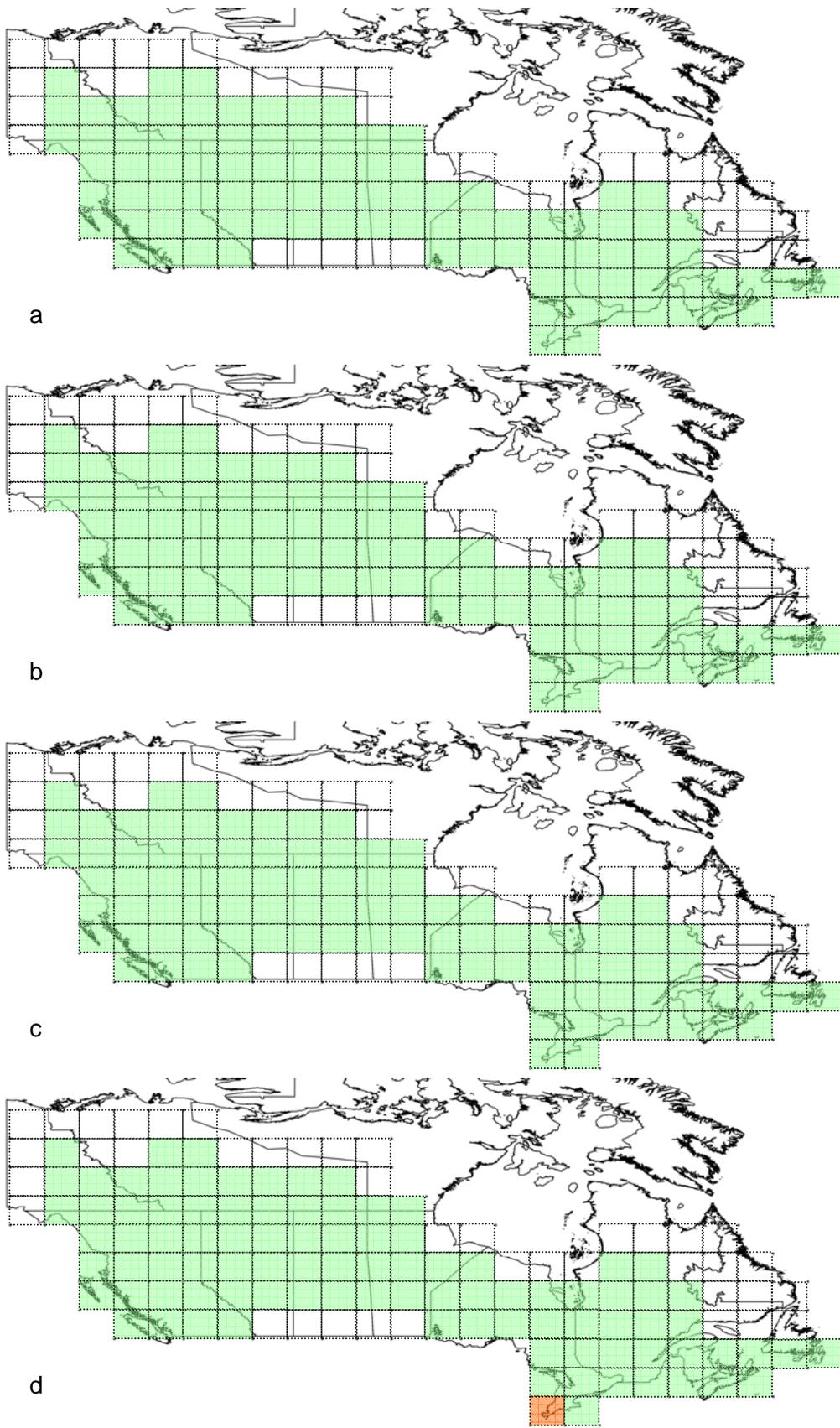


Figure 10.

Green shading:
host tree range and
potential range of
EWW in Canada.

Orange shading:
Potential range of
EWW univoltinism
[at least 2500
degree-days over
6.8°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-low
emissions
(SRES B2)** scenario,
based on **HadCM3**
projections.

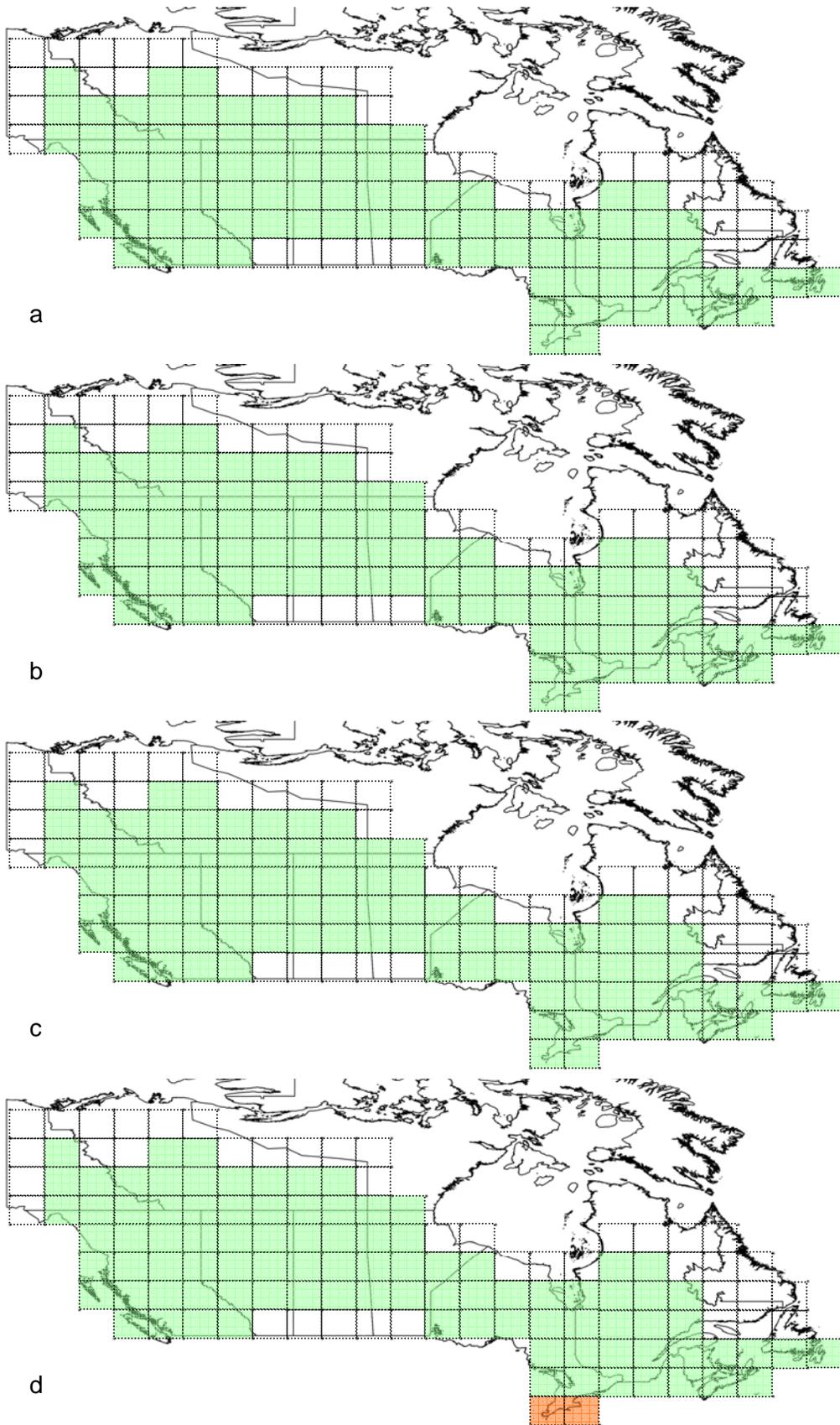


Figure 11.

Green shading:
host tree range and
potential range of
EWW in Canada.

Orange shading:
Potential range of
EWW univoltinism
[at least 2500
degree-days over
6.8°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-high
emissions
(SRES A2)** scenario,
based on **HadCM3**
projections.

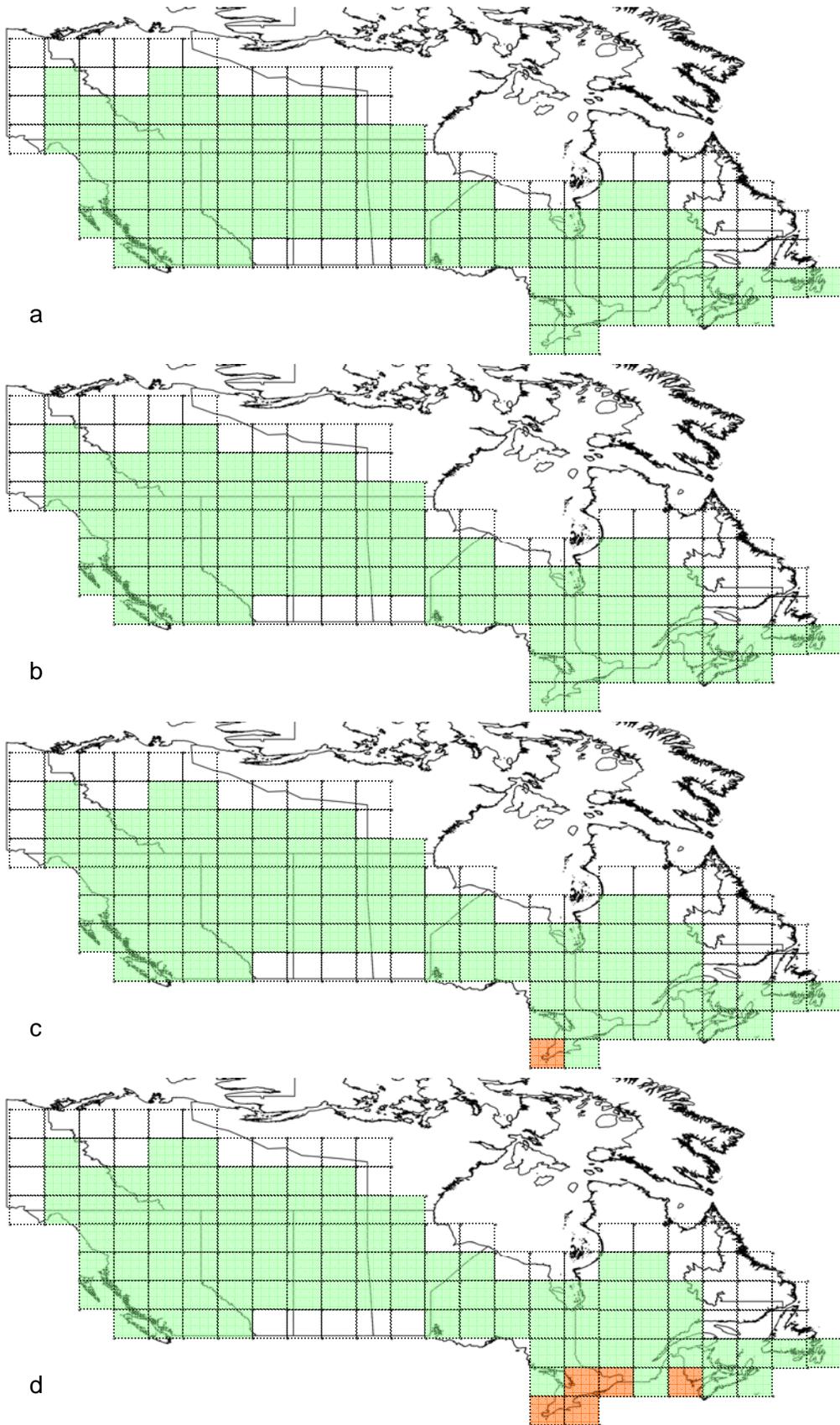


Figure 12.

Green shading:
host tree range and
potential range of
EWW in Canada.

Orange shading:
Potential range of
EWW univoltinism
[at least 2500
degree-days over
6.8°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **high emissions
(SRES A1)** scenario,
based on **HadCM3**
projections.

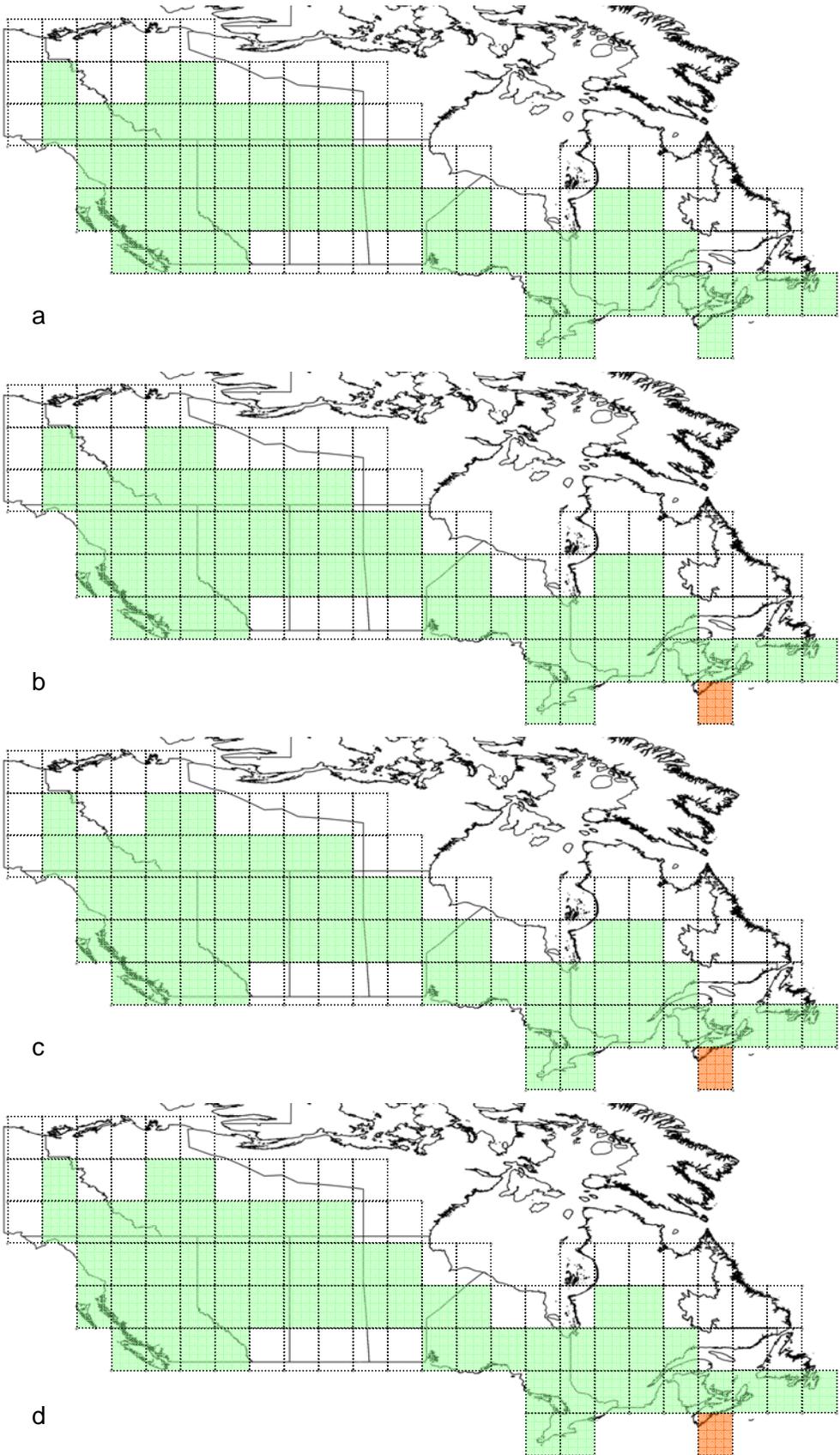


Figure 13.

Green shading:
host tree range and
potential range of
EWW in Canada.

Orange shading:
Potential range of
EWW univoltinism
[at least 2500
degree-days over
6.8°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-low
emissions
(SRES B2)** scenario,
based on **CGCM2**
projections.

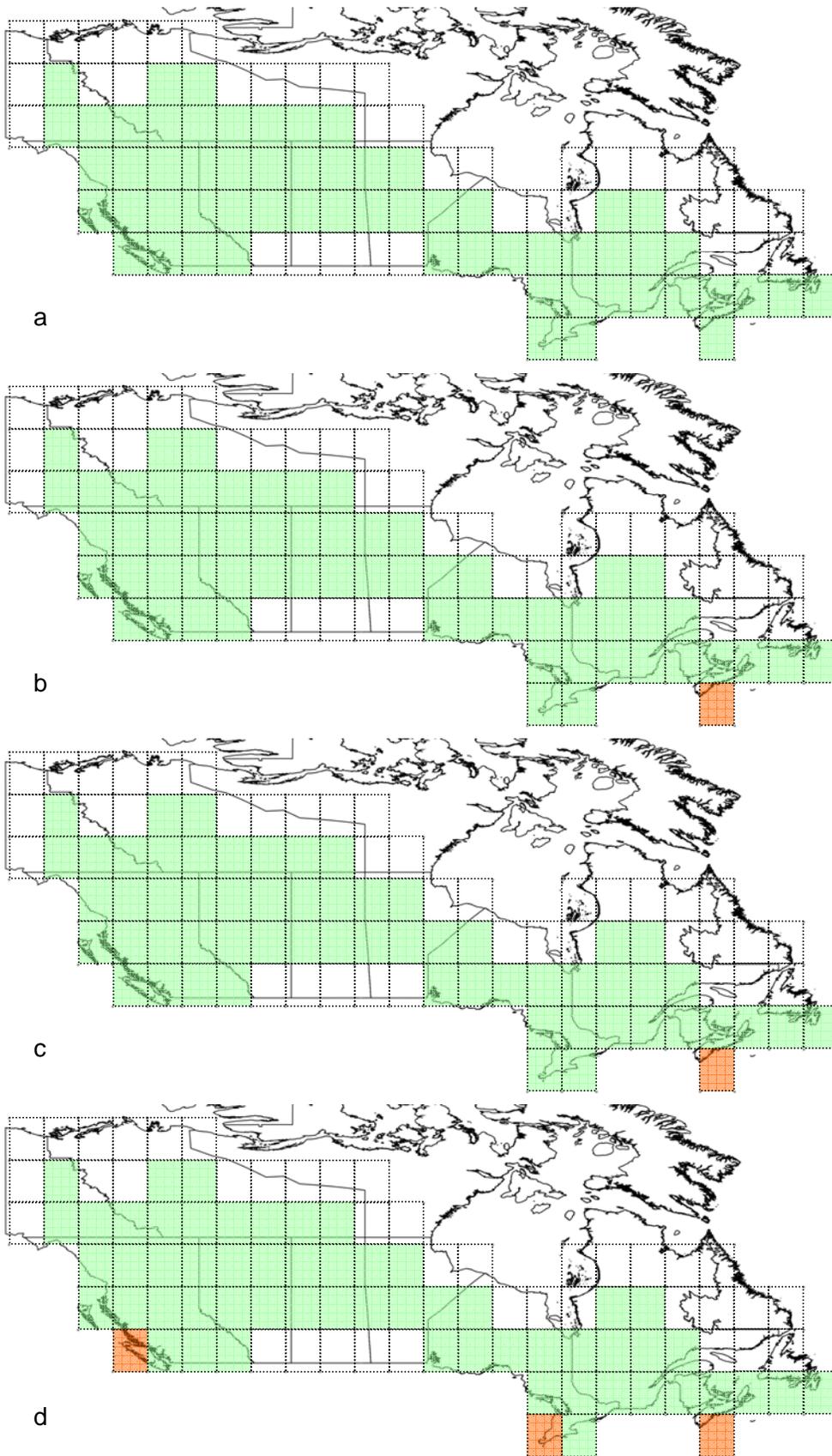


Figure 14.

Green shading:
host tree range and
potential range of
EWW in Canada.

Orange shading:
Potential range of
EWW univoltinism
[at least 2500
degree-days over
6.8°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-low
emissions
(SRES A2)** scenario,
based on **CGCM2**
projections.

5.5 Asian gypsy moth (*Lymantria dispar*, Asian biotype) (AGM)

The Asian gypsy moth (AGM) belongs to the same species as its European counterpart, yet differs in important features that give it the potential to be even more of a threat to North American forests than its cousin (Wallner 2000). Unlike the female adults of the European biotype, which is already a major defoliator of hardwood forests in the northeastern U.S. and southeastern Canada (Wallner 2000, Matsuki 2001), AGM females can fly. AGM also has a wider host range (more potential hosts); its preferred host species are oaks (*Quercus* spp.), poplars (*Populus*), lindens (*Tilia* – in Canada, basswood), birches (*Betula*), larches (*Larix*), and elms (*Ulmus*), and it will also attack other conifers if mixed in with these species.

AGM has been introduced several times into North America, and so far eradication efforts near the source of the introductions have been successful, and have prevented the insect from becoming established. In Canada, AGM was reported infesting trees in Vancouver in 1991 and was subsequently eradicated (Liebhold *et al.* 1995). Future introductions are very likely (Gray and Keena 2005), and AGM has been deemed a very high risk organism for establishment and spread in North America, with potentially severe environmental consequences (Wallner 2000).

Outbreaks of AGM are episodic, rather than cyclical, and last from 1 to 3 years (Elkington and Liebhold 1990, Wallner 2000). Trees can be killed if weakened by repeated defoliation or other stresses, and accordingly, reductions in vigour and growth caused by defoliation can make trees vulnerable to attack by other insects and pathogens. Warm, dry weather is one factor associated with the occurrence of outbreaks (Elkington and Liebhold 1990).

AGM is native to temperate and boreal forests throughout Asia, as far north as 60 degrees latitude. Climate conditions in its native range suggest that it may be capable of successful establishment in colder climates than the European gypsy moth, and it also has less stringent diapause requirements (V. Nealis, pers. comm.), meaning that it can also establish where winters are not cold or long enough for its European counterpart (Gray and Keena 2005). Very cold temperatures (<-25°C) cause high egg mortality, but egg masses can survive if insulated by snow (eggs are laid on rocks and logs as well as on tree trunks and branches), so extremely cold winter temperatures may limit populations but do not guarantee their complete absence (Elkington and Liebhold 1990). Small mammals are important predators of gypsy moth larvae in general, and while avian predation exerts strong controls on gypsy moth in parts of Asia, this does not occur in North America with the European biotype (Elkington and Liebhold 1990). A nuclear polyhedrosis virus (NPV) is often responsible for the collapse of outbreak-level populations of gypsy moth; the susceptibility of the larvae to the virus is affected by chemistry of the foliage they are feeding on. Higher levels of tannins seem to protect the larvae from the virus (Elkington and Liebhold 1990), possibly explaining the insect's preference for oak. If climate or atmospheric change alters tannin production in the gypsy moth's hosts, this could influence its susceptibility to an important control mechanism.

The Asian gypsy moth is a univoltine species (one generation per year) (J. Regniere, pers. comm.), requiring 1700 degree days above 1°C to complete its life cycle (estimated by Matsuki *et al.*, 2001, from data on developmental rates of each life stage, and knowledge of the northern limits of AGM in its native range). Climate conditions in parts of Canada are already suitable for the insect to become established; climate change expands its climatic envelope to encompass most of the range of even its northernmost hosts (*Populus* spp.) by 2080 (Figures 15 to 20). Matsuki *et al.* (2001), using a more detailed model of the insects' climate requirements and tolerances, predicted its potential range in North America under current climate conditions; our 'baseline' climate envelope corresponds reasonably well to their predictions. Gray and Keena (2005) note that the seasonality of the insect's development must be satisfied for the AGM to establish successfully; for example, eggs need to hatch coincidentally with new foliage emergence. If climate change resulted in a disruption of these patterns, it could conceivably have a detrimental effect on AGM populations.

The European biotype has contributed to reduced oak abundance in forests within its range in North America, resulting in less acorn mast available to wildlife. Also, frass inputs have contributed to nitrogen losses from forest soils, increasing nitrogen in surface waters; AGM is likely to cause similar ecological disruptions (Wallner 2000).

Oak wood has a high specific gravity, and hence stores more carbon per unit volume than many of the hardwoods and softwoods with which it is commonly associated (particularly ash or pine; sugar maple wood is similar in density to oak wood). Also, oak litter is associated with slower decomposition rates compared to associates such as maple and ash, which could result in a larger forest floor C pool.

Widespread poplar mortality in the north could favour associated conifer species such as spruce, possibly resulting in greater susceptibility to fire and less C storage over the forest landscape, although larger soil C stocks could result from increased conifer dominance due to changes in decomposition dynamics.

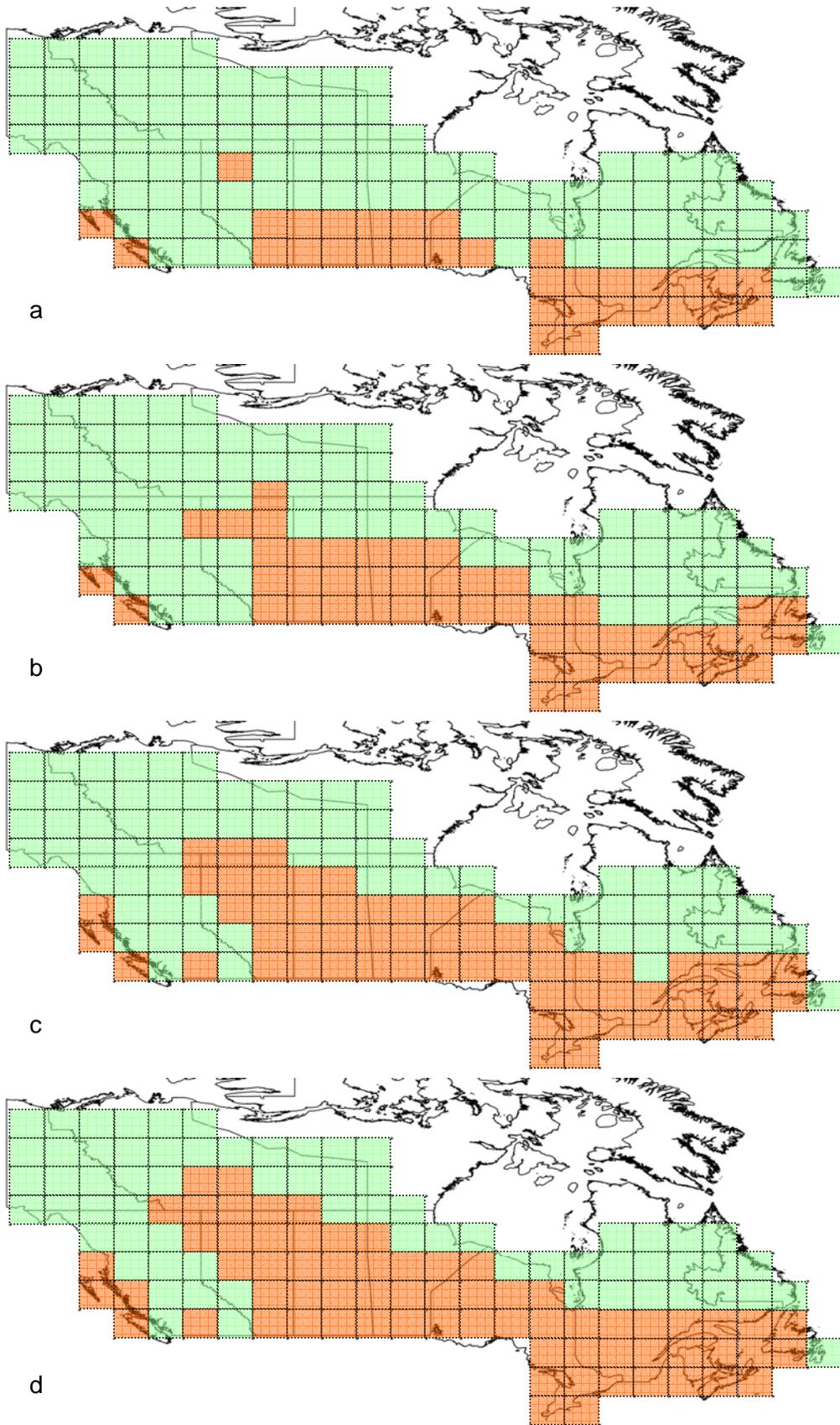


Figure 15.

Green shading:
AGM host tree range
in Canada.

Orange shading:
Potential range of
AGM [equivalent to
potential range of
univoltinism; at least
1700 degree-days
over 1°C] (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **low emissions
(SRES B1)** scenario,
based on **HadCM3**
projections.

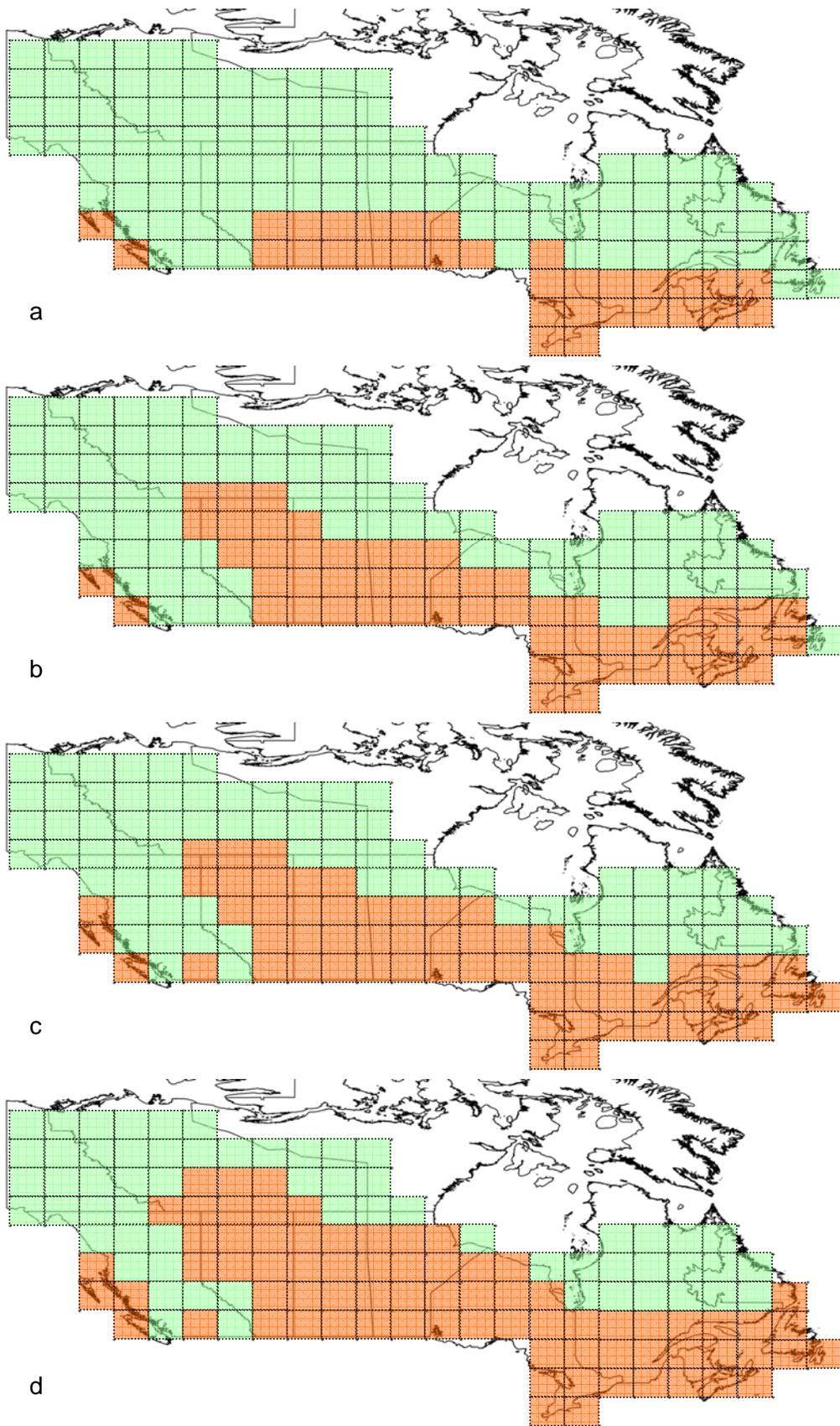


Figure 16.

Green shading:
AGM host tree range
in Canada.

Orange shading:
Potential range of
AGM [equivalent to
potential range of
univoltinism; at least
1700 degree-days
over 1°C] (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under a **medium-low
emissions
(SRES B2)** scenario,
based on **HadCM3**
projections.

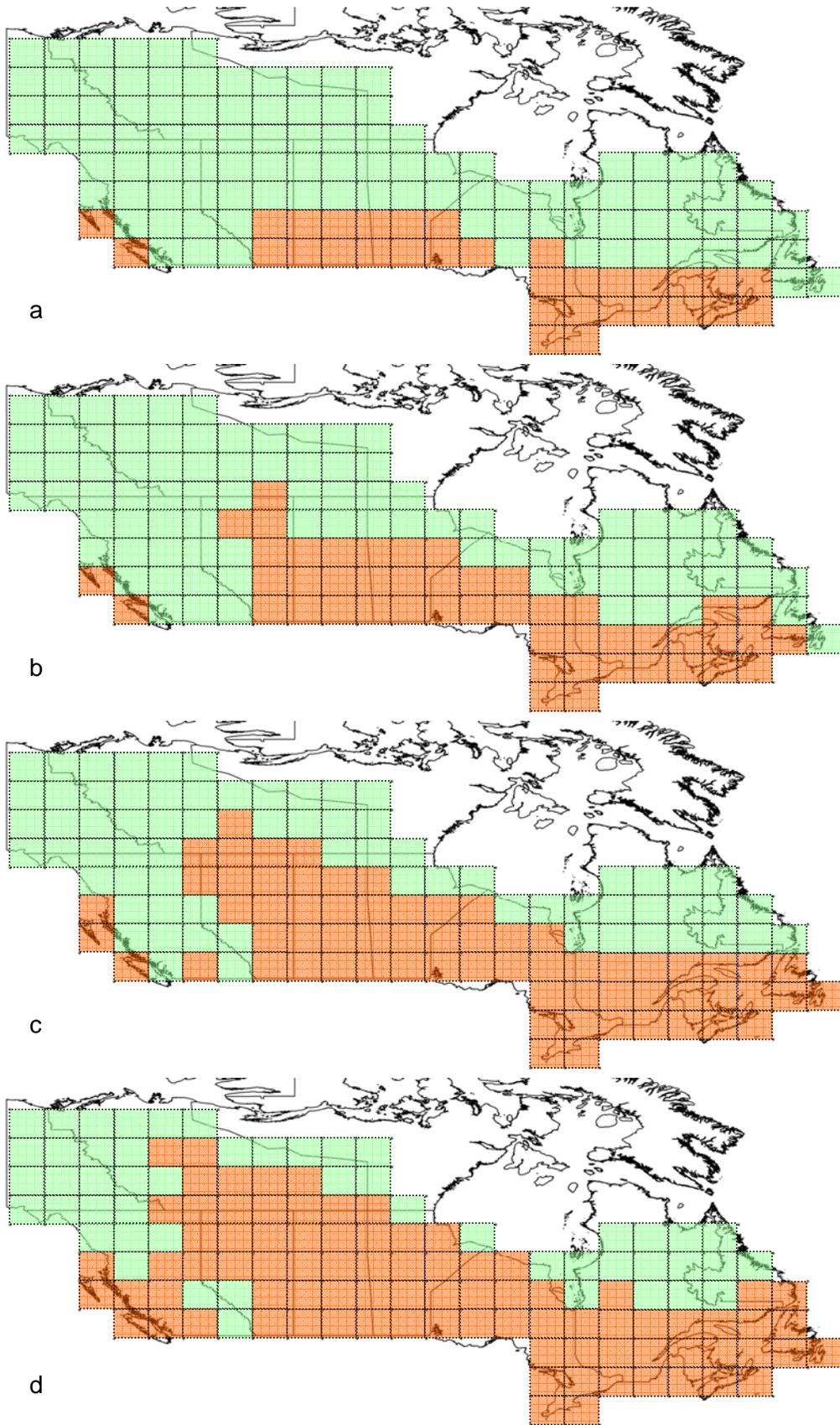


Figure 17.

Green shading:
AGM host tree range
in Canada.

Orange shading:
Potential range of
AGM [equivalent to
potential range of
univoltinism; at least
1700 degree-days
over 1°C] (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-high
emissions
(SRES A2)** scenario,
based on **HadCM3**
projections.

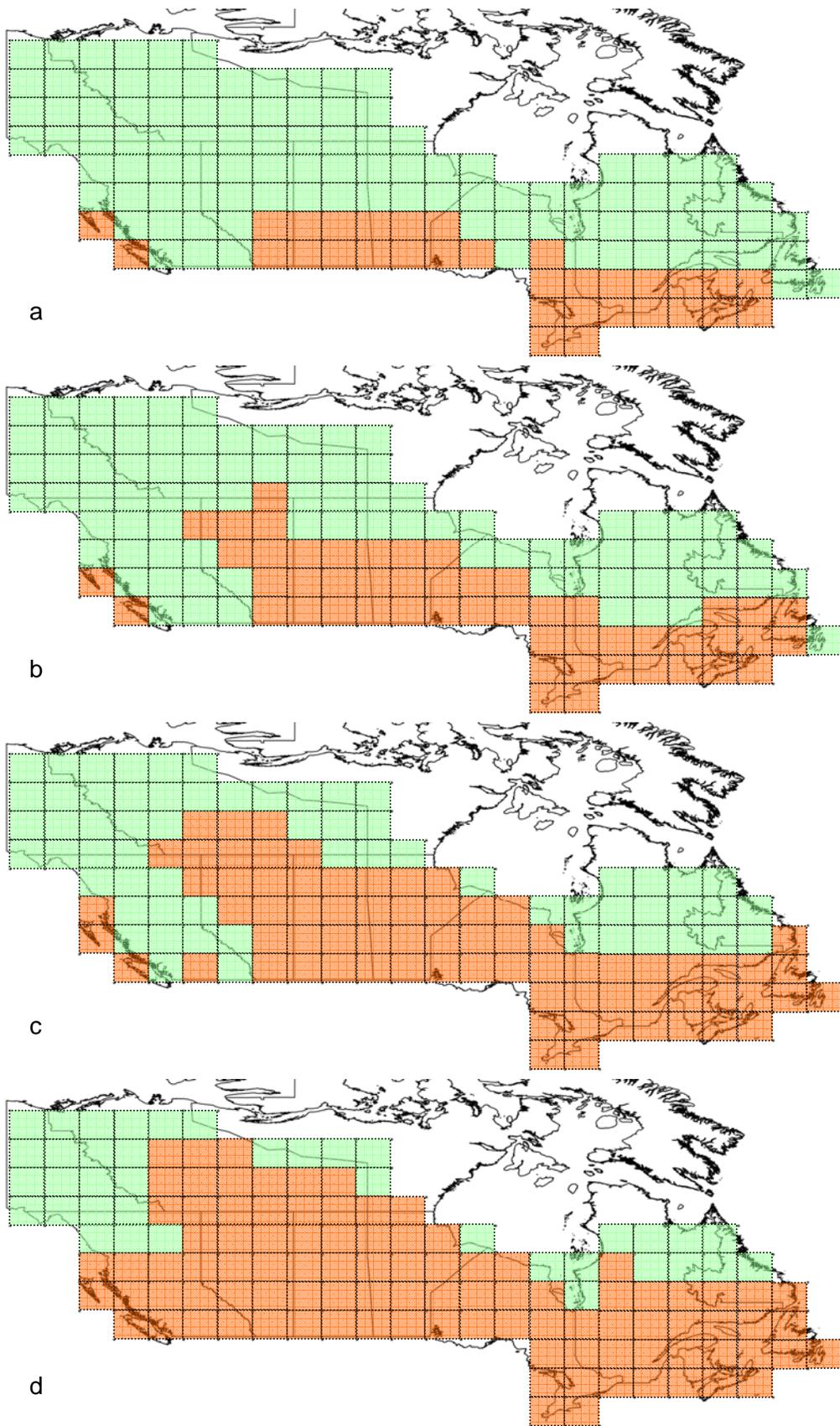


Figure 18.

Green shading:
AGM host tree range
in Canada.

Orange shading:
Potential range of
AGM [equivalent to
potential range of
univoltinism; at least
1700 degree-days
over 1°C] (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **high emissions
(SRES A1)** scenario,
based on **HadCM3**
projections.

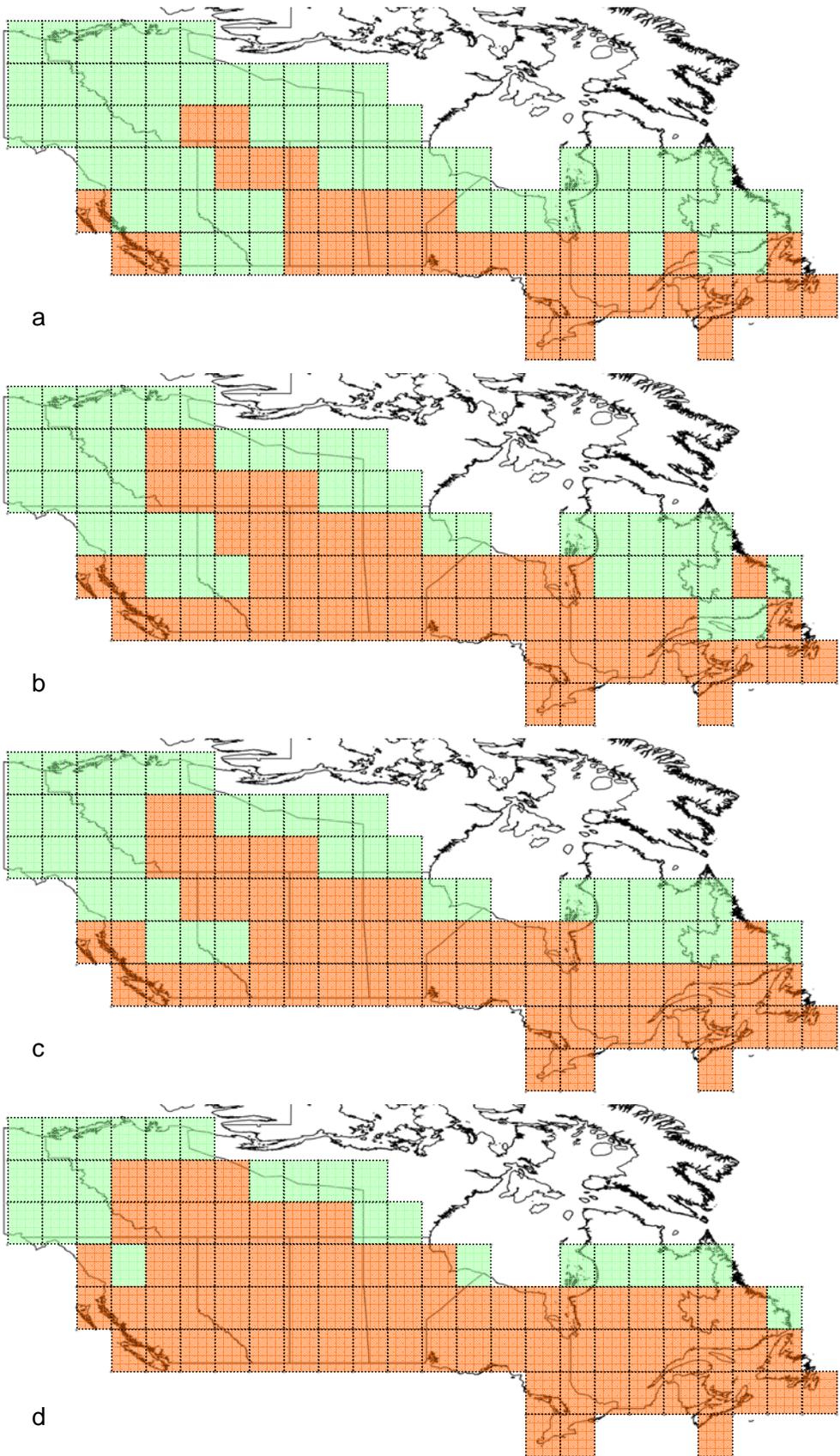


Figure 19.

Green shading:
AGM host tree range
in Canada.

Orange shading:
Potential range of
AGM [equivalent to
potential range of
univoltinism; at least
1700 degree-days
over 1°C] (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under a **medium-low
emissions
(SRES B2)** scenario,
based on **CGCM2**
projections.

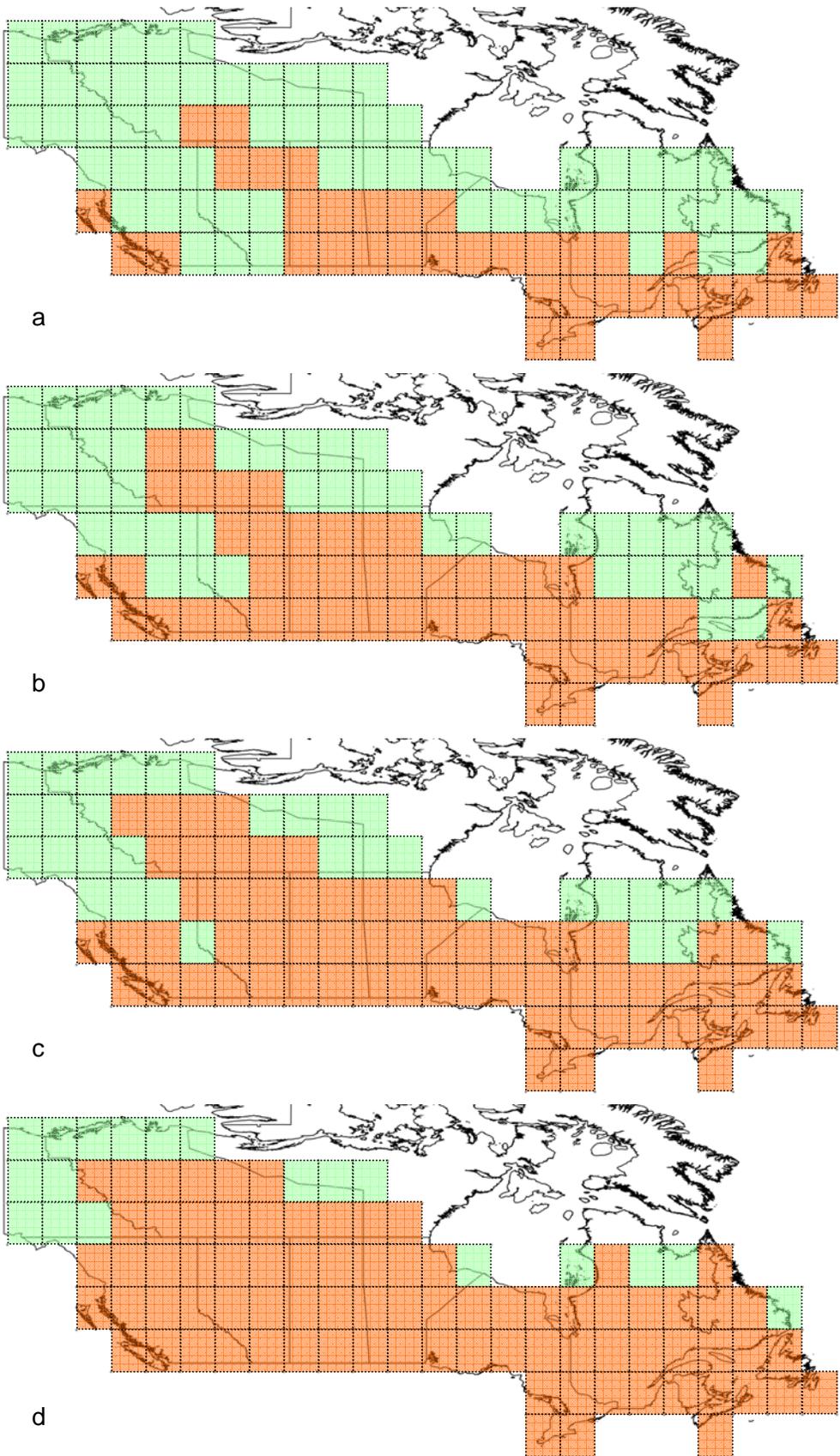


Figure 20.

Green shading:
AGM host tree range
in Canada.

Orange shading:
Potential range of
AGM [equivalent to
potential range of
univoltinism; at least
1700 degree-days
over 1°C] (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-high
emissions
(SRES A2)** scenario,
based on **CGCM2**
projections.

5.6 Asian long-horned beetle (*Anoplophora glabripennis*) (ALB)

The Asian long-horned beetle (ALB) was found infesting urban trees in the north Toronto area in 2003 (Smith 2005); since then many trees have been destroyed in an attempt to eradicate the beetle, and although infestations are still occurring, attempts to limit their spread have been fairly successful thus far (CFIA 2005). ALB has, however, been identified as posing a high risk of establishment and spread in North America (Cavey 2000). Its preferred host genera (*Acer*, *Populus*, and *Salix*) are represented by important tree species in Canada, and its range of hosts appears to be wider than in its native Korea and China; it has attacked *Betula* spp. (birches) and *Fraxinus* spp. (ashes) in North America (Cavey 2000), and there is evidence that red oak (*Quercus rubra*) may also be a potential host (Morewood *et al.* 2005), though this is in dispute (P. deGroot, pers. comm.)

ALB attacks healthy as well as weakened trees (Humphreys 1998), causing mortality in 3 to 5 years (Smith 2005). Adult beetles feed on the bark of twigs, but the main damage is caused by the larvae, which initially feed on cambium tissue, then tunnel into sapwood and even heartwood to feed as they mature, disrupting the vascular system of the tree (Cavey 2000). The relatively long-lived adults (up to two months) can fly several hundred metres in one flight, but seldom take advantage of this ability and are thus relatively slow-spreading (Cavey 2000). The complete life cycle requires 1 to 2 years in its native range, and all life stages except the adult can survive overwintering (CFIA 2005). Based on the climate in its native range, which extends north to Beijing in China, MacLeod *et al.* (2002) estimated that ALB requires 1400 degree days above 5°C to complete a generation. Using a similar approach, Peterson and Scachett-Pereira (2004) determined that a minimum average annual temperature of 2.5°C was required for ALB to become established. These authors predicted that current climate conditions put eastern Canada at risk from ALB, while its potential current range in western Canada would be limited by climate. Our baseline (1990) climate envelope (Figures 21 to 26) matches reasonably well with predictions from Peterson and Scachett-Pereira (2004), but does show a stronger presence of ALB in the south parts of western Canada. By 2080, our predicted high-emissions-scenario climate envelope for this insect extends over much of its host range, except in the far northwestern parts of Canada. It must be noted that there is some observational evidence (J. Turgeon, pers. comm.) that ALB currently requires more than one year to produce one generation in the Toronto area, and that the degree day requirements may therefore be greater than those estimated by Macleod *et al.* (2002).

The establishment and spread of ALB could have serious implications for both temperate forests, in which one of its preferred hosts (*Acer saccharum*) is a major component, and boreal forests, which include another of its preferred hosts (*Populus* spp.) as important and widespread species.

In the boreal forest, attacks on poplar species, particularly *P. tremuloides*, may result in accelerated succession of mixed stands to conifer stands if spruce and fir species are released from competition and increase their dominance in affected stands. A change in microclimate and decomposition dynamics may result in larger soil C pools in the more conifer-dominated stand, but the reduction in hardwood abundance may increase stand flammability and fire risk. If other hardwood associates such as white birch (*Betula papyrifera*) are present and increase in abundance in response to the decline in poplar, different wood densities may influence the amount of C stored in tree biomass (e.g. *B. papyrifera* wood contains about 300 kgC/m³, while *P. tremuloides* wood contains about 200 kgC/m³).

The largest impacts on temperate forests in Canada would likely stem from attacks on sugar maple, a dominant species in the Great-Lakes-St. Lawrence forest region of Ontario, as well as in Quebec and New Brunswick. Sugar maple influences nutrient cycling processes in forests through its effect on soil carbon to nitrogen ratios and nitrification rates (Venterea 2002; Lovett *et al.* 2002). Increased abundance of associated species such as American beech and red oak could result in higher C to N ratios, slower decomposition and less nitrogen leaching and loss from forest soils, possibly resulting in larger soil C stocks. Replacement of sugar maple with white ash would have less of an effect on nutrient cycling rates (Lovett *et al.* 2002).

ALB is considered a major forest pest in China, especially of poplar plantations (Cavey 2000). More consideration may need to be given to the increasing use of poplar plantations for C sequestration and land reclamation programmes in Canada if this pest escapes eradication efforts and spreads in Canada.

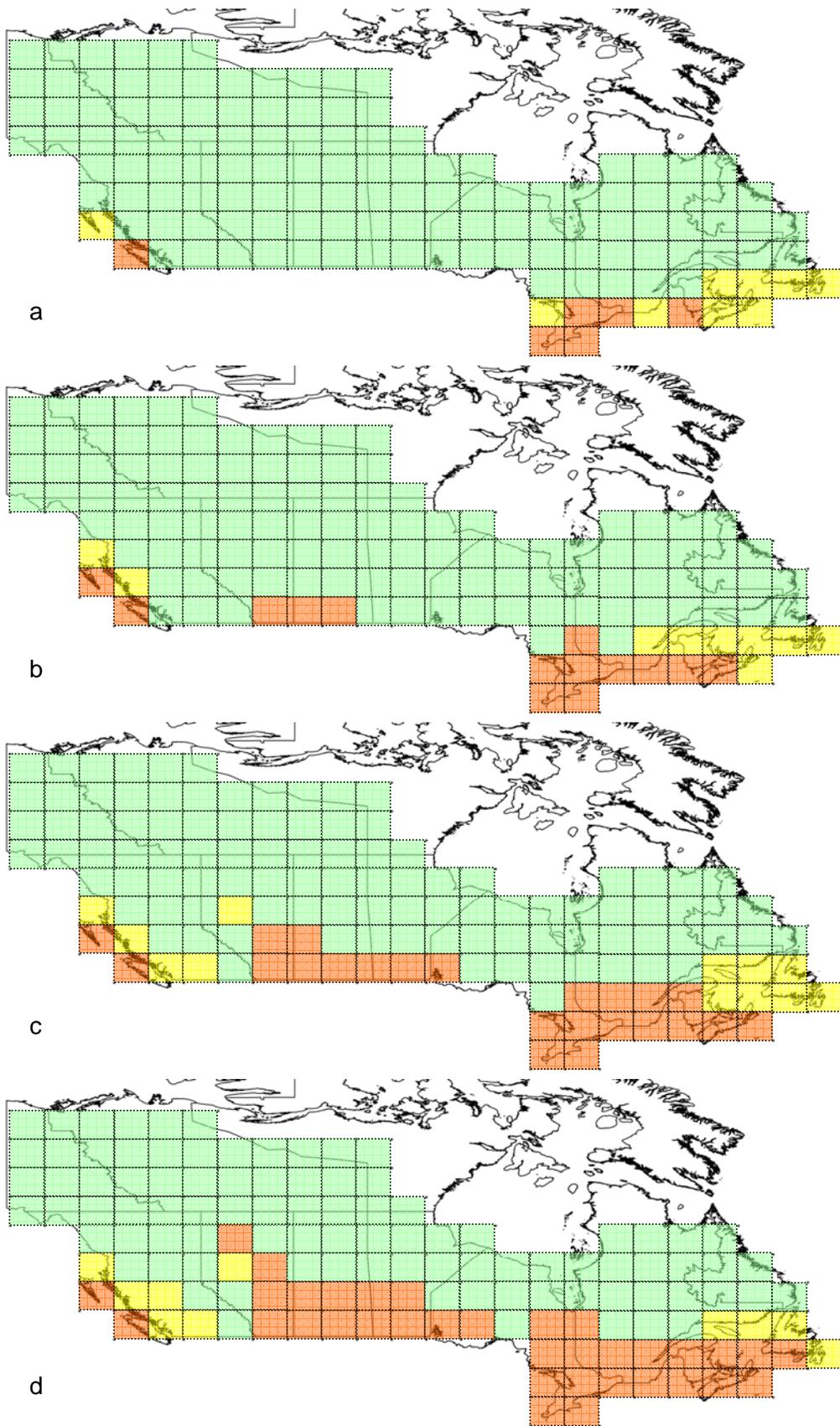


Figure 21.

Green shading:
ALB host tree range
in Canada.

Yellow shading:
Potential range of
ALB [mean annual
temperature at least
2.5°C. Orange
shading: potential
range of ALB
univoltinism [at least
1400 degree days
>5°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **low emissions**
(SRES B1) scenario,
based on **HadCM3**
projections.

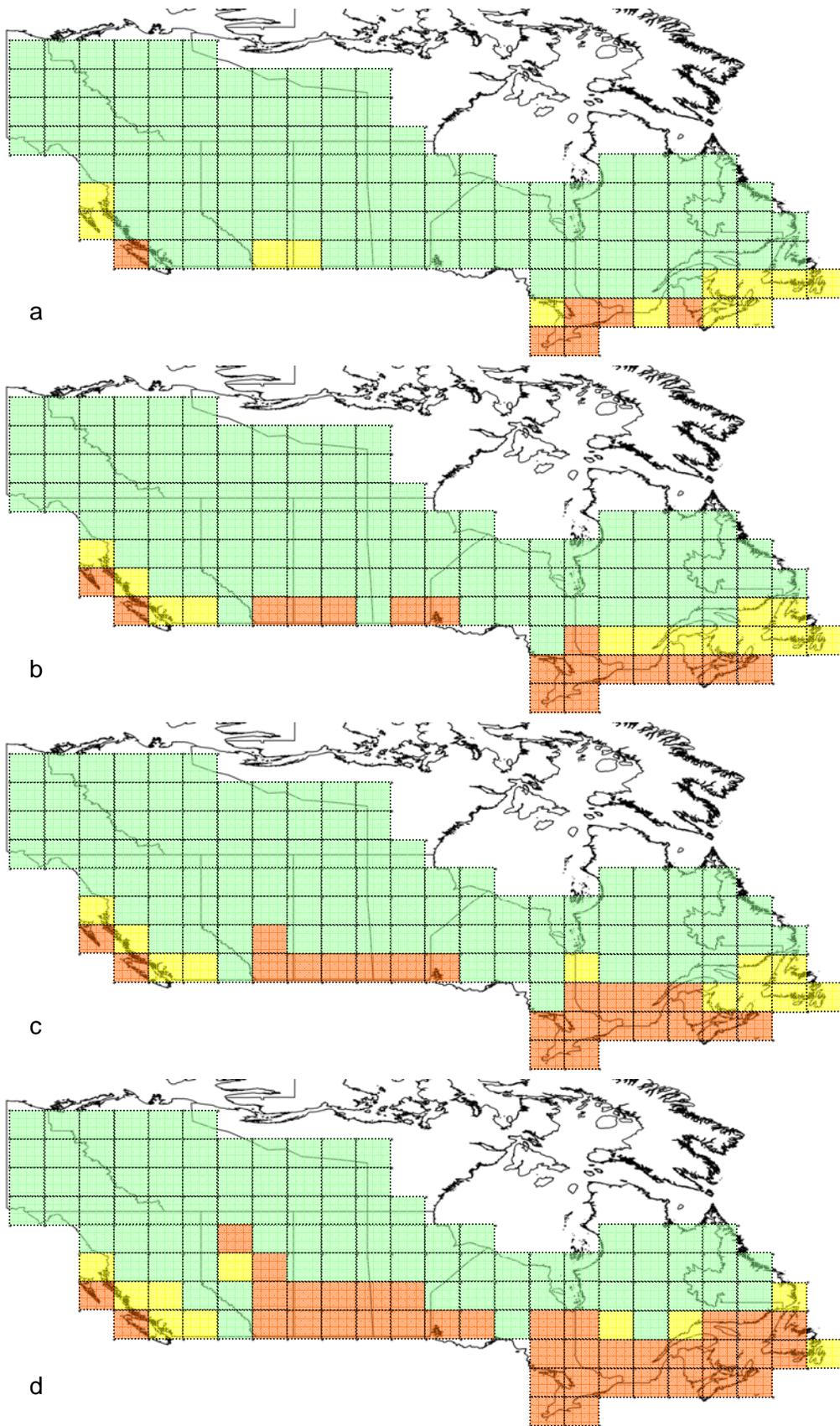


Figure 22.

Green shading:
ALB host tree range
in Canada.

Yellow shading:
Potential range of
ALB [mean annual
temperature at least
2.5°C. Orange
shading: potential
range of ALB
univoltinism [at least
1400 degree days
>5°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-low
emissions
(SRES B2)** scenario,
based on **HadCM3**
projections.

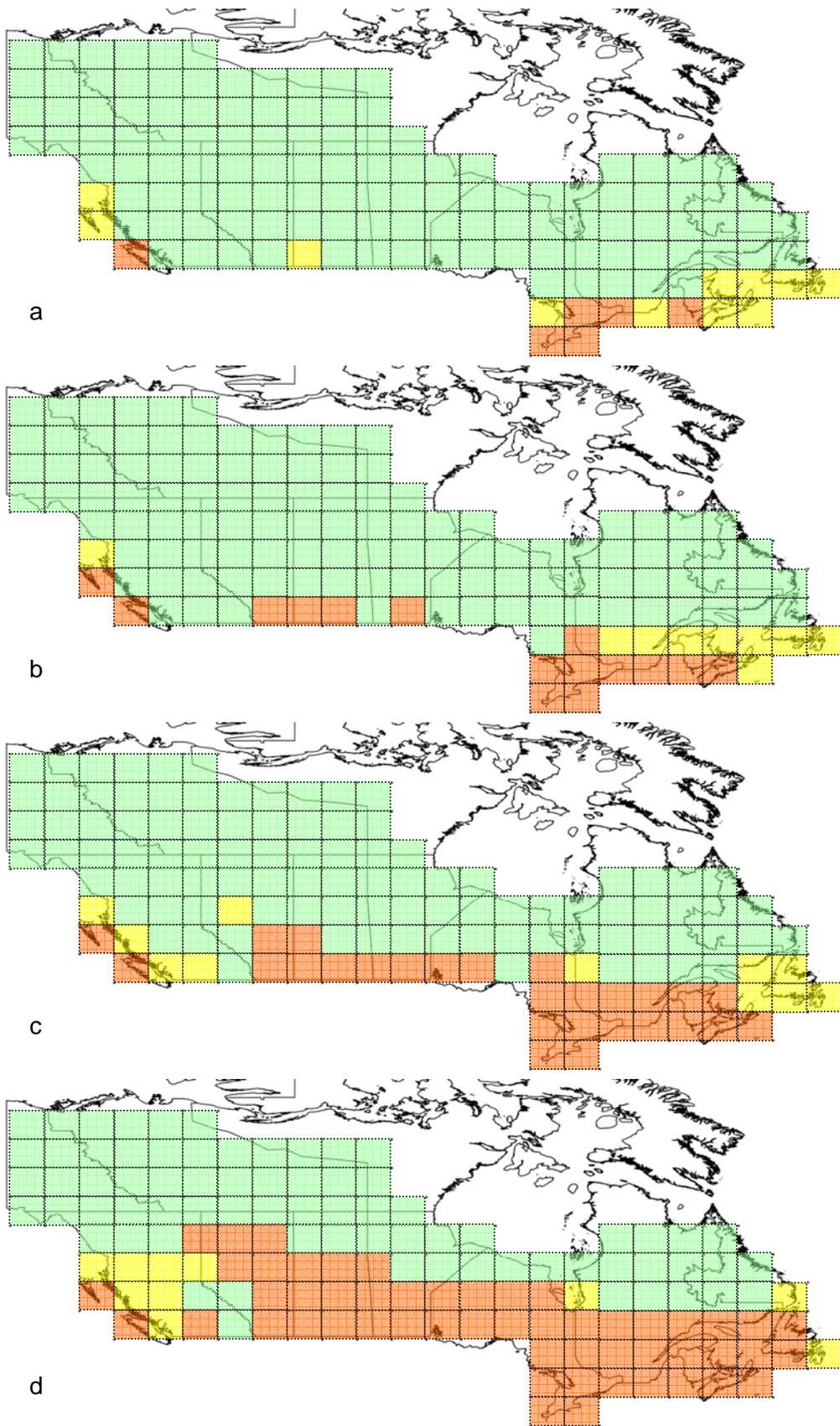


Figure 23.

Green shading:
ALB host tree range
in Canada.

Yellow shading:
Potential range of
ALB [mean annual
temperature at least
2.5°C. Orange
shading: potential
range of ALB
univoltinism [at least
1400 degree days
>5°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-high
emissions
(SRES A2)** scenario,
based on **HadCM3**
projections.

Figure 24.

Green shading:
ALB host tree range
in Canada.

Yellow shading:
Potential range of
ALB [mean annual
temperature at least
2.5°C. Orange
shading: potential
range of ALB
univoltinism [at least
1400 degree days
>5°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **high emissions
(SRES A1)** scenario,
based on **HadCM3**
projections.

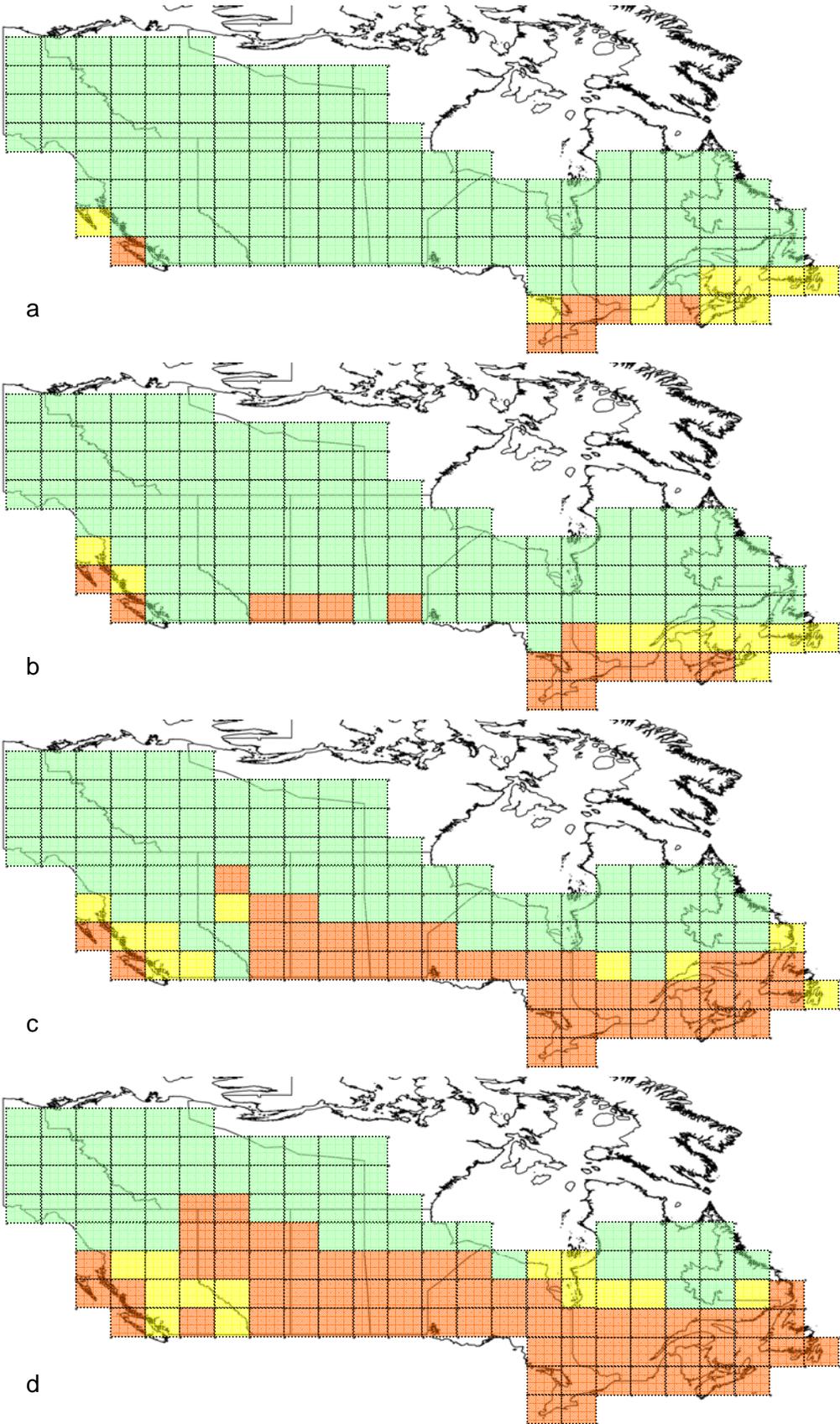


Figure 25.

Green shading:
ALB host tree range
in Canada.

Yellow shading:
Potential range of
ALB [mean annual
temperature at least
2.5°C]; orange
shading: potential
range of ALB
univoltinism [at least
1400 degree days
>5°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-low
emissions
(SRES B2)**
scenario, based on **CGCM2**
projections.

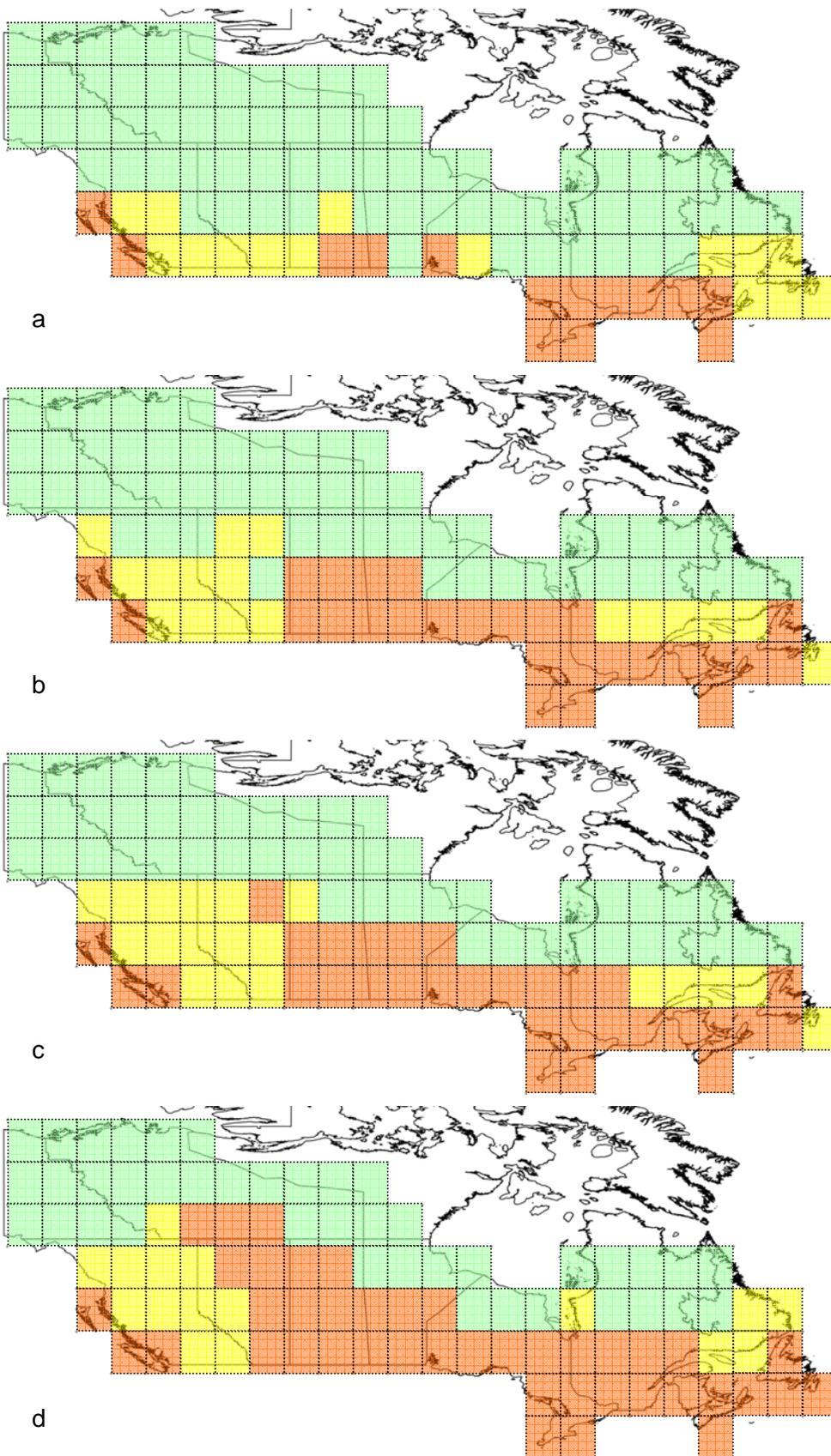
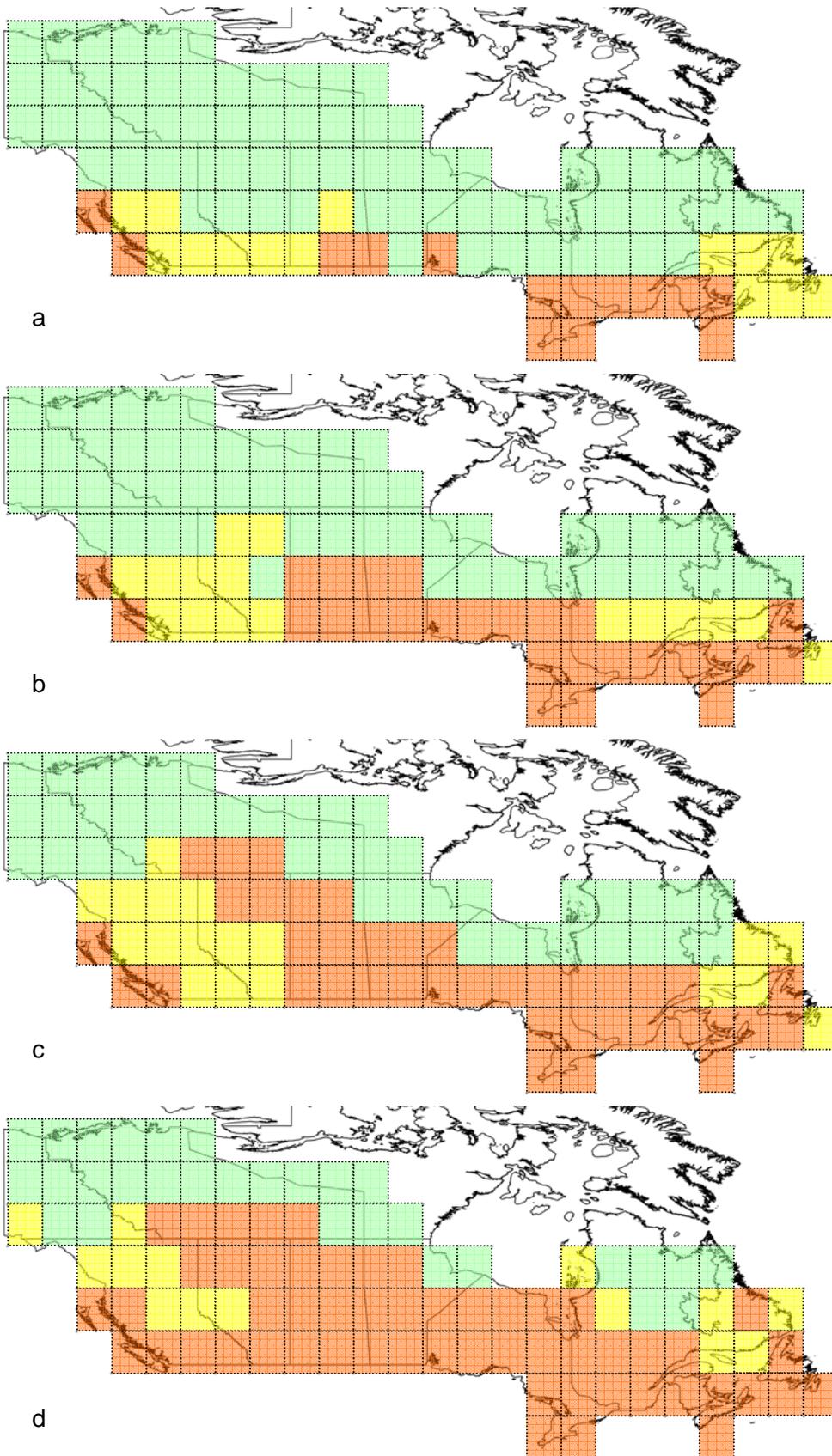


Figure 26.

Green shading:
ALB host tree range
in Canada.

Yellow shading:
Potential range of
ALB [mean annual
temperature at least
2.5°C]; orange
shading: potential
range of ALB
univoltinism [at least
1400 degree days
>5°C] in (a) 1990,
(b) 2020, (c) 2050,
and (d) 2080 under
a **medium-high
emissions
(SRES A2)**
scenario, based on **CGCM2**
projections.



5.7 Emerald ash borer (*Agrilus planipennis*) (EAB)

The emerald ash borer (EAB) is a wood-boring insect; its larvae feed on the cambial tissues (phloem and outer sapwood) of ash trees (*Fraxinus* spp.), usually causing tree mortality within three years. The short-lived adults consume a small amount of tree foliage (Haack *et al.* 2002). The insect occurs in low densities in its native range (Gould *et al.* 2005), which includes northeastern China, Taiwan, Korea, Japan, and far eastern Russia (Haack *et al.* 2002), and is not considered to be a major pest of ash in these regions. *A. planipennis* may also attack walnut (*Juglans* spp.) and elm (*Ulmus* spp.) (though there is no evidence of this in North America).

The ash borer's invasion of North America was discovered in 2002 in Michigan; populations were soon found attacking ash trees in southwestern Ontario, and by the end of 2004 the insect's distribution had greatly expanded in a stratified diffusion pattern (Muirhead *et al.* 2006). The risk of further establishment and spread of *A. planipennis* in North America has been rated as very high (Ciesla 2003). The insect can fly relatively long distances (>1km) and has a high reproductive potential. Despite the low density of ash trees in Canadian forests, EAB's directed host-searching ability (Ciesla 2003) is high. In its native range, the borer typically attacks trees in forests with relatively open canopies or along forest edges, while North American populations will also attack trees in denser-canopied forests (Haack *et al.* 2002).

Since its establishment in North America, major efforts have been underway to increase understanding of the ash borer's biology and climatic requirements. Investigations into the insect's thermal requirements are ongoing (Lyons and Jones 2005). In China, one generation per year is typically produced; two years are required for development in colder climates (Haack *et al.* 2002). In the currently infested parts of southern Ontario, univoltinism has been observed in EAB.

All five species of ash (*Fraxinus* spp.) found in Canada appear to be vulnerable to *A. planipennis* (Haack *et al.* 2002), although there is some evidence that blue ash (*Fraxinus quadrangulata*) may be resistant to the borer, and research efforts are underway to investigate this (G. Otis pers. comm.). Although ash species are rarely dominant and typically make up no more than 3-4% of the wood volume in forests where they are found, their loss due to EAB has the potential to cause major changes in forest biodiversity and stand dynamics in these ecosystems (MacFarlane and Meyer 2005). All ash species have been identified as providing a food source for wildlife, with ash seeds eaten by small mammals, wood ducks, songbirds, and other birds. Moose, deer and beaver commonly feed on ash shoots. Ash leaves are higher in nutrients and decompose faster than many of its potential replacement species (Melillo *et al.* 1982; Burns and Honkala 1990), and so the loss of ash may alter organic matter turnover and nutrient cycling rates.

White ash is only moderately shade tolerant, and grows most vigorously as a pioneer tree in open conditions. If scattered ash trees are lost from the forest matrix, resulting in small gaps being formed, it is likely that these gaps will be filled in with other more shade-tolerant species. Alterations in species composition can have effects on forest C stocks through wood density differences among tree species (Bunker *et al.* 2005), and the decomposition dynamics of tree litter (e.g. Bhatti and Apps 2002). Green ash is an important prairie woodland species (MacFarlane and Meyer 2005), and also commonly planted in shelterbelts on agricultural land, which were found to store a significant amount of C (Kort and Turnock 1999). Ash species such as blue and pumpkin ashes are rare in Canada, and may be made regionally extinct by EAB infestation, with a concomitant decrease in biodiversity.

5.8 Hemlock woolly adelgid (*Adelges tsugae*) (HWA)

Hemlock woolly adelgid (HWA) is an aphid-like insect that feeds on the xylem parenchyma cells of small twigs at the base of hemlock (*Tsuga* spp.) needles, depleting the tree's carbohydrate reserves and resulting in leaf drop due to desiccation (McClure *et al.* 2001). HWA attacks hemlocks of all sizes and age classes, causing a gradual decline in tree health, resulting in tree death within 5 to 15 years (Stadler *et al.* 2005). Native to China, Japan, India, and Taiwan, where it is relatively innocuous

(McClure *et al.* 2001), the adelgid was introduced into western North America in the 1920s (CFIA 2005). Little damage has been done to Western Hemlock (*Tsuga heterophylla*), which shows resistance to HWA, but since its introduction into the eastern U.S. in the 1950s, the adelgid has been steadily decimating populations of eastern hemlock (*T. canadensis*) and Carolina hemlock (*T. caroliniana*). HWA has infested hemlock forests as far north as southern parts of Maine and New Hampshire, as well as one county in upstate New York, but has not yet expanded its range into eastern Canada. It is thought that its spread is limited at its northern edge by cold winter temperatures (Parker *et al.* 1999), although there is evidence that HWA is adapting to increasingly cold temperatures (McClure and Cheah 2002). In areas where HWA is established, the occurrence of below-normal winter temperatures have resulted in high mortality of the insect (McClure and Cheah 2002).

A. tsugae produces two generations per year, with feeding occurring in both summer and winter. Winged and wingless adults are produced; the winged forms disperse and lay eggs on spruce (*Picea* spp.) trees, while the wingless form (covered in the white waxy substance that is responsible for the adelgid's common name) reproduce on hemlock. In North America, no suitable spruce hosts exist, and eggs deposited on spruce trees do not survive (CFIA 2005). This lack of a spruce host exerts some control over North American HWA populations, as populations are reduced on hemlock as its nutritional status declines due to infestation (McClure *et al.* 2001). Besides this unfulfilled requirement for an alternate host, and abiotic factors such as periodic temperature extremes, the adelgid is without natural controls (host resistance, natural enemies) in eastern North America. However, the testing of a biocontrol agent (a lady beetle imported from China) has shown some promising results to date (McClure *et al.* 2001).

The insect's coldhardiness declines over the winter, so it is particularly susceptible to very low temperatures in late winter/early spring. In lab experiments, most HWA individuals were killed at temperatures below -30°C in January, -25°C in February, and -20°C in March, with 100% mortality occurring at temperatures of -35°C in any month (Parker *et al.* 1999). In the field, however, some individuals may be able to survive these extremely low temperatures: in its native range it is known to persist at high elevations where temperatures drop below -35°C (McClure *et al.* 2001), and during extreme cold in its North American range, HWA mortality has been found to be lower where branches are covered with an insulating layer of snow (McClure and Cheah 2002). The length of time of exposure to low temperatures seems to have little influence on mortality rates; in the lab, individuals that could withstand low temperatures for short times were not likely to die if exposed for longer times (Parker *et al.* 1999). Climate warming that results in milder winters could lead to more widespread outbreak conditions in the woolly adelgid's present range (Parker 1997), as well as range expansion.

Eastern hemlock, a slow-growing, long-lived, late-successional species (Burns and Honkala 1990) is a more important species ecologically than its abundance would suggest (McClure *et al.* 2001, Ellison *et al.* 2005). The most shade-tolerant of all North American tree species, its dense evergreen crown provides important shelter for wildlife in winter, and moderates temperatures under its canopy in summer. Eastern hemlock reaches its northern limit in Canada in south-central Ontario, southern Quebec, New Brunswick, and Nova Scotia, as part of the Great-Lakes-St. Lawrence and Acadian forest regions.

In many of the forest types in which hemlock is found it is likely to be replaced by the hardwoods with which it is most often associated, such as sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), yellow birch (*Betula alleghaniensis*), red maple (*Acer rubrum*), and red oak (*Quercus rubra*). A previous decline in hemlock populations that occurred 5000 years ago in Ontario, likely due to outbreaks of an insect or pathogen, resulted in increased abundance of these hardwood species (Fuller 1998). All of these likely replacement species have denser wood than hemlock (Gonzales 1990), meaning that for a given wood volume, more carbon would be sequestered in the hardwoods (e.g. Bunker *et al.* 2005). However, the less dense crowns and deciduous nature of the likely replacement species will alter the microclimate at the forest floor by allowing more sunlight to penetrate the canopy, possibly resulting in increased decomposition rates and the release of soil C into the atmosphere. More

snow accumulation and lack of an evergreen canopy to protect from wind will mean reduced habitat value for some species of wildlife.

Fuller (1998) determined that hemlock populations took 1000-2000 years to recover from the disturbance that resulted in their dramatic decline millennia ago. In more recent times a relatively large body of information has been collected on the ecological effects of eastern hemlock loss from forests in the northeastern U.S., ranging from effects on nutrient cycling to effects on aquatic ecosystems. Hemlock mortality was found to increase rates of decomposition and nitrogen mineralization (Jenkins *et al.* 1999; Kizlinski *et al.* 2002), and to alter species composition as hardwood species filled the gaps, shifting the forest structure from conifer to hardwood-dominated (Jenkins *et al.* 1999). Homogenization of forest composition and structure has been noted as already-present species replace the lost hemlock (Tingley *et al.* 2002).

A study of the impacts of hemlock removal on bird communities found that some species abundances were greatly reduced in areas with high hemlock mortality, suggesting the possibility of local bird species extinction (Tingley *et al.* 2002). The authors noted that while the gaps formed in the canopy can temporarily attract disturbance-loving species and increase overall bird species richness, when gaps fill in with hardwoods, those species move on, leaving the community without the strongly hemlock-associated species, and reducing diversity over the longer-term. Hemlock trees form 'islands' of unique, multilayered, evergreen habitat in hardwood dominated forests. The dense shade cast by hemlock crowns can also influence aquatic communities in forest streams (Snyder *et al.* 2002).

5.9 Sudden oak death (*Phytophthora ramorum*)

P. ramorum, recently identified as the causal agent of the 'sudden oak death' phenomenon in forests of coastal California and Oregon (Rizzo *et al.* 2002), is a fungus-like organism (Kingdom: Chromista) likely introduced from an unknown source location into both North America and Europe (CFIA 2006). The organism has a wide range of host species in numerous genera of trees, shrubs, and herbaceous plants. Infection by *P. ramorum* can be lethal to some hosts, killing trees by forming cankers on main stems, and non-lethal or sub-lethal to others, infecting leaves and twigs (CFIA 2006). It is thought that the species for which the pathogen is non-lethal act as intermediate hosts, filling an important role in the organism's epidemiology by acting as sources of inoculum. In fact, there is evidence that dispersal propagules are produced on the foliage of infected intermediate hosts, but not on the stems of hosts that are mortally infected (Rizzo and Garbelotto 2003). The range of hosts is likely wider than that confirmed so far (CFIA 2006).

The worldwide list of *P. ramorum* hosts covers over 40 plant genera (Rizzo and Garbelotto 2005); in Canada, several tree species, as well as numerous understory shrubs may be susceptible. *P. ramorum* in its lethal form infects oaks (*Quercus* spp.) in the red oak group and also has been reported to cause seedling mortality in Douglas fir (*Pseudotsuga menziesii*). Canada is home to 10 oak species, 5 of which are in the susceptible group (*Q. rubrum*, *Q. velutina*, *Q. palustris*, *Q. ellipsoidalis*, and *Q. shumardii*) including red oak, which is widespread in southeastern Canada (southern Ontario, southern Quebec, and New Brunswick). Other susceptible oaks such as pin and black oaks are limited to southern Ontario, in the already highly impacted and fragmented Carolinian forest region. In Western Canada, Douglas fir is a widely planted commercial species, and Garry oak, the only native oak to B.C., is part of a unique habitat type in this region. In the U.S., Garry oak has not been found to be a host for *P. ramorum* in the wild, but does become infected under laboratory conditions (CFIA 2006). Other likely hosts for *P. ramorum* include important hardwood species in Canada such as sugar maple (*Acer saccharum*), American beech (*Fagus americanus*), and ash (*Fraxinus* spp.) among others; these have been shown to be susceptible to infection under controlled conditions (CFIA 2006).

The current range of established *P. ramorum* in North America covers oak and tanoak (*Lithocarpus*) forests along a 750 km swath of California and Oregon coastline. The climate in which it is presently found is characterized as maritime, with moderate summer temperatures and frequent summer fog (CFIA 2006). Affected areas are generally within 30 km of the coast. In Canada, the only analogous

climate occurs in southern coastal B.C.; however, this may not represent the full extent of environments the pathogen is able to exploit. Contrary to early assumptions that *P. ramorum* would not be able to survive the harsh winters in most parts of Canada, its spores have more recently been found to be capable of surviving temperatures as low as -15°C (CFIA, citing Marra, unpublished data, based on spores taken from leaves of nursery plants in the northeastern U.S.). In the UK, spores on leaf litter or incorporated into the soil were able to survive temperatures of -9°C; in lab-based experiments, all spores were killed when temperatures were held at -25°C for several hours (DEFRA 2004). It is therefore possible that winter temperatures commonly experienced over large parts of Canada may not be limiting to this pathogen, particularly if snow acts as an insulator for propagules contained in leaf litter on the forest floor (CFIA 2006). In addition, some models of potential *P. ramorum* distribution in the U.S. have identified the northeast as a high risk area.

Forests that contain intermediate host species as well as 'terminal' hosts (species on which stem cankers are produced, likely causing mortality) will be most impacted by *P. ramorum*. In tanoak (in California; not present in Canada), the pathogen attacks and kills tree in all age classes, however in true oaks, mature trees are more likely to be infected and killed. *P. ramorum*-induced tree mortality is patchy; within a forest stand, 100-300m patches of killed trees can be surrounded by stands of undiseased trees of susceptible species (Rizzo and Garbelotto 2003). The presence of an intermediate shrub host seems to be the factor most closely related to the occurrence of *P. ramorum* mortality in California's coastal oak forests (Rizzo and Garbelotto 2005). Edge effects may also be important, influencing the interception of propagules in wind-driven rain (Rizzo and Garbelotto 2005); this has implications for the susceptibility of forest fragments in human altered landscapes such as southern Ontario.

P. ramorum was introduced into Canada in 2003 and 2004 on nursery stock imported from California into B.C. Subsequent eradication measures were successful, and prevented the pathogen from becoming established. The recently revised pest risk assessment for *P. ramorum* in Canada determined the overall risk from this organism to be high in British Columbia, and medium in Ontario, Quebec, and the maritime provinces. Included in the updated PRA was an assessment of potential future *P. ramorum* distribution in a changing climate; the detailed bioclimatic analysis revealed little change in the pathogen's range under different climate change scenarios. The authors emphasize, however, that because the bioclimatic predictions are based on the current climate conditions in which the organism is found, which may not represent its current potential range, results must be interpreted with caution (CFIA 2006).

The loss of susceptible species will alter forest structure, with trickle-down effects on the forest ecosystem, for example increased inputs of dead wood to the forest floor, potentially affecting soil nutrient status (Rizzo and Garbelotto 2005), as well as potential effects on biodiversity (Brasier 2003; CFIA 2006). Monahan and Koenig (2006) predict declines in abundance of up to 70% in oak-dependent bird populations in California coastal forests. Regeneration of susceptible species after *P. ramorum*-induced tree mortality will be affected by the presence of persistent sources of inoculum in associated non-lethally-infected plants, potentially altering successional patterns. Forest species composition may also be altered as competition dynamics change among intermediate host species that may exhibit varying degrees of inhibited growth (Rizzo and Garbelotto 2003).

In Canada, the loss of oak species in particular may have ramifications for forest C stocks, as well as impacts on wildlife. Acorns are a valuable food form many species of birds and mammals, and so a reduction in oak abundance would affect habitat value. Oak wood has a high specific gravity, and hence stores more carbon per unit volume than many of the hardwoods and softwoods with which it is commonly associated. Also, oak litter is associated with slower decomposition rates compared to associates such as maple and ash, which could result in a larger forest floor C pool.

The authors of the recent pest risk assessment for *P. ramorum* in Canada speculate that the establishment of this organism in Canada would lead to loss of species diversity, loss of productivity, reduction of wildlife habitat, increased fire hazard, and changes to water and soil quality (CFIA 2006).

6. Conclusions and recommendations

Climate change will almost undoubtedly affect the ability of non-native insect and pathogen species to establish and spread in Canada; rising temperatures in particular will likely benefit invading insects and pathogens and expand their potential range. For some pests, the degree of future warming, which varies by emissions scenario, will likely be a significant factor in determining the geographic area over which they can become established in Canada. For others, the present climate is not limiting over their host's range; in these cases, their success may be increased in other ways, such as by increasing survival rates of overwintering life stages, or the number of generations produced annually, depending on the degree of warming projected to occur. More factors are involved in the success of an invading pest than just temperature, however, and more information is needed on other climatic requirements of potential invaders as well as on host-pest interaction dynamics that will influence the success, and resultant impacts, of the pest.

The vast areal extent of the boreal forest and its importance to Canada's C budget mean that non-native pest effects on the dominant species of this forest region will have a greater impact on Canada's overall forest C stocks than will the disturbance caused by pests of less widespread host species. Pest-induced stand-replacing disturbance (i.e. high rates of mortality of the dominant tree species) will likely have a greater effect on forest C stocks than smaller-scale disturbances (or 'matrix' disturbances) due to the impact on age-class structure over the landscape. It has been suggested that if pest outbreaks result in greater susceptibility of affected forests to fire, this could outweigh any direct outbreak effects on forest C, however research is needed to better understand interactions between insect outbreaks and fire.

Temperate forest ecosystems in Canada may also be severely disrupted if certain non-native pests become established and spread. The loss of individual species from the forest matrix can have immediate direct effects and well as longer-term indirect effects on nutrient cycling patterns, decomposition rates, and wildlife habitat, through changes in microclimate, redistribution of nutrients and organic matter, and ultimately through changes in species composition and forest structure (Hunter 2001). The establishment and spread of non-native pests in Canada's temperate forests could have important implications for forest biodiversity in Canada, as several tree species would be at risk of local or regional extinction (including eastern Hemlock and the rarer ash species).

Indirect effects of non-native pest invasions such as the impact of control measures are outside of the scope of this report, but also must be considered. GHG emissions resulting from the production, transportation, and application of pesticides used in control efforts, and the impacts of pesticides on non-target organisms, must be considered (Fleming 2000). The only option for eradicating invasive pests before they become widespread in Canada has often been the harvest and destruction of infested and nearby trees, which has its own environmental ramifications (Cavey 2000). Kislinski *et al.* (2002) found that salvage cutting of infested trees can exacerbate the impacts of pest outbreaks of forest ecosystems.

Maintaining large areas of intact forest ecosystems, where natural ecological processes are allowed to unfold, as well as striving to maximize diversity (at all scales) in managed forests, may provide the best long-term insurance against invasive insects and pathogens. Monospecific plantations are often the hardest hit by pest outbreaks (Woods *et al.* 2005; Cavey 2000), and forest fragmentation can worsen outbreaks due to alterations in natural enemy populations (Roland 1993).

Ayres and Lombadero (2005) offer general suggestions for research that would improve our understanding of the effects of climate change on forest insects and pathogens. Among other things, they recommend more work on developing and testing general principles that will help predict the responses of pests to climate change, more research into the direct and indirect effects of temperature and moisture change on pest insects and pathogens, and experimental studies on the interactions between tree physiology and forest pathogens (Ayres and Lombadero 2005).

In Canada, and elsewhere, there is a lack of data on many aspects of the effects of outbreaks on forest C including CO₂ production caused by forest insect outbreaks, and changes in decomposition rates due to microclimate alterations; however efforts are underway to model the effects of climate change on the outbreak dynamics of native insects and the resultant impacts on forest C sequestration (MacLean *et al.* 2002). Fleming (2000) lists the knowledge gaps that need to be filled in order for better predictions to be made of the effects of climate change on insect disturbance regimes (similar gaps exist for pathogens) in Canada: more information is needed on interactions between insects and other disturbances under climate change, the effects of insect outbreaks on successional pathways, tree growth, and carbon sequestration rates, and the effects of outbreaks on soil C dynamics. Also, species-specific forest inventories are needed in order to make better, more quantitative predictions of the potential impacts of non-native pests (e.g. McKenney *et al.* 2003).

Recommendations:

- Encourage research in several areas in order to improve our understanding of the potential threats and impacts of non-native forest pests. Research priorities include the following areas:
 - Biology, phenology, and ecology of non-native pest species that pose a threat to Canada's forests
 - Roles and functions of insects and pathogens (native and non-native) in Canada's forest ecosystems
 - Interactions between pest-induced disturbance and other forest disturbances, notably fire, but also anthropogenic disturbances such as the conversion of natural forest to plantations
 - Stand-level and landscape-level effects of tree species loss on ecosystem functioning and forest C stocks
 - More detailed and finer-scaled bioclimatic modeling of pest species' potential ranges in Canada under current and future climate, using data from lab studies, field observations, and knowledge of climatic conditions in its native range, as these become available
- Develop forest inventory data that allows for the determination of the relative abundances of individual tree species within a given forest type and/or geographical area; this will improve assessment of the impact of a given forest pest.
- Facilitate the sharing of knowledge among forest entomologists, forest ecologists, climate change biologists, and specialists of particular non-native species, by holding a meeting or symposium to discuss Canada-specific issues related to the threats and impacts of non-native forest pests. Foster communication among researchers and agencies with different but overlapping interests and mandates.

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Appendix A: Expert opinion survey

Canadian researchers in the areas of forest entomology, pathology, and ecology were surveyed as a means to gather expert opinion on the threats and impacts of non-native pests to Canada's forest ecosystems and C stocks. The list of experts contacted and their affiliations are as follows:

Eric Allen, Canadian Forest Service, Pacific Forestry Centre
Pierre Bernier Canadian Forest Service, Laurentian Forestry Centre
Allan Carroll, Canadian Forest Service, Pacific Forestry Centre
Steve Colombo, Ontario Ministry of Natural Resources, Ontario Forest Research Institute
Dave DeYoe, Ontario Ministry of Natural Resources, Ontario Forest Research Institute
Richard Fleming, Canadian Forest Service, Great Lakes Forestry Centre
Brad Hawkes, Canadian Forest Service, Pacific Forestry Centre
Ted Hogg, Canadian Forest Service, Northern Forestry Centre
Dave MacLean, Faculty of Forestry and Environmental Management, University of New Brunswick
Taylor Scarr, Ontario Ministry of Natural Resources
Jan Volney, Canadian Forest Service, Northern Forestry Centre
Richard Wilson, Ontario Ministry of Natural Resources

Most of those contacted sent a response, however only a small proportion specifically responded to the survey questions. Many of the responses indicated hesitance in answering the survey questions due to lack of knowledge about non-native pests. One respondent went as far as to say that the reaction of non-native pests to new environments would be "anyone's guess". Another suggested the thought-provoking consideration of invasive species as "pioneers of ecosystem change" in a changing climate. The few responses that were received are summarized following each survey question listed below.

Survey questions and responses:

1. In a changing climate, how do you think the effects of non-native 'pests' on Canada's forest carbon stocks will compare, in magnitude, to the effects of native forest pests and fire disturbance?

-in the medium-term (next 30 years), exotic pests will have a small effect on forest C stocks relative to disturbance from fire and native insects; in the long-term (100 years), the effect could be more substantial

-fire will likely continue to be the most important disturbance affecting forest C stocks, although if the expansion into new regions of native pests such as mountain pine beetle were considered, the relative importance of invasive species would grow

-depends on type of insect; defoliators that don't cause high levels of tree mortality will have less impact on fuel loads and potential fire behaviour than bark beetles that can cause high mortality levels

-potential C loss due to fire is higher in bark beetle-killed stands than live stands, as more dead branchwood and downed wood exist to be consumed

2. Which will be the most important mechanism by which non-native pests will impact forest ecosystems and forest carbon stocks: a) the alteration of tree species composition due to the loss of particular tree species by non-stand-replacing insect or pathogen disturbance, or b) the alteration of forest age structure over the landscape by insects or pathogens that initiate stand replacement due to large-scale pest-induced tree mortality, or c) the increased risk of fire in pest-damaged forests.

-b) and c) are likely to be most important

-all will be important but a) is most likely

-depends on fire occurrence and level of pest damage

-b) will be most important and could have a positive effect on C storage if no fire occurs to release C from pest-damaged forest

-c) will be important if stand-replacing insect outbreak occurs where fire frequency high and fire size large

Survey questions and responses, continued:

3. Which of the following non-native forest insects/pathogens has the greatest potential to impact Canada's forest ecosystems in a changing climate and why? Are there others that you would include on this list?

- Sudden oak death (*Phytophthora ramorum*)
- European woodwasp (*Sirex noctilio*)
- Asian gypsy moth (*Lymantria dispar*)
- Brown spruce longhorn beetle (*Tetropium fuscum*)
- Asian long-horned beetle (*Anoplophora glabripennis*)
- Emerald ash borer (*Agrilus planipennis*)
- Hemlock woolly adelgid (*Adelges tsugae*)

-southern pine beetle (*Dendroctonus frontalis*) should be added to the list as its range is expanding northwards and will likely expand into Canada
-expansion of mountain pine beetle (*Dendroctonus ponderosae*) into eastern boreal forests
-difficult to answer as the threat may come from as yet unknown pests

4. What do you think are the main knowledge gaps in making accurate predictions about the impacts of non-native pests on Canada's forest ecosystems under climate change?

-the effects of climate change on the rates of population growth and spread of non-native pests
-basic biology of many pests, as well as host-pest interactions
-role of climatic factors governing outbreaks
-population biology/host interaction process models to predict potential extent and stand mortality
-interaction with new potential hosts

-other comments:
-risk assessment a more reasonable goal than prediction
-prevention of the introduction of non-native pests, and early containment of introduced pests, should be an important focus

5. Please attach a level of certainty to the following statements: (1=low certainty, 5=high certainty)

a) non-native insects and pathogens will have less difficulty becoming established and spreading in Canada as climate change progresses

-range of certainty levels: 4-5

b) forest tree species in Canada will be lost due to non-native pest invasions

-range of certainty levels: 1-4
-comments: considering functional loss, not necessarily complete loss

c) non-native pest-induced forest disturbance will lead to a decline in Canada's forest carbon stocks

-range of certainty levels: 3-5

d) the roles of insects and pathogens in the functioning of Canada's forest ecosystems are well understood

-range of certainty levels: 1-3
-comments: the low certainty value indicates high certainty that the roles of these organisms in our forest ecosystems are not well understood