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Climate, trees, pests, and weeds: Change, uncertainty, and biotic stressors in eastern U.S. national park forests

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ABSTRACT

The US National Park Service (NPS) manages over 8900 km^2 of forest area in the eastern United States where climate change and nonnative species are altering forest structure, composition, and processes. Understanding potential forest change in response to climate, differences in habitat projections among models (uncertainty), and nonnative biotic stressors (tree pests and diseases and invasive plants) are vital for forward-looking land management. In this research, we examined potential changes in tree habitat suitability using two climate scenarios ('least change' and 'major change') to evaluate uncertainty in the magnitude of potential forest change. We further used nonnative tree pest and plant data to examine strengths and spatial patterns of these stressors and their correlations with projected changes in tree habitat. Analyses included 121 national parks, 134 tree species (from the US Forest Service Climate Change Atlas), 81 nonnative tree pests (from the US Forest Service Alien Forest Pest Explorer Database), and nonnative vascular plant presence data from each park. Lastly, for individual tree species in individual parks, we categorized potential habitat suitability change (from late 20th century baseline to 2100) into three change classes: large decrease (<50%), minor change (50–200%), and large increase (>200%). Results show that the potential magnitude of forest change (percentage of modeled tree species in the large decrease and large increase classes, combined) varies from 22% to 77% at individual parks. Uncertainty (the percentage of tree species in differing change classes across climate scenarios) varies from 18% to 84% at parks. Nonnative plant species comprise from <10% to about 50% of the flora at parks. The number of nonnative tree pest species ranges from 15 to 70 among parks. Potential forest change, uncertainty, and nonnative pests and plants have significant positive correlations, illustrating the broad scope of potential compounding effects and future changes in many eastern forests. The combination of rapid climate change and nonnative stressors may accelerate decline of some tree species and inhibit other species from occupying climatically suitable habitat. Stewarding forests for continuous change is a challenge for park managers. Understanding and anticipating projected rates and directions of forest change and nonnative biotic stressors should facilitate monitoring and management efforts on park lands and across the broader landscape.

1. Introduction

Global change agents, including rising temperatures, pollution, fragmentation, and nonnative insect pests, diseases, and invasive plants, are altering ecosystem processes, structure, and composition (Vitousek, 1994; Grimm et al., 2013). Forest managers are dealing with both rapid directional change and multiple uncertainties (Heller and Zavaleta, 2009). Understanding potential directions of change as well as sources of uncertainty is vital for effective land management (Harris et al., 2012). For example, interpretations of climate-vegetation models to project shifts in habitat suitability due to climate change can inform natural resource planning and management actions (Monahan et al., 2013). Many tree species are keystone or foundation species and shifts in forest composition and structure will affect other trophic levels within the ecosystem (Ellison et al., 2005). Biotic stressors, such as nonnative tree pests and nonnative invasive plant species, can exacerbate climate stress and alter compositional trajectory of forests (Dukes et al., 2009; Sturrock et al., 2011). Furthermore, climate and forest changes may have cascading effects on resource

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management, forest operations, and recreation. This research focuses on two global change factors, climate and nonnative species, and examines potential landscape-scale forest change, uncertainty, and presence of nonnative biotic stressors for national parks in the eastern half of the continental U.S. These forest lands, specifically selected for protection of significant natural and cultural resources, are imbedded within a matrix of other ownerships, and analyses may be pertinent to forests under other jurisdictions.

The US National Park Service (NPS) manages over 8900 km^2 of forest area in the eastern U.S. (east of the 100th meridian). Many of these national parks were authorized up to 100 years ago (e.g., Acadia National Park in 1916 and Shenandoah, Great Smoky Mountains, and Mammoth Cave National Parks in 1926) and have relatively high protection status among public lands in the U.S. However, land use legacies and other anthropogenic stressors have had long-term influences on these landscapes. Current overstory composition in many eastern forests is partially a result of the combination of logging, grazing, farming, and other settlement activities. European settlement activities impacted over 80% of the land that became Great Smoky Mountains National Park, including 40% utilized for corporate logging (Pyle, 1988). Logging and fire, followed by successful fire suppression campaigns, shifted forest composition in many areas of the East from pine to oak and others from oak to maple and associated mesic species (Whitney, 1987; Abrams and Nowacki, 1992). Logging and associated fires on Isle Royale in Lake Superior shifted the dominant tree species from fire-sensitive *Abies balsamea* (balsam fir) to *Populus tremuloides* (quaking aspen) and *Betula papyrifera* (paper birch) before Isle Royale National Park's creation (Janke et al., 1978). This succession of settlement and protection within parks often resulted in large continuous tracts of fairly even-aged stands at high risk to other stressors such as insects and disease.

Introduction of nonnative organisms has further transformed eastern forest landscapes. Discovered in the U.S. in the early 1900s, the nonnative disease chestnut blight (*Cryphonectria parasitica*) spread rapidly, decimating *Castanea dentata* (American chestnut) trees across the East (Anagnostakis, 1987). Once common and often an overstory dominant, *C. dentata* was primarily replaced by *Quercus rubra* (red oak), *Quercus prinus* (chestnut oak), and *Acer rubrum* (red maple) in forests of Shenandoah and Great Smoky Mountains National Parks (Woods and Shanks, 1959; Karban, 1978). However, species such as oaks are also susceptible to numerous nonnative pests. Gypsy moth (*Lymantria dispar*), a nonnative insect first introduced in the late 1800s, defoliates several eastern tree species and has caused major defoliations and diebacks when combined with other stressors (Davidson et al., 1999). For example, 28% of oaks within Morristown National Historical Park, New Jersey died after gypsy moth defoliation (Kegg, 1971). Similarly in Shenandoah National Park, gypsy moths defoliated over 43,000 hectares in 5 years and killed 54% of oak trees used as winter dens by black bears (Kasbohm et al., 1996). Hemlock woolly adelgid (*Adelges tsugae*) has already devastated *Tsuga canadensis* (eastern hemlock) stands in several NPS managed areas (Abella, 2014a) and caused cascading impacts to avian and arthropod species at Delaware Water Gap National Recreation Area and Shenandoah National Park (Allen et al., 2009; Rohr et al., 2009). These and numerous other tree pests and diseases in the eastern US threaten dozens of trees species (Liebhold et al., 2013).

Invasive nonnative plant species can also be major concerns in eastern forests (Allen et al., 2009) by reducing abundance and diversity of native species (Vilà et al., 2011). Management of nonnative species in forests generally leads to increases in native plant cover and diversity and improved establishment and recruitment of native tree species, though responses are varied and infested areas may require intensive management to achieve desired effects (Loh and Daehler 2008; Abella, 2014b). Other stressors, including tree pests, can facilitate nonnative plant invasions (e.g., hemlock woolly adelgid at Delaware Water Gap National Recreation Area; Eschtruth and Battles 2009) and warming temperatures are likely to enable many invasive species to expand their ranges (Dukes et al., 2009).

As described above, national park forests experienced major changes both before and after establishment as protected areas. Climate is changing rapidly and many parks across the NPS (81% of natural resource parks) are already experiencing climatic conditions at the warm extremes of their historical range of variability (Monahan and Fisichelli, submitted for publication), and climate change is correlated with measurable responses by birds, mammals, and vegetation within national parks (Moritz et al., 2008; Tingley et al., 2009; Dolanc et al., 2013). Climate change affects all tree life stages, from seed development, germination, and emergence (Walck et al., 2011; Fisichelli et al., 2014a) to seedling growth and recruitment (Fisichelli et al., 2012, 2014b) to survival of overstory trees (Allen et al., 2010). As the rate of climate change accelerates and impacts compound over time, forests are likely to incur more widespread changes in overstory composition and related changes to other biota.

The objective of this study was to examine regional patterns of projected forest changes and nonnative biotic stressors within landscapes encompassing eastern national park forests. Specifically, we assessed changes in potential tree habitat suitability under two climate scenarios ('least change' and 'major change') to evaluate one important source of uncertainty in the magnitude of potential forest change. We further used nonnative tree pest and plant data to examine strengths and spatial patterns of these stressors and their correlations with projected changes in tree habitat. We discuss potential forest change and uncertainty, how biotic stressors may accelerate forest change and constrain adaptive capacity, and appropriate adaptation strategies.

2. Methods

2.1. Study areas

A total of 121 NPS management units (hereafter 'parks') were included in analyses. These parks are located across the eastern US from Maine to Minnesota to east Texas and north Florida and have >4 hectares of forest area (based on National Land Cover Database 2006; see Fig. 1 for locations and Appendix A Table A.1 for unit names). The Appalachian National Scenic Trail, due to its length (stretching from Georgia to Maine), was divided into four separate units for analyses (deep South, southern Virginia, Mid-Atlantic, and New England).

2.2. Climate data

We evaluated a potential range of future climatic conditions using two general circulation models (Parallel Climate Model [PCM] and HadleyCM3 [Had]) and two greenhouse gas emissions trajectories (B1 and A1FI) that bracket the probable range of future greenhouse gas emissions (see Iverson et al., 2008 for details). The PCM combined with the B1 scenario presents a 'least change' climate scenario based on dramatic cuts in greenhouse gas emissions and modest climatic changes, and the Had-A1FI combination represents a 'major change' scenario under high greenhouse gas emissions. These two scenarios project an increase in mean annual temperature of 3–6 °C (5.4–10.8 °F) over the 21st century in the eastern US and varied changes in precipitation (−27% to + 75%), depending on geographic location and climate model (values are compared with the 1961–1990 baseline; Appendix A Fig. A.1).

2.3. Suitable habitat modeling

The projections of climate and suitable habitat for tree species come from the US Forest Service Climate Change Tree Atlas (Prasad et al., 2007; Iverson et al., 2008). Projections of tree habitat suitability changes in response to climate are based on climate projections and the relationships between environmental factors, including climatic variables, and individual species' abundance and distribution. The statistical model used in these analyses, the DISTRIB model, uses an ensemble of regression tree techniques to correlate tree abundance to 38 environmental predictor variables (including 7 climate, 5 elevation, 21 soil, and 5 land use variables) across the eastern US (Iverson et al., 2008). Future climate projections (PCM-B1 and Had-A1FI) relate modeled climate values to local observations through probability functions (Iverson et al., 2008).

US Forest Service Forest Inventory and Analysis (FIA) data from the period 1980–1993 were used to calculate mean importance values (IVs: average of relative stem density and relative basal area) within 20×20 km grids for 134 eastern tree species (Prasad et al., 2007). Model runs used mean monthly climate normals for 1961–1990 and a future 30-year time period ending in 2100. The average number of tree species per park with modeled habitat (present and/or future) varied from 23 to 99 species (mean = 74). Model output indicates *potential* suitable habitat for a tree species, and not where the species may occur at a particular point in time. Additional analyses using 30-year time periods ending in 2040 and 2070 were run for a subset of parks (Voyageurs National Park in Minnesota, Acadia National Park in Maine, Catoctin Mountain Park in Maryland, Congaree National Park in South Carolina, and Buffalo National River in Arkansas) to illustrate change in potential suitable habitat over time across the 21st century. For sufficient sample sizes (FIA plots) and due to variations among model runs and climatic predictors, we buffered the area surrounding each park to include 40 DISTRIB cells.

2.4. Forest change and uncertainty

We used the DISTRIB model to quantify potential forest change and uncertainty in eastern park landscapes. For analyses, we categorized each tree species' ratio of future (2100) to baseline (1990) habitat suitability for each climate scenario ('least change' and 'major change') into three classes: large decrease, minor change, and large increase (based on Prasad et al., 2007; Swanston et al., 2011). For common species, future:baseline ratios of habitat suitability <0.5 (i.e., >50% reduction in habitat) were classed as large decreases, ratios >2.0 (i.e., >100% increase in suitable habitat) denote large increases, and all values in between were in the minor change class. For rare species within a park (i.e., baseline IVs <5) we used more conservative estimates of change: ratio cutoff values were 0.2 and 8.0, for large decrease and large increase classes, respectively. Species with no baseline habitat but with new potential future suitable habitat were classified as large increases. To quantify the mean magnitude of potential change that park forests may undergo, the percentage of modeled species for each park in the large decrease and large increase classes was averaged across the two climate scenarios. To quantify uncertainty in forest change, we compared habitat projections from the two climate scenarios and calculated percentage of species in each park in differing change classes by climate scenario (e.g., a species may have minor change in suitable habitat under the PCM-B1 but a large decrease under the Had-A1FI model).

2.5. Biotic stressors

In addition to projections of forest change in response to climate, we assessed present levels of two nonnative biotic stressors: nonnative plants and tree pests and diseases (i.e., not native to the continental US). We used the vascular plant species lists for each park contained in the NPS Inventory and Monitoring Program's NPSpecies database (https://irma.nps.gov/NPSpecies/) to determine percentage of nonnative plant species comprising a park's flora. The NPSpecies database contains species lists compiled and certified by park managers and researchers based on species observations, vouchers, and other occurrence evidence. The percentage of nonnative plant species provides a general measure of stress that nonnative plants may have on native plant communities and is not intended to portray species-specific invasiveness and direct local impacts.

Presence of nonnative forest insects and diseases (hereafter referred to as tree pests) was derived from the US Forest Service Alien Forest Pest Explorer (AFPE) Database (http://foresthealth.fs.usda.gov/portal/Flex/APE). Occurrence data are available at county or state level spatial scales, depending on pest species. We considered a park within the infested area for a pest if the park boundary intersected an infested zone. Thus, pests may not be present within a park at the time of mapping, but at minimum, parks are at high risk of becoming infested in the near future. The total number of tree pests for each park was used as a general measure of stress. Pest host tree data were used to determine potential suitable parks for tree pests, and for a subset of tree pests (12 species known to cause major damage/mortality to tree species, http://foresthealth.fs.usda.gov), we calculated the number of parks within and outside infested areas. Additionally, future tree habitat suitability was compared between parks within and outside infested areas for a subset of tree and pest associations (7 of the above 12 host – pest associations which had $\geqslant$10% of parks in infested areas). Pearson's correlation coefficients among change, uncertainty, and biotic stressor variables were also calculated. All statistical analyses were run using the R software package (v.3.0, R Core Team, 2013).

3. Results

3.1. Tree habitat change and uncertainty

Climate change is projected to create significant alterations in potential tree habitat suitability at parks across the eastern US (Fig. 1a, see Appendix A Table A.1. for park-level values). The mean magnitude of potential forest change – the percentage of modeled species for each park in the large decrease and large increase classes – varies from 22% to 77% at individual parks. Not surprisingly, this trend significantly correlates with projected change in mean annual temperature ($r = 0.62$, p-value <0.001) and has a general south to north gradient of increasing potential change, with upper Midwest and northeast parks exhibiting the greatest potential change. For example in the south, Cumberland Island National Seashore in Georgia and Vicksburg National Military Park in Mississippi each have <37% of modeled species with large changes in potential habitat, while in the North, Sleeping Bear Dunes National Lakeshore in Michigan and Acadia National Park in Maine each have >70% of modeled species in large change classes.

Uncertainty – the percentage of species with differing change classifications between the 'least change' and 'major change' scenarios – varies from 18% to 84% (Fig. 1b). Uncertainty also significantly correlates with the difference in temperature projections between 'least change' and 'major change' scenarios ($r = 0.19$, p-value = 0.04) and mean magnitude of potential change ($r = 0.31$, p-value < 0.001, Table 1). Ocmulgee National Monument in Georgia and Timucuan Ecological and Historical Preserve in Florida each have fewer than 30% of modeled species in differing

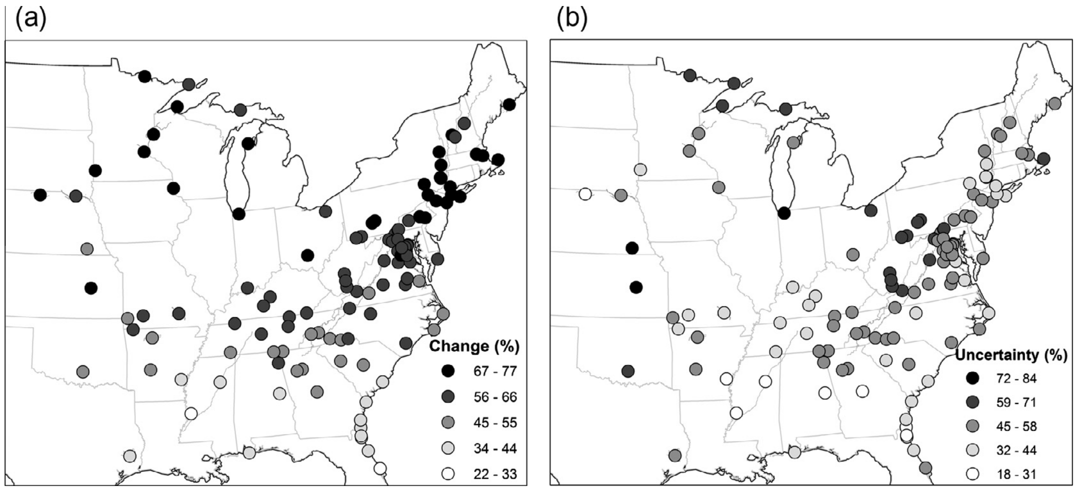

Fig. 1. Projected change and uncertainty in tree habitat suitability at 121 national parks in the eastern U.S. (a) Projected change denotes the average percentage of modeled tree species at each park projected to undergo large change (>50% decrease or >100% increase) in habitat suitability by 2100. (b) Uncertainty in habitat projections due to magnitude of climate change is calculated as the percentage of species within a park in differing habitat suitability change classes by climate scenario ('least change' and 'major change', see Methods for further details).

Table 1
Pearson's correlation coefficients among forest change, uncertainty, and biotic stressors in eastern US national park forests. Forest change is the average percentage of modeled tree species at each park projected to undergo large change (>50% decrease or >100% increase) in habitat suitability by 2100. Uncertainty in forest change is calculated as the percentage of species within a park in differing habitat suitability change classes by climate scenario ('least change' and 'major change'). Nonnative plants is percentage of the local flora comprised by nonnative plant species and tree pests is the number of nonnative tree insects and diseases present within areas including each park. Significance codes for tests of associations among paired variables: *p*-value <0.001 '***', <0.05 '*', ⩾0.05 'NS'.

	Uncertainty	Nonnative plants	Tree pests
Forest change	0.31***	0.36***	0.72***
Uncertainty		0.09 (NS)	0.30***
Nonnative plants			0.55***

change classes by climate scenario. Conversely, Grand Portage National Monument in Minnesota and Indiana Dunes National Lakeshore in Indiana have approximately 70% of modeled species in differing change classes by climate scenario. These results suggest that change in future tree habitat suitability and forest composition may be greater at northern parks. However, rate and direction of change is more dependent on actual greenhouse gas emissions and associated climate change at northern than southern parks.

Change and uncertainty for selected tree species at five parks illustrate the breadth of habitat suitability projections for presently common species (Fig. 2). Patterns of change and uncertainty vary from large reductions in potential habitat under both climate scenarios (Fig. 2a, d, g, j and m) to major uncertainty dependent on emissions and climate (Fig. 2b, e, h, k and n) to substantial increases in potential habitat in both scenarios (Fig. 2c, f, i, l and o). For example, at Acadia National Park, *A. balsamea* (balsam fir) is projected to lose considerable habitat under both climate scenarios, *Acer saccharum* (sugar maple) is projected to gain habitat in the future, and potential habitat for *T. canadensis* (eastern hemlock) depends on the magnitude of climate change, with the species retaining current habitat under the 'least change' scenario but losing substantial habitat under 'major change'. Projections for individual species also differ by park, as shown by *A. saccharum* losing habitat at Buffalo National River in Arkansas but gaining habitat at Acadia.

3.2. Biotic stressors

Biotic stressor levels (nonnative flora and tree pests) vary broadly across parks but also display broad south-north geographical patterns and significant correlations with mean magnitude of potential forest change (Table 1, Fig. 3, Appendix Table A.1). Nonnative plant species comprise from <10% to about 50% (mean = 20%) of the local flora at parks. For example, nonnative plants comprise 10% of the flora at Big Thicket National Preserve in Texas and >40% at Gateway National Recreation Area in New York and Valley Forge National Historical Park in Pennsylvania. Similarly, the number of nonnative tree pest species varies from 15 to 70 (mean = 38), with <20 species at both Pea Ridge National Military Park in Arkansas and Chickasaw National Recreation Area in Oklahoma, and >50 at Delaware Water Gap National Recreation Area in New Jersey and Pennsylvania and the New England section of the Appalachian National Scenic Trail.

The biotic stressors positively correlate with one another ($r = 0.55$) and tree pests positively correlate with tree habitat uncertainty ($r = 0.30$, Table 1). Thus, parks with high levels of modeled change and high uncertainty generally have high numbers of biotic stressors.

The 121 eastern NPS parks in this study occur within infested areas of 81 nonnative tree pests. Twelve pest species with major eastern tree species as hosts are shown in Table 2. Emerging threats still absent or rare in eastern forests include sudden oak death (*Phytophthora ramorum*, 0 parks) and Asian longhorned beetle (*Anoplophora glabripennis*, 3% of parks with suitable tree hosts in infested areas). Widely established pests include chestnut blight (*C. parasitica*, 100% of potential parks) and Dutch elm disease (*Ophiostoma novo-ulmi*, 100% of parks). Other pests currently expanding across the East include hemlock woolly adelgid (*A. tsugae*), already infesting areas that include 78% of parks containing its host tree, *T. canadensis*, and emerald ash borer (*Agrilus planipennis*) in 25% of park areas containing *Fraxinus* spp. trees.

Individual host tree future habitat suitability varied significantly by pest presence/absence for 4 out of 7 tree species analyzed (p-values <0.05), but there were large overlaps in the ranges of habitat ratios between parks within and outside infested areas (Fig. 4). *Cornus florida* (pest = dogwood anthracnose; t-test = −2.43, p-value = 0.02), *Quercus* spp. (pest = gypsy moth; t-test = −2.01, p-value = 0.04), and *Pinus strobus* (pest = white pine blister rust; t-test = −3.83, p-value <0.001) each have slightly higher

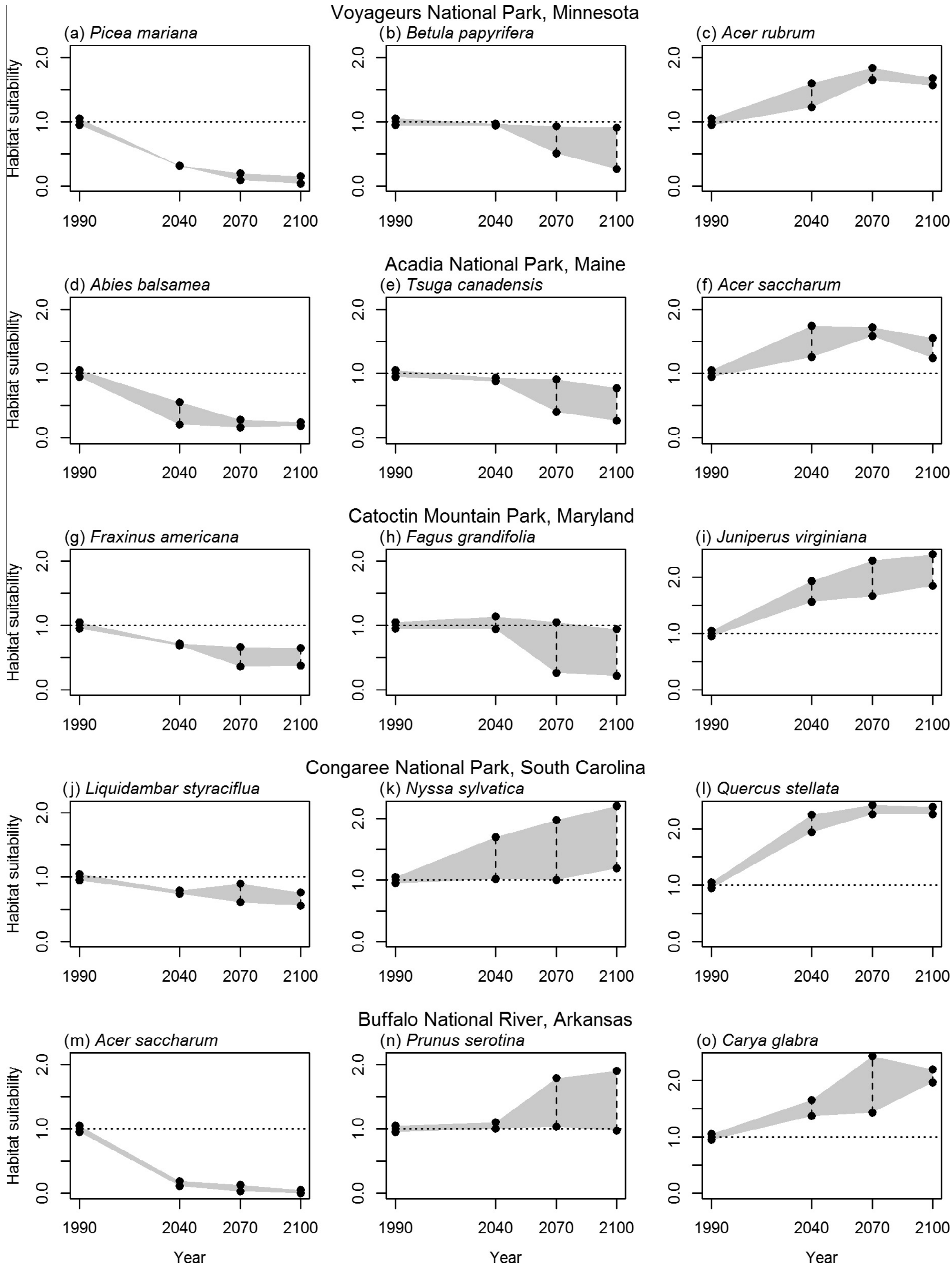

Fig. 2. Projected change and uncertainty in tree habitat suitability for selected common species at five national parks: (a–c) Voyageurs National Park in Minnesota; (d–f) Acadia National Park in Maine; (g–i) Catoctin Mountain Park in Maryland; (j–l) Congaree National Park in South Carolina; and (m–o) Buffalo National River in Arkansas. Ratios of future to late 20th century habitat suitability for three time periods are from the 'least change' and 'major change' climate scenarios (black circles) and gray shading denotes the uncertainty based on differences in projections between climate scenarios. The first column of subplots (a, d, g, j, m) shows species likely to lose substantial habitat, the second column (b, e, h, k, n) shows species with large uncertainty, and the third column (c, f, i, l, o) has species likely to gain habitat in the future.

future:baseline habitat suitability ratios in parks within infested areas. In contrast, *Fraxinus* spp. show the opposite trend, with future habitat greater at parks without emerald ash borer (t-test = 2.88, p-value = 0.004). Some tree species show major decreases in potential suitable habitat regardless of tree pest presence (*A. balsamea* (Fig. 4a)), others exhibit moderate decreases

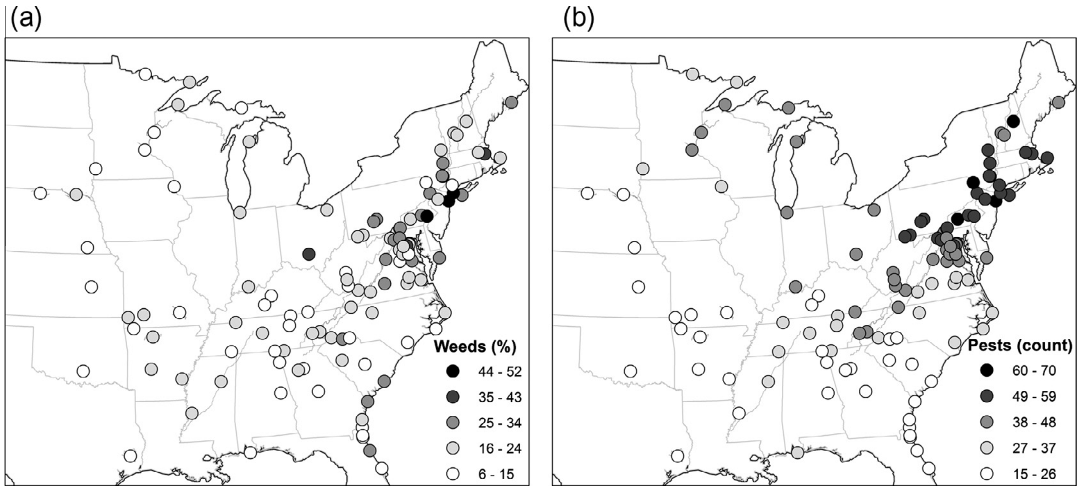

Fig. 3. Nonnative biotic stressors in eastern U.S. national park forests. (a) Nonnative plant species as percentage of the local park flora and (b) Total number of nonnative tree pest (insects and diseases).

Table 2
Presence of 12 nonnative tree pests and major host trees at eastern U.S. national parks (121 potential parks).

Nonnative tree pest	Major host trees	Parks with host trees	Parks in infested zone
Asian longhorned beetle	Numerous species (e.g., *Acer*, *Betula*, *Salix*, *Ulmus*)	121	4 (3%)
Balsam woolly adelgid	*Abies balsamea, A. fraseri*	16	9 (56%)
Beech bark disease	*Fagus grandifolia*	98	35 (36%)
Chestnut blight	*Castanea dentata*	105	105 (100%)
Dogwood anthracnose	*Cornus florida*	101	64 (63%)
Dutch elm disease	*Ulmus* spp.	108	108 (100%)
Emerald ash borer	*Fraxinus* spp.	120	30 (25%)
Gypsy moth	Numerous species (e.g., *Betula*, *Larix*, *Populus*, *Quercus*)	121	60 (50%)
Hemlock woolly adelgid	*Tsuga canadensis*	59	46 (78%)
Sudden oak death	*Quercus* spp.	114	0 (0%)
White pine blister rust	*Pinus strobus*	79	63 (80%)
Winter moth	Numerous species (e.g., *Betula*, *Populus*, *Prunus*, *Quercus*)	121	8 (7%)

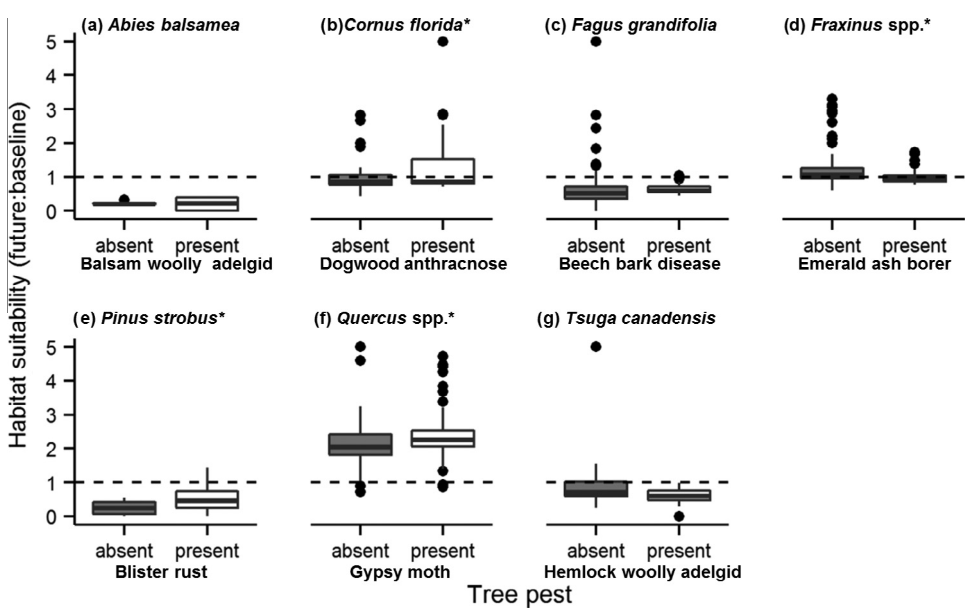

Fig. 4. Future (2071–2100) to baseline (1961–1990) ratio of tree habitat suitability for tree pest hosts at parks within zones where their pest species is present and absent. Boxes denote the interquartile range (IQR), the bold horizontal line within each box is the median value, whiskers extend 1.5 × IQR, and points are extreme values. *Species with significant differences in habitat projections by pest presence/absence (p-value <0.05).

(*Fagus grandifolia* (Fig. 4c)), and some genera have increased future habitat suitability across most parks (*Quercus* spp. (Fig. 4f)). Potential impacts of pests at the local scale are illustrated by *T. canadensis* projections for Acadia National Park and *F. grandifolia* at Catoctin Mountain Park (Fig. 2e and h). Acadia currently does not have hemlock woolly adelgid and Catoctin lacks beech bark disease (*Nectria coccinea*). For the duration that these parks remain pest free, host tree habitat and abundance may be tied to climatic conditions; however, when these pests arrive, they may strongly limit tree survival and expansion, regardless of climate scenario.

4. Discussion

4.1. Change, uncertainty, and biotic stressors

Eastern forests experienced tremendous changes in the past from direct and indirect human influences and coming decades will likely bring continued changes due to multiple global change factors including climate change and nonnative biota. Potential forest change, uncertainty, and nonnative pests and plants are positively correlated in eastern parks, illustrating the broad scope of future changes and potential compounding effects in many forests. Past change on the landscape has been heterogeneous and future change is also likely to be variable though relationships uncovered here may be useful to regional and local forest planning. The combinations of shifting global change factors challenge managers to formulate and maintain desired forest conditions, and crucial steps to facilitate this process include understanding, anticipating, and detecting forest changes.

Areas with greater potential change, uncertainty and stressors may require significant revisions and added flexibility to management goals. For example, northeastern parks generally have high rates of potential change and also already have high numbers of forest pests. Thus, forest decline and change may be rapid in this region, though many tree species may not be available or able to quickly fill potential future habitat due to migration limits and forest pests. At Acadia National Park the balsam woolly adelgid may accelerate decline of a tree species near its southern range limits (*A. balsamea*), while arrival of the Asian long-horned beetle could slow expansion of temperate hardwood species projected to gain suitable habitat. Mature forests require several decades for development, whereas overstory trees can be rapidly killed by pest outbreaks and other disturbances (e.g., more frequent or severe wind and ice storms). Thus, reductions in mature forest canopy may outpace overstory replacement rates at these parks with high stressor levels and potential change, leading to an increase in young stands dominated by a small number of tree species. Parks with greater potential change and uncertainty also suggest that managing towards a specific desired future forest condition will be especially challenging given uncertainties in rates of future climate change.

Parks with less potential change, uncertainty, and current stressors, such as many areas in the south central portion of the East, may be well positioned to achieve specific climate-informed adaptation goals. Relatively low stressor levels and potential rates of forest change indicate that intact, productive, and healthy forests may be achievable through available management actions. The large number of species with minor changes in habitat suitability suggests that species adapted to future conditions may already be present on many landscapes. Furthermore, the low levels of nonnative stressors indicates that limiting or eradicating nonnative species could be achievable in many areas (Abella, 2014b). The baseline period for habitat suitability projections in this study was 1961–1990 and warming since that time has been heterogeneous across parks in the eastern U.S. (Monahan and Fisichelli *In revision*). Continued slow rates of climate change at some parks could help forests adapt to changing conditions and conversely, rapid future changes could strongly alter tree performance and adaptation options.

Land managers make decisions every day in the face of uncertainty, including uncertainties in budgets, staffing, biological and ecological systems, and climate change. Many of these uncertainties are not quantifiable, but availability of a broad range of future climate projections facilitates an understanding of one area of climate change uncertainty. In this research we use 'least change' and 'major change' climate scenarios to bracket a plausible range of futures and facilitate detection of potential changes common to both scenarios as well as changes that depend on the specific climate future. For example, many trees species show large change in habitat suitability (either decreases or increases) under both climate scenarios, particularly tree species near their range limits (e.g., *Picea* spp. at Voyageurs and *A. saccharum* at Buffalo National River). It is important to emphasize that large changes in potential habitat are projected for many species even under the 'least change' scenario, which reflects a greenhouse gas emissions pathway with substantial future decreases in emissions (IPCC, 2007). Due to past greenhouse gas emissions, residence time of these gases in the atmosphere, and slow climate feedbacks, continued and significant climate change is expected in the coming decades, regardless of future emissions (Hansen et al., 2013), and this is likely to manifest in changes to eastern park forests. In addition to comparisons across climate scenarios, interpretations should also weigh model reliability for individual species (see Iverson et al., 2008). These analyses use current biotic stressor levels and potential future tree habitat; many biotic stressors are expanding across the landscape and interactions between a changing climate and biotic stressors may alter their impacts over time. Lastly, additional factors to consider include forest dynamics at fine spatial scales (e.g., individual stands within a park) and future interactions among additional global change stressors such as pollution and habitat fragmentation.

4.2. Forest adaptation strategies

Successful climate change adaptation requires development of locally appropriate goals and strategies, collaboration with managers from neighboring jurisdictions, and includes a spectrum of strategies from resisting to actively directing change (Millar et al., 2007; NFWPCAP, 2012). National park managers can incorporate climate change adaptation strategies in routine management actions such as fire management, nonnative plant eradication, and deer management. For example, prescribed fires can promote specific tree species deemed desirable and adapted to future conditions (e.g., oaks and pines in central and northern parks favored under a warmer and drier climate). Nonnative plant management could foster forest transitions among desired native species by limiting this stressor in areas likely to experience the greatest change (e.g., areas projected to shift from boreal to temperate forest). Conversely, to meet a goal of preserving boreal tree species, a manager could focus nonnative plant eradication in potential climate refugia (e.g., northern and lower slope positions) to reduce competition and facilitate tree species persistence. Similarly, management of specific pest species can focus on parks where host species will retain suitable habitat in the future (e.g., *T. canadensis* at Acadia National Park). Preventing intense browse damage by wildlife species (e.g., deer, elk, and rabbits) could enable palatable trees likely to gain potential habitat in the future, such as *Quercus* and *Acer*, to more rapidly respond to changing climatic conditions (Fisichelli et al., 2012).

4.3. Detecting change

Detecting early indications of forest response to climate change can include measuring shifts in tree phenology, establishment, recruitment, survival, and local distributions. Seedling establishment beyond existing distributions of adult trees can occur relatively rapidly, whereas detectable range shifts of trailing-edge populations likely will be slower to develop, due to the longevity of overstory trees (Jump et al., 2009). This means that leading-edge range expansion of warm-adapted tree species may become detectable before decline of cold-adapted species, such as for temperate and boreal species, respectively, at Voyageurs and Acadia National Parks. Local ecotonal boundaries, where competing tree species occur at/near their ecophysiological tolerances, are likely locations to exhibit such early signs of climate-mediated change (Parmesan et al., 2005; Fisichelli et al., 2014b). Trees growing near their physiological limits are also likely less resistant to the combination of climate and non-climate stressors and thus changes in mortality rates of overstory trees may signal initial stages of forest change (Allen et al., 2010). For example, *A. saccharum* at Buffalo National River may show increased mortality rates in the near-term as temperatures continue to warm and soil moisture becomes more limiting. Also, changes in biotic stressors should be monitored, as virulence over time may increase or decrease and thus require a change in intervention strategies.

Uncertainty in climate patterns, changing distributions of forest pests and hosts, and greater anthropogenic changes to the landscape will likely complicate management of eastern forests. Analyses such as this will help managers understand and anticipate future changes, thereby facilitating development and implementation of adaptive practices that address the challenge to "manage for continuous change", as identified by the NPS Blue Ribbon Advisory Board (NPS AB, 2012). Our results indicate the broad scope of future changes, and correlations among global change factors suggest actions to address these changes and illustrate the need for further research to elucidate potential interactions of climate, trees, pests, and weeds.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2014.04.033.

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