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Review

Forest health conditions in North America[☆]

Borys Tkacz [a,*], Ben Moody [b], Jaime Villa Castillo [c], Mark E. Fenn [d]

[a] *USDA Forest Service, Forest Health Protection, 1601 North Kent Street, RPC7-FHP, Arlington, VA 22209, USA*
[b] *Canadian Forest Service, Ottawa, ON, Canada*
[c] *Comision Nacional Forestal, Zapopan, Jalisco, Mexico*
[d] *USDA Forest Service, Pacific Southwest Research Station, Riverside, CA, USA*

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The forests of North America continue to face many biotic and abiotic stressors including fragmentation, fires, native and invasive pests, and air pollution.

Abstract

Some of the greatest forest health impacts in North America are caused by invasive forest insects and pathogens (e.g., emerald ash borer and sudden oak death in the US), by severe outbreaks of native pests (e.g., mountain pine beetle in Canada), and fires exacerbated by changing climate. Ozone and N and S pollutants continue to impact the health of forests in several regions of North America. Long-term monitoring of forest health indicators has facilitated the assessment of forest health and sustainability in North America. By linking a nationwide network of forest health plots with the more extensive forest inventory, forest health experts in the US have evaluated current trends for major forest health indicators and developed assessments of future risks. Canada and Mexico currently lack nationwide networks of forest health plots. Development and expansion of these networks is critical to effective assessment of future forest health impacts.
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1. Introduction

The forests of North America provide a variety of benefits including water, recreation, wildlife habitat, timber and other forest products. The health of these forests is directly related to their capacity to increase or maintain productivity while maintaining resistance to biotic and abiotic stressors (McLaughlin and Percy, 1999). The health and sustainability of North American forests have been assessed in the context of Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests developed under the Santiago Agreement (Anon., 1995a,b; USDA, 2004; NRCan, 2006; SEMARNAT, 2003). This paper highlights and updates the findings of these assessments regarding the most damaging forest stressors affecting North American forests in recent years, including fires, native and invasive pests, fragmentation, and air pollution, and provides some projections of future risks. It also contrasts the approaches to monitoring utilized in Canada, the United States and Mexico and provides some recommendations for future enhancements.

2. Forest cover and fragmentation

Forests cover 677.5 million hectares or nearly one third of the total land area of North America (UNFAO, 2005). This represents over 17 percent of the total global forest area, with Canada and the United States (US) ranking third and fourth, respectively, among countries with the largest forest area. The total area of forests in Canada, the US and Mexico has been relatively stable over the last 15 years (Table 1).

☆ This is an expanded version of a paper presented at the International Union of Forest Research Organizations Conference: Impacts of Air Pollution and Climate Change on Forest Ecosystems, held in Riverside, CA in September 2006 (Tkacz et al., 2007).

* Corresponding author. Tel.: +1 703 605 5343.
E-mail address: btkacz@fs.fed.us (B. Tkacz).

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Table 1
Total forest area (million hectares) by country and year (UNFAO, 2005)

Year	Canada	Mexico	US
1990	310	69	298.6
2000	310	65.5	302.3
2005	310	64.2	303

Although the total forest area in the US has been relatively stable, some portions of the country have been increasingly fragmented. Fragmentation of forests may lead to changes in ecological processes, reduction in biological diversity and the spread of invasive species from disturbed edges. Even small openings may introduce these impacts deeper into the forest. Analyses of high-resolution forestland cover maps derived from satellite imagery indicated that large portions of the forestland in the US were fragmented with about 44 percent being within 90 m of the forest edge, 62 percent within 150 m of forest edge, and less than 1 percent being more than 1230 m from the forest edge (Fig. 1; USDA, 2004). However, where forest existed, it was dominant. Seventy-two percent of all forest was in landscapes that were at least 60 percent forested. About half the fragmentation consisted of small (less than 7.3 hectares) perforations in interior forest areas. More detailed assessments of forest edge revealed that there were about 31.4 million km of forest–nonforest edge in the US (Coulston et al., 2005a). Equal amounts of forest edge were anthropogenic (forest edge with urban or agricultural land-cover types) and semi-natural (forest edge with water, wetland, barren, grassland, or shrubland). Most anthropogenic forest edge was found in the eastern US, whereas semi-natural forest edge was more common in the western US. The conversion of forest land to agricultural and urban uses, was concentrated in southern Canada, and was likely the main cause of endangerment and extinction of several forest-associated species. Forestry operations and other types of anthropogenic disturbances, such as oil and gas extraction and mining (e.g. in Alberta and Saskatchewan) can also create deforested areas, on a smaller scale, and may also impact biodiversity (NRCan, 2006). Temporary removal of forest cover in Canada through harvesting averages 900,000 hectares per year and has been relatively stable for several years (CCFM, 2006). Estimates of the annual area of forest in Canada permanently converted to nonforest for reasons such as urbanization, agriculture, and forest road construction, range from 55,000 to 80,000 hectares per year (CCFM, 2006).

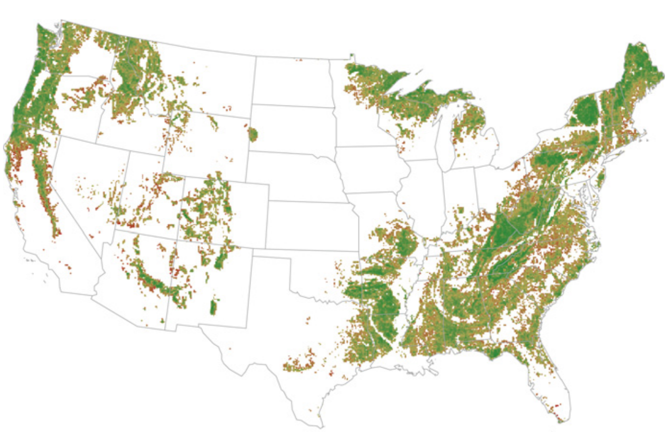

Fig. 1. This map shows the relative amount of "interior" forest at 7-ha scale shaded from low (red) to high (green) for areas containing >60% forest overall.

3. Forest fires

Fire is a major disturbance agent in many forests of North America. Many forest ecosystems are adapted to particular fire frequencies and intensities (Hardy et al., 1998). The annual amount of forest area burned varies depending on weather conditions, fuel loading, and forest stand conditions (SEMARNAT, 2003; USDA, 2004; NRCan, 2006). The total area of forest fires by country and year from 1990 until 2005 are presented in Fig. 2. Many years of fire suppression have resulted in increased fuel loads and dense forests resulting in increased risks of catastrophic, stand-replacing fires (Covington et al., 1994; Brown, 1995). The recent increase in number, size and severity of fires in the western US has also been linked to recent climatic changes. Large fire (>400 hectares) frequency and total area burned have increased markedly since the mid 1980s in strong association with increased spring and summer temperatures and an earlier spring snowmelt (Westerling et al., 2006). The total area burned in the US in 2006 was the largest in the last 46 years (National Interagency Coordination Center: http://www.nifc.gov/stats/fires_acres.html). In Canada, the 2005 forest fires represented a typical year with 7438 fires, close to the 10-year average of 7496, and 1.7 million hectares of area burned, below (70.8 percent) the 10-year average of 2.4 million hectares (NRCan, 2006). In 2006, Canada recorded 9713 fires covering 2.08 million hectares compared to the 10-year average of 7445 fires and 1.96 million hectares (Johnston, 2006).

4. Forest pests

Forest insects and pathogens are biotic disturbance agents that can be either beneficial or detrimental to forests. While they play critical roles in forest ecosystems (such as providing

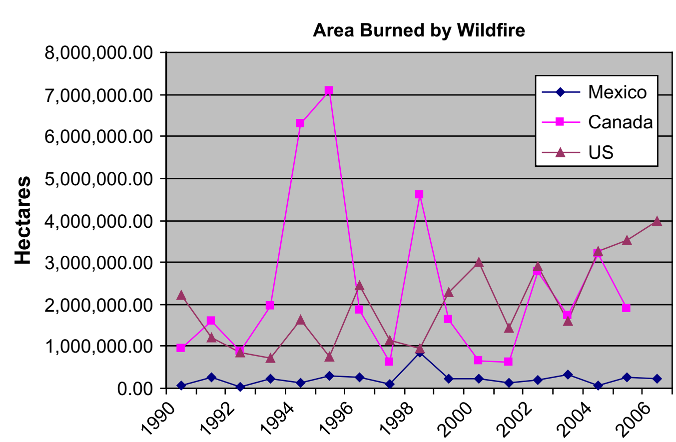

Fig. 2. Historic trend in forest fires for Canada, Mexico, and the US.

for wildlife habitat), they can be devastating when populations reach high levels. Outbreaks can lead to damaging levels of defoliation or mortality under suitable climatic and site conditions. Pests can also be beneficial to the forest, for example, by suppressing unwanted invasive plants or weeds.

4.1. Canada

Historic trends in four of the most important forest pests for Canada are presented in Fig. 3 (NRCan, 2006). The spruce budworm (*Choristoneura fumiferana*) is the most damaging insect pest of spruce and fir species in Canada. In 2004, a total of 755,325 hectares were defoliated by this insect, the lowest level in the past 10 years and significantly lower than in peak years which have reached 20 million hectares affected. The large aspen tortrix (*C. conflictana*) caused significant defoliation to aspen mainly in Alberta, reaching its largest infested area of 6.0 million hectares in 2003. The mountain pine beetle (*Dendroctonus ponderosae*) has caused increasing levels of tree mortality in British Columbia, most likely due to changing climate. By 2004, the beetle had infested 700,000 hectares of mature lodgepole pine (NRCan, 2006). The mountain pine beetle outbreak is the largest ever seen in North America with more than 8.7 million hectares affected by 2005. Approximately 450 million cubic meters of pine have been killed (CCFM, 2006). Damage by the western spruce budworm (*C. occidentalis*), a significant pest of Douglas fir in British Columbia, has increased steadily from 123,638 hectares defoliated in 2001 to 624,000 hectares defoliated in 2004 (NRCan, 2006). Defoliation of trembling aspen caused by the forest tent caterpillar (*Malacosoma disstria*) has decreased in recent years from a peak of nearly 15 million hectares in 2001 (NRCan, 2006). Another climate-change related disease, (redband) needle blight caused by *Dothistroma septosporum*, defoliated and killed lodgepole pine in British Columbia. The disease incidence is favored by warmer spring temperatures (Woods et al., 2005). Other devastating forest pathogens include white pine blister rust, Dutch elm disease (resulting in 13,823 tree removals in 2005), root and butt rots, and dwarf mistletoes especially on conifers in western Canada (Pines, 2006). Drought conditions caused tree mortality across Canada especially in Ontario (82,785 hectares in 2005). Several summer storms resulted in blowdown and tree damage in several areas, the most serious in Ontario (515,746 hectares in 2005) (Evans et al., 2006).

4.2. United States

Historic trends in forest pest activity in the US are presented in Fig. 4 (USDA, 2006, 2007). Mountain pine beetle (*D. ponderosae*) outbreaks increased in area throughout the western US from 2003 through 2005, following several years of drought. Lodgepole pine forests have been affected the most. Southern pine beetle (*D. frontalis*) populations remain at low levels since 2003. Treatment strategies now focus on prevention and restoration. Alaska experienced a large outbreak of spruce beetle (*D. rufipennis*) in the 1990s with mortality levels exceeding 90 percent in many areas (USDA, 2006). Recently, favorable weather conditions (mild winters and warm summers) have led to increasing populations in Arizona, Colorado, Montana, Utah, and Wyoming. Since its introduction in 1869, gypsy moth (*Lymantria dispar*) has spread to 17 States and the District of Columbia (USDA, 2006). The current area infested is 25 percent of the total susceptible area. Current management strategy focuses on slowing the spread along the advancing front of the infestation. In recent years, the effect of the biocontrol fungus, *Entomophaga maimaiga*, is evident.

In addition to gypsy moth, several other invasive forest pests are threatening forests in the US (USDA, 2006, 2007). The hemlock woolly adelgid (*Adelges tsugae*), a native of Asia, continues to spread in eastern hemlock forests. Since its introduction in 1924 it has spread to hemlock forest from southeastern Maine to northeastern Georgia and west to eastern Tennessee. Biological control agents have been released in an attempt to control populations. The emerald ash borer (*Agrilus planipennis*), also a native of Asia was first reported killing ash trees in the Detroit area of Michigan and Windsor area of Ontario (Canada) in 2002. Since then, infestations have been found throughout lower Michigan and neighboring areas in Ontario (Canada), northwest Ohio, and northern Indiana. In 2006, infestations were also found in the

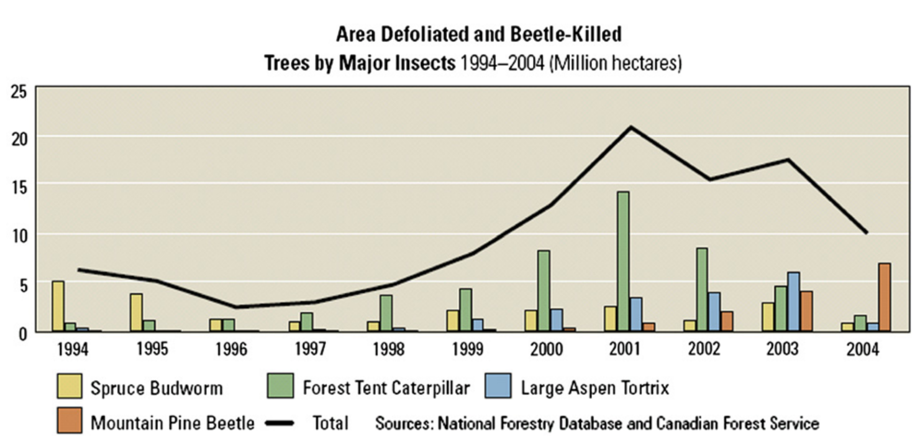

Fig. 3. Historic trends in forest pest activity in Canada.

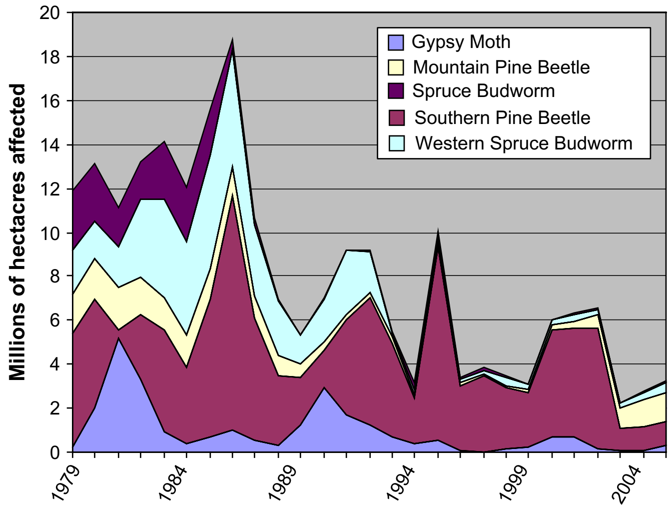

Fig. 4. Historic trends in forest pest activity in the US.

Chicago, Illinois area. The European woodwasp (*Sirex noctilio*) has recently been found infesting pine trees in New York State and Ontario. Introductions of this insect into other countries have resulted in significant mortality levels in pine plantations. Monterey, lodgepole, ponderosa, jack and most species of southern pines (especially loblolly) are known to be susceptible. The susceptibility of other North American conifers is not known. A new disease called ‘‘sudden oak death’’ (caused by *Phytophthora ramorum*) is killing thousands of tanoak (*Lithocarpus densiflorus*), coast live oak (*Quercus agrifolia*) and California black oak (*Q. kelloggii*) in coastal areas of California. An isolated infestation, discovered in Oregon, is being treated with the goal of eradication (USDA, 2007). National surveys of oak forests have not found infestations outside California and Oregon.

4.3. Mexico

Historic trends in forest pest activity in Mexico are presented in Fig. 5. Recent bark beetle activity includes Douglas-fir beetle (*D. pseudotsugae*) in Durango, roundheaded pine beetle (*D. adjunctus*) in Chihuahua and Oaxaca, southern pine beetle (*D. frontalis*) in Guerrero, Oaxaca, and Chiapas, and Mexican pine beetle (*D. mexicanus*) throughout central Mexico. The Mexican pine beetle is the bark beetle with the broadest distribution in Mexico, affecting 3000 hectares in eight states in 2004. Defoliators in Mexico include *Lophocampa alternata* in Puebla and Chihuahua, *Zadiprion falsus* in Durango and Jalisco, and *Pterophylla beltrani* in Tamaulipas. The infestation of *P. beltrani* scaled up from 200 hectares in 2004 to 1700 hectares in 2005, affecting mostly mixed oak vegetation types. A survey of oak mortality in central Mexico revealed the presence of *Phytophthora cinnamomi*, an exotic forest pathogen. Confirmed locations include Arrayanal and Colima in 2001, Tierra Colorada and Jalisco in 2004, and Tecoanapa and Guerrero in 2005. The pink hibiscus mealy bug, *Maconellicoccus hirsutus*, was first reported in Mexico affecting teak (*Tectona grandis*) plantations in January 2004. It is distributed in up to 10,000 hectares in Valle de Banderas, Nayarit and adjacent Jalisco State. This insect also affects mango, guava, soursop, ornamental shrubs and at least 38 other wild plants species. Teak blight, an exotic disease to Mexico, was detected in December 2004. This disease is also found in Costa Rica, Nicaragua, El Salvador, Belize, Honduras, Guatemala, and Panama. The causal agent is the fungus *Olivea tectonae*, a parasitic disease of teak widely distributed in Asia. This disease may cause serious losses in nursery production. The presence of this disease in young plantations may cause growth losses up to 30 percent.

5. Air pollution

5.1. Nitrogen and sulfur deposition

Air pollutants, including sulfur, nitrogen, and tropospheric ozone, can have significant cumulative effects on forests. Canada and the US have cooperated in monitoring air pollutant deposition, concentrations and effects under the 1991 Canada–United States Air Quality Agreement (EC, 2004). Spatial distribution of wet sulfate and wet nitrate deposition is presented in Figs. 6 and 7, respectively. The maps show that

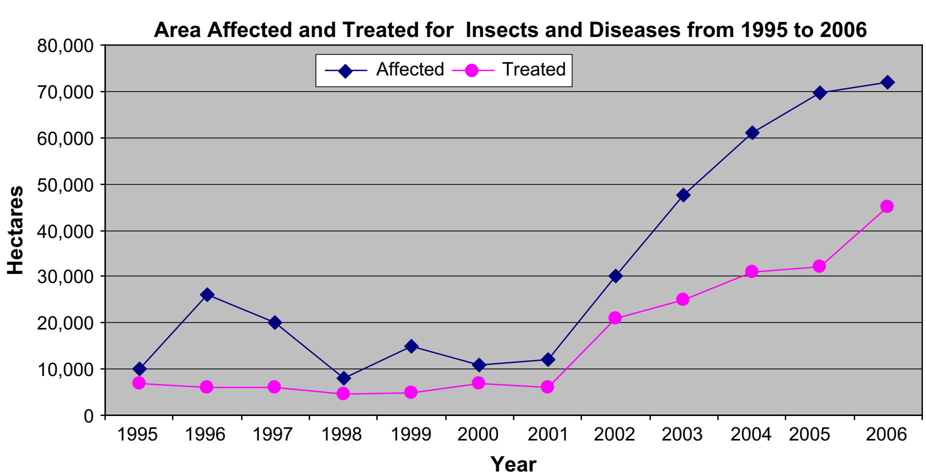

Fig. 5. History of forest pest activity in Mexico.

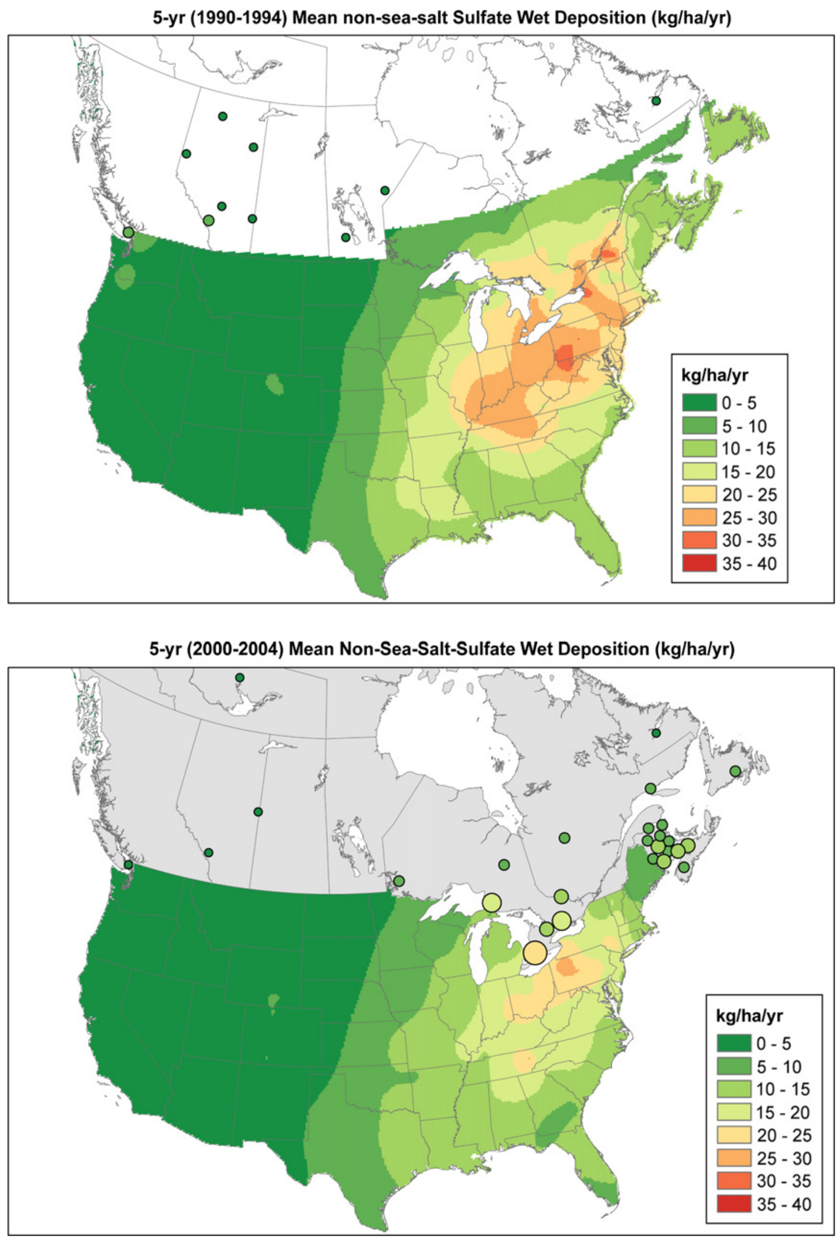

Fig. 6. (a) Mean sulfate wet deposition for 1990–1994. (b) Mean sulfate wet deposition for 2000–2004.

wet sulfate deposition remains highest in eastern North America, and the gradient follows an axis running from the confluence of the Mississippi and Ohio through the lower Great Lakes (EC, 2007). Comparison of the 2000–2004 sulfate deposition map (Fig. 6b) with the 1990–1994 map (Fig. 6a) shows significant reductions in wet sulfate deposition in eastern United States and much of eastern Canada (EC, 2007). The pattern for wet nitrate deposition (Fig. 7a and b) shows a similar south-to northeast axis but the high-deposition area is focused around the lower Great Lakes. Reductions between the two periods were more modest than for sulfate (EC, 2007). No data were available for Quebec and Newfoundland and Labrador.

For the US, cumulative distribution functions and frequency distributions were used to estimate the percent forest by region of the country exposed to specific levels of air pollution (Coulston et al., 2004) In the North and South, approximately 50 percent of the forest was exposed to wet sulfate deposition of more than 15 kg ha^{-1} yr^{-1} for 1994–2000 (Fig. 8) compared to the Pacific Coast and Rocky Mountain regions where approximately 50 percent of the forests received less than 2 kg ha^{-1} yr^{-1}. Nitrate wet deposition was highest in the North where approximately 50 percent of the forests received an annual average input of more than 13 kg ha^{-1} yr^{-1}. The North and South regions experienced the highest ammonium deposition rates. However, it should be emphasized that wet deposition monitoring does not include deposition inputs from fog or dry deposition, which are the major input forms in arid zones or areas of significant deposition in cloudwater or fog. Thus, notwithstanding the value of the long term data available from the National Atmospheric Deposition Program (NADP) monitoring

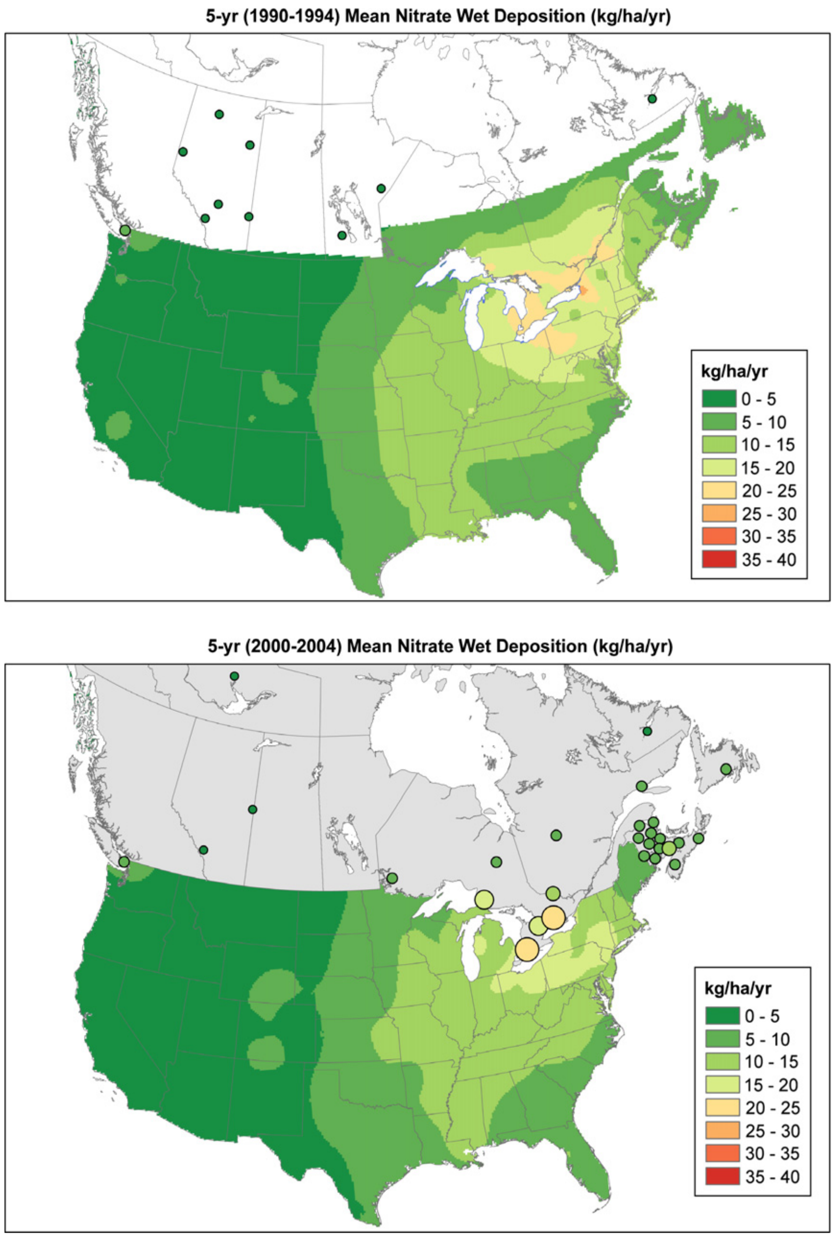

Fig. 7. (a) Mean nitrate wet deposition for 1990–1994. (b) Mean nitrate wet deposition for 2000–2004.

network, the data are limited when it comes to identifying polluted areas in semiarid climates. For example, the highest N deposition in all of North America occurs in the Transverse Ranges of southern California, yet NADP maps show this area as having very low N deposition (Fig. 7). As a further example, deposition of NHx is elevated in the Central Valley, western Sierra Nevada mountains, and in the San Bernardino Mountains of California, and in southeastern Idaho because of agricultural ammonia emissions (Bytnerowicz et al., 2002; Fenn et al., 2003c). Again, this is not at all apparent from the NADP wet deposition data.

5.2. Ozone

Ozone is the most pervasive air pollutant in North America and global background concentrations are increasing. In the western US, ponderosa and the closely-related Jeffrey pine are by far the most important species impacted by ozone, in addition to a few tree and understory species used as biomonitors (Campbell et al., 2007); whereas in the east there is a broader mix of ozone-sensitive forest species (Skelly et al., 1997; Chappelka and Samuelson, 1998; Smith et al., 2003). Furthermore, peak ozone levels in the eastern US are generally less than in southern California and in the southern Sierra Nevada. Ozone is more of a regional problem in several parts of the eastern US.

Ozone concentrations were relatively high across much of the South from 1994 to 2000 with only 10 percent of the forests exposed to ozone index concentrations of less than 6 ppm-h yr^{-1} (Fig. 8). Although most of the Pacific Coast region forests were exposed to relatively low ozone index, 10 percent of the forested area experienced exposure between

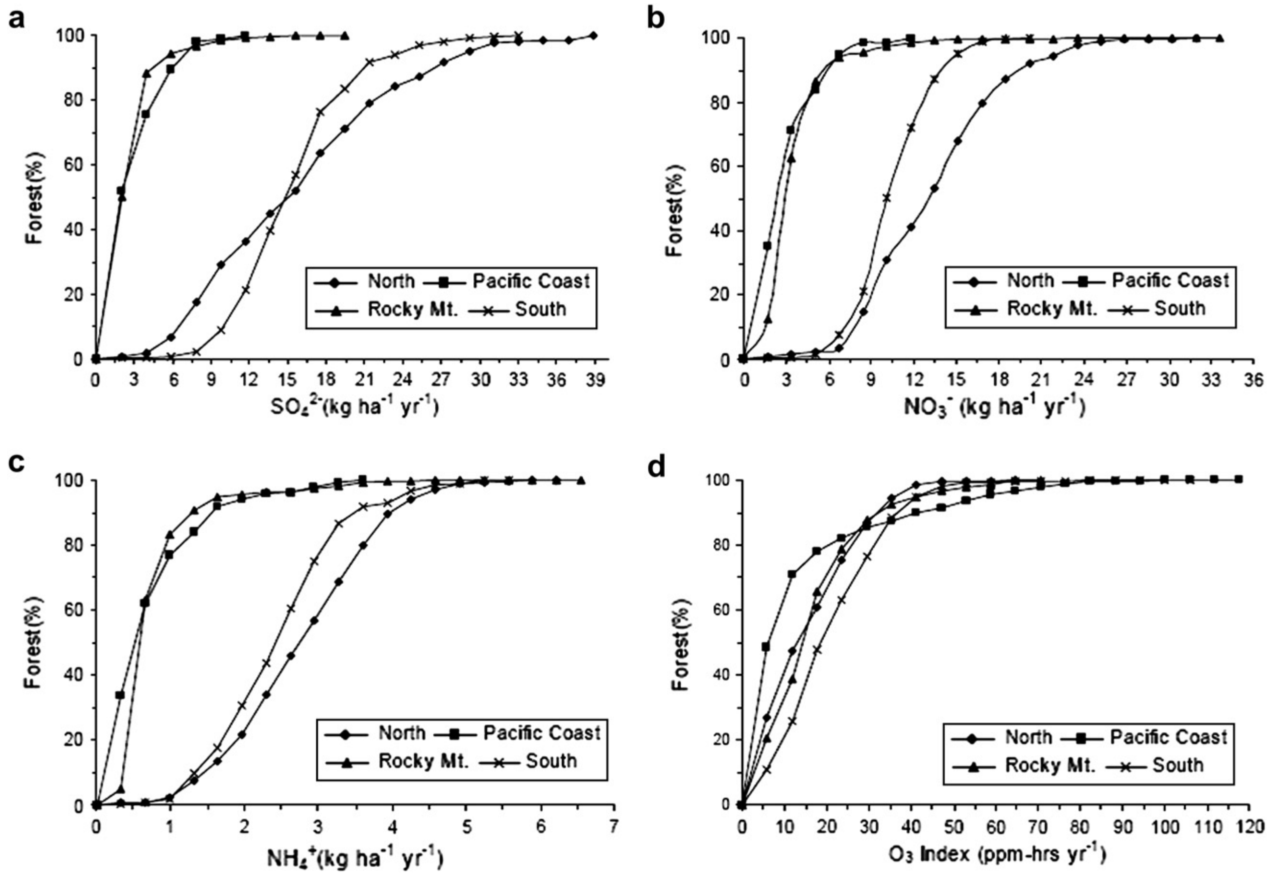

Fig. 8. Cumulative distribution or frequency distributions of the percent of forests exposed to specific levels of air pollutants in the US from 1994 to 2000 (deposition units are for wet deposition of N and S).

41.2 and 117.8 ppm-h yr^{-1}. In these California forests exposed to elevated ozone and N deposition, the combined effects of both pollutants, causes dramatic physiological and phenological disturbance, including increased susceptibility to bark beetles (Jones et al., 2004). This also leads to increased stand densification, fuel loading and fire risk (Grulke et al., in press). In the case of the most widely distributed North American tree species, trembling aspen (*Populus tremuloides*), CO_2 and O_3, singly or in combination affected productivity, physical and chemical leaf defenses, with stimulating effects from O_3 on performance of key defoliators like the forest tent caterpillar. Because of these changes in plant quality, insect and disease populations were altered (Percy and Ferretti, 2004). Recently, Percy et al. (2007) developed exposure-based regression models that predict aspen growth change in response to ambient O_3 concentrations. The growing season fourth-highest daily maximum 8 hour concentration (Fig. 9) was the best single indicator of aspen stem cross-sectional growth. Predicted growth loss for aspen ranged from 11 to 25 percent in the Great Lake States, Ontario, Quebec, and southwestern Nova Scotia and from 15 to 25 percent in Mexico, Arizona, Colorado, and Utah (Fig. 10).

Significant ozone injury has been reported from several case studies in the eastern US. Prominent areas of documented ozone problems include Acadia National Park along the Atlantic coast of Maine, the Allegheny Plateau region of north central Pennsylvania, parts of the Great Lakes region, Shenandoah National Park in Virginia, the Blue Ridge Mountain in Virginia and Great Smoky Mountains National Park. Field studies indicate that ambient ozone levels in parts of the southeastern US cause growth reductions in mature pine trees and in loblolly pine in particular. The estimated amount of pine growth reduction is highly variable (ranging from 0 to 30 percent) depending on pine species, individual tree variability, year, microsite conditions, and ozone concentrations, although the decrease in aboveground forest growth in the eastern US at ambient ozone levels is usually reported to be in the range of 0 to 10 percent (Chappelka and Samuelson, 1998). However, in a recent study from several forest stands in eastern Tennessee stem growth of most species was reduced 30–50 percent in a high ozone year as determined with high resolution electromechanical dendrometers (McLaughlin et al., 2007a). Furthermore, ozone exposure was related to increases in whole-tree canopy conductance, depletion of soil moisture and reduced late-season streamflow (McLaughlin et al., 2007b), thus highlighting the broader ecological impacts of ozone on forested ecosystems.

Three regions of Canada are known to have elevated levels of ozone: The Fraser Valley area of British Columbia, the Windsor to Québec City corridor, and the southern Atlantic region (Fuentes and Dann, 1994; Krzyzanowski et al., 2006). Ozone concentrations have trended upward near the more

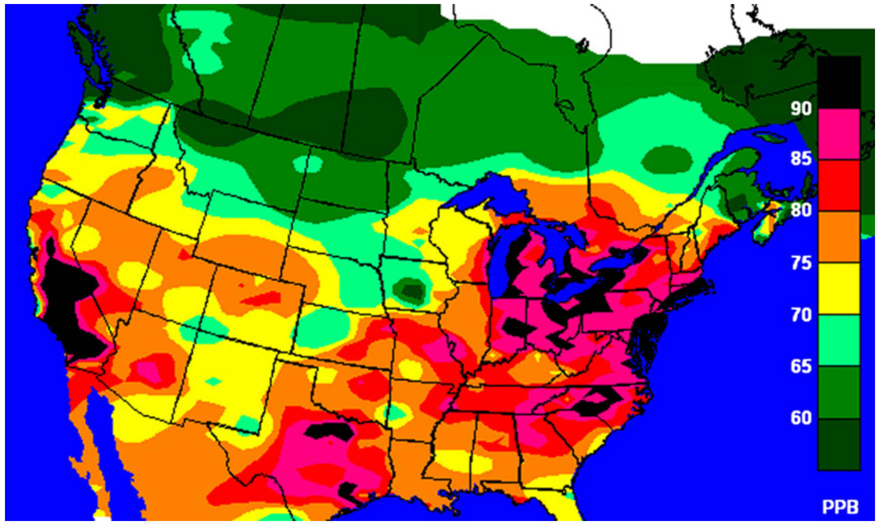

Fig. 9. Spatial distribution of North American surface O_3 calculated using the North American ambient air quality standard metric form 3-year (2001–2003) average of the annual 4th highest daily maximum 8-h average O_3 concentration.

populated regions near Vancouver, the Atlantic provinces, and Ontario (Hales, 2003). Symptoms of ozone damage have been observed on eastern white pine in southern New Brunswick. In Ontario, white pine has exhibited chlorotic dwarfism. In the Fraser River valley during the 1990s 14 species of woody plants were observed to have ozone-like symptoms (NAFC, 2003; CFS, 1999). High ozone exposures are well known in Mexico City and ozone injury to urban trees and trees in forests located south and southwest of the city have been well documented (Miller et al., 2002). Ozone concentrations remain high in the Valley of Mexico, but have been declining since the mid 1990s (Fernández-Bremauntz, 2008). The main forest species sensitive to ozone are *Pinus hartwegii* and *Prunus serotina* var. *capuli*, but ozone is also believed to contribute to the decline problem in stands of *Abies religiosa* in the Desierto de los Leones national park southwest of Mexico City (Alvarado-Rosales and Hernández-Tejeda, 2002).

6. Critical loads for N and S deposition

Canada has recently calculated critical loads (CL) exceedance of acidifying compounds for forest soils in eastern Provinces (Fig. 11) (CCFM, 2006). Generally in Canada, exceedance would be greater if nutrient depletions associated with tree harvesting are considered. The lowest percentage area of exceedance is in Prince Edward Island (3.5 percent of the mapped area); highest exceedances occur in eastern Ontario and southern Québec. Preliminary indications are that more than 48 percent of the upland forest area in Ontario and Quebec and more than 35 percent of the upland forest of Nova Scotia and insular Newfoundland receive acid deposition in excess of the critical load.

In Canada, approximately 40 million hectares of Ontario's forests receive deposition of sulfur and nitrogen in excess of the critical load. If nutrient removals through forest harvesting

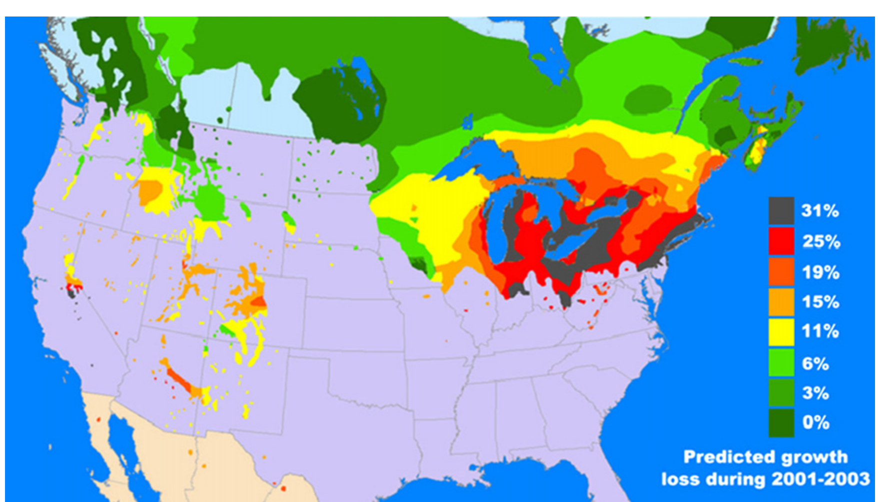

Fig. 10. Estimation of trembling aspen growth loss across North America due to 3-year (2001–2003) average of the ambient annual 4th highest daily maximum 8-h average O_3 concentration.

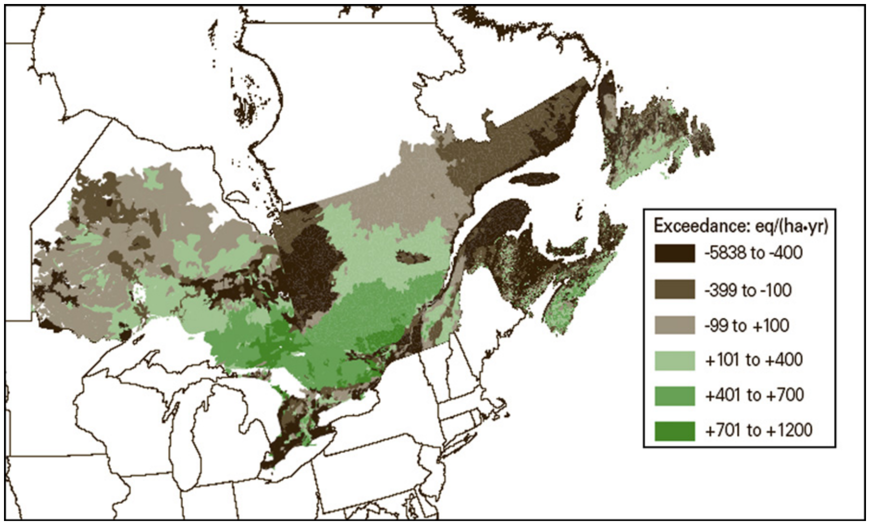

Fig. 11. Critical load exceedance of acidifying compounds in Canada.

are taken into account, the area of exceedance of critical load increases to approximately 45 million hectares and the magnitude of the exceedance also increases (Watmough et al., 2004). In Quebec, researchers have found that areas subjected to critical load exceedance experience a 30 percent reduction in forest growth. Most of the research plots where deposition has exceeded CL are located in nutrient-poor sites in the Laurentian Mountains of the Canadian Shield, and in the Appalachian range of southeastern Quebec. The researchers concluded that further reductions in national and international sulfate and nitrate emissions rates should be undertaken to protect Quebec forests from excessive soil acidification (Ouimet et al., 2001).

Work on and interest in developing and implementing critical loads for the protection of natural resources from air pollution is increasing in the US (Porter et al., 2005). We are not aware of any work on CL per se in Mexico, but it is anticipated that CL evaluations will be done, at least for forests outside of Mexico City that are impacted by N and S deposition (Fenn et al., 2002). In a recent national scale assessment, McNulty et al. (2007) estimated that about 15 percent of the US forest soils in the conterminous United States exceed the acidic deposition loading by more than 250 eq ha^{-1} yr^{-1} and that these sites are predominantly in the northeastern US and portions of southern California. In a more detailed and intensive site-specific data collection and CL calculation process, the CL exceedances for acidity were determined for six New England states and five eastern Canadian provinces. In an estimated 36 percent of the forested land area, atmospheric deposition levels of N and S were in excess of the CL (Forest Mapping Group, 2007). Critical loads for N as a nutrient (eutrophication effects) also occur in areas of the country where N deposition levels are elevated, sites that often coincide with soil acidification impacts. In a review by Fenn et al. (1998), so called N saturated sites in the US are identified. Key areas affected by excess N deposition are in the Northeast (particularly the Adirondacks and Catskills), portions of the Appalachians, high elevation catchments in the Colorado Front Range, and mountains in southern and central California. The empirical CL for N as a nutrient or eutrophication effects in mixed conifer forests in southern and central California (based primarily on nitrate leaching) is estimated to be 17 kg ha^{-1} yr^{-1} (Fenn et al., 2008), and chaparral catchments have a similar empirical CL (Fenn et al., 2003b). In Northeastern forests, elevated nitrate leaching typically begins to occur with N deposition above 10 kg ha^{-1} yr^{-1}, although the response is variable because of other factors affecting N loss (Aber et al., 2003). This suggests a conservative empirical CL for nitrate leaching of approximately 10 kg ha^{-1} yr^{-1} in NE forests. In high elevation catchments in the Colorado Front Range the empirical CL for nitrate leaching is estimated to be 4 kg ha^{-1} yr^{-1} as wet deposition (Williams and Tonnessen, 2000), or 5–8 as total N deposition (Fenn et al., 2003a).

Critical loads for N deposition have recently been determined based on detrimental effects on lichen species community composition (Glavich and Geiser, in press). Because of the extreme sensitivity of some lichen species to N deposition and the enhanced growth of nitrophilous species, the CL values for lichen community changes are unusually low. N deposition as low as 2–5 kg ha^{-1} yr^{-1} from urban and agricultural emissions has impacted lichen communities in the Columbia River Gorge and in forests of the Pacific Northwest and throughout much of California (Jovan and McCune, 2006; Fenn et al., 2007, 2008). Similarly, effects on subalpine vegetation in the Colorado Rockies have been documented with N deposition as low as 5–8 kg ha^{-1} yr^{-1} (Fenn et al., 2003a). Sensitive organisms such as diatoms in high elevation aquatic ecosystems appear to be impacted at even lower N deposition levels. Baron estimated a CL of 1.5 kg N ha^{-1} yr^{-1} as wet deposition for the alteration of diatom assemblages in alpine lakes in Rocky Mountain National Park (Baron, 2006). Because of their high level of sensitivity to N deposition, lichens and diatoms are increasingly being used in the US and elsewhere to determine the CL for aquatic and terrestrial effects of N deposition (Baron, 2006; Fenn et al., 2008;

Glavich and Geiser, in press). The lichen data for this CL work are primarily from the Forest Health Monitoring (FHM) and Forest Inventory and Analysis (FIA) databases of the US Forest Service. The lichen indicator described below is proving to be very useful in identifying areas exposed to and affected by atmospheric deposition. Regarding coordinated monitoring networks, the ICP Forests program (International Cooperative Programme on Assessment and Monitoring of Air Pollution Effects on Forests) established in Europe in 1985 and affiliated programs have collected an overwhelming amount of data and information on CL and CL exceedances (Lorenz et al., 2008). This is in stark contrast to the lack of such a coordinated monitoring, modeling and CL mapping effort in North America, although many individual projects and case studies have been done in the US, and to a larger extent in Canada, as summarized above.

7. Monitoring forest health indicators

In the US, the FHM (http://fhm.fs.fed.us) and the FIA (http://fia.fs.fed.us) programs monitor a suite of forest health indicators to determine the effects of air pollution and other stressors (Stolte, 2001). Data on these indicators are collected on systematically located forest health ground plots integrated with the national forest inventory system (Bechtold and Patterson, 2005). The standard forest inventory plots are located within 2400 hectare hexagons covering most forested lands in the conterminous US[1] and southeastern Alaska. Each plot is remeasured once every 5–10 years on a rotating panel basis. Additional forest health measurements are collected on a 1/16th subset of the standard inventory plots. Forest health indicators include measurements of crown condition, tree mortality, tree damage, soil condition, downed woody material, vegetation structure and diversity, lichen communities, and ozone injury (Riitters and Tkacz, 2004). Results from selected forest health indicators are presented below. Unfortunately, Canada and Mexico currently lack a similar national-level routine monitoring system for forest health. In Canada, the monitoring of national forest health began in 1984 through the Acid Rain National Early Warning System (ARNEWS) (D'Eon et al., 1994). This program was designed to monitor long-term changes in forests that were attributed to pollution and acid deposition, until the closure of ARNEWS in 2000.

7.1. Crown conditions

Tree crowns provide one of the earliest indicators of biotic or abiotic stress on trees. Stressors, such as drought, insects, pathogens and air pollutants, can impact the amount, distribution, and condition of foliage in tree crowns. The crown variables collected on forest health ground plots in the US include: crown ratio—the total length of the tree bole divided into the length of the tree that has living branches; crown density—the blocking of sunlight by the branches and foliage of the crown; foliage transparency of crowns—the amount of sky the can be seen through the foliage part of the crown; and crown dieback—the death of the sun-exposed, growing twigs at the top of the tree crown (Schomaker et al., 2007). To assess amount and distribution of unhealthy crowns in the US, crown dieback and foliar transparency measurements were used to calculate a composite crown index. This index represents the theoretical reduction of individual tree foliage relative to an ideal, fully foliated tree having the same crown diameter, live crown ratio, and crown density (Coulston et al., 2005b). In all ecoregions of the US, less than 15 percent of the basal area was associated with unhealthy crowns. Ecoregion sections having greater than 10 percent average basal area associated with unhealthy crowns were mostly located in the Interior West (Fig. 12). The most likely causes of these unhealthy crowns are droughts and insects or disease outbreaks that have affected these ecoregions (Coulston et al., 2005b).

7.2. Tree mortality

Tree mortality on forest health plots in the US has been assessed using two indices: MRATIO—ratio of annual mortality volume to annual gross growth volume, and DD/LD—the ratio of the average dead tree diameter to the average live tree diameter (Coulston et al., 2005b). An MRATIO of >1 indicates that mortality exceeds growth and that live volume is decreasing. For stands that are not naturally senescing due to old age, a high MRATIO (>0.6) may indicate the activity of acute mortality agents, such as insects, diseases, or fire. High DD/LD ratios (much greater than 1) also indicate senescence or the activity of acute mortality agents. Fig. 13 presents the two mortality indices by ecoregion section of the US. The highest mortality ratios occurred in Idaho and western Washington, most likely related to fire activity. Ratios were also relatively high in parts of the interior West, most likely due to the effects of drought and bark beetle outbreaks. In the East, high mortality ratios occurred in northern Minnesota and Wisconsin, most likely related to a large blowdown and insect outbreaks. High mortality ratios in the Adirondacks of New York are difficult to interpret due to limited data over a short time span. Better assessments of mortality will possible in the future as more plots are remeasured.

7.3. Soil condition

Forest soils are critical components of forest ecosystems. The soil condition indicator collects information on physical and chemical properties of soil on measured plots (Ambrose and Conkling, 2007). Information collected on the forest health plots can be used to assess soil conditions related to forest health and the deposition of atmospheric pollutants. Mean soil pH is responsive to air pollution and precipitation chemistry and is an important indicator of forest ecosystem health (Bailey et al., 2005). The mean soil pH value for measured plots is 4.8 with acidic tendency of soil pH most clear east of the Mississippi River (Fig. 14). The distribution

[1] As of 2008, all states in the conterminous US are included in the annualized inventory with the exception of Nevada and Wyoming.

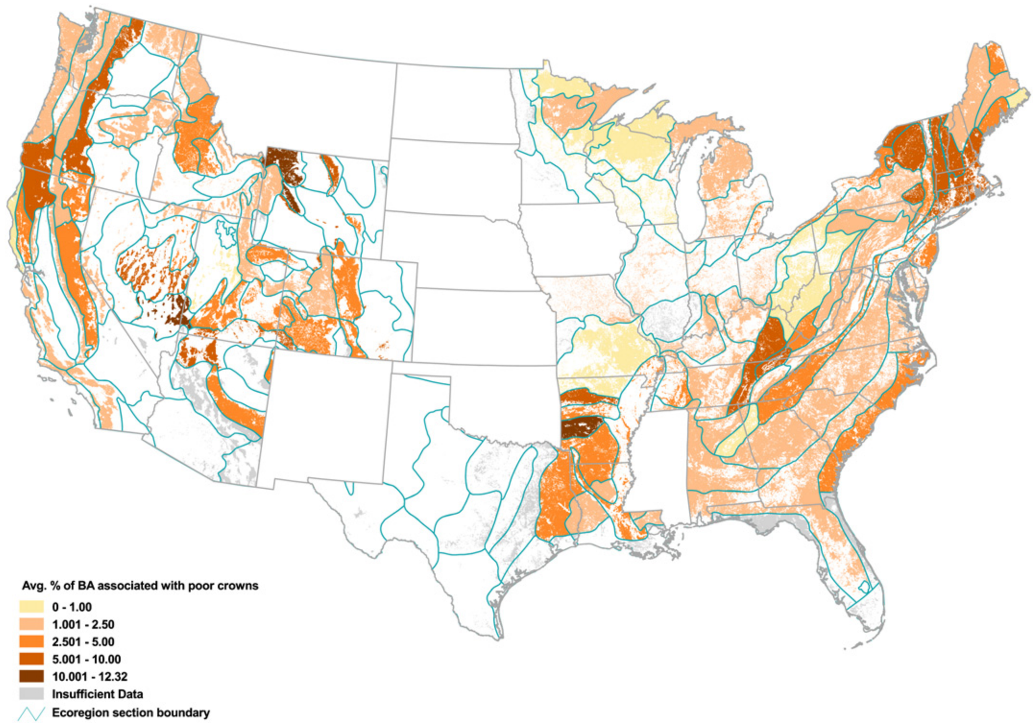

Fig. 12. Average percent of plot basal area associated with poor crown condition by ecoregion section.

of acidic soils in the US coincides with previous assessments conducted by the National Atmospheric Deposition Program (NADP, 2005). Effective cation exchange capacity (ECEC), calculated as the sum of exchangeable bases (sodium, potassium, magnesium, and calcium) and aluminum in soils, indicates the soil's capacity to store critical nutrients (Ambrose and Conkling, 2007). The southeastern US tended to have the greater proportion of forest soils with low effective cation exchange capacity levels (Fig. 15). These soils are predominantly highly weathered ultisols that are low in organic matter (Fig. 16). Total soil carbon content is generally the highest in the northeastern and northern US where decay rates are very

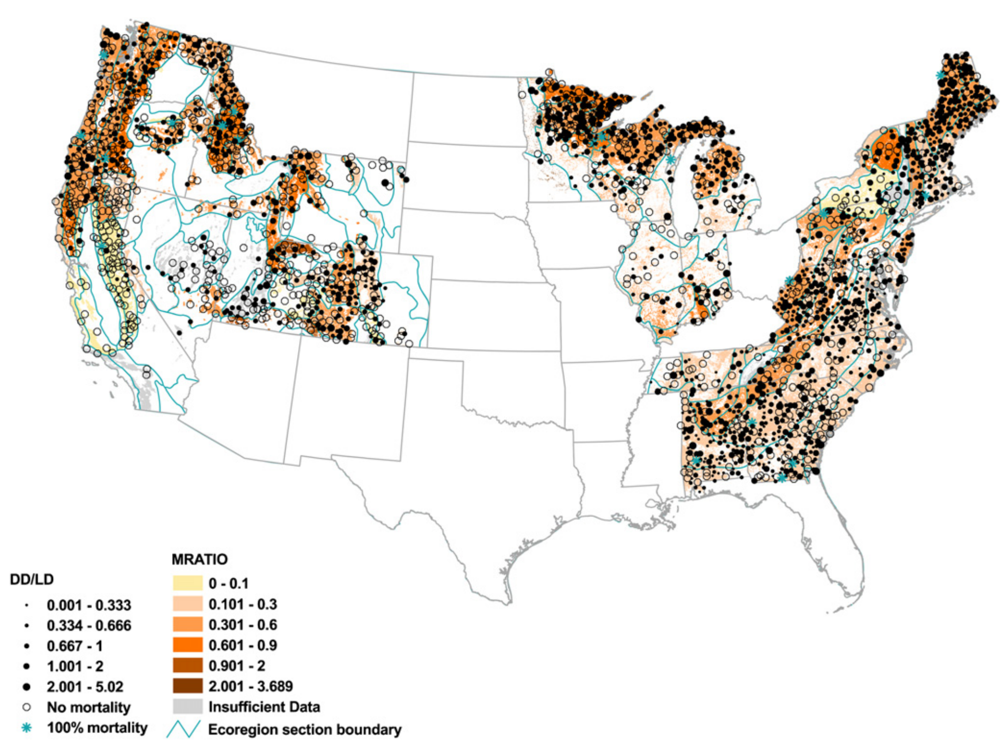

Fig. 13. Tree mortality expressed as MRATIO and DDLD by ecoregion section.

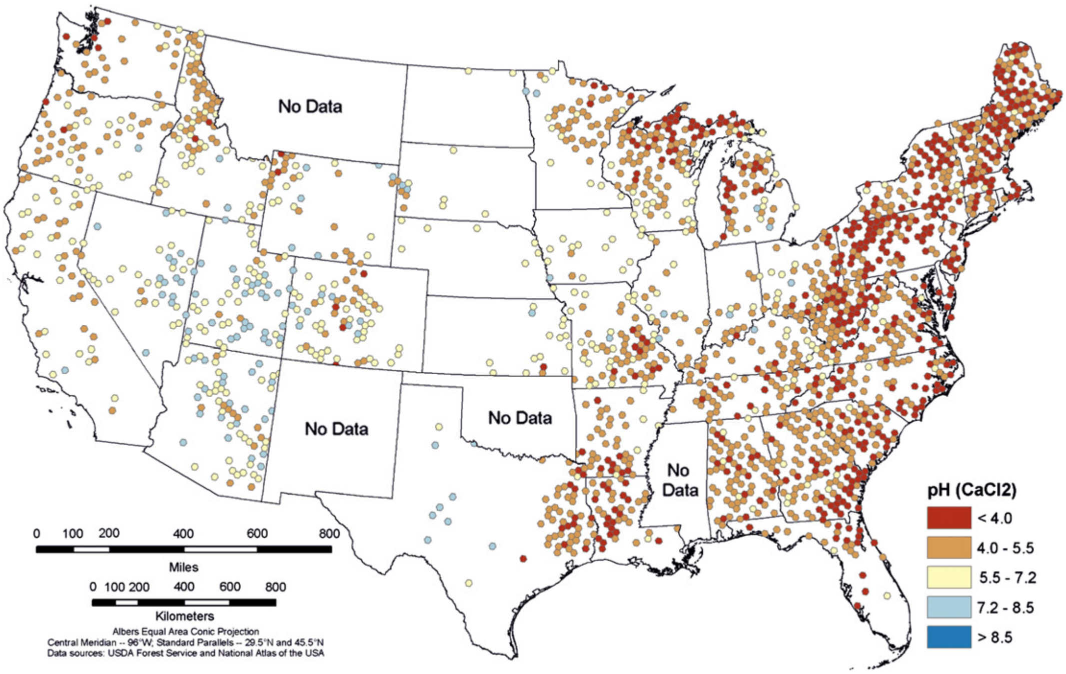

Fig. 14. National map of observations of soil pH in the top 10 cm of soil (2001–2003) by Environmental Monitoring and Assessment Program (EMAP) hexagon. Soil pH was measured in a $CaCl_2$ solution.

low (Fig. 16). Soil carbon content information from forest health plots can be used to assess carbon sequestration rates and amounts for forest soils in the US (Ambrose and Conkling, 2007).

7.4. Ozone biomonitoring

A number of factors complicate the evaluation of ozone impacts on forest ecosystems, such as: visible injury is not always correlated with growth impacts; many environmental factors affect plant sensitivity to ozone (e.g. light, nutrition, moisture); wide genetic variation in ozone sensitivity, even within the same species; topographic effects on ozone exposure; year to year variation in the severity of ozone injury; moisture status of plant affects stomatal conductance and ozone uptake into plant tissue; and similar symptoms sometimes caused by other abiotic and biotic agents. Considerable evidence suggests that a general plant response to ozone is to

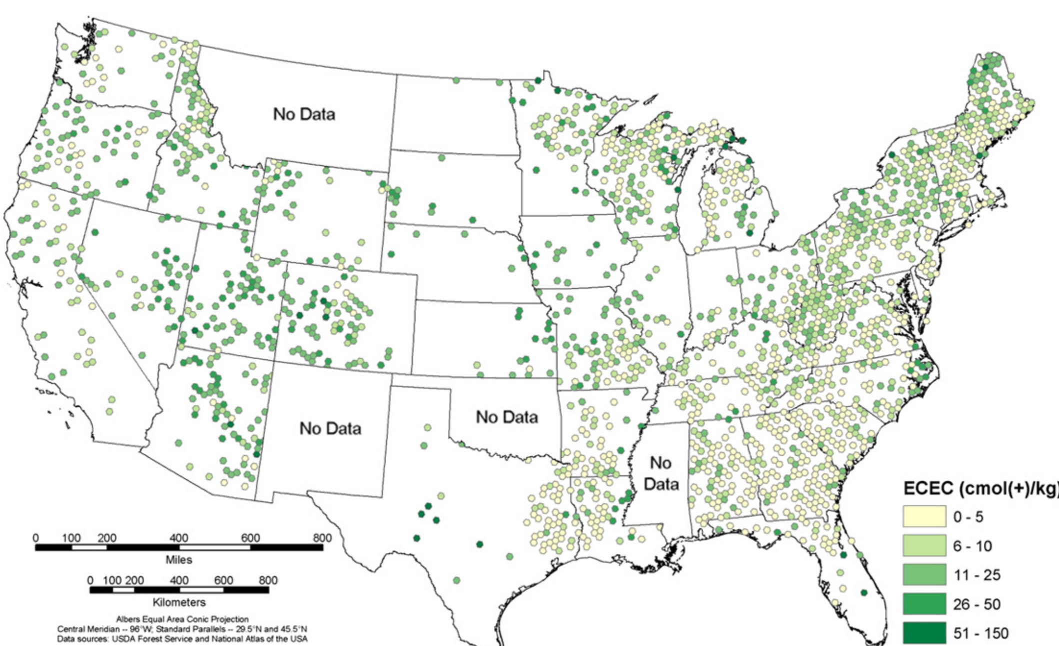

Fig. 15. National map of effective cation exchange capacity in the top 10 cm of soil (2001–2003) by Environmental Monitoring and Assessment Program (EMAP) hexagon.

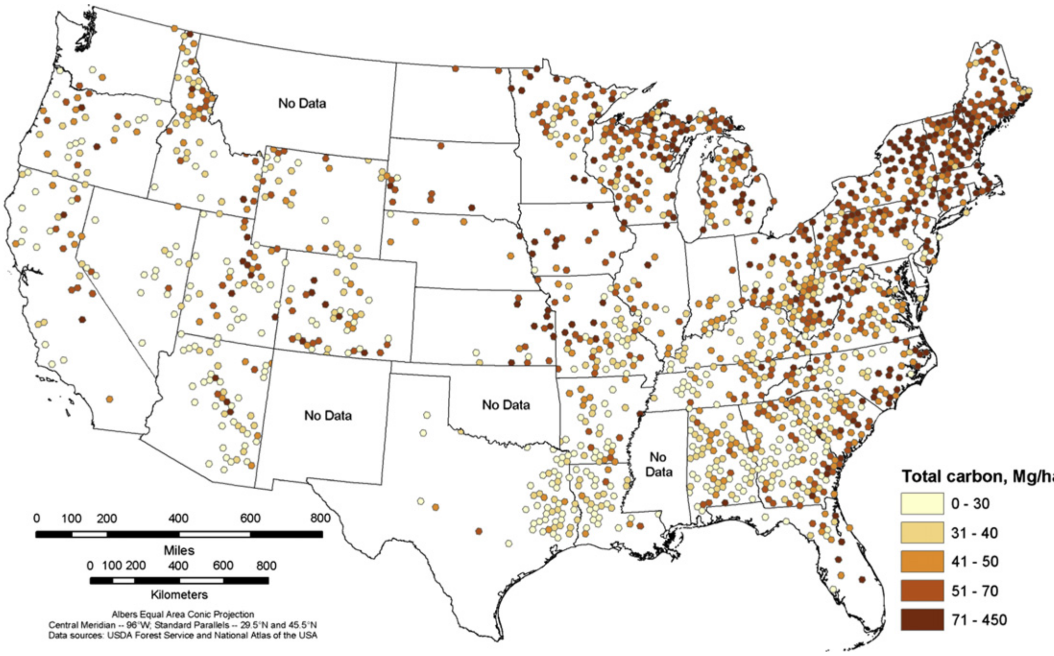

Fig. 16. Total soil carbon—forest floor and top 20 cm of soil (2001–2003) by Environmental Monitoring and Assessment Program (EMAP) hexagon.

allocate additional resources to aboveground tissues such as foliage and woody plant parts, resulting in lower root growth. However, in most field studies the effects of ozone exposure on roots are not evaluated and the long-term ecological consequences of reduced root growth are not well known. Experience has shown the importance of long-term biomonitoring in conjunction with air quality data in order to ascertain the spatial patterns and severity of ozone impacts (Smith et al., 2003). Biomonitoring results are important for identifying impacted areas and as a guide for where more in depth studies of ozone effects or silvicutural prescriptions are needed.

The effects of ozone on forest ecosystems are monitored by assessing damage to ozone-sensitive species on ozone biomonitoring sites located in forests throughout the US. The severity of foliar injury is assessed according to an injury score—0 to 4.9 for no or minute injury, 5 to 15 for light to moderate injury, 15 to 25 for moderate to severe injury, and greater than 25 for severe injury. Spatial interpolations of plot injury scores for the period from 1999 to 2002 are presented in Fig. 17 (Ambrose and Conkling, 2007). This analysis shows that the highest foliar injury occurred in the Mid-Atlantic and the Southeast, with significant injury recorded in Southern

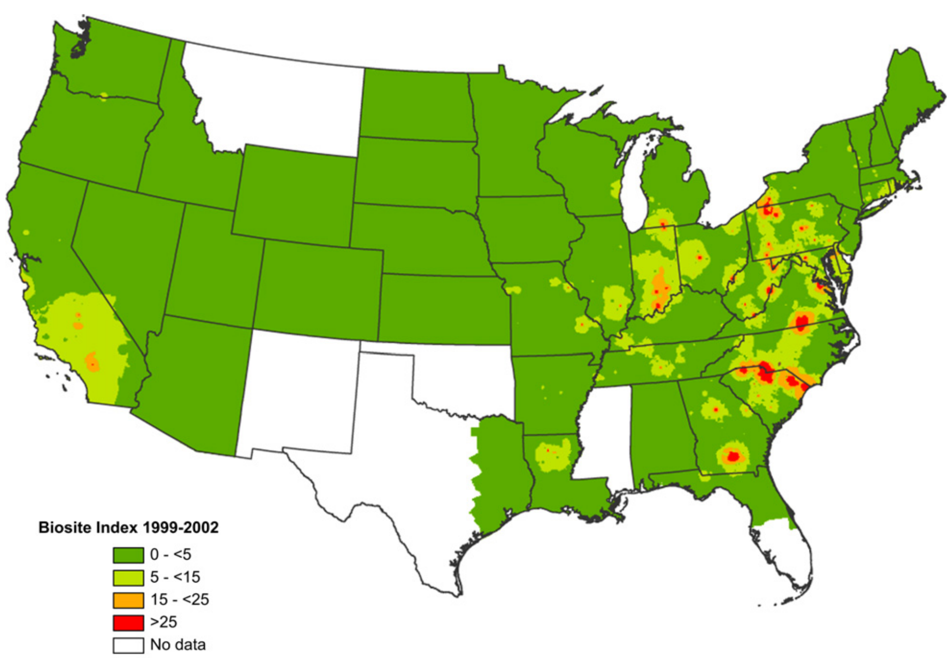

Fig. 17. Ozone biosite scores for 1999–2002.

California. Ozone injury occurrence is believed to be more widespread in the eastern US than in the West because of the more mesic conditions in the East which foster greater plant uptake of ozone (Smith et al., 2003). In California ozone injury was reported from 25 to 37 percent of the monitoring sites from 2000 to 2005 (Campbell et al., 2007). In the western US other than California, ozone injury has been reported from one monitoring site in Utah and from a site in Washington within the Columbia River Gorge approximately 160 km east of Portland (Campbell et al., 2007; Smith et al., 2003).

7.5. Lichens

The FHM and FIA programs also monitor effects of air pollution and climate on lichen communities. Biotic indices have been developed based on lichen community data along air pollution and climate gradients (Geiser and Neitlich, 2007; Jovan and McCune, 2005). Fig. 18 presents spatial interpolation of lichen index scores across Washington and Oregon showing a decrease of air pollution sensitive lichens near major metropolitan areas. (Geiser and Neitlich, 2007). Non-metric multidimensional scaling was used by Jovan and McCune (2005) to characterize landscape-level trends in lichen community composition from monitoring plots in northern and central California forests. Two macroclimatic gradients were found related to temperature/elevation and moisture. As the permanent lichen plots are re-measured, changes in lichen community composition may provide an early warning of changing climatic conditions. Understanding the differential plant sensitivities allows Canada to monitor the impact of air pollution throughout the environment. This is the basis of Environment Canada's Ecological Monitoring and Assessment (EMAN) Network community lichen monitoring. EMAN consist of linked organizations and individuals involved in ecological monitoring in Canada to detect, describe, and report on ecosystem changes (Thormann, 2006). In Europe, specific lichen indicator species, or all lichens, as indicators of sulfur dioxide pollution are monitored in five 10×10 cm quadrat sampling frames hung 1.5 m above ground on all cardinal points of different tree species (Asta et al., 2002). This method is efficient in some parts of Canada (e.g., Nova Scotia) where the sites are in protected areas and can be used to assess air quality in areas not accessible to air monitoring stations. Nova Scotia has air quality monitoring stations across the province which measure particulates, sulfur dioxides, nitrous oxides and ozone. Data from these measured pollutants can be correlated with changes in lichen presence and abundance over time to provide a better understanding of how certain pollutants may affect ecosystems. Because lichens are closely tied to overall biodiversity of ecosystems, a change in diversity of lichens over time may signify impending impacts to other groups of organisms (Cameron, 2003). An arboreal lichen survey in the city of Hamilton, Ontario, Canada, to assess relative local air quality, showed that air quality generally improved with increasing distance from the city core, as indicated by an increase in lichen biodiversity (http://www.eman-rese.ca/eman/ecotools/protocols/terrestrial/lichens/metal.html).

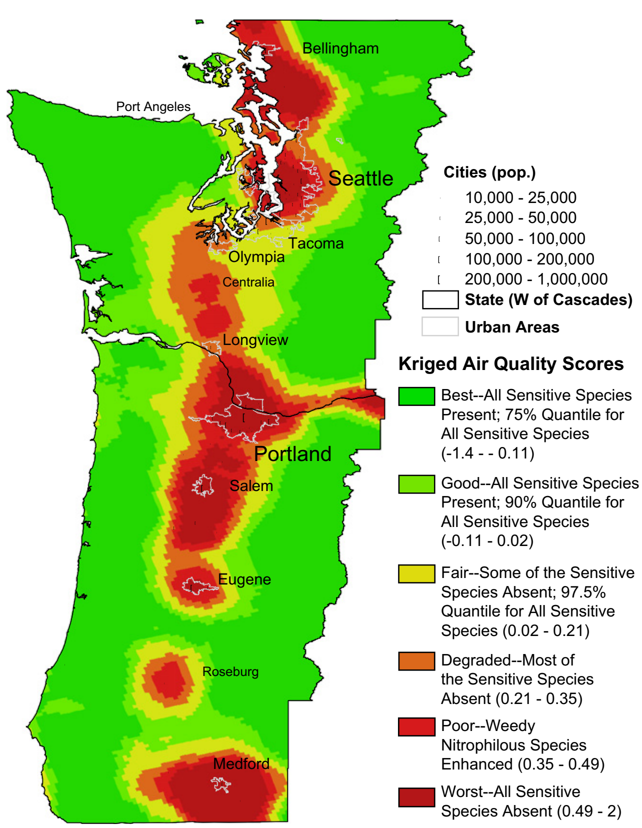

Fig. 18. Lichen index scores for Washington and Oregon.

8. Risk mapping

The US Forest Service and partners have recently completed a national risk assessment that maps potential future risk of tree mortality due to insects and diseases (Krist et al., 2007). Risk was defined as an expectation that 25 percent or more of the standing live basal area of trees greater than 1 inch in diameter will die over the next 15 years due to insect and disease activity. According to this assessment, more than 23 million hectares of forest land are at risk (Fig. 19). Most of this risk can be attributed to 11 risk agents including: bark beetles of western conifers, oak decline, southern pine beetle, root diseases, and gypsy moth. This strategic assessment provides a useful tool in developing broad prevention strategies.

9. Future challenges, monitoring needs and data gaps

Management of forest ecosystems under a changing climate call for approaches that increase resistance and resiliency of forests to disturbances (Millar et al., 2007). In the future, timely detection, analysis, and reporting of adverse changes in forest health must be improved in order to facilitate adaptive management responses. To increase our understanding of the adverse changes in forest health, we need to expand our evaluations of the extent, severity and dynamics of forest stressors and enhance our understanding of the likely effects of changing climates on forest ecosystems. Our continued

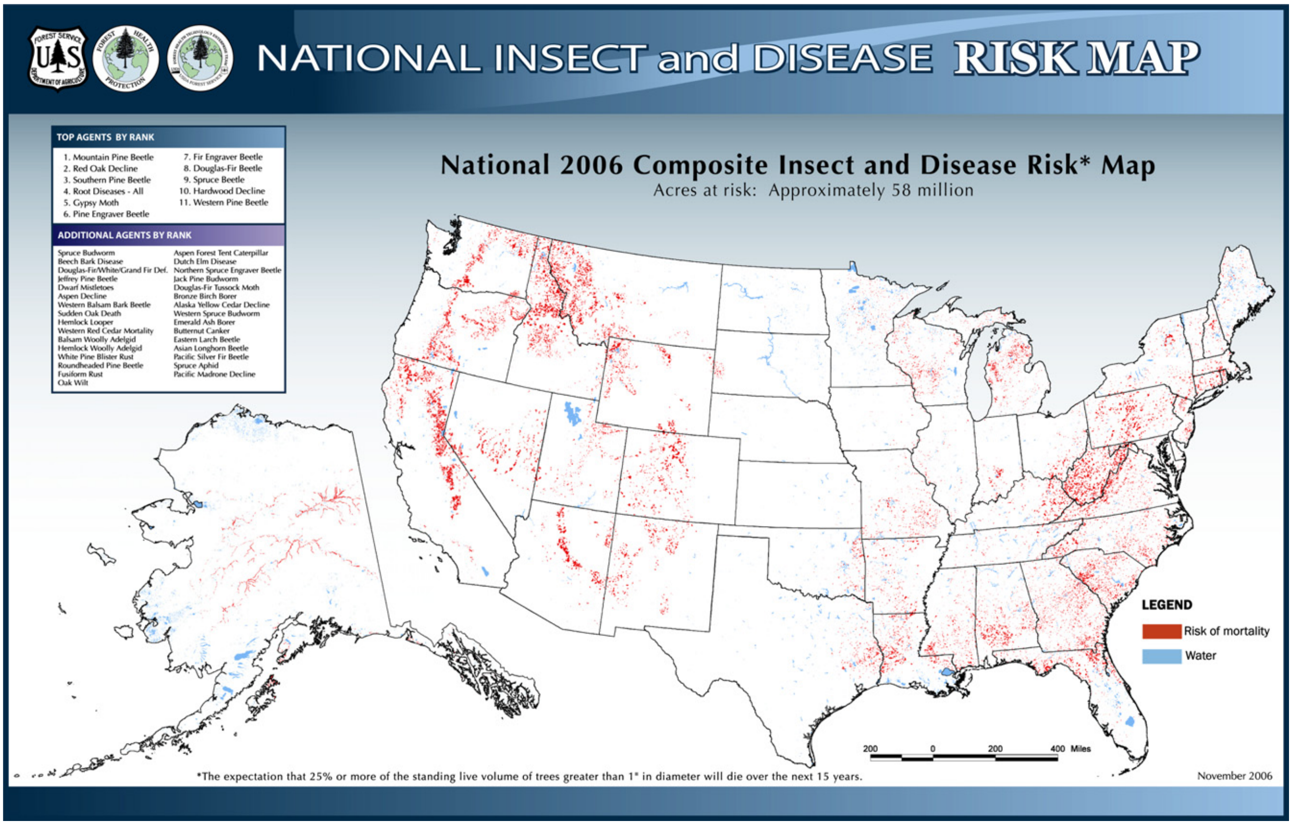

Fig. 19. Forest insect and disease risk map.

development of national and regional risk assessments that include a range of possible climate scenarios will promote development of more effective strategies to improve resistance and resiliency of forests to disturbances. Perhaps the greatest future challenge is maintaining and enhancing nationwide networks of forest health assessment plots. Consistent, large-scale, long-term data sets, such as the FIA and FHM networks in the US, provide critical baselines that allow detection of changes in forest health indicators (Riitters and Tkacz, 2004). Canada and Mexico would certainly benefit from development of similar networks, while in the US the challenge is maintaining a stable level of support for the existing network in the face of budget shortfalls. These forest health networks should also be linked with process-level research studies to facilitate development and validation of ecosystem models and risk assessments of future changes in forest ecosystems due to climate change.

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