



Forest Health Monitoring in Acadia National Park

Northeast Temperate Network 2006-2013 Summary Report

Natural Resource Report NPS/NETN/NRR—2014/777



ON THE COVER

Camilla Seirup and Mary Short sampling a forest health plot in Acadia National Park
Photograph by: Kate Miller, NPS

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February 2014

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

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Please cite this publication as:

Miller, K. M., B. R. Mitchell, P.J. Curtin, and J. S. Wheeler. 2014. Forest health monitoring in Acadia National Park: 2006-2011 summary report. Natural Resource Report NPS/NETN/NRR—2014/777. National Park Service, Fort Collins, Colorado.

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Executive Summary

The 2013 field season marked the Northeast Temperate Network (NETN) Inventory and Monitoring Program's 8th consecutive year of forest health monitoring in Acadia National Park (ACAD), and nearly all 176 forest plots in ACAD have been sampled twice. The goals of the forest health monitoring program are to monitor status and trend in forest composition, structure and function, and to interpret and report forest condition to park managers. This report summarizes results from the last 8 years of monitoring in ACAD.

Methods

Permanent forest plots are sampled on 4-year intervals in ACAD using a rotating panel design. In ACAD, one of the four panels is sampled each year, and each panel contains 44 plots. Plots were randomly located on Isle au Haut (IAH), Mount Desert Island (MDI) and Schoodic Peninsula (Schoodic) using generalized random tessellation stratified sampling. The permanent plot design was adapted from the U.S. Forest Service's Forest Inventory and Analysis Program (FIA), and consists of a 15 m x 15 m square plot with smaller nested plots located within. Tree and stand measurements are made in the full square plot. Tree regeneration is measured within three 2-m radius circular microplots embedded within each square plot. Coarse woody debris (CWD) is assessed using line intersect sampling along three 15-m transects originating at plot center. Understory diversity is monitored within eight 1-m² quadrats, and soil samples are obtained from a location adjacent to the plot.

To interpret forest condition, we developed an Ecological Integrity Scorecard, which interprets the condition of forested ecosystems by assessing a suite of compositional, structural, and functional metrics in relation to their natural or historical range of variation. For each metric, assessment points are defined that distinguish acceptable or expected conditions from undesired conditions based on current scientific understanding of their natural or historic range of variation. Integrity is interpreted using three categories: Good, Caution, and Significant Concern. "Good" represents acceptable or expected conditions; "Caution" indicates a problem may exist; and "Significant Concern" indicates undesired conditions that may need management action. Metric calculations are based on the most recent 4-year cycle of data collected from all plots.

Results and Discussion

Ecological Integrity

Ecological integrity metrics indicate that forests in ACAD are approaching mature stages of forest succession, and overall forest condition is good. Anthropogenic land use surrounding forest plots is minimal, and the majority of forest area occurs in forest patches > 50 ha. ACAD forests are largely composed of native species, and only three out of 176 plots have detected indicator invasive plant species. Beech bark disease and balsam woolly adelgid (*Adelges piceae*) are the only major tree pests or pathogens that have been observed in plots. ACAD forest soils are robust to N saturation, but vulnerable to acidification. Abundance of tree regeneration is adequate, and patterns of regeneration reflect those of earlier successional and mature forest stages. Impacts from deer browse on tree regeneration and understory herbaceous species are minimal. Late-successional structure, such as coarse woody debris and large snags is below desired levels, particularly in the eastern section of MDI and on Schoodic. As forests are left to succeed, late-successional structure is expected to increase over time.

Tree growth rates are lower than the regional growth rates calculated from the FIA program. Lower growth rates are likely due, in part, to the higher proportion of late-successional and mature plots in ACAD than the surrounding matrix of forestlands. Overall tree mortality was rated Good for all units except Schoodic, which had a relatively high mortality rate for red spruce (*Picea rubens*). Forests on Schoodic are primarily even-aged spruce – fir stands that are beginning to break up and move into later successional forest, and this may partially explain the high mortality rate. If the elevated red spruce mortality rate on Schoodic continues or increases over time, this may indicate a more serious forest health concern, such as soil acidification. Balsam fir (*Abies balsamea*) and big tooth aspen (*Populus grandidentata*) also had elevated mortality rates. Higher mortality rates for balsam fir are likely influenced by the presence of balsam woolly adelgid. Big tooth aspen mortality is likely the result of forest succession.

2010-2013 Ecological Integrity Scorecard

Metric Type	Metric	IAH	MDI-East	MDI-West	Schoodic
Landscape Context ¹	Anthropogenic Land Use	Good	Good	Good	Good
	Forest Patch Size ²				
Structure	Structural Stage Distribution	Caution	Sign. Conc. ⁶	Caution	Caution
	Snag Abundance	Good	Caution	Good	Sign. Conc.
	Coarse Woody Debris	Caution	Caution	Caution	---
Composition	Tree regeneration ³				
	Tree condition & forest pests ⁴				
	Indicator invasive species	Good	Good	Good	Good
Function	Soil Chemistry: acid stress ⁵	Good	Caution	Caution	Good
	Soil Chemistry: N saturation ⁵	Good	Good	Good	Good
	Tree growth and mortality rates	Caution	Good	Good	Caution

¹Based on 2010 analysis

²Metric was not rated for ecological integrity, due to presence of naturally occurring woodland habitats. Pie charts represent the proportion of forest patches in the following patch size categories: >50 ha (dark green), 10-50 ha (green), <10 ha (light green).

³Tree regeneration is rated overall for abundance (Good) and pattern compared to late-successional forests (Caution). "—" indicates too few plots to rate the metric

⁴Pie charts represent the proportion of plots in each ecological integrity rating category.

⁵Based on 2009-2012 sample data

⁶Significant Concern is abbreviated as Sign. Conc.

Forest Composition and Structure

Tree diameter distributions across all park units are typical of second-growth spruce – fir forests, with few trees represented in large diameter size classes (i.e., > 40 cm DBH), and many trees in smaller size classes. Seedling densities and composition are also consistent with patterns expected for spruce – fir forests that are reaching maturity; regeneration is abundant, and is primarily composed of shade-tolerant species that are capable of reaching the canopy in small canopy gaps. Red spruce is the dominant species across all strata, and this is particularly the case on IAH and Schoodic. Jack pine (*Pinus banksiana*) on Schoodic, and pitch pine (*Pinus rigida*) on IAH are also common canopy species in conifer woodlands, but are uncommon in the regeneration layer.

Tree composition is more diverse on MDI than IAH and Schoodic. This is particularly true for eastern MDI, which, due to the 1947 fire, has a sizeable component of early successional forest. The early successional species big tooth aspen and birch species (*Betula papyrifera* and *B. populifolia*) are relatively common canopy species in this section of the park. However, red spruce, balsam fir, and northern hardwood species occur at higher seedling densities than birch and aspen, indicating, as expected, that stands are shifting to later successional forest species.

The understory in ACAD is dominated by woody species. On IAH and Schoodic, shrubs make up the greatest percent cover, and this is primarily from ericaceous species, such as black huckleberry (*Gaylussacia baccata*) and low bush blueberry (*Vaccinium angustifolium*), both of which are often abundant in conifer woodlands. Tree seedlings make up a large percentage of cover in the ground layer across all units. While fern, forb and graminoid species are consistently present in the ground layer, they typically occur as only a few individuals per species. These patterns are expected for spruce – fir forests, which tend to support a seedling bank of advanced tree regeneration in the ground layer and generally lack a well-developed herbaceous layer.

Many of the species that are common to the forest canopy are relatively long-lived and once established in the canopy, may be somewhat resistant to stressors caused by climate change. In contrast, seedlings are relatively short-lived and more sensitive to environmental changes, and the results from this analysis serve as an important baseline to track future changes in forest composition across strata.

Forest Soil Chemistry

Results from the ordination of soil A horizon samples suggest forest soils in ACAD, which have high organic content, low pH, and low base saturation, are the most vulnerable to acidification in NETN, despite receiving the lowest annual acid deposition rates in the network. Low stream acid neutralizing capacity (ANC) also indicates high vulnerability to acidification for terrestrial and aquatic systems in ACAD. Forest soils in ACAD are less vulnerable to N saturation and NO_3^- export due in part to high soil organic content. Low stream NO_3^- concentrations provide further evidence that ACAD is robust to N saturation. It will be important to track soil chemistry through time to see if soils are continuing to respond to, or are recovering from acid deposition, and to relate patterns of soil chemistry with indicators of forest health.

Forest Inventory and Analysis Comparisons

While over 60% of the Ecological Subsection that includes ACAD is forested, forestlands outside of ACAD are largely under private ownership and in timber production, whereas forests

in ACAD are reserved from timber production. The difference in management regimes has resulted in forests in ACAD that are distinct from the surrounding matrix of forestlands. The proportion of plots in late-successional forest and mature forest plots is higher in every park unit than the surrounding matrix. Indicators of older forest structure, such as coarse woody debris, large snags, and large live trees are often more abundant in ACAD park units than the surrounding matrix of more intensively managed forests. These results suggest that forests in ACAD parks may represent regionally significant, older forest habitat, particularly as forests in ACAD are left to succeed.

Tree Ring Analyses

In 2013 NETN began coring trees adjacent to forest plots on western MDI to better understand how stand age relates to ecological integrity, and to determine how tree diameter at breast height (DBH) relates to tree age. Results supported our assumption that larger DBH trees tend to be older than smaller trees, but only for co-dominant trees. However, there was a lot of variability in the data, indicating that DBH is not the best predictor of tree age. In most cases, stand ages derived from tree cores supported the structural stage designated by our structural stage ecological integrity metric, but the metric may be a bit conservative at classifying late-successional plots. In contrast, snag abundance and coarse woody debris (CWD), which are expected to be higher in older forests, varied greatly across the plots, and there were no clear patterns with stand age. However, with only 16 plots of age data, our analysis is currently limited by sample size, especially for CWD and snags, which tend to be rare features on the landscape.

Comparisons between co-dominant and intermediate tree ages within the same plot revealed that plots with co-dominant trees less than 80 years old at DBH also had similar ages for intermediate trees. This follows patterns expected for mature forests, which are generally composed of a single-aged cohort of trees. On plots with co-dominant trees over 80 years old at DBH, intermediate trees tended to be several decades or more younger, which is suggestive of later successional forest.

Overall, results from the first year of coring revealed valuable information about forest stands on western MDI, and we expect to learn more as more plots are cored. As time permits, it will also be important to expand this project to all units of ACAD to increase sample size and examine if patterns hold up across the park.

Conclusions

Forest condition in ACAD is threatened by a range of stressors, including invasive plants, exotic pests and pathogens, acid deposition, and stress induced by climate change. While late-successional structure (e.g., coarse woody debris and snags) is below desired levels, the overall condition of forests in ACAD is good, especially compared to the surrounding matrix of more heavily managed forestlands, and compared to other parks in NETN. Continued invasive species management and early detection efforts are vital to maintain forest condition, as invasive species have the potential to drastically alter forest composition and structure, and may respond more favorably than native species to climate change. As long as forests are left to mature, and management activities do not reduce the potential for development of late-successional structure (e.g., cutting and/or removing coarse woody debris), we expect the ecological integrity of forests in ACAD to continue to improve and be regionally significant in their low level of invasive plants, and extent of older forest structure and habitat.

Acknowledgements

We are grateful for the efforts of numerous individuals who have contributed to the overall success of the forest health monitoring program and have made this report possible. We are indebted to Geri Tierney for developing the forest ecological integrity scorecard and the forest monitoring protocol, and for her continued input. We are grateful to Adam Kozlowski for his hard work developing and maintaining the forest database. We are grateful to Ed Sharron for his help with the format of this report, and for incorporating our results into exceptional outreach materials. Thanks to Fred Dieffenbach for making it possible to compare NETN forest data with US Forest Service Forest Inventory and Analysis data. We would like to thank the 2013 forest crew, including Emilie Schuler, Camilla Seirup, and Mary Short, for their hard work and dedication to the forest monitoring program. We would also like to thank the Resource Management staff in Acadia, particularly David Manski, for their feedback and continued support. Many thanks to Ivan Fernandez and Stephanie Perles for helping us frame and interpret the forest soil analyses. Finally, we thank Jim Comiskey for his helpful suggestions on the report and continued collaboration.

Introduction

The Northeast Temperate Network (NETN) of the National Park Service (NPS) Inventory and Monitoring Program developed a long-term monitoring program for forest resources in response to the identification of Forest Vegetation as a high-priority vital sign for the network (Mitchell et al. 2006). This program also provides data for three additional high-priority vital signs: Forest Soil Condition, White-tailed Deer Herbivory, and Landscape Context. The overall goal of the forest monitoring program is to assess status and trends in the composition, structure, and function of NETN forested ecosystems in order to inform management decisions affecting those systems.

Adapted from the U.S. Forest Service (USFS) Forest Inventory and Analysis Program (FIA), the NETN forest protocol monitors a representative suite of site and vegetation measures in an extensive network of randomly located permanent plots (Tierney et al. 2013). The following parks are monitored in this program: Acadia National Park (ACAD), Marsh-Billings-Rockefeller National Historical Park (MABI), Minute Man National Historical Park (MIMA), Morristown National Historical Park (MORR), Roosevelt-Vanderbilt National Historic Sites (ROVA), Saint-Gaudens National Historic Site (SAGA), Saratoga National Historical Park (SARA), and Weir Farm National Historic Site (WEFA). Roosevelt-Vanderbilt National Historic Sites includes Eleanor Roosevelt National Historic Site (ELRO), Home of Franklin D. Roosevelt National Historic Site (HOFR), and Vanderbilt Mansion National Historic Site (VAMA). Currently NETN has completed two sampling cycles for the plots that have been established throughout the network.

An Ecological Integrity Scorecard is used to facilitate reporting and interpreting forest condition in NETN parks. The Ecological Integrity Scorecard examines a suite of compositional, structural, and functional metrics in relation to their natural or historical range of variation (Tierney et al. 2009). NETN recognizes that “ecological integrity” may not be the primary goal of park resource management, particularly at historical parks and historic sites where cultural resource management may take precedence. Nevertheless, being able to compare the condition of park resources to ecological integrity benchmarks is valuable because it provides a deeper understanding of park condition, as well as a consistent baseline to assess management goals.

The first section of this report is the Ecological Integrity Scorecard calculated from data representing the most recent full cycle of plot sampling (2010 to 2013). The following sections contain important results from previous network-wide annual reports, including a section on forest structure and composition, comparisons between ACAD forest plots and USFS FIA plots, and an analysis of forest soil chemistry. We also include a new section containing preliminary results from a tree coring project on the western section of Mount Desert Island (MDI).

Methods

NETN monitors forest health in ACAD using a network of 176 randomly located permanent plots on Isle au Haut (IAH), Mount Desert Island (MDI), and Schoodic Peninsula (Schoodic; Figure 1). Plots are sampled on a four-panel rotation, with one panel sampled each year and each plot sampled every 4 years. Plots were randomly located using generalized random tessellation stratified sampling (GRTS; Stevens and Olsen 2004). GRTS points that were primarily forested or succeeding to forest were established as forest plots, as long as they were ≥ 15 m from a road (including a carriage road), a perennial stream or water body large enough to create a canopy opening, a mowed area, or the park boundary. Plots in areas receiving management, such as invasive species removal, are included in the target population as long as plots are not being targeted for the management.

The permanent plot design is illustrated in Figure 2. Tree and stand measurements are made within fixed-area, square plots ($15 \times 15 \text{ m}^2$). Tree regeneration is measured within three 2-m

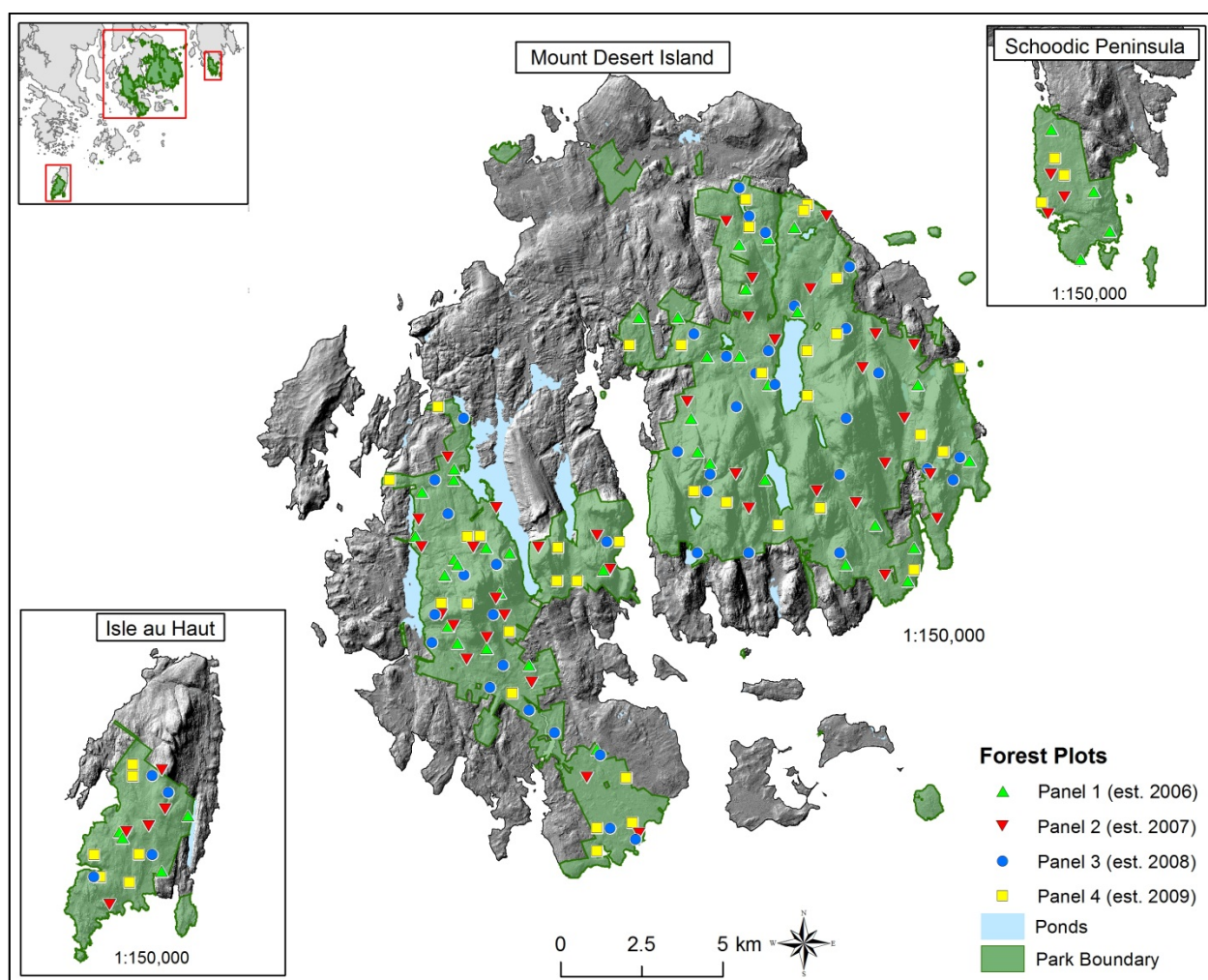


Figure 1. Map of forest plots in Acadia National Park overlaid on 2010 LiDAR bare earth imagery.

radius circular microplots embedded within each square plot. Coarse woody debris (CWD) is assessed using line intersect sampling along three 15-m transects originating at plot center. Understory diversity is monitored within eight 1-m² quadrats, and soil samples are obtained from a location adjacent to the plot. For a more detailed description of the forest protocol, including background information, field methods, and sample design, refer to Tierney et al. (2013).

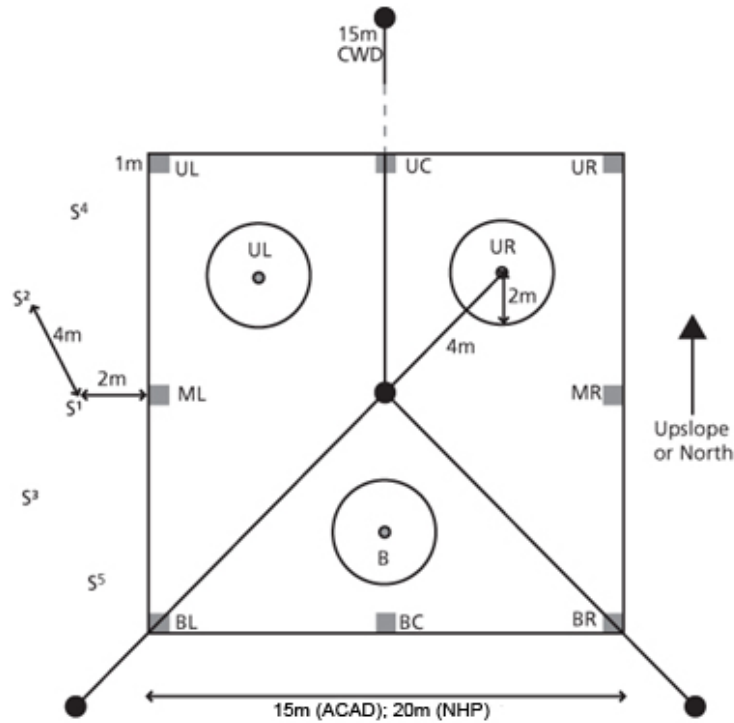


Figure 2. Plot layout showing square tree plot with three nested 2-m radius regeneration microplots, eight 1 m² vegetation quadrats, and three 15-m coarse woody debris (CWD) transects. S^x is location of soil sample.

Ecological Integrity Scorecard

The Ecological Integrity (EI) scorecard uses the concept of “ecological integrity” to interpret the condition of forested ecosystems by assessing a suite of compositional, structural, and functional metrics in relation to their natural or historical range of variation (Tierney et al. 2009). For each metric, assessment points are defined that distinguish acceptable or expected conditions from undesired conditions based on current scientific understanding of their natural or historic range of variation (Bennetts et al. 2007). Integrity is interpreted for a broad audience using three categories: Good, Caution, and Significant Concern. “Good” represents acceptable or expected conditions; “Caution” indicates a problem may exist; and “Significant Concern” indicates undesired conditions that may need management action. Caution is equivalent to the Moderate Concern rating used in State of the Park reports. For some metrics, NETN does not define conditions that merit Significant Concern because current knowledge is insufficient to justify this rating. Ecological Integrity assessment points are summarized in Table 1. For a more detailed discussion of a metric’s justification, calculation and confidence, refer to the Ecological Integrity Reporting SOP in Tierney et al. (2013).

Metric calculations are based on the most recent full cycle (four panels) of data collected (2010-2013). Tree condition is the only exception to this approach, as annual changes in tree condition may reflect a pest or disease outbreak. Therefore, the tree condition metric will consider both the 4-year average and the most recent year of plot data.

The following metrics are reported overall and by park unit for ACAD using 2010-2013 data: structural stage distribution, snag abundance, coarse woody debris, tree regeneration, tree condition and forest pests, indicator invasive species, and tree growth and mortality rates. In addition we include landscape context metrics from Miller et. al. (2011), and soil chemistry metric results using 2009-2012 data.

Landscape Context Background

Prior to European settlement, the northeastern U.S. was primarily forested and dominated in large part by climax forest (Lorimer and White 2003). Over the last 350 or so years, large shifts in land use in the northeastern U.S. have caused radical changes in forest habitat starting with widespread logging and land clearing and later reforestation following agricultural abandonment (Foster et al. 1998). Currently the northeastern U.S. is primarily forested, but the forests typically are highly fragmented and impacted by anthropogenic land use and human disturbance. Negative impacts from fragmentation and human land use on forest condition include reduction in interior forest habitat, increased risk of exotic species invasion, loss of biodiversity, and altered forest structure and composition (Austen et al. 2001, Boulinier et al. 2001, Fahrig 2003, Lundgren et al. 2004, Harper et al. 2005, Honnay et al. 2005). In this section we characterize forest patch size and adjacent anthropogenic land use to examine the extent that surrounding landscape may be influencing forest condition in ACAD.

Landscape context was examined by delineating forest patch size at the park level, and adjacent land use analyses were performed at the level of the forest plot. All spatial analyses were

Table 1. Metrics and ratings for evaluating the ecological integrity of forest ecosystems in ACAD.

Metric type	Metric	Metric Rating		
		Good	Caution	Significant Concern
Landscape context	Forest patch size ¹	> 50 ha	10-50 ha	0.5 to < 10 ha
	Anthropogenic land use	< 10%	10 - 40%	> 40%
Structure	Structural stage	≥ 50% late-successional structure	< 50 late- successional structure	< 50% combined mature <i>and</i> late-successional structure
	Snag abundance	> = 10% standing trees are snags and > = 10% med-lg trees are snags ²	< 10% standing trees are snags or < 10% med-lg trees are snags	< 5 med-lg snags/ha
	Coarse woody debris ratio ³	> 15% live tree volume	5 - 15% live tree volume	< 5% live tree volume
Composition	Tree regeneration	Stocking index ≥ 20 and sapling: small seedling ratio ≤ 0.11	Stocking index ≥ 20 and sapling: small seedling ratio > 0.11	Stocking index < 20
	Tree condition and forest pests	Foliage problem < 10% <i>and</i> no Priority 1 or Priority 2 pests ⁴ <i>and</i> BBD ⁴ ≤ 2	Foliage problem 10 - 50% <i>or</i> Priority 2 pest present <i>or</i> BBD > 2	Foliage problem > 50% <i>or</i> Priority 1 pest present
	Biotic homogenization	No change	Increasing homogenization ⁵	
	Indicator species: invasive exotic plants	< 0.5 indicator species per plot	0.5 to < 3.5 indicator species per plot	3.5 or more indicator species per plot
	Indicator species: deer browse	No decrease in frequency of most browse-sensitive species	Decrease in frequency of most browsed species <i>or</i> increase in frequency of browse-avoided species	Decrease in frequency of most browsed species <i>and</i> increase in frequency of browse-avoided species
Function	Tree growth and mortality rates	Growth ≥ 60% mean <i>and</i> Mort ≤ 1.6%	Growth < 60% mean <i>or</i> Mort > 1.6% ⁶	
	Soil chemistry: acid stress	Soil Ca:Al ratio > 4	Soil Ca:Al ratio 1 - 4	Soil Ca:Al ratio < 1
	Soil chemistry: nitrogen saturation	Soil C:N ratio > 25	Soil C:N ratio 20 - 25	Soil C:N ratio < 20

¹ Forest patch size is not rated for ACAD, due to naturally occurring woodland habitats.² Med-lg trees are ≥ 30 cm diameter-at-breast-height.³ CWD metric ratings were developed for forested habitats, and do not include plots in woodland habitats in ACAD.⁴ Priority 1 pests are Asian longhorned beetle, emerald ash borer, hemlock woolly adelgid, and sudden oak death. Priority 2 pests are balsam woolly adelgid, butternut canker, elongate hemlock scale, and beech bark disease (BBD).⁵ Significant Concern rating has not been defined for these metrics.

performed using recent leaf-on orthophotography, were delineated at the 1:6,000 scale or finer, and incorporated vegetation map delineations where appropriate. For more detail on the justification and methods for generating forest patch size and anthropogenic land use data, refer to the Landscape Context SOP in Tierney et al. (2013).

Forest patch size was defined as an area of continuous medium to high-canopy (≥ 8 m height) forest vegetation with at least 60% overall canopy closure and at least 0.5 ha in areal extent. Roads wide enough to register as a break in the canopy on an orthophoto were not mapped as part of a forest patch, and often fragmented an otherwise large forest patch into several smaller patches. Woodlands, though naturally occurring in ACAD, were not included in forest patch delineations. This definition of a forest patch is very similar to that used by the U.S. Forest Service Forest Inventory and Analysis (FIA) program, with the exception that FIA's minimum patch size is 0.4 ha and uses a minimum stocking percent instead of a minimum stand height and density (Riemann and Tillman 1999). To be included in the analysis, forest patches had to contain at least 0.5 ha of area within one of the major park units currently being monitored with forest plots (e.g., small islands in ACAD were not included in the analysis). However, where there was no visible break in the canopy, forest patches extended beyond the park boundary.

Forest patch size ratings were derived from Kennedy et al. (2003), which reviewed the literature to determine the minimum patch area required by a variety of taxa. Minimum patch sizes ranged from 1 ha for invertebrates, 10 ha for small mammals, up to 50 ha for most bird species, and over 200,000 ha for large mammals. Considering their relatively small size, NETN parks are not capable of independently supporting large mammal populations, and thus our ratings are based on the needs of birds, which therefore also meet the needs of small mammals and invertebrates. Currently, forest patch size at ACAD is calculated but not rated because naturally-occurring woodlands and forests historically co-exist at this park as a patchwork and naturally limit forest patch size.

Adjacent anthropogenic land use (ALU) was delineated at the plot level, and was calculated as the percentage of anthropogenic land use within the local or immediate neighborhood surrounding each plot. Plantations of exotic tree species, maintained open fields, active farmlands, roads, and residential sites were all considered anthropogenic land use. Native tree plantations, abandoned fields, woodlands, and early successional forests were considered natural land use. Neighborhoods (i.e., plot buffers) of different plots commonly overlapped in parks. While the mean % ALU was unbiased, the lack of independence between plots did not allow for straight-forward standard error estimates.

Adjacent anthropogenic land use ratings were derived from theoretical models that examined the combined impacts of habitat loss and fragmentation. Simulations examining connectivity and proportion of natural habitat found 60% natural habitat to be an important threshold; likelihood of a continuous landscape sharply declined below 60% natural habitat (McIntyre and Hobbs 1999, O'Neill et al. 1997). Therefore, we use $> 40\%$ anthropogenic land use to define a "Significant Concern" rating for this metric. Ratings for this metric are defined in Table 1.

Results and Discussion

Forest patch size varied across ACAD, but over 80% of the forest area across all units occurred in patches that were more than 50 ha in size (Figure 3). The largest patch was on western MDI

and was 940 ha (Table 2). Woodlands were most common on eastern MDI and Schoodic, resulting in smaller forest patch sizes in these units. Forest patches were most fragmented by roads on the eastern side of MDI, where road density in and adjacent to the park was highest and clearly visible on orthophotos (Appendix A: Figures 1-4). Over 20% of the area delineated as forest patch extended outside the boundary of the park, and several of the largest patches on eastern MDI and IAH that extended well beyond the park boundary are not protected by easements.

Plots in the eastern section of MDI and on Schoodic had the highest proportion of neighborhood ALU, but plot buffers still averaged over 95% natural habitat and all units were rated Good for this metric (Table 3). Roads and parking areas were the most common features delineated as anthropogenic land use.

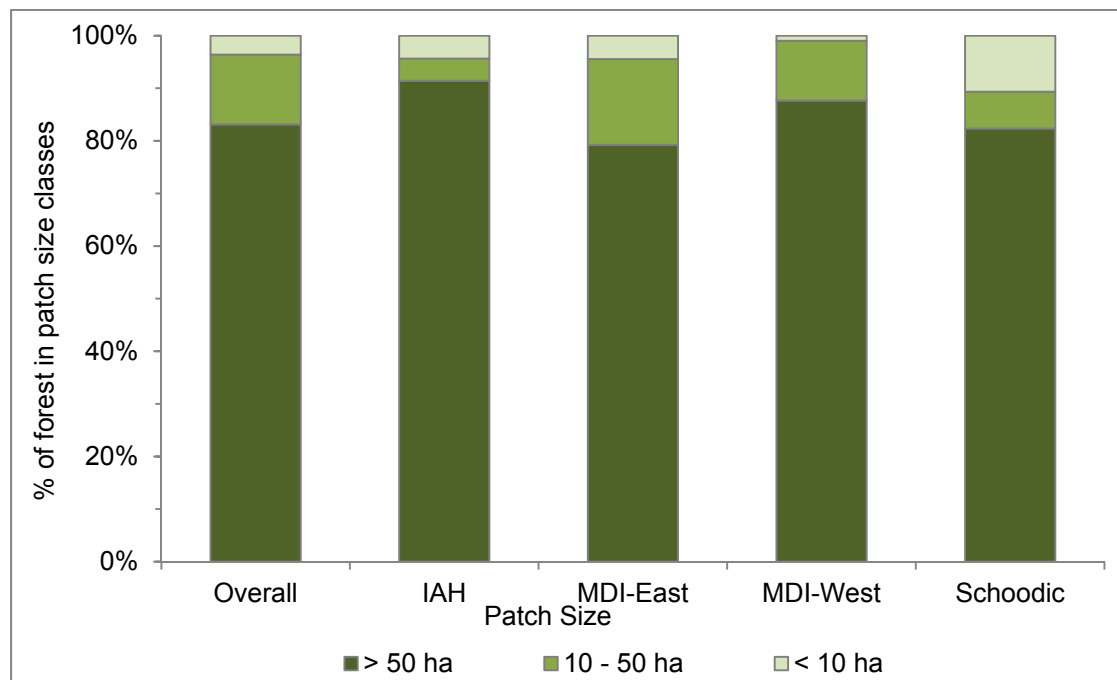


Figure 3. Proportion of forest in each patch size category by park unit.

Table 2. Forest patch statistics for each park unit. % of Patches Outside of Park is the average percentage of forest patch area that is outside the park boundary. Patch sizes were delineated using USDA National Agriculture Imagery Program orthophotos taken in 2009.

Unit	Patch Size (ha)					# Patches	% of Patches Outside Park
	Minimum	25% Quartile	Median	75% Quartile	Maximum		
Overall	0.54	2.28	7.73	32.46	939.88	247	23.69
IAH	0.55	1.42	2.55	13.30	850.30	19	30.09
MDI-E	0.60	2.27	7.49	29.10	734.95	151	25.23
MDI-W	0.54	4.87	13.76	60.17	939.88	55	22.84
Schoodic	0.55	0.88	3.75	8.50	197.70	22	6.95

Table 3. Anthropogenic land use statistics and ratings for each park unit. Anthropogenic Land Use Percent is the percent of anthropogenic land use within the specified buffer surrounding each forest plot. Plot Maximum and Minimum report the maximum and minimum individual plot values.

Unit	Buffer radius (m)	# Plots	Anthropogenic Land Use Percent			Rating
			Plot Average	Plot Maximum	Plot Minimum	
Overall	400	175	2.9%	31.7%	0%	Good
IAH	400	19	0.9%	2.6%	0%	Good
MDI-E	400	82	4.1%	31.7%	0%	Good
MDI-W	400	65	1.8%	13.6%	0%	Good
Schoodic	400	9	3.8%	17.4%	1.1%	Good

Structural Stage Distribution Background

Prior to European settlement, the northern hardwood and spruce – fir forests of the northeastern US were predominantly in late-successional and old growth stages of forest succession (Frelich and Lorimer 1991, Lorimer and White 2003). Natural disturbance regimes consisted primarily of frequent, small-scale disturbances (< 20% canopy removal), and stand replacing disturbances were rare (as high as 1,000-7,500 years for inland northern hardwood forests). Presettlement forests were primarily composed of shade-tolerant tree species, and characterized by a multi-aged or multi-cohort age structure, abundant standing dead and downed trees at varying stages of decay, and an understory composed of advanced regeneration of shade-tolerant tree species (Mosseler et al. 2003). The onslaught of logging and land-clearing for agriculture post-European settlement constituted a major disruption to the natural processes and disturbance regimes of these forests (Foster et al. 1998), and the second-growth forests of today are structurally and compositionally different from late-successional and old growth forests (Scheller and Mladenoff 2002, Mosseler et al. 2003, Burton et al. 2009).

The distribution of structural stages is important for maintaining a full complement of native species, which vary in their dependence upon different successional stages. Human alteration and management have greatly changed the structural stage distributions of eastern forests such that younger forests are far more abundant, and amount of late-successional and old-growth forest has been greatly reduced compared to presettlement times (Lorimer and White 2003, Mosseler et al. 2003, Whitman and Hagan 2007). These distributions will likely be further affected by altered disturbance regimes coincident with global climate change and outbreaks of exotic pests and pathogens (Dale et al. 2001). Comparison of existing distributions with those expected under natural disturbance regimes provides an indicator of altered disturbance regimes as well as habitat availability.

Structural stage distribution is calculated from tree size and canopy position measurements, using a method similar to that of Frelich and Lorimer (1991), but substituting basal area for exposed crown area (Goodell and Faber-Langendoen 2007). Plots are classified as pole, mature, or late-successional based on relative basal area of live canopy trees within pole, mature and large size classes. In ACAD, size classes are defined as 10-19.9 cm DBH for pole, 20-34.9 cm DBH for mature, and ≥ 35 cm DBH for large.

We assign assessment points based on the expected percentage of late-successional forest stages across the landscape as compiled by Frelich and Lorimer (1991) and Lorimer and White (2003), and ratings are based on proportion of plots that are late-successional and mature forest. Plots classified in the field as woodland, which have $\leq 60\%$ canopy cover due to harsh growing conditions, were not included in this metric. Expected structural stage distributions are based on the Acadian Low-Elevation Spruce-Fir-Hardwood Forest at ACAD, and are defined in Table 1.

Results and Discussion

Levels of late-successional forest were below desired levels across all units of ACAD (Table 4). Only a small percent of plots were classified as late-successional forest in ACAD, and all park subunits but eastern Mount Desert Island (MDI) were rated Caution. Eastern MDI had a much higher proportion of plots in the pole stage due to the 1947 fire, and due to a lack of late-successional forest, was rated Significant Concern. As long as forests are left to succeed under natural disturbance regimes, this metric is expected to improve across all park units over time.

While not part of the metric, it is notable that roughly 20% of the plots on IAH and Schoodic are in woodlands, which tend to occur in areas with exposed bedrock, and have $< 60\%$ canopy cover due to the harsh growing conditions. Plots in woodlands are not expected to succeed into later successional forest, and may be more sensitive to anthropogenic stressors due to already stressed growing conditions.

Coarse Woody Debris and Snag Abundance

Background

Dead wood, in the form of fallen coarse woody debris (CWD) and standing dead trees (snags), is an important structural feature of forests that provides habitat for many taxonomic groups, including mammals, birds, herpetofauna, lichens, fungi, and insects. In addition, tip-up mounds caused by wind-throw contribute to forest soil turnover and create important microhabitats for vegetation such as tree seedlings and bryophytes (Jonsson and Esseen 1990, Ulanova 2000, Samonil et al. 2010). Silviculture, land management, and hazard tree removal can reduce the quantity or quality of these features. However, thoughtful land management can maintain or enhance snags and CWD (Keeton 2006).

Density of snags and volume of CWD in mature and late-successional stands vary substantially across ecosystems and with site conditions (Tyrrell et al. 1998). However, positive relationships between live and dead tree density can be used to indicate expected snag levels, and positive relationships between live tree volume and volume of CWD can be used to indicate expected CWD levels. In addition, large-diameter snags provide habitat for a greater number of vertebrate

Table 4. Structural stage distribution of plots in ACAD park units. Woodland and mosaic plots are excluded from this metric. Schoodic was not rated for this metric due to insufficient sample size.

Unit	# Plots	% Late-successional	% Mature and Late-successional	Rating
Overall	138	11.6	60.1	Caution
IAH	12	0.0	66.7	Caution
MDI-East	68	8.8	42.6	Significant Concern
MDI-West	54	18.5	77.8	Caution
Schoodic	4	0.0	100.0	---

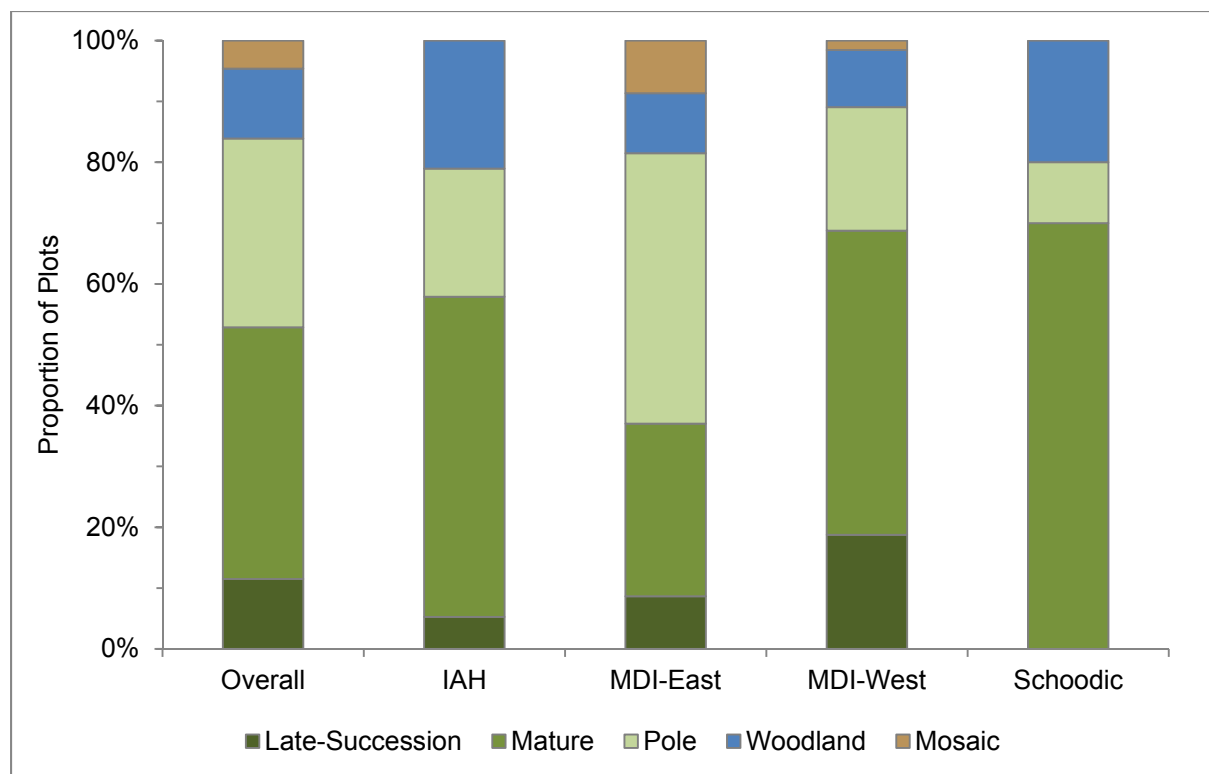


Figure 4. Proportion of plots in each structural stage (Late-Succession, Mature, and Pole) and field stand structure class (Mosaic and Woodland), based on data collected between 2010 and 2013. Structural stages are based on proportion of basal area by size class. Field stand structure classes are determined in the field and not classified into structural stages. Mosaic plots are composed of two or more distinct structural classes, each covering at least 25% of the plot.

species (Cline et al. 1980, DeGraaf and Shigo 1985) and persist longer than smaller-diameter snags (Morrison and Raphael 1993, Garber 2005). Therefore density of large-diameter snags is also an important indicator of habitat quality and availability. These metrics assess the density of snags and volume of CWD in relation to live tree density and volume. CWD metric ratings are based on CWD ratio, which is the ratio of CWD volume to live tree volume and is expressed as a percent. The CWD metric calculation requires a stand height measurement for each plot to calculate live tree volume. Therefore, plots missing stand height measurements are excluded from this metric calculation. Plots in woodland habitats are also excluded from the CWD metric, as expected levels of CWD volume to live tree volume are unknown. Snag abundance metric ratings are based on the percent of standing trees that are snags and are calculated on all plots. Since NETN plots are not large enough to accurately estimate rare features such as snags, snag abundance is evaluated at the scale of the park or park subunit, and requires at least 10 plots to be adequately estimated. To facilitate comparisons of our CWD metric calculations with other studies, we also report volume of CWD using both metric and English units. Assessment points for CWD and Snag Abundance metrics are defined in Table 1.

Results and Discussion

The CWD ratio in the western section of Mount Desert Island (MDI) was very close to the 15% assessment point that distinguishes between Good and Caution, and has fluctuated slightly over and under 15% during the last 8 years of reporting (Table 5). The switch between Good and

Caution does not likely represent an actual trend in CWD, but rather reflects variability in the data. The western section of MDI and all other park units were rated Caution for CWD.

Ratings for Snag Abundance were more variable across park units (Table 6). All units of ACAD had a relatively high snag density. However, due to a lack of medium to large snags (diameter at breast height [DBH] ≥ 30 cm), the eastern section of MDI was rated Caution and Schoodic was rated Significant Concern. IAH and western MDI were the only units to receive Good ratings.

Forested systems in ACAD are relatively young, and are likely to fall short of desired late-successional levels of CWD and snags for some time (decades to centuries). As long as forests are left to mature and management activities (e.g., cutting and/or removing coarse woody debris) do not reduce the potential for development of CWD and large diameter snags, we expect these metrics to improve with time.

Tree Regeneration

Background

This metric assesses the quantity and structure of tree regeneration in the forest understory, which will impact future canopy structure. Regeneration can be affected by a variety of stressors, including invasive species, acid deposition, and climate change. Most notably, sustained, selective browsing by a historically high population of white-tailed deer is currently impacting seedling establishment, growth, and composition in parts of the Midwest and Northeast U.S. (Cote et al. 2004). In the coastal spruce – fir forests that dominate ACAD, deer browse is not the primary concern for regeneration, as red spruce (*Picea rubens*) and balsam fir (*Abies balsamea*) are not preferred browse species (Uchytel 1991, Sullivan 1993). Our main concerns about regeneration in spruce – fir forests are 1) whether regeneration abundance is adequate to replace the canopy, and 2) whether regeneration patterns in spruce – fir forests are consistent with those expected in late-successional forests.

To assess regeneration, NETN developed a tree regeneration metric specifically for ACAD using data collected in late-successional spruce – fir stands in coastal Maine (Miller et al. 2013a). To assess regeneration abundance, the metric uses a stocking index that was developed by McWilliams et al. (2005), and can be applied to all stand types in ACAD. The stocking index awards and sums points for native tree seedlings by size class within 2-m radius circular microplots as follows: one point for each seedling 15-30 cm, two points for each seedling 30-100 cm, 20 points for each seedling 100-150 cm, and 50 points for each seedling or sapling > 150 cm tall but less than 10 cm diameter at breast height (DBH). To assess patterns of regeneration, we use a ratio of sapling to small (15-100 cm) seedling density, and only include plots in upland spruce – fir forest. The expectation is that regeneration in late-successional stands is abundant and, while both are present, small seedlings are more abundant than saplings. To account for the high spatial variability in regeneration, this metric requires a minimum of ten plots to be rated, and is summarized at the group level. The assessment points for tree regeneration in ACAD are defined in Table 1.

In addition to the regeneration metric in ACAD, we calculate stem density by size class to examine regeneration structure, and use a deer browse index to track impacts of deer browse in each plot. The deer browse index was adapted from Brose et al. (2008) and Perles et al. (2010),

Table 5. Coarse woody debris (CWD) metric ratings, ratios, and volume calculations for park units based on data collected from 2010 to 2013. Metric ratings are based on CWD ratio, which is the ratio of coarse woody debris volume to live tree volume and is expressed as a percent. The CWD ratio is calculated for each plot in a group and then averaged, and will differ from the mean CWD divided by the mean live tree volume. Only plots with complete stand height data are included in the analysis. Groups with fewer than 10 plots were not rated for CWD, and woodland plots are not included in the ACAD ratings. SE refers to 1 standard error around the mean.

Unit	# Plots	Rating	CWD Ratio (%)		Metric CWD Volume (m ³ /ha)		English CWD Volume (ft ³ /acre)		Live Tree Volume (m ³ /ha)	
			Mean	SE	Mean	SE	Mean	SE	Mean	SE
Overall	152	Caution	11.96%	1.23%	23.49	2.26	335.64	32.36	210.79	9.84
IAH	15	Caution	11.31%	2.15%	21.85	5.02	312.30	71.79	178.40	28.40
MDI-East	73	Caution	10.02%	1.88%	15.90	2.80	227.20	39.95	187.91	16.21
MDI-West	57	Caution	14.79%	2.09%	34.13	4.29	487.80	61.37	250.66	11.56
Schoodic	7	---	10.68%	2.69%	19.43	6.27	277.64	89.57	194.28	48.46

Table 6. Snag abundance ratings and calculations for park units based on data collected from 2010 to 2013. Metric ratings are based on the percent of standing trees (i.e., ≥ 10 cm diameter at breast height) that are snags. M-L Snags refers to medium to large (≥ 30 cm diameter at breast height) diameter snags. SE refers to 1 standard error around the mean.

Unit	# Plots	Rating	# of Snags/ha		# of M-L Snags/ha		% of Trees as Snags	
			Mean	SE	Mean	SE	All Sizes	M-L Snags
Overall	175	Caution	170.92	10.71	7.87	1.56	15.90%	9.09%
IAH	19	Good	119.30	16.67	11.70	5.73	12.94%	14.29%
MDI-East	82	Caution	147.42	14.57	5.42	1.79	14.60%	7.46%
MDI-West	64	Good	199.30	19.40	11.11	3.13	16.58%	10.32%
Schoodic	10	Significant Concern	280.00	47.80	0.00	0.00	25.10%	0.00%

and is a qualitative assessment of deer impact based on the presence/absence of deer signs (e.g., scat, deer trails, browse, etc.). The deer browse index is a five-point index ranging from unnaturally low impacts (level 1) due to location in a deer enclosure, to exceptionally high deer browse impacts (level 5; Table 7).

Results and Discussion

Based on the regeneration metric for ACAD, forests across all subunits and successional stages in ACAD have sufficient levels of regeneration and all units of ACAD were rated Good for the stocking index (Table 8). In contrast, patterns of regeneration reflect those of earlier stages of forest succession, with a higher abundance of saplings than is expected for late-successional forest (Figure 5). Therefore, all units of ACAD were rated Caution for the sapling:small seedling ratio. The exception is Schoodic, which was not rated for the sapling:small seedling ratio due to insufficient sample size for upland spruce – fir forests. As forests mature and move towards more uneven-aged stands, we expect the sapling:seedling density ratios across ACAD subunits to approach the ratio observed in reference stands.

Deer browse impacts were relatively low across all park units (Table 9). The seven plots that were rated High for the deer browse index were either close to a road or hiking trail (n=5), or near an open fen complex (n=2), where edge habitat was present. Deer browse impacts on plots in interior forest were low to moderate.

Table 7. Index of deer browse impacts assessed for each plot in ACAD (adapted from Brose et al. 2008 and Perles et al. 2010).

Level	Browse Impact	Description
1	None	Plot located inside deer enclosure, and no browse.
2	Low	No observed browse or sign of deer activity on plot
3	Moderate	Evidence of browse and signs of deer activity on the plot
4	High	Browse evidence common; and non-preferred species show signs of browse
5	Very High	Browse evidence omnipresent; browse-resistant and non-preferred species show signs of heavy browsing and browse line evident.

Table 8. Regeneration metric summaries for ACAD based on data collected from 2010-2013. Standard error (SE) for the ratio accounts for the uncertainty in the estimates of sapling and seedling abundance. Mosaic and forested swamp plots were excluded from the stocking index, and only upland forests were included in the sapling:small seedling ratio.

Unit	# Plots	Stocking Index			Sapling: Small Seedling Ratio			
		Mean	SE	Rating	# Plots	Mean	SE	Rating
Overall	172	142.06	14.04	Good	133	0.46	0.09	Caution
IAH	19	143.47	29.76	Good	14	0.91	0.41	Caution
MDI-East	80	114.42	14.40	Good	62	0.54	0.17	Caution
MDI-West	63	179.64	31.85	Good	52	0.41	0.11	Caution
Schoodic	10	123.73	27.91	Good	5	0.16	0.08	---

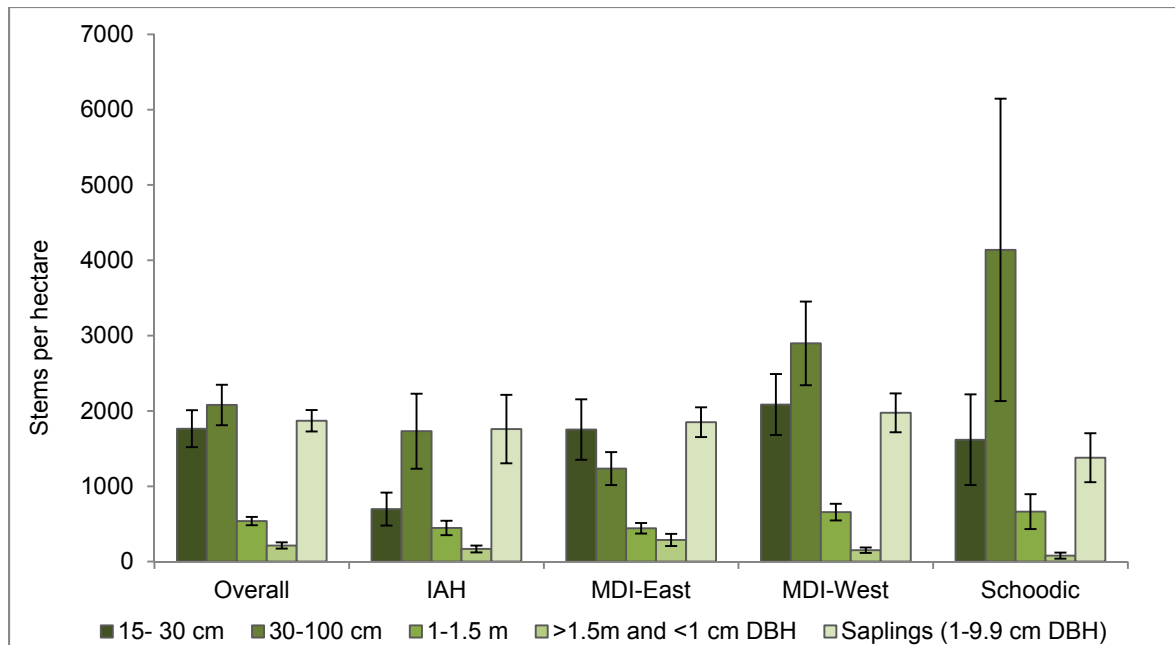


Figure 5. Mean seedling and sapling density (stems/ha) based on data collected from 2010-2013. Error bars denote ± 1 SE around the mean. Seedlings in the smallest two size classes make up the small seedling class for the sapling:small seedling ratio.

Table 9. Deer browse index results from data collected from 2010-2013. Levels are defined in Table 7.

Unit	Deer Browse Index			Number of Plots Per Level				
	# Plots	Mean	SE	None	Low	Moderate	High	Very High
Overall	175	2.50	0.05	0	93	75	7	0
IAH	19	2.42	0.16	0	11	7	1	0
MDI-East	82	2.52	0.06	0	40	40	2	0
MDI-West	64	2.47	0.07	0	37	24	3	0
Schoodic	10	2.60	0.22	0	5	4	1	0

Tree Condition and Forest Pests

Background

This metric assesses tree condition based on qualitative observations of specific tree health problems and canopy foliage condition. As the season progresses, most trees develop minor foliage problems. However, more extensive damage to canopy foliage may be indicative of tree health problems within a species or across a region, and could be related to any combination of soil chemistry, climate, pathogens, or other drivers and stressors. In particular, exotic pests and pathogens have the potential to severely impact forest tree composition as well as structure and function. Several species of exotic pests pose serious threats to northeastern forests if they advance into the region; these NETN “Priority 1” pests are Asian longhorned beetle (*Anoplophora glabripennis*; ALB), emerald ash borer (*Agrilus planipennis*; EAB), hemlock woolly adelgid (*Adelges tsugae*; HWA), and sudden oak death (*Phytophthora ramorum*; SOD). Other forest pests can cause problems that are not quite as severe, and/or may already be widespread in the NETN region. These NETN “Priority 2” pests are balsam woolly adelgid (*Adelges piceae*; BWA), beech bark disease (BBD), butternut canker (*Sirococcus clavignenti-*

juglandacearum; BC), and elongate hemlock scale (*Fiorinia externa*; EHS). As we learn more about the distribution and impacts of winter moth (*Operophtera brumata*), a recently introduced pest in the Northeast, we may add this species to one of the priority lists.

To calculate this metric, specific tree health problems are tallied per plot, and average percent foliage condition is calculated for the plot and for each tree species represented by two or more individuals. Good condition is defined as minor foliage problems (< 10%) averaged across a plot and within individual species, minor damage from BBD (BBD index ≤ 2), and no evidence of damage by Priority 1 or other Priority 2 pests and pathogens. More extensive foliage problems (10-50%), more extensive BBD damage (BBD index > 2), or evidence of damage by one of the other Priority 2 pests and pathogens indicate cause for concern, and are rated Caution. Foliage damage above 50% averaged across a stand or within an individual species (represented by at least two individuals) indicates a more serious problem. Likewise, evidence that Priority 1 pests have reached NETN forests indicates a significant concern which might still be controlled by management action.

Results and Discussion

Between 2010 and 2013, only four plots were rated Significant Concern in ACAD for tree condition and forest pests, and these were all the result of elevated insect herbivory on hardwoods (Figure 6). Balsam woolly adelgid (BWA) and beech bark disease (BBD) are the only pests/pathogens on the priority list that have been detected in ACAD forest plots since 2006. BWA has been detected in five plots on eastern Mount Desert Island (MDI), 13 plots in the western section of MDI, and one plot on Schoodic. BBD has been detected in six plots on eastern MDI, and one plot on western MDI. Tree condition ratings for 2013 tended to have slightly more plots in the Caution category than the 4-year average. If this pattern continues, this would indicate a forest health concern.

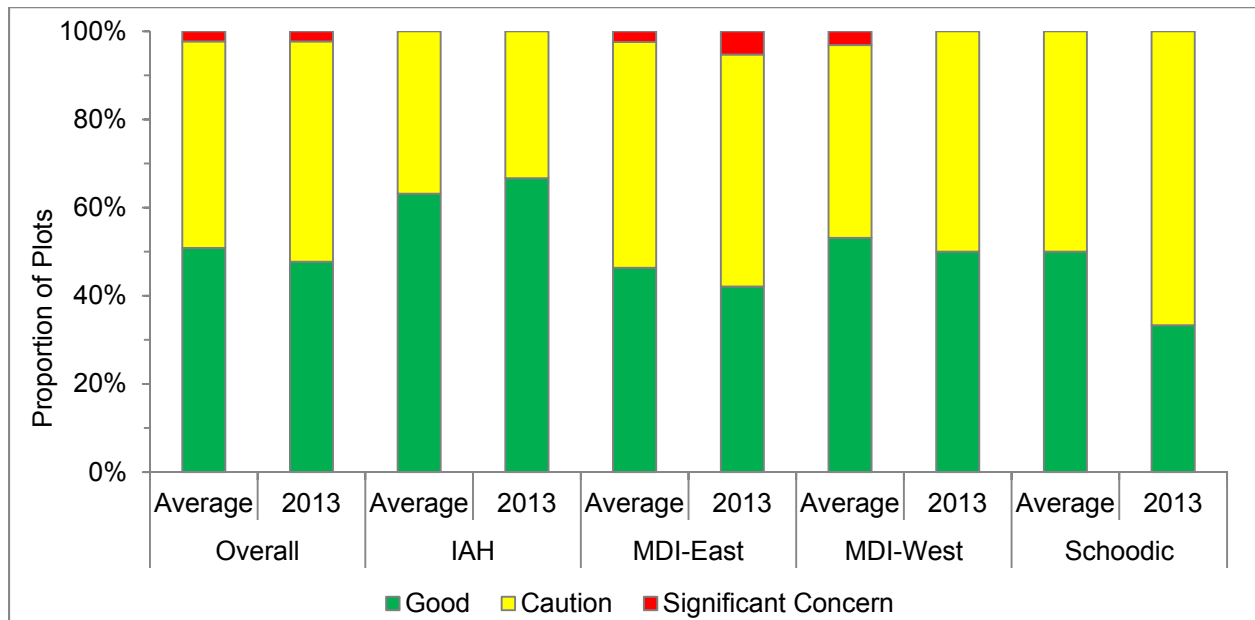


Figure 6. Proportion of plots receiving Good (green), Caution (yellow), and Significant Concern (red) ratings for tree condition for each park unit. Average bars include 4 years (2010 – 2013) of data, and 2013 bars only include data collected in 2013.

While no Priority 1 species have been detected in ACAD forest plots, the range of several pest species continues to expand and threaten forests in ACAD. As of December 2013, the two closest known populations of Emerald Ash Borer (EAB) are southern New Hampshire and western Massachusetts (Figure 7). EAB is also present in southern Ontario and southern Quebec, but spatial data were not available for Figure 7. After the trap trees and prism traps from 2013 are processed by the multiple state and federal agencies monitoring for EAB, the known range may expand further. The range of hemlock woolly adelgid (HWA) also continues to expand into northern New England, including several locations on planted eastern hemlock (*Tsuga canadensis*) trees on MDI. While HWA has not been detected on naturally occurring eastern hemlock in ACAD, the proximity of HWA to the park is of great concern.

The range of winter moth (*Operophtera brumata*) now includes MDI. Winter moth is a non-native insect that feeds on the leaves of a range of hardwood and deciduous shrub species, including maples, oaks and blueberries (U.S. Forest Service 2012). Repeated defoliation by winter moth can increase tree mortality either directly, or through increased susceptibility to

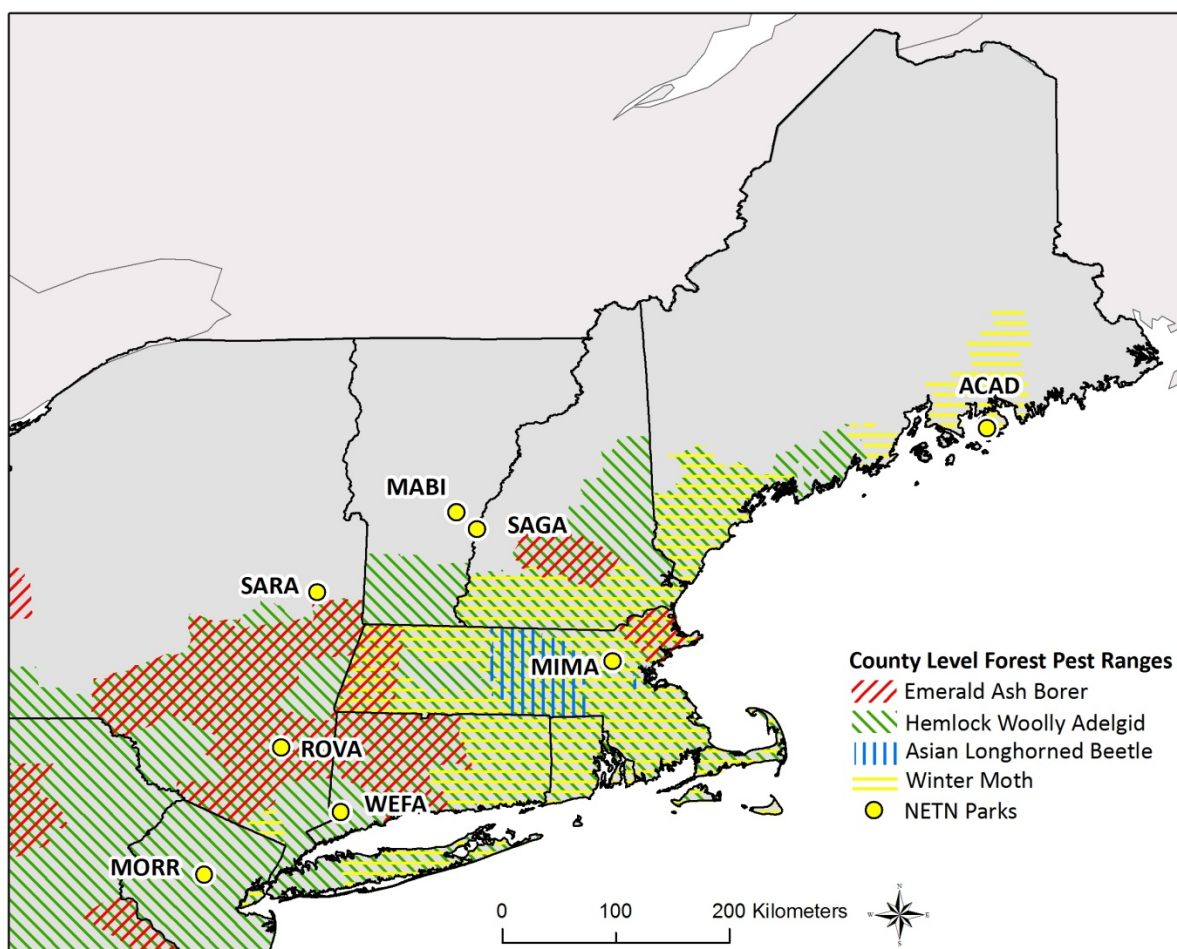


Figure 7. Exotic forest pest detections near NETN parks. Pest ranges were mapped at the county level and were downloaded from the Alien Forest Pest Explorer (<http://www.nrs.fs.fed.us/tools/afpe/maps/>), and the Cooperate EAB Project (<http://www.emeraldashborer.info>). Asian Longhorned Beetle range only reflects the most recent detections that have not yet been eradicated.

other stressors (e.g., drought, secondary defoliators, and fungal pathogens). Winter moths are best identified when the pale brown male moths emerge and swarm in November, a time of year when few other insects are out. Identifying winter moth during the growing season is much more difficult, and the pest may not be detected in ACAD forest plots in the early stages of infestation.

While the range of Asian longhorned beetle (ALB) has not expanded at the rate of EAB and HWA in New England, ALB remains a serious threat to ACAD forests. The ALB can infest a range of host species, including red maple (*Acer rubrum*) and paper birch (*Betula papyrifera*), which are the two most common hardwood species in ACAD. An ALB infestation in ACAD has the potential to cause cascading effects throughout the ecosystem. Continued vigilance and outreach are vital to prevent the spread of ALB and all pests that threaten forests in ACAD.

Invasive Exotic Plant Indicator Species

Background

Invasive exotic species have the potential to impact structure, composition, and function of forested ecosystems, and are one of the leading threats to biodiversity and ecological integrity of ecosystems worldwide (Mooney et al. 2005). Early detection of invasive exotic plants is an important NETN vital sign that has been incorporated into several long-term monitoring protocols. Status and trends of invasive exotic species are monitored on NETN forest plots, and the Eastern Rivers and Mountains Network (ERMN) and NETN are implementing a separate protocol for early detection of invasive exotic species and forest pests.

To better understand the threat posed by invasive exotic plants, we developed a list of 22 indicator plant species that are highly invasive in northeastern forest, woodland, and successional habitats (Appendix B). Included species are those that may dominate and/or persist under shaded conditions (i.e., in forest or woodland), or that may disrupt forest succession. Several exotic species such as knapweeds (*Centaurea* spp.) and celandine (*Chelidonium majus*) were not used as indicator species, despite being present in several parks, because they are not known to suppress forest succession and/or dominate under a shaded canopy. Therefore, total abundance of exotic species may be higher than reported here for indicator species.

Results and Discussion

With indicator species found in only three out of 175 plots, all units of ACAD were rated Good for indicator invasive exotic species (Table 10). All three plots with detections occurred on MDI East, with glossy buckthorn (*Rhamnus frangula*) occurring in two plots, and Japanese barberry (*Berberis thunbergii*) in one. While currently at low levels in ACAD forests, both of these species have the potential to form dense thickets that can reduce understory diversity, impact forest regeneration, and alter nutrient cycling (Silander and Klepeis 1999, Fagan and Peart 2002, Poulette and Arther 2012). In addition to ecosystem impacts, recent studies suggest that exotic shrub thickets can have an indirect effect on human health. By creating a more favorable humid microclimate, exotic shrub thickets can harbor higher densities of blacklegged ticks (*Ixodes scapularis*) than uninvaded forests (Elias et al. 2006, Williams and Ward 2010). Research has also found higher incidences of blacklegged ticks that are infected with human pathogens, such as Lyme disease, in exotic shrub thickets due to altered movement and habitat use by mammals that disperse ticks (Williams et al. 2009, Allan et al. 2010). It is therefore critical to both ecosystem and human health that park managers continue to manage and prevent the introduction and spread of these species in ACAD.

Table 10. Indicator invasive exotic plant species ratings for ACAD park units. Metric ratings are based on the mean number of indicator species found per plot (Mean #/ Plot) from 2010 to 2013.

Park	Group	# of Plots	# Plots with Indicator Invasives	Mean #/ Plot	Rating
ACAD	Overall	175	3	0.02	Good
	IAH	19	0	0.00	Good
	MDI East	82	3	0.04	Good
	MDI West	64	0	0.00	Good
	Schoodic	10	0	0.00	Good

Tree Growth and Mortality Rates

Background

Tree growth and mortality rates are important indicators of tree health and vitality. Tree growth rates can decline in response to environmental factors or anthropogenic stress, and tree mortality is often preceded by some years of reduced tree growth (Ward and Stephens 1997, Pedersen 1998, Dobbertin 2005). Decreased growth or elevated mortality rate in trees of a particular species can indicate a particular health problem for that species (Duchesne et al. 2003, Hyink and Zedeker 1987); while altered vital rates for multiple species across a region may indicate a regional environmental stress, such as acid deposition (Steinman 2004, Dobbertin 2005).

To account for differences in growth and mortality rates due to site characteristics, stand structural stage, and by tree species and size and crown class (i.e., tree position in canopy), both rates are calculated only for canopy trees (i.e., dominant, co-dominant, or intermediate crown classes) in stands that have succeeded to a mature or late-successional stage (based on the structural stage metric). To examine patterns of tree growth, we use relative basal area growth (% basal area / year), which is calculated as the percent change in basal area from successive diameter at breast height (DBH) measurements on individual trees, converted to an annual rate. Relative basal area growth rate is then averaged for canopy trees across the plot, and also by species within the plot. Metric ratings are based on regional growth rates, and are derived from U.S. Forest Service Forest Inventory and Analysis (FIA) data collected from plots in the same Ecological Subsection as ACAD.

Annual mortality rate is calculated from successive observations as the percent of stems that died between sample events (not including trees that were cut) and is converted to an annual basis. Annual mortality rate is calculated for each plot and for individual species within a plot, and then averaged by park or group of plots. Assessment points for this metric are based on annual mortality rates that have been documented in old-growth forests, which range from 0.3 to 1.6% (Busing 2005, Runkle 2000, Woods 2000).

Growth and mortality rates presented in this section represent nearly all (eight plots were established during the second cycle) of the monitoring plots in ACAD. Species growth and mortality rates were only calculated for species with five or more individuals in a unit, and overall growth and mortality rates include all species. It is also important to note that DBH measurements were checked for potential errors that could skew growth rate calculations. Trees

that differed more than 6 cm between surveys were likely the result of a data entry error, and were omitted from the analysis.

Results and Discussion

Overall growth rates were lower for all park units than FIA average growth rates, and IAH and Schoodic had growth rates that were less than 60% of the overall FIA growth rate (Table 11). Balsam fir (*Abies balsamea*) had a relatively high growth rate, especially on western MDI. Interestingly, balsam fir also had a high mortality rate overall, and on western MDI (Table 12). The high mortality rate for balsam fir is likely due, in part, to the balsam woolly adelgid, and reasons for the high growth rate are unclear. Paper birch (*Betula papyrifera*), yellow birch (*B. alleghaniensis*), eastern white cedar (*Thuja occidentalis*), and black spruce (*Picea mariana*) had relatively low growth rates. Paper birch is mostly found in stands that originated after the 1947 fire. Early successional birch and aspen species in these stands are being replaced by more shade tolerant species, such as red spruce and balsam fir, and the low growth rate for paper birch is not surprising. The slow growth rates for black spruce and eastern white cedar are also expected, as these species mostly occur in wetlands where harsh growing conditions inhibit growth.

The high mortality rate of striped maple (*Acer pensylvanicum*) is only based on five individuals of this species, two of which died between the first and second survey cycle, and is not of great concern. The high mortality rate of red spruce on Schoodic is potentially a concern, as red spruce is the most abundant tree species in ACAD. Many of the plots in closed-canopy forest (i.e., not woodland) on Schoodic are even-aged stands, where stem exclusion may be the reason for increased mortality. If the elevated mortality rate for red spruce continues or increases over time, this may indicate a more serious forest health concern. For example, acid deposition may be a factor contributing to red spruce decline (DeHayes et al. 1999).

Table 11. Average growth rates (% basal area / year) for the most common tree species in each park unit and all species combined. The growth rate calculation includes canopy trees (i.e., dominant, co-dominant, or intermediate crown classes) in stands that have succeeded to a mature or late-successional stage (based on the structural stage metric). U.S. Forest Inventory and Analysis (FIA) data were derived from plots located within the same Ecological Subsection as ACAD, and data were from 2007-2011.

Latin	Common	Overall	IAH	MDI-East	MDI-West	Schoodic	FIA
<i>Abies balsamea</i>	balsam fir	3.10	---	2.13	3.33	---	4.21
<i>Acer rubrum</i>	red maple	1.67	0.52	1.93	1.66	---	2.40
<i>Betula alleghaniensis</i>	yellow birch	0.86	0.79	0.59	---	---	---
<i>Betula papyrifera</i>	paper birch	0.98	---	1.66	0.81	---	1.48
<i>Fraxinus americana</i>	white ash	2.53	---	---	---	---	---
<i>Larix laricina</i>	tamarack	1.42	---	1.33	---	---	---
<i>Picea mariana</i>	black spruce	1.12	0.62	---	0.89	---	---
<i>Picea rubens</i>	red spruce	1.96	2.31	2.04	1.83	1.15	3.14
<i>Pinus resinosa</i>	red pine	1.70	---	1.10	---	---	---
<i>Pinus strobus</i>	eastern white pine	2.69	---	2.80	2.17	---	---
<i>Populus grandidentata</i>	bigtooth aspen	2.18	---	2.17	---	---	2.50
<i>Quercus rubra</i>	northern red oak	2.38	---	2.38	---	---	---
<i>Thuja occidentalis</i>	eastern white cedar	1.01	---	0.81	1.08	---	1.84

Latin	Common	Overall	IAH	MDI-East	MDI-West	Schoodic	FIA
Overall growth rate		1.78	1.69	1.96	1.72	1.52	2.84
Unit rating		Good	Caution	Good	Good	Caution	

Aspen (*Populus* spp.) had the greatest mortality rate, ranging from around 5% in ACAD overall to 12.5% on western MDI. Conversely, the mortality rate of aspen in FIA plots was very low. Aspens are early successional, shade intolerant species, and as forests mature aspens decline and are replaced by more shade tolerant species (e.g., red spruce, balsam fir and red maple [*Acer rubrum*]). Many stands in ACAD, particularly those that originated after the fire in 1947, are at this stage of succession, and the high mortality rate is not of great concern.

Analyses suggest that forests in FIA plots are primarily composed of younger stands that are managed more intensively for forest products (See Forest Inventory and Analysis Comparisons). Forests managed for timber or pulpwood tend to favor earlier successional species, and often the management goal is to maximize tree growth and minimize mortality (McGee et al. 1999). This may explain, in part, why the growth rates in FIA plots tend to be higher than in ACAD, and the mortality rates tend to be lower in FIA plots. Based on the tree growth and mortality rates, ACAD overall, and both sections of MDI were rated Good for this metric (Table 13). IAH and Schoodic had lower than expected growth rates, and Schoodic had higher than expected mortality rates, and both units were rated Caution. We will continue to track tree growth and mortality, and over time, we hope to have a better understanding of the dynamics influencing these rates.

Table 12. Average annual percent mortality rate for dominant species in each unit. The mortality rate calculation includes only plots in mature and late-successional stands, and only canopy trees (i.e., dominant, co-dominant and intermediate crown class). U.S. Forest Service Forest Inventory and Analysis (FIA) data were derived from plots located in the same Ecological Subsection as ACAD, and data were from 2007-2011.

Latin	Common	Overall	IAH	MDI-East	MDI-West	Schoodic	FIA
<i>Abies balsamea</i>	balsam fir	3.77	---	2.08	3.87	---	4.91
<i>Acer pennsylvanicum</i>	striped maple	8.33	---	---	---	---	---
<i>Acer rubrum</i>	red maple	0.29	0.00	0.00	0.31	---	0.68
<i>Betula alleghaniensis</i>	yellow birch	0.00	0.00	0.00	0.00	---	---
<i>Betula papyrifera</i>	paper birch	1.58	---	1.28	2.36	---	2.75
<i>Fagus grandifolia</i>	American beech	0.00	---	0.00	---	---	---
<i>Fraxinus americana</i>	white ash	0.00	---	---	---	---	---
<i>Larix laricina</i>	Tamarack	1.56	---	1.92	---	---	---
<i>Picea mariana</i>	black spruce	1.48	0.00	---	3.09	0.00	---
<i>Picea rubens</i>	red spruce	0.97	0.72	1.59	0.67	4.51	0.54
<i>Pinus resinosa</i>	red pine	0.00	---	0.00	---	---	---
<i>Pinus strobus</i>	eastern white pine	0.00	---	0.00	0.00	---	---
<i>Populus</i> spp.	aspen species	5.11	---	1.66	12.50	---	0.36
<i>Quercus rubra</i>	northern red oak	0.35	---	0.35	---	---	---
<i>Thuja occidentalis</i>	eastern white cedar	0.00	---	0.00	0.00	---	0.58
Overall mortality rate		1.02	0.45	0.88	1.07	4.57	1.16
Unit rating		Good	Good	Good	Good	Caution	

Table 13. Overall growth and mortality rate summaries and metric rating for each park unit. Growth rates that are less than 60% of the regional mean are labeled as "-". Growth and mortality rates that are within the desired ranges established by NETN are labeled "()". Mortality rates that are greater than 1.6% are labeled "+".

Unit	Growth Rate	Mortality Rate	Rating
Overall	()	()	Good
IAH	—	()	Caution
MDI-East	()	()	Good
MDI-West	()	()	Good
Schoodic	—	+	Caution

Soil Chemistry

Background

NETN monitors soil chemistry to understand the effects of atmospheric deposition on forest health, which is a special concern for forests in the Northeast that tend to grow on thin soils with low soil buffering capacity. Acid deposition occurs when emissions derived from the combustion of fossil fuels and industrial processes react with water and oxygen in the atmosphere to produce acidic anions of sulfur (S) and nitrogen (N), and then fall to earth in dry or wet form (Connolly et al. 2007). Impacts of atmospheric deposition include soil acidification, base cation leaching, increased availability of toxic aluminum (Al), and altered N cycling (Bailey et al. 2005, Connolly et al. 2007). Acid deposition has also dramatically increased inputs of N and S to forested ecosystems in the Northeast, and there is concern that excess N may “saturate” forest soils, causing excess nitrification and N leaching and exacerbating the effects of acidification (Aber et al. 1998). All of these impacts can reduce plant growth and increase susceptibility of trees to other stresses (Bullen and Bailey 2005).

NETN also monitors for the presence of earthworms on each plot. Earthworms are not native to ACAD, and their invasion has the potential to drastically alter forest soils and understory communities (Hale et al. 2006, Frelich et al. 2006).

At each plot, we collect a composite of up to three soil samples from each O and A horizon per plot, and inspect the forest floor for the presence of earthworms. Post collection, soil samples are oven dried at 33 °C within 2 weeks of collection, and then sent to the Analytical Laboratory at the University of Maine for chemical analysis. We currently rate soil chemistry based on the ratio of exchangeable Ca to Al, which has been used as an indicator of acid stress on forest soils (Cronan and Grigal 1995), and the ratio of total C to total N, a primary indicator of N status (Aber et al. 2003, MacDonald et al. 2002). Soil metrics are rated on median values for all park samples (i.e., samples collected for each plot), rather than means, due to the non-normal distribution of values typical in soil chemistry samples.

At the time of this report, the chemical analysis results were not available for the 2013 survey. We therefore include results using data from 2009 to 2012, and which were already reported in Miller et al. (2013b). More detailed analyses of soil chemistry that were conducted in 2011 are included in the Forest Soil Chemistry chapter of this report. We also include boxplot summaries of the last 4 years (2009-2012) of soil chemistry data by O and A horizons in Appendix D. Soil chemistry summaries in Appendix D were corrected for horizon, using 20% TC as the break between O horizons ($\geq 20\%$ TC) and A horizons ($< 20\%$ TC). Where the % TC check resulted in

duplicate samples of the same horizon, we adjusted soil chemistry variables by calculating weighted averages using the depth collected for each sample as weights.

Results and Discussion

Based on the median Ca:Al ratio, soil acidification may be an issue in ACAD, particularly on eastern MDI where median ratios are below 3 (Table 14). All units of ACAD have high C:N ratios, suggesting that forest soils in ACAD are robust to N saturation. It will be important to track soil chemistry over time, and to relate trends in soil chemistry to indicators of forest health to improve our soil chemistry metrics.

With the exception of eastern MDI, ACAD park units are largely uninvaded by earthworms. Earthworms have been detected in 10 plots on eastern MDI. The acidic nature of most soils in ACAD may provide resistance to earthworm invasion (Frelich et al. 2006). We will continue to track earthworm invasion and impacts of earthworm invasion on forest health over time.

Table 14. Soil chemistry ratings for NETN parks from 2009 to 2012, and earthworm data from 2010-2013. % of Plots with Earthworms are based on a rapid search for signs of earthworms, and may not detect earthworms at low levels. Plots recorded as inconclusive for earthworms were treated as an absence.

Unit	% of Plots with Earthworms	Median Ca:Al	Ca:Al Rating	Median C:N	C:N Rating
Overall	5.71	3.30	Caution	32.49	Good
IAH	0.00	4.06	Good	33.57	Good
MDI-East	12.20	2.65	Caution	29.87	Good
MDI-West	0.00	3.47	Caution	34.56	Good
Schoodic	0.00	4.81	Good	40.82	Good

Forest Composition and Structure

Background

The composition and abundance of species across forest strata provide important information about the current and future forest. To examine patterns in forest structure and composition, we summarized tree diameter distributions, density of seedlings by size class, and understory species composition.

For each park unit in ACAD, we estimated density by species of trees (≥ 10 cm diameter at breast height [DBH]) by 10 cm size classes and seedlings and saplings by height class to examine compositional patterns in the forest strata. Understory species composition was examined by calculating species diversity and average percent cover by plant guild. Species diversity was calculated at the plot level, and includes all species detected on a plot. Average quadrat percent cover was calculated from percent cover class data collected in quadrats, and is an estimate of abundance. Using the midpoints of each cover class, percent cover was averaged over the eight quadrats per plot, and then averaged at the park level. Plant guilds include exotic species, ferns, forbs, graminoids, shrubs and trees. The exotic species category includes all species that are not native to the eastern U.S., as designated by NatureServe, and includes more species than the invasive exotic plant indicator species metric. The fern category contains vascular plants that reproduce by spores and includes ferns and fern allies (e.g., club moss [*Lycopodium* spp.]). Forbs are defined as herbaceous flowering plants, but do not include grasses and sedges. The graminoids group includes grasses (Family: Poaceae), sedges (Family: Cyperaceae) and rushes (Family: Juncaceae). Trees are woody plants that have a central vertical trunk and are capable of reaching the forest canopy. Remaining woody species are shrubs.

Results and Discussion

Tree Diameter Distributions

Tree diameter distributions across all park units are typical of second-growth spruce – fir forests (Figures 8 and 9), with few trees represented in large diameter size classes (i.e., > 40 cm DBH), and many trees in smaller size classes (Kenefic et al. 2005, Solomon and Gove 1999). With over 60% of the naturally vegetated area in ACAD mapped as spruce – fir forest (Lubinski et al. 2003), the dominance of red spruce (*Picea rubens*) across all park units and diameter classes is expected. Forest habitats on IAH and Schoodic are primarily spruce – fir forest and pine woodland (Lubinski et al. 2003), and this is also reflected in the diameter distributions (Figure 8). Other than red spruce, black spruce (*Picea mariana*), jack pine (*Pinus banksiana*) on Schoodic, and pitch pine (*Pinus rigida*) on IAH are the most common species in these units.

While red spruce is the dominant tree species on Mount Desert Island (MDI), tree composition is more diverse in both sections of MDI (Figure 9). This is particularly true for eastern MDI, which, due to the 1947 fire that burned nearly half of eastern MDI (Figure 10), has a sizeable component of early successional forest. The early-successional species big tooth aspen (*Populus grandidentata*) and birch species (*Betula papyrifera* and *B. populifolia*) were the first to occupy the canopy post-fire (Barnicle 1984), and they remain relatively common canopy species in this section of the park. Eastern white pine (*Pinus strobus*), red maple (*Acer rubrum*), and northern red oak (*Quercus rubra*) are also fairly common on the richer areas of eastern MDI. Eastern white cedar (*Thuja occidentalis*), which primarily occurs in forested swamps, is most abundant on western MDI and is also found on eastern MDI and Schoodic.

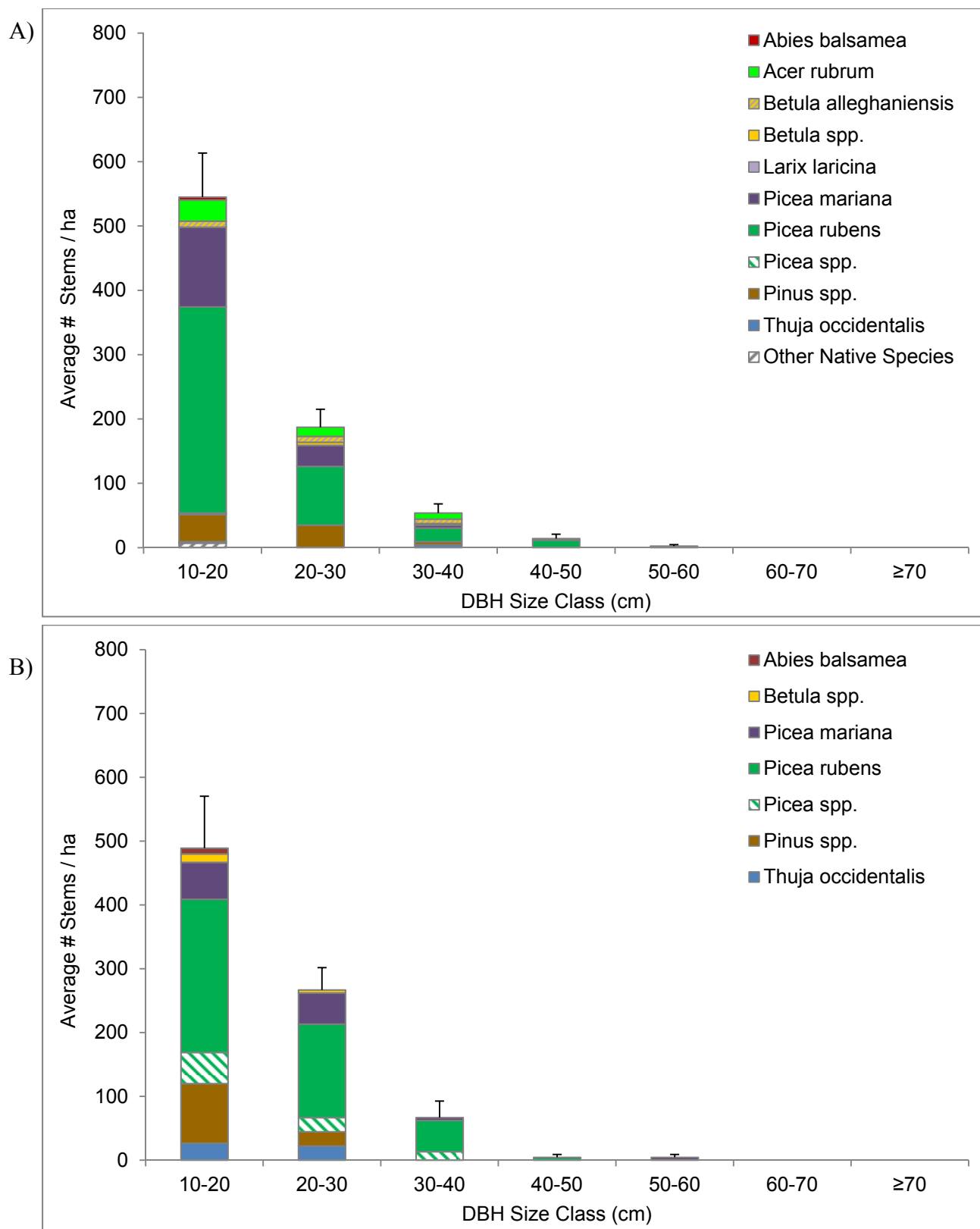


Figure 8. Tree diameter distribution of forest plots in A) Isle au Haut (n = 19) and B) Schoodic Peninsula (n = 10). Error bars represent +1 SE around the overall mean stem density for each DBH size class.

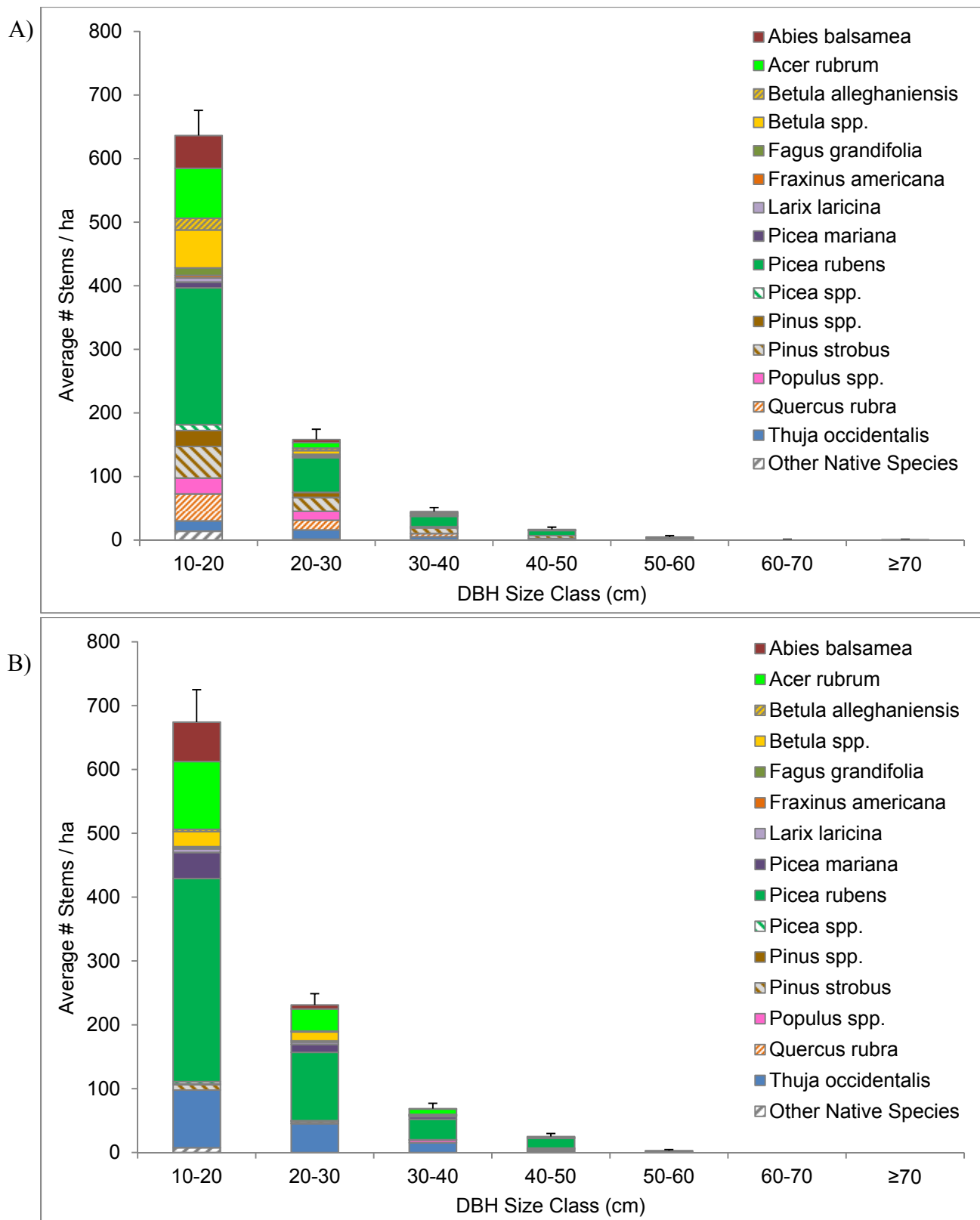


Figure 9. Tree diameter distribution of forest plots in A) eastern MDI (n = 82) and B) western MDI (n = 64). Error bars represent +1 SE around the overall mean stem density for each DBH size class.

Tree Regeneration

Overall, seedling densities and species composition are consistent with patterns expected for spruce – fir forests that are reaching mature stages of succession; regeneration is abundant and is primarily composed of shade-tolerant species, particularly red spruce, that are capable of reaching the canopy in small canopy gaps (Figures 11 and 12). The seedling layer on IAH is almost entirely dominated by red spruce and black spruce (Figure 11), which were also the dominant canopy species. Patterns are similar on Schoodic, although balsam fir is also common, and seedling densities are much higher. Pine species (pitch pine on IAH; jack pine on Schoodic), which are common canopy species in woodland habitats, are lacking in the regeneration layer.

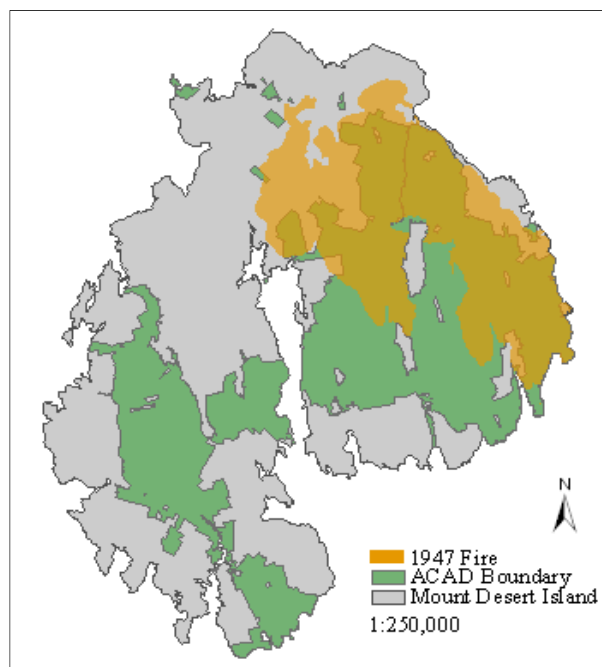


Figure 10. Map of MDI and ACAD showing extent of the 1947 fire.

The composition of regeneration on eastern section of MDI differs somewhat with the composition of the canopy (Figure 12). This is primarily the result of the 1947 fire. Much of the area that burned was stand replacing, and the early successional species bigtooth aspen (*Populus grandidentata*) and birch (*Betula papyrifera* and *B. populifolia*) were the first to occupy the canopy post-fire (Barnicle 1984). As is expected, these stands are shifting to later successional forest species, with red spruce, balsam fir, and northern hardwood species occurring at higher seedling densities than birch and aspen.

Understory Composition

Exotic species percent cover is nearly absent from all park units (Figure 13A), and plot richness of exotic species is also very low (Figure 13B). These results indicate that ACAD is largely composed of native species. The understory in ACAD is dominated by woody species (Figure 13). On IAH and Schoodic, shrubs make up the greatest percent cover, and this is primarily from ericaceous species, such as black huckleberry (*Gaylussacia baccata*) and low bush blueberry (*Vaccinium angustifolium*), both of which are often abundant in pine woodlands. Averaging around 10% cover across all park units, tree seedlings and overhanging branches (up to 1.5 m) are also abundant in the ground layer (Figure 13A).

While ferns, forbs, and graminoids are present in the ground layer, they are much less abundant than woody species. Species richness in both sections of MDI is highest in the tree guild (Figure 13B). Patterns of richness on IAH and Schoodic are similar, with forbs, shrubs, and trees having similar average plot richness. On all but Schoodic, plots average at least one fern species per plot, and all units average at least one graminoid species per plot. These results suggest that while the understory is dominated by woody species, herbaceous (ferns, forbs, and graminoids) species are consistently present, but at low levels. These patterns are expected for spruce – fir

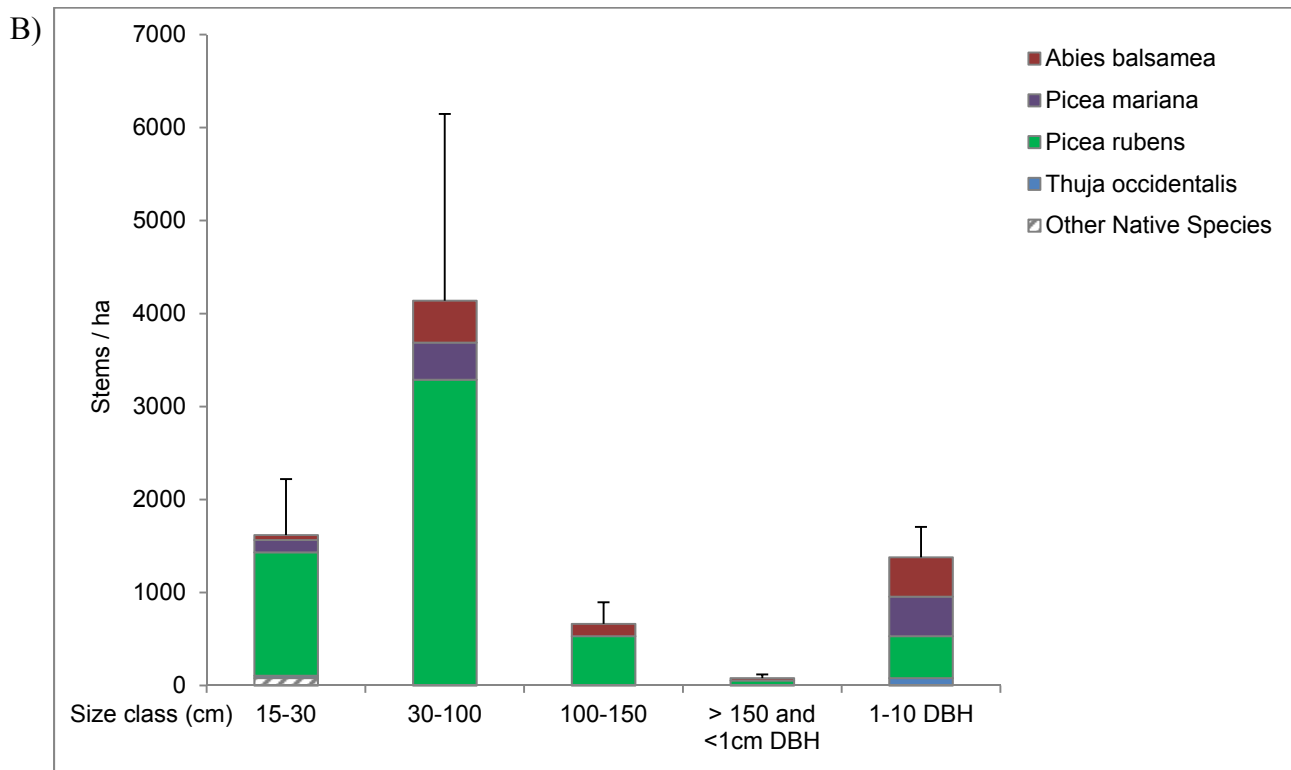
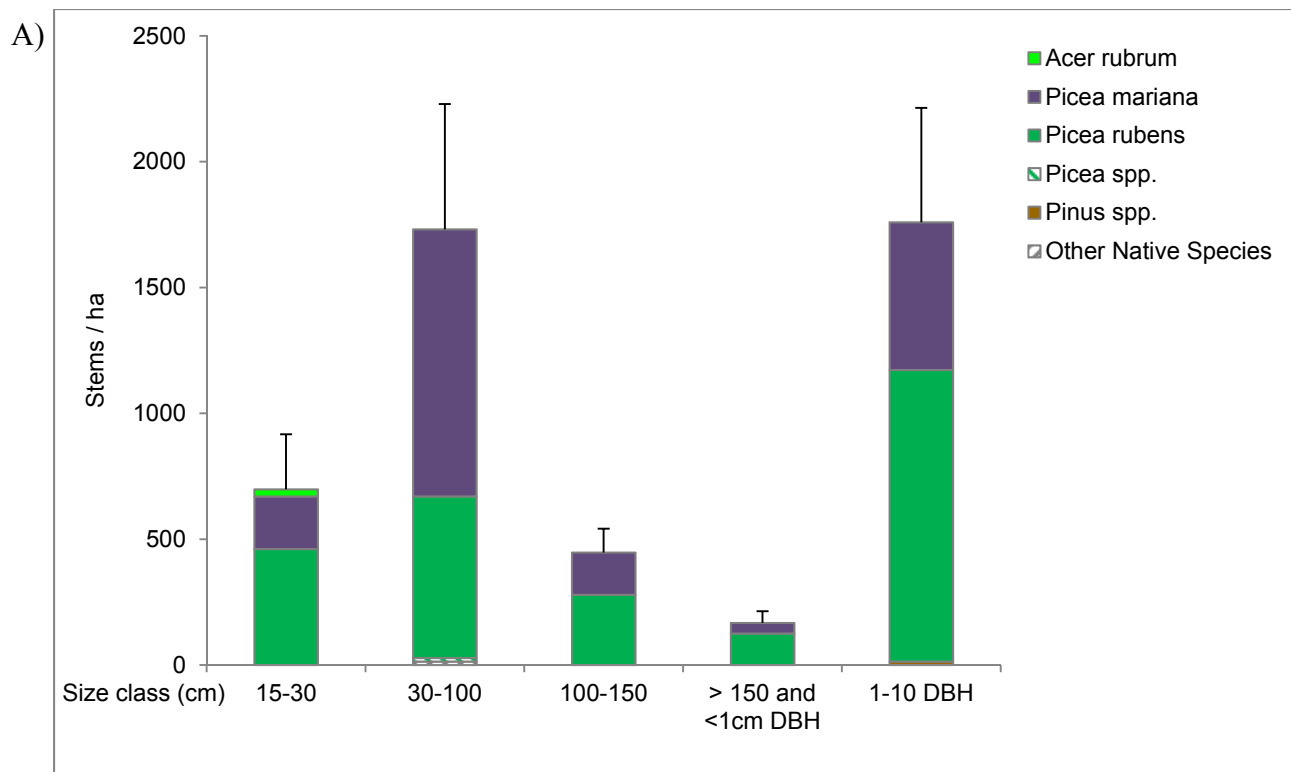


Figure 11. Density of seedlings and saplings in A) Isle au Haut ($n = 19$) and B) Schoodic peninsula ($n = 10$). Classes 15-30, 30-100, 100-150 and > 150 cm (for individuals with diameter at breast height < 1 cm) reflect seedling height, and 1-10 DBH reflects sapling diameter at breast height. Error bars represent + 1 SE around the mean.

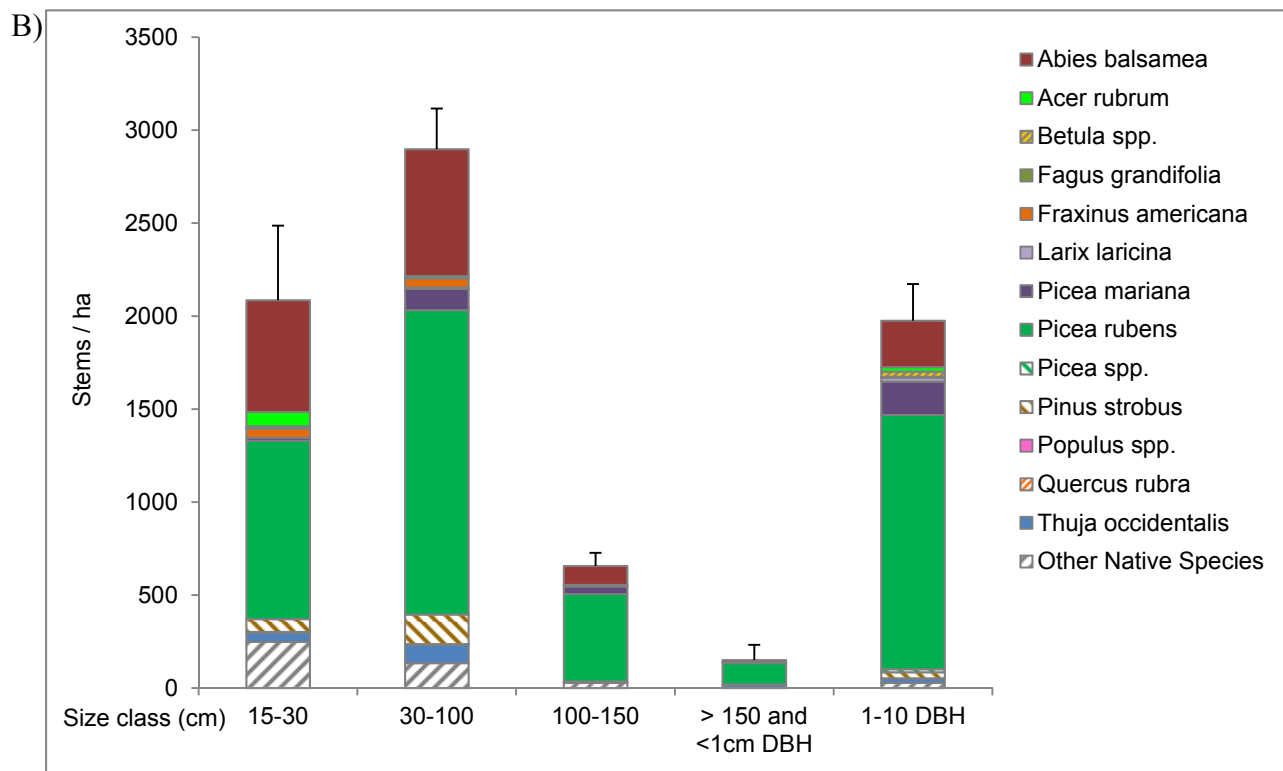
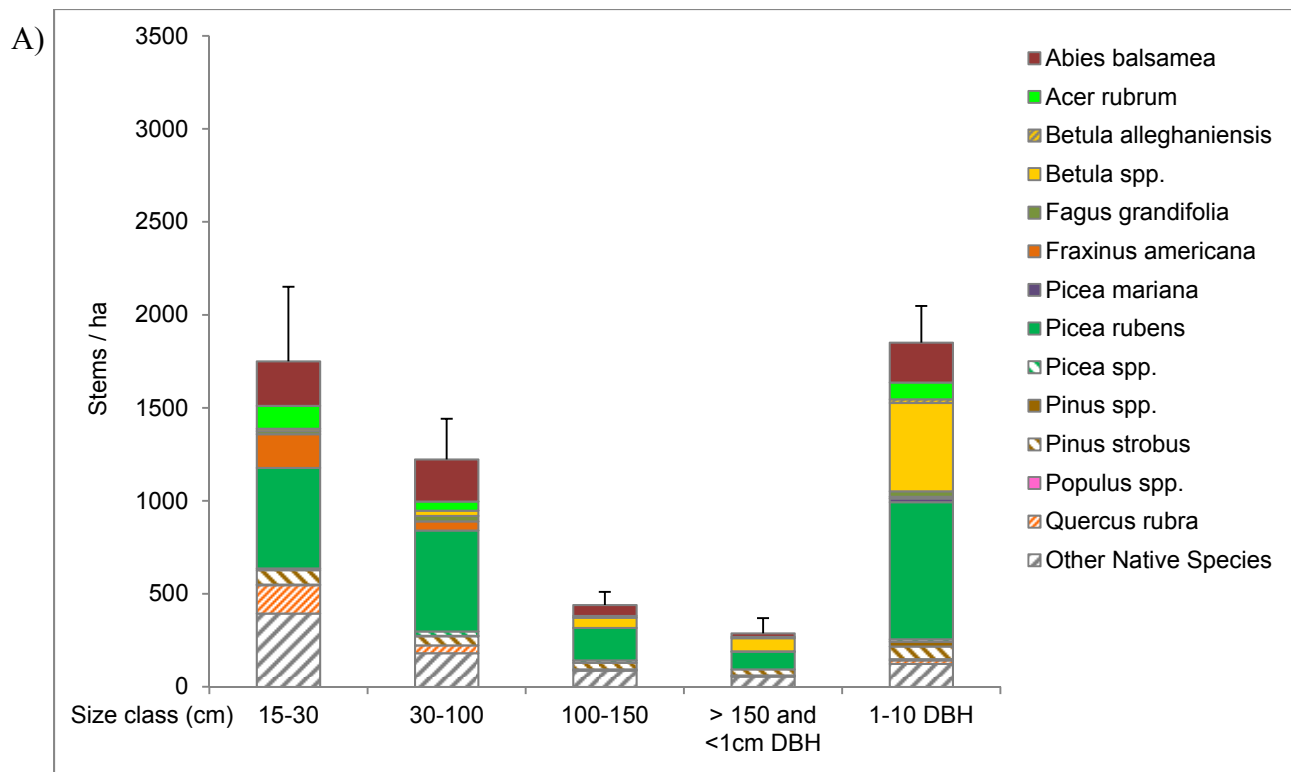


Figure 12. Density of seedlings and saplings in A) eastern Mount Desert Island (n = 82) and B) western Mount Desert Island (n = 64). Classes 15-30, 30-100, 100-150 and > 150 cm (for individuals with diameter at breast height < 1) reflect seedling height, and 1-10 DBH reflects sapling diameter at breast height. Error bars represent + 1 SE around the mean.

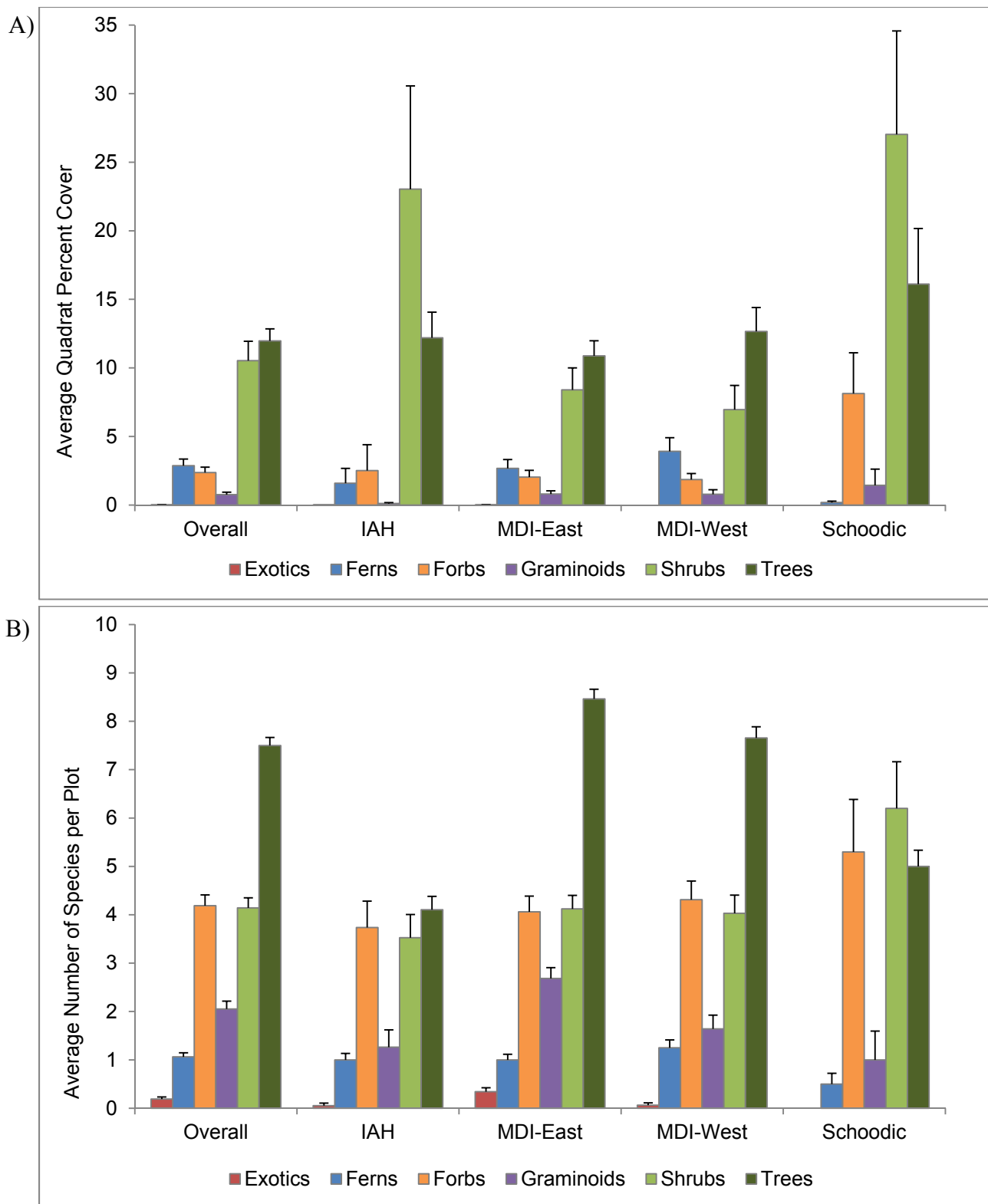


Figure 13. Average quadrat percent cover by plant guild (A) and average species richness per plot by plant guild (B) for data collected from 2010 to 2013. The Exotics guild contains any species not native to Maine. Remaining guilds only contain native species. Error bars represent + 1 SE around the mean.

forests, which tend to support a seedling bank of advanced tree regeneration in the ground layer and generally do not have a well-developed or rich herbaceous layer.

Conclusions

Many of the species that are common to the forest canopy are relatively long-lived and once established in the canopy, may be somewhat resistant to stressors caused by climate change. In contrast, seedlings are relatively short-lived and more sensitive to environmental change (Fisichelli et al. 2013, Zhu et al. 2014). Therefore changes in forest composition caused by climate change or other stressors may first be detected by changes in the regeneration and groundlayer, and current results serve as an important baseline to compare with future compositional trends across the forest strata.

Forest Inventory and Analysis Comparisons

Introduction

The U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) program has been monitoring forest resources in the Eastern U.S. since the 1930s. The primary focus of FIA has been on attributes related to timber supply, such as forest cover, ownership, and tree volume. In the 1990s, the FIA program went through a major revision which included standardizing plot configurations, measurement protocols, and sample designs across the country, and adding a Forest Health Monitoring component to a subset of monitoring plots (McRoberts et al. 2005). The newly revised FIA protocols were implemented in 2000, and plots have been sampled on a roughly 5-year interval.

The FIA program collects and reports data on status and trends in forest resources across the country, examining metrics such as forest area and ownership, species composition, tree volume, tree growth and mortality, and tree health. Plots are sampled on a three-tiered approach. Phase 1 monitoring consists of remote sensing to determine extent of forest and nonforest, and all potential plots are sampled by Phase 1. Phase 2 monitoring consists of ground plots that are sampled in the field, and plots occur at the scale of 1 to 6,000 acres (2,400 ha) in the Northeastern U.S. (McRoberts et al. 2005). A subset of Phase 2 plots are analyzed for additional forest health variables, such as soil chemistry, vegetation structure and dead woody material, and these data are collected on about one plot per 96,000 acres (39,000 ha). FIA protocols have undergone extensive development by the USFS with academic and other government agency partners and have been used to develop a substantial and ongoing database revealing trends in forest resource capacity and health across regions and the country. For these reasons, the NETN Long-term Forest Monitoring Protocol was based on the FIA program (Tierney et al. 2013). This allows us to take advantage of the larger dataset to examine regional trends and compare NPS forests with the surrounding matrix of forest lands that are under different management regimes and protection.

Methods

To compare forests in NETN parks with the surrounding forest matrix, we developed an MS Access database that contains FIA plot data and calculates metrics such as forest cover, land ownership, and metrics of forest ecological integrity (Tierney et al. 2013). Metrics are calculated from FIA plots that are in the same ecological subsection as each NETN park (U.S. Forest Service 2007), and are based on the publicly available fuzzed, swapped FIA plot locations (Figure 8). ACAD is located in the Maine Eastern Coastal Ecological Subsection (211Cb). This subsection consists of glacially scoured, often granitic bedrock, with gently rolling topography. More fog and precipitation occur in this subsection than other eastern coastal regions, and the wet, cool climate supports forests that are primarily dominated by spruce and fir (McNab et al. 2007).

Results and Discussion

Land Cover

Based on area in potential and actual FIA plots, over 60% of the subsection containing ACAD is forested (Table 15). Of these forestlands, over 85% are under private ownership, and only 4% are reserved from timber production. ACAD is also primarily forested (Lubinski et al. 2003), but forests in the park are reserved from timber production.

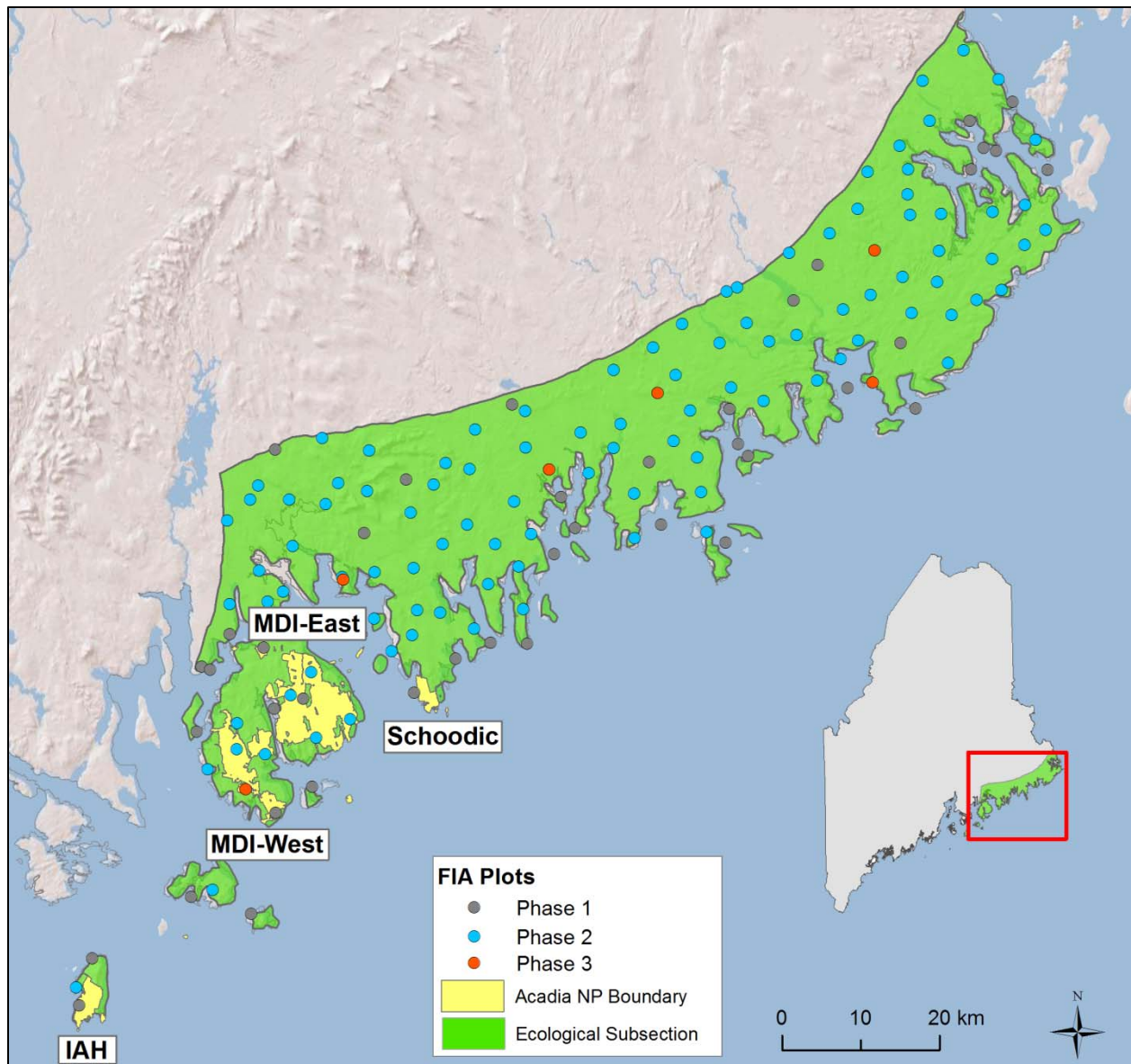


Figure 14. Map of the Ecological Subsection (Maine Eastern Coastal- 211Cb) used for comparisons between FIA data with ACAD park units. FIA plots represent the publicly available fuzzed, swapped plot locations, and not the true plot location.

Table 15. Percent of forestland overall, by timber production status and ownership, based on area in potential and actual FIA plots summed across the Ecological Subsection. Total is the percent of forestland. Reserved is the percent of forestland that is out of production from timber harvesting, and can be under any type of ownership. Ownership is the percent of forestland under each type of owner.

Maine Eastern Coastal Ecological Subsection		
% Forestland	Total	64.3%
	Reserved	4.0%
Ownership	Federal	6.9%
	State/Local	6.7%
	Private	86.4%

Structural Stage Distribution

The proportion of plots that are late-successional was higher for all ACAD units (except Schoodic) than in FIA plots (Figure 9). The combined proportion of late-successional and mature forest plots was also higher across all park units than the surrounding matrix of forestland.

ACAD park units, particularly western MDI, also had consistently higher density of live trees over 30 cm DBH than the surrounding matrix of forestlands (Figure 10).

Coarse Woody Debris and Snag Abundance

Coarse woody debris (CWD) volume was considerably higher in ACAD park units than the surrounding matrix of forestlands (Table 16). Live tree volume was consistently higher in ACAD than the nearby FIA plots, which may suggest that ACAD forests are more heavily stocked. It is also possible that the differences in live tree volume are a result of protocol differences. FIA measures stand height for all live trees on a plot, and volume estimates for each plot are calculated using the height and diameter at breast height measurement of each tree. In NETN, we only measure the height of three co-dominant trees per plot, and multiply the co-dominant stand height by constants for dominant, intermediate, and subcanopy trees to calculate live tree volume. Also, note that the metrics presented for FIA in Table 16 only include data from Phase 3 plots, which only consists of six plots in the Maine Eastern Coastal Ecological Subsection, and should be treated as preliminary.

Density of snags across all units in ACAD is higher than surrounding forestlands (Table 17). Density of snags ≥ 30 cm DBH was more variable across ACAD. Western MDI and IAH had the highest density of snags ≥ 30 cm, and snag density was higher in these units than in FIA plots. Schoodic had the highest percent of standing trees that are snags, but all are below 30 cm DBH.

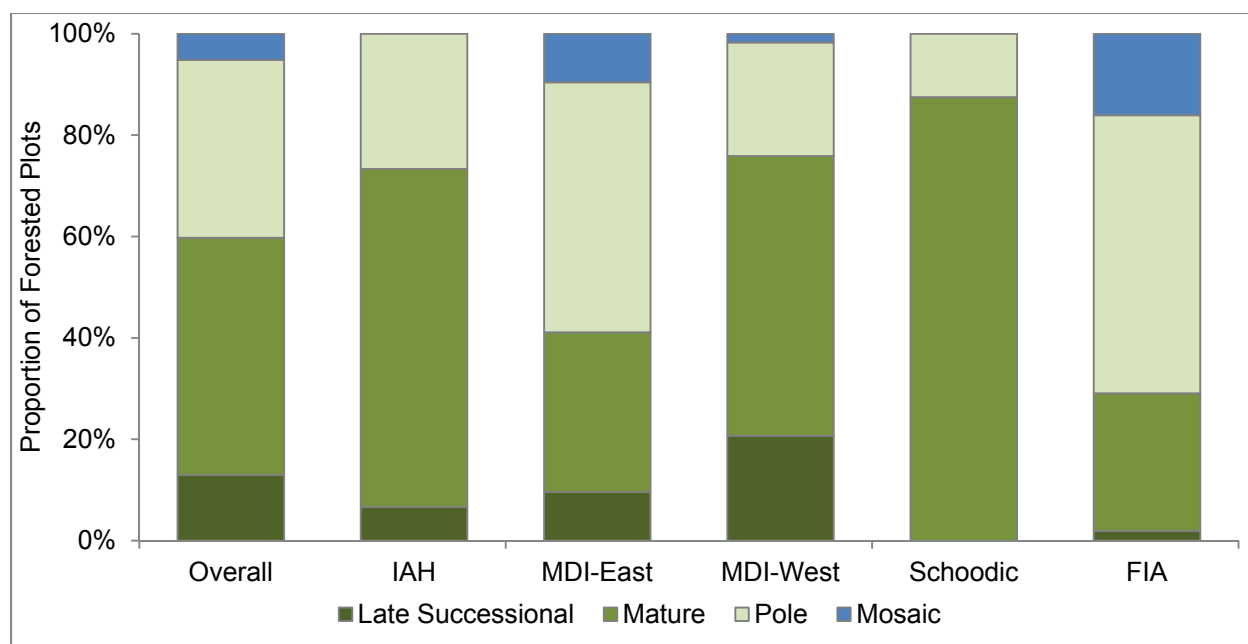


Figure 15. Percent of forest plots by structural stage in ACAD park units and FIA plots within the same Ecological Subsection.

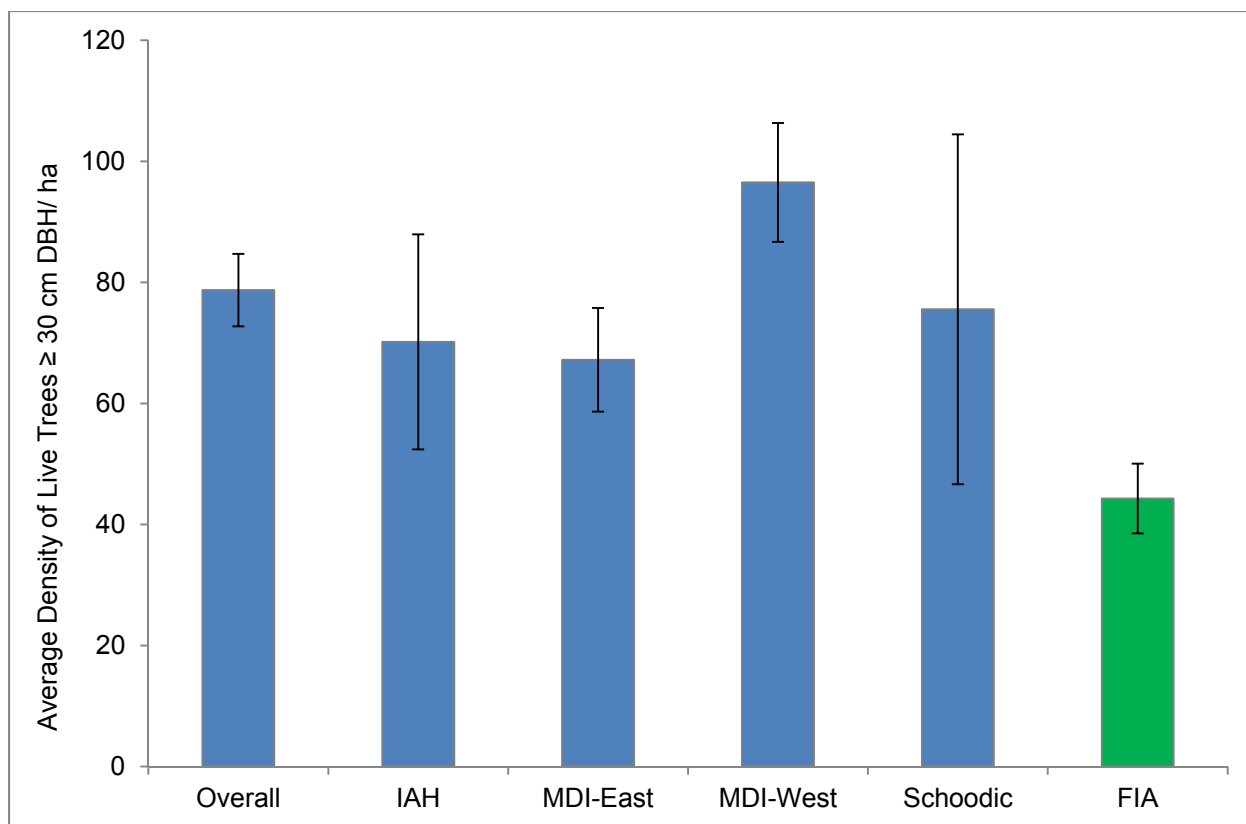


Figure 16. Average density of live trees per hectare that are ≥ 30 cm DBH ACAD park units and FIA plots located in the same Ecological subsection. Error bars represent ± 1 SE around the mean.

Table 16. Coarse woody debris metrics for ACAD park units and FIA plots within the same Ecological Subsection. The CWD Ratio is the ratio of CWD volume to live tree volume, and is expressed as a percent.

Group	# Plots	CWD Volume (m3/ha)		Live Tree Volume (m3/ha)		CWD: Live Tree Volume	
		Mean	SE	Mean	SE	Mean	SE
Overall	152	23.49	2.26	210.79	9.84	11.96	1.23
IAH	15	21.85	5.02	178.40	28.40	11.31	2.15
MDI-East	73	15.90	2.80	187.91	16.21	10.02	1.88
MDI-West	57	34.13	4.29	250.66	11.56	14.79	2.09
Schoodic	7	19.43	6.27	194.28	48.46	10.68	2.69
FIA	6	2.52	1.43	113.32	47.95	16.10	11.20

Table. 17. Snag abundance metrics for ACAD park units and FIA plots within the same Ecological Subsection, and based on the most recent full survey cycle from both datasets. % Snags is the percent of standing trees that are snags, and includes trees >10cm DBH (All Sizes), and trees > 30 cm DBH (M-L Snags). SE refers to 1 standard error around the mean.

Group	Snags/ha		# Snags ≥30 cm /ha		% of Trees as Snags	
	Mean	SE	Mean	SE	All Sizes	≥30cm
Overall	170.92	10.71	7.87	1.56	15.88	9.09
IAH	119.30	16.67	11.70	5.73	12.94	14.29
MDI-East	147.42	14.57	5.42	1.79	14.60	7.46
MDI-West	199.30	19.40	11.11	3.13	16.58	10.32
Schoodic	280.00	47.80	0.00	0.00	25.10	0.00
FIA	99.31	0.48	6.86	1.51	16.04	13.80

Conclusions

Forestlands surrounding ACAD are largely under private ownership and in timber production, whereas forests in ACAD are reserved from timber production. The difference in management regimes has resulted in forests in ACAD that are distinct from the surrounding matrix of forestlands. The combined proportion of plots in late-successional and mature forest is higher across all park units than surrounding forestlands. Indicators of older forest structure, such as coarse woody debris, large snags, and large live trees tend to be more abundant in ACAD than the surrounding matrix. Tree growth rates are also lower in ACAD than in the surrounding matrix of forestlands. These results suggest that forests in ACAD provide older forest habitat that may be regionally significant along the Down East Coast of Maine.

Forest Soil Chemistry

Background

NETN monitors soil chemistry to understand the effects of atmospheric deposition on forest health, which is a special concern for forests in the northeast that tend to have thin soils with low soil buffering capacity. Acid deposition occurs when emissions derived from the combustion of fossil fuels and industrial processes react with water and oxygen in the atmosphere to produce acidic anions of sulfur (S) and nitrogen (N), and then fall to earth in dry or wet form (Connolly et al. 2007). Impacts of atmospheric deposition include soil acidification, base cation leaching, increased availability of toxic aluminum (Al), and altered N cycling (Bailey et al. 2005, Connolly et al. 2007). Acid deposition has also dramatically increased inputs of N and S to forested ecosystems in the northeast, and there is growing concern that excess N may “saturate” forest soils, causing excess nitrification and N leaching and exacerbating the effects of acidification (Aber et al. 1998). All of these impacts can reduce plant growth and increase susceptibility of trees to other stresses (Bullen and Bailey 2005).

At each plot, NETN collects a composite of soil samples from each O and A horizon, because these upper soil layers are where the bulk of fine roots are located and nutrient uptake occurs. In most cases, the composited sample for each layer is composed of three replicate samples. In extremely rocky or root-laden habitats, there are occasions where only one or two samples can be collected. If no soil horizons are apparent, NETN collects a composite of the upper soil containing fine roots to a maximum depth of 10 cm. The thickness of each horizon is recorded for each sample, and the presence or absence of earthworms is recorded for each plot.

Post collection, soil samples are oven dried, and then sent to the Analytical Laboratory at the University of Maine for chemical analysis. Soil pH is measured in distilled water. Organic matter is measured by loss on ignition at 550 °C. % total N (TN) and % total C (TC) are measured by combustion analysis at 1,350 °C. Exchangeable acidity is extracted in potassium chloride and measured by titration. Exchangeable cations are extracted in ammonium chloride and measured by inductively coupled plasma optical emission spectrometry (ICP-OES). Effective cation exchange capacity is calculated by summation of milliequivalent levels of Ca, K, Mg, and Sodium (Na), and acidity. From the chemical analysis results provided by the University of Maine Analytical Laboratory, several additional variables are calculated, including the soil quality index developed by Amacher et al. (2007; Table 18). Samples were corrected for horizon, using 20% TC as the break between O horizons ($\geq 20\%$ TC) and A horizons ($< 20\%$ TC). Where the % TC check resulted in duplicate samples of the same horizon, we adjusted soil chemistry variables by calculating weighted averages using the depth collected for each sample as weights.

Parks included in this analysis are Acadia National Park (ACAD), Marsh-Billings-Rockefeller National Historical Park (MABI), Minute Man National Historical Park (MIMA), Morristown National Historical Park (MORR), Roosevelt-Vanderbilt National Historic Sites (ROVA), Saint-Gaudens National Historic Site (SAGA), Saratoga National Historical Park (SARA), and Weir Farm National Historic Site (WEFA).

Data Analysis

Non-metric Multidimensional Scaling (NMDS) ordinations were used to examine major gradients in forest soil chemistry across network parks, and to relate these gradients to stressors

Table 18. Soil chemistry variables used in Non-metric Multidimensional Scaling ordinations, and codes used for each soil variable on ordination figures.

Soil Chemistry Variable	Code	Soil Chemistry Variable	Code
Soil pH	soil pH	Carbon: Nitrogen ratio	C:N
Effective cation exchange capacity	ECEC	Nitrogen: Phosphorus ratio	N:P
% Base Saturation	BaseSat	% Total Carbon	TC
Calcium: Aluminum Ratio	Ca:Al	% Total Nitrogen	TN
Calcium % Saturation	Ca%Sat	% Loss of Ignition	LOI
Aluminum % Saturation	Al%Sat	Phosphorus (mg/kg)	P
Calcium (mg/kg)	Ca	Exchangeable acidity (meq/100g)	acidity
Aluminum (mg/kg)	Al	Iron (mg/kg)	Fe
Magnesium (mg/kg)	Mg	Manganese (mg/kg)	Mn
Potassium (mg/kg)	K	Zinc (mg/kg)	Zn
Sodium (mg/kg)	Na	Soil Quality Index	SQL

(e.g., acid deposition, N saturation, and invasive species), land use history, environmental gradients, and forest condition. To examine potential relationships between soil chemistry and other gradients, a matrix of environmental variables was assembled to include values derived for each permanent monitoring plot or park, depending on the data source.

For each plot, elevation, aspect, topographic position, slope, location (X and Y coordinates), vegetation type, bedrock geology, native plant species richness, stocking index (McWilliams et al. 2005), and average cover of invasive exotic indicator plant species were compiled. Total N and S deposition rates for each park were extracted from Sullivan et al. (2011). Water quality parameters collected from park waterbodies were averaged across each park, including pH, acid neutralizing capacity (meq/L), and nitrate concentrations ($\mu\text{eq/L}$). In all but WEFA, water quality data were collected from park streams. WEFA water data were taken from Weir Pond because there are no streams in the park.

Prior to each ordination, soil chemistry variables were standardized to remove the influence of different scales and units. Ordinations were performed using the metaMDS function in the R vegan package (Oksanen et al. 2011, R Development Core Team 2011). After experimenting with multiple combinations of distance matrices (Euclidean, Bray-Curtis, Manhattan, and Kulczynski), dimensions ($k=1-3$), and transformations, we found the settings that resulted in the best (i.e., low stress, converged model) and most interpretable models were with $k=2$ dimensions, Bray-Curtis distance matrix, and manual standardization: $(x_i - \min_x) / (\max_x - \min_x)$. NMDS ordinations were performed on 1) the full soil community dataset, 2) the A horizon for all NETN parks, and 3) the O horizon for ACAD. All analyses are based on data collected between 2007 and 2010.

NMDS results are displayed in 2-dimensional ordination plots to aid interpretation of patterns. Forest plots that are close together in the ordination plot have similar soil chemistry, and forest plots that are far apart have dissimilar soil chemistry. The location of soil chemistry variables in the ordination plot were calculated using weighted averages of the plot scores for each variable. In other words, plots near the soil pH variable in an ordination plot all tend to have relatively high soil pH, and plots that are far away from the soil pH variable tend to have relatively low soil

pH. In addition, soil chemistry variables that are close together in the ordination plot tend to be positively correlated, and variables far apart tend to be negatively correlated.

To assess associations between the ordinations and environmental variables, multiple additional functions were used from the vegan package in R. The envfit function was used to test the significance of environmental variables and groups in the ordination. Because many of the environmental variables did not have a linear response, the ordisurf function was also used. The function ordisurf uses thinplate splines in a generalized additive model (GAM) to predict and plot the surface of each variable in the 2-D ordination space. To assess distinct park groupings we used the ordiellipse function which plotted 95% confidence regions for the standard error of the centroid of each group. The ordisurf, ordiellipse, and envfit functions do not influence the ordination results, but provide a visual reference for how environmental variables and factors relate to the ordination. Soil ordination interpretations are based on dozens of ordisurf, ordiellipse, and envfit operations on multiple environmental and soil variables. To save space, we only present the most relevant or illustrative ordination plots. The full suite of ordination plots, R code, and data are available upon request. Results presented here focus on relationships between soil chemistry and impacts from atmospheric deposition.

Results and Discussion

NMDS ordinations were successfully performed on the full soil community dataset (stress = 0.063), A horizon samples for all parks (stress = 0.077), and O horizon samples from ACAD (stress = 0.152); stress values below 0.3 indicate a successful ordination. The first major gradient in the ordination of the full soil community dataset related to soil horizon (Figure 11). The centroids of the O and A horizon were significantly different in the ordination space (envfit $p = 0.001$), and there was very little overlap between O and A horizon samples. This was expected, as chemical properties of the O and A horizons are determined by different soil processes. With such a distinct separation in horizons, and with numerous plots represented twice on opposite ends of the ordination, assessing environmental variables at this level of ordination was not ideal. Therefore, assessment of environmental gradients focused on individual soil horizons. We focused on A horizons for the network-wide analysis because there were considerably more A horizon samples per park (except at ACAD). Environmental gradients were also not as strong in the O horizon ordinations, and may be due to the strong influence of plant species-specific leaf litter and other micro-site conditions on the O horizon that may obscure the environmental gradients. We did, however, examine patterns in the O horizon for ACAD, because there are more O horizon samples in ACAD, and the vegetation in the park is more homogenous.

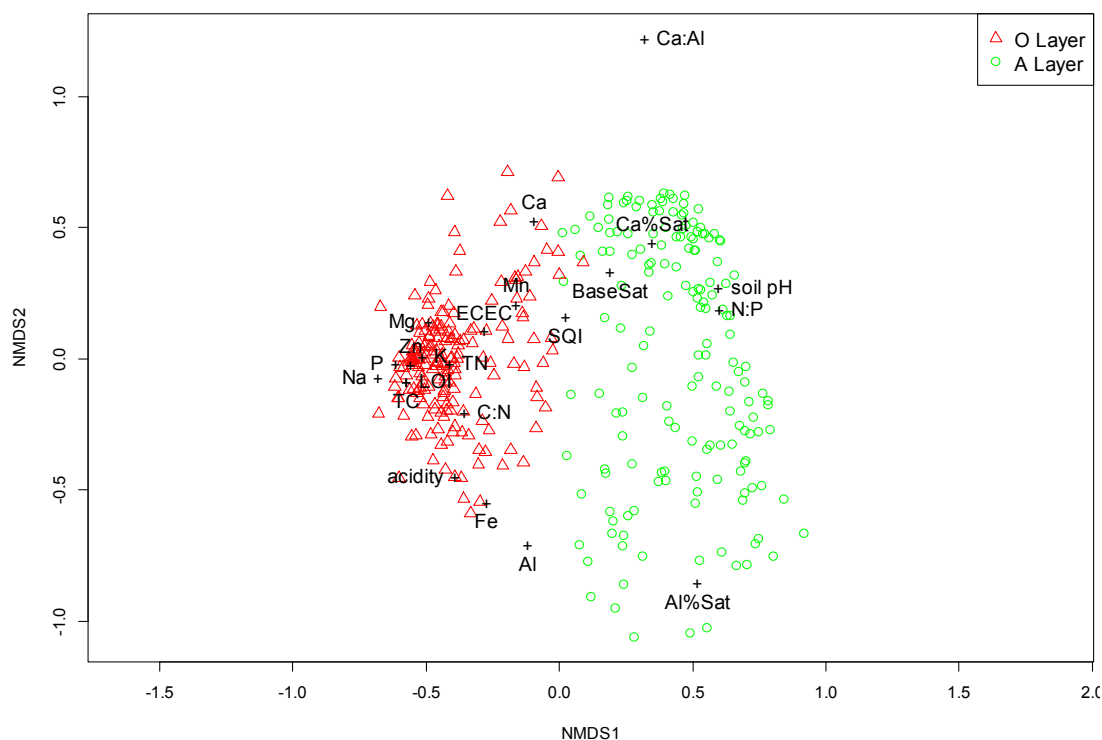


Figure 17. NMDS ordination of O and A horizon samples collected between 2007 and 2010 in NETN parks. Red and green symbols represent plots. The weighted average score for each soil chemistry variable is plotted as a black “+”.

Analysis of A horizon samples for all NETN parks

Ordination results indicate that A horizon soil chemistry varies considerably across (and sometimes within) NETN parks, and follows two major gradients reflected in the NMDS axes (Figure 12). The gradient in NMDS Axis 1 relates to the coupled negative relationship between Ca and Al. Plots on the negative (left) side of Axis 1 have A horizons with lower pH, higher Al and Fe concentrations, and higher Al % saturation (Appendix E: Figure E-1). Alternatively, plots on the positive (right) side of Axis 1 have richer A horizons with higher soil pH, higher base cation concentrations, and higher base cation % saturation. Average quadrat species richness is also correlated with Axis 1, with greater diversity associated with richer soils ($r^2 = 0.122$, $p < 0.001$).

The second major gradient in the A horizon follows NMDS Axis 2. Plots on the negative (bottom) end of Axis 2 have high % Total C, organic C (LOI), % Total N and P, and a lower pH than plots on the positive (high) end of Axis 2 (Appendix E: Figure E-3). Relating these patterns to each park, we find that soils in ACAD are acidic with relatively high Al, Fe and organic matter. On the other extreme, soils in SARA have high base saturation and base concentration, higher pH and lower C and N content. Soils in MABI and MORR also appear to be relatively rich, although C content is higher in MORR. Soil chemistry in remaining parks is more variable with plots represented across each gradient.

The gradients in Axis 1 and Axis 2 can also be related to risk of impacts from atmospheric deposition. Axis 1 provides information about the relative risk of each park to soil acidification,

such that soils on the positive side of Axis 1 are well buffered and resistant to acidification and soils on the negative end of Axis 1 have lower buffering capacity and are more vulnerable to acidification. The Ca:Al ratio and acid neutralizing capacity (ANC) of park streams provide further evidence of acidification vulnerability along Axis 1. Low values of the Ca:Al ratio (i.e., $\text{Ca:Al} < 4$) have been used as an indicator of acid stress for trees (Cronan and Grigal 1995, and Fenn et al. 2011), and this ratio increases from left to right on Axis 1. ANC also increases from left to right on Axis 1. To relate these results to each park, Axis 1 was divided into five categories (bins) of vulnerability. Bins were chosen based on ranges of the Ca:Al ratio that have been linked to levels of acidification risk, and natural breaks in % base saturation and Al concentration (Appendix E: Figure E-2). Distribution of plots along the acidification gradient were then summarized as pie charts, and overlaid on a map of 2011 wet deposition of S and N (Figure 13).

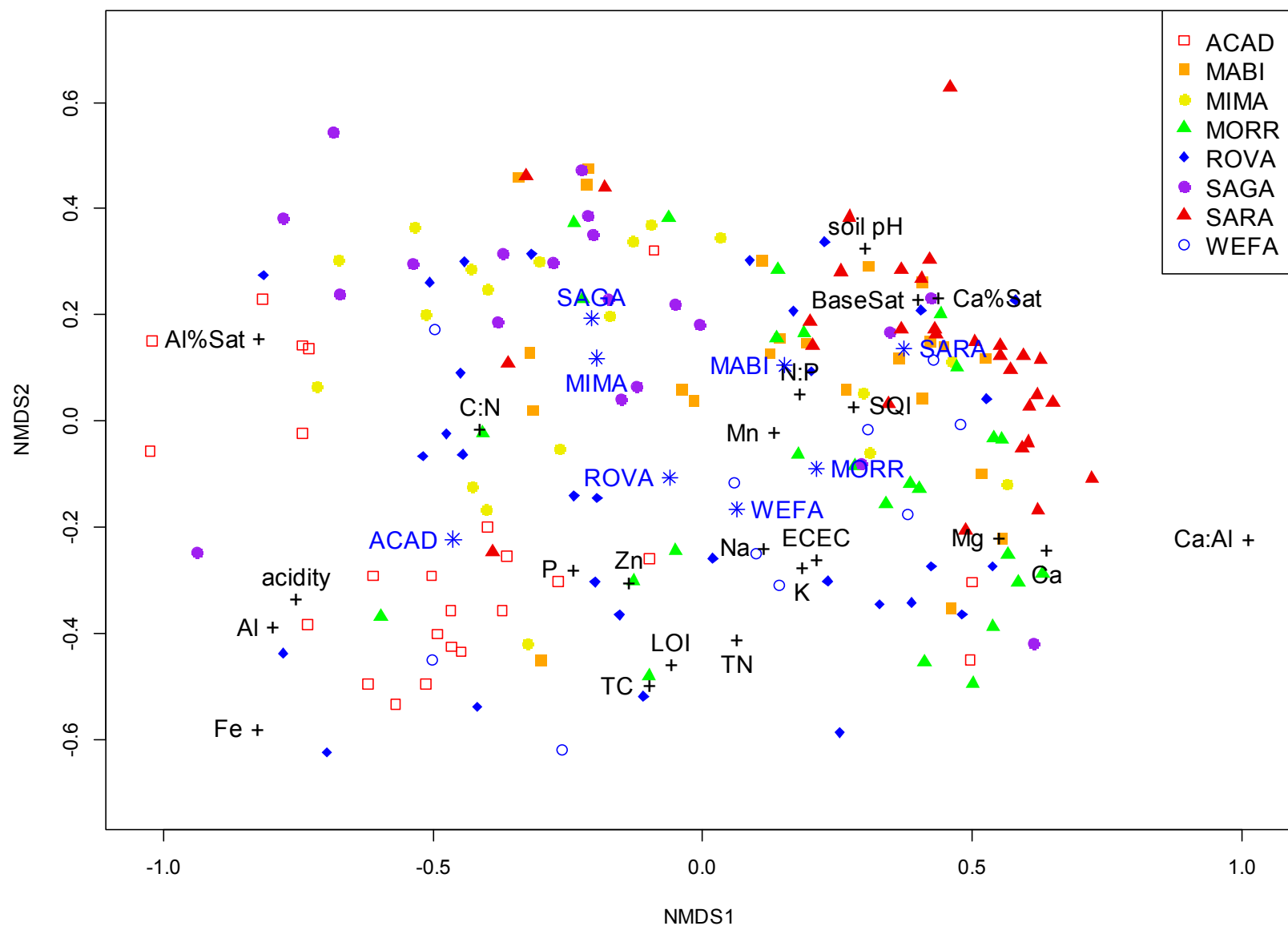


Figure 18. NMDS ordination on A horizon samples collected between 2007 and 2010 in NETN parks. Colored symbols represent plots. The weighted average score for each soil chemistry variables is plotted as a black "+". The blue "*" represents the centroid for each park.

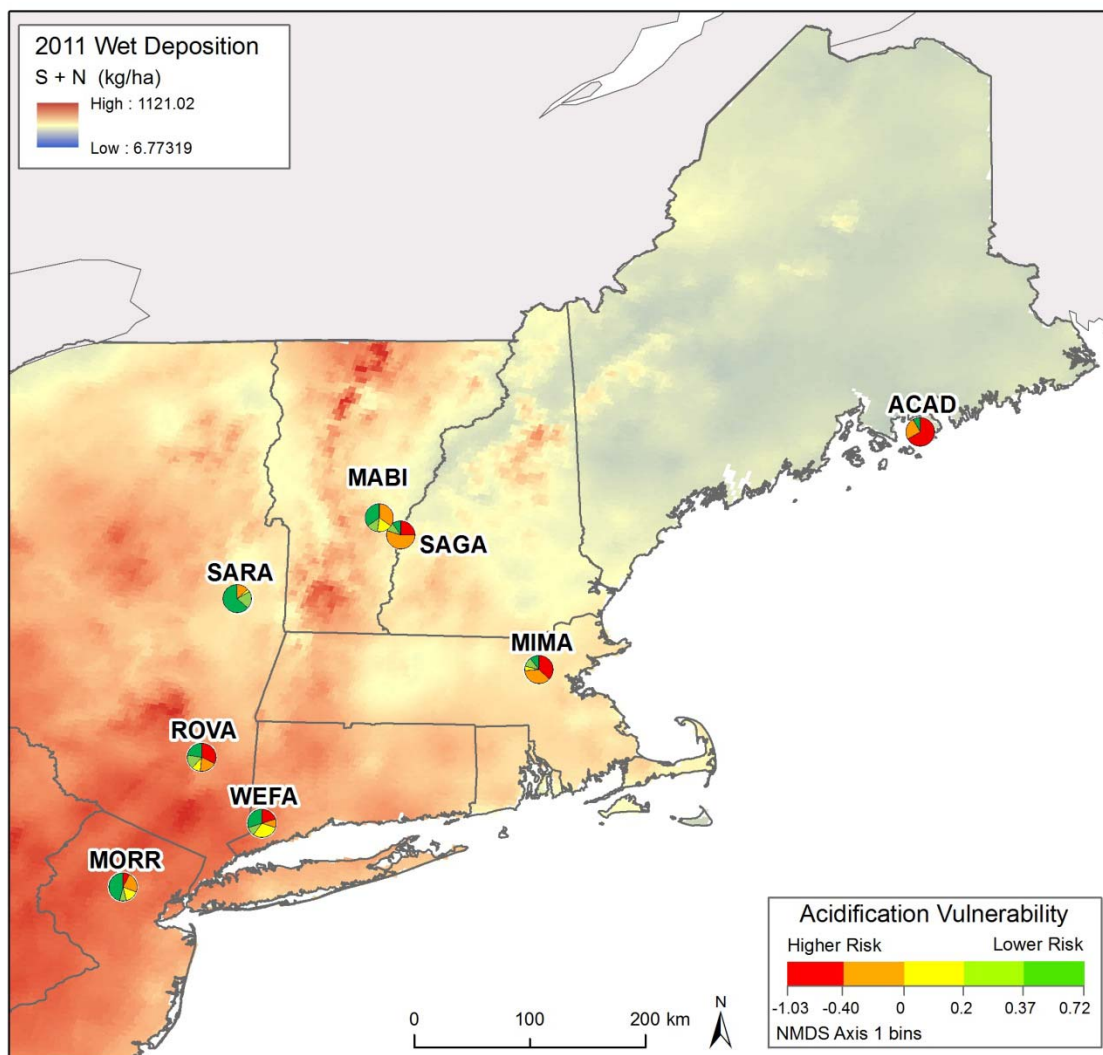


Figure 19. Relative vulnerability of A horizon soils to acidification based on A horizon NMDS Axis 1. NMDS Axis 1 was divided into five bins, with the most negative bin representing highest risk (red), and the most positive bin representing lowest risk (green). Pie charts represent the proportion of plots in each NMDS Axis 1 bin. Background map displays 2011 S and N deposition rates for the eastern U.S. (National Atmospheric Deposition Program 2012).

Despite receiving the highest S and N deposition rates, the majority of plots in MORR are at low risk of acidification. The soils in SARA and MABI appear most resistant to acidification. ACAD soils are most vulnerable, with over 90% of plots falling in the two higher risk categories. Over 70% of plots in MIMA, and 80% of plots in SAGA also fell in the two higher risk categories of acidification vulnerability. Plots in ROVA and WEFA were relatively evenly spread across Axis 1.

The patterns in Axis 2 describe a positive association of total C, total N and P and organic C. This association was not surprising, as they are all products of the decomposition of organic material. Interestingly, parks with a more recent history of agriculture or timber harvesting (e.g., MABI, MIMA, SAGA and SARA) fall almost exclusively on the positive end of Axis 2

suggesting A horizons in these parks have low organic content and lower N and P. Parks with a less intensive history of disturbance, including MORR, ROVA and WEFA, have plots scattered across Axis 2.

While the patterns in Axis 2 are informative, relating these patterns to risk of acid deposition impacts is more complicated than with the first axis. To help interpret the results in Axis 2, we rely on two indicators of N saturation risk, including soil C:N ratio (Aber et al. 2003, MacDonald et al. 2002), and stream NO_3^- concentration (Fenn et al. 2011). The C:N ratio indicates N status of forest soils such that ratios below 20 are at risk of increased nitrification and NO_3^- export. Fenn et al. (2011) suggest that stream NO_3^- concentrations above 20 $\mu\text{eq/L}$ indicate sites where N saturation and NO_3^- export may be occurring (assuming N inputs from other sources in the stream are minimal).

Based on A horizon C:N ratios, stream NO_3^- concentrations, and deposition rates, ACAD appears to be the least at risk of N saturation (Figure 14). SARA is on the other extreme with the lowest average C:N ratio, and the only average stream NO_3^- concentration above the 20 $\mu\text{eq/L}$ threshold. This suggests that soils in SARA are most vulnerable to N saturation, and it is possible represent thresholds proposed in Fenn et al. (2011) such that C:N ratios below 20 indicate risk of nitrification, and stream nitrate values above 20 indicate possible nitrate leaching. that N saturation and NO_3^- leaching are occurring in SARA. However, SARA is surrounded by agricultural lands, and NO_3^- inputs from agriculture cannot easily be separated from those of soil leaching. ROVA also has a fairly high stream NO_3^- concentration, and as with SARA surrounding land use may be a contributing factor.

A horizons in MABI, MIMA and SAGA also tend to have low C and C:N ratios at or below 20 (Figure 14), indicating park-wide vulnerability to N saturation. Plots in MORR, ROVA and WEFA vary in their vulnerability to N saturation, and C:N ratios in these parks may be related to forest type and landform. With low stream NO_3^- concentrations, N saturation and NO_3^- export does not appear to be an issue in most NETN parks. However, if N deposition rates were to increase, the low C:N ratios for most NETN parks suggests N saturation may occur. These results serve as an important baseline that, over time, will allow us to understand relationships between soil chemistry, acid deposition, and stream chemistry and to relate these trends with indicators of forest health (e.g., tree growth and mortality rates).

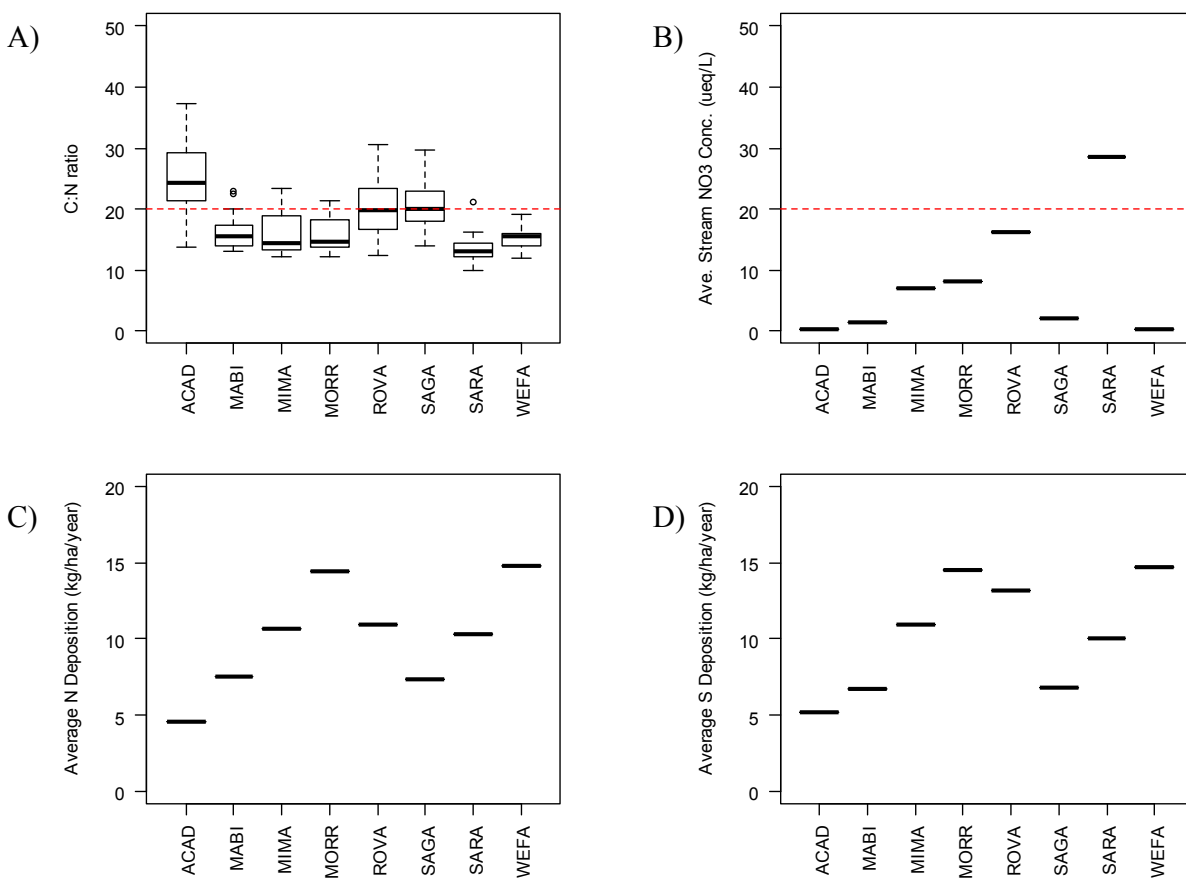


Figure 20. Boxplots of A) C:N ratios of A horizons by park, B) Average stream nitrate concentrations (µeq/L), C) Average annual N deposition (kg/ha), and D) Average annual S deposition (kg/ha). Red lines

Analysis of Acadia National Park soil chemistry

There were no distinct groupings by vegetation type, park subunit, or fire history in the NMDS ordination of O horizon samples (Figure 15). While plots burned in the 1947 fire tended to have slightly lower organic C (LOI) and % total C than plots on eastern MDI that didn't burn, there were few other differences in soil chemistry between burned and unburned plots. Soils do appear to be more homogenous and similar to each other on Isle au Haut (IAH) and Schoodic Peninsula (Schoodic) than on Mount Desert Island (MDI). While larger sample size may partially explain the greater variability in soil chemistry on MDI, greater diversity of habitats and broader range of elevations on MDI is also likely a factor.

Patterns in the first axis of the ordination on ACAD O horizon samples are similar to the A horizon analysis of all NETN parks. Plots on the negative (left) side of Axis 1 are more acidic with greater Al concentration and Al % saturation. These plots also tend to have a Ca:Al ratio below 1, indicating elevated risk of acid stress. Plots on the positive (right) side of Axis 1 tend to have O horizons with greater soil fertility and Ca:Al ratios above 5.

To examine how patterns in Axis 1 relate to topography and geographic location in ACAD, NMDS Axis 1 was broken into five categories of vulnerability to acidification. Vulnerability

categories were based on ranges of the Ca:Al ratio that have been linked to levels of acidification risk and natural breaks in % base saturation and Al concentration (Appendix F: Figure F-2). Each plot was assigned a risk based on its Axis 1 score, and was mapped over a bare ground LiDAR digital elevation model. Acidification vulnerability assignments for each plot from the network-wide analysis of A horizon samples were also plotted on these maps. It is important to note that the acidification vulnerability assignments on O and A horizons are based on different analyses and therefore not directly relatable. However vulnerability categories were based on similar ranges of the Ca:Al ratio, % base saturation and Al concentration in both horizon analyses and should be roughly comparable.

Maps of acidification vulnerability indicate soils in each subunit run the full range of acidification risk in the O horizon, and tend to have higher vulnerability assignments in the A horizon (Figures 16-19). Acidification vulnerability does not appear to differ between areas in the park that burned in 1947 and areas that did not burn (Figure 17). Higher elevation plots tend to be more vulnerable to acidification ($r^2 = -0.215$, $p < 0.005$) in the O horizon, and this is most evident on eastern MDI where most of the high elevation sites occur in ACAD. However, the negative correlation between elevation and Axis 1 is weak. The weak correlation is due to numerous plots at lower elevations that are also high risk, including a cluster of plots with relatively high acidification vulnerability on the northern end of western MDI. It is unclear why so many plots in this area are vulnerable, but may relate to forest type, stand age, and/or land use history.

These results serve as a valuable baseline for forest soil chemistry in ACAD, and it will be important to continue tracking trends in soil chemistry to see if our current understanding is correct and to better understand how soil chemistry relates to indicators of forest health, such as tree condition, growth, and mortality.

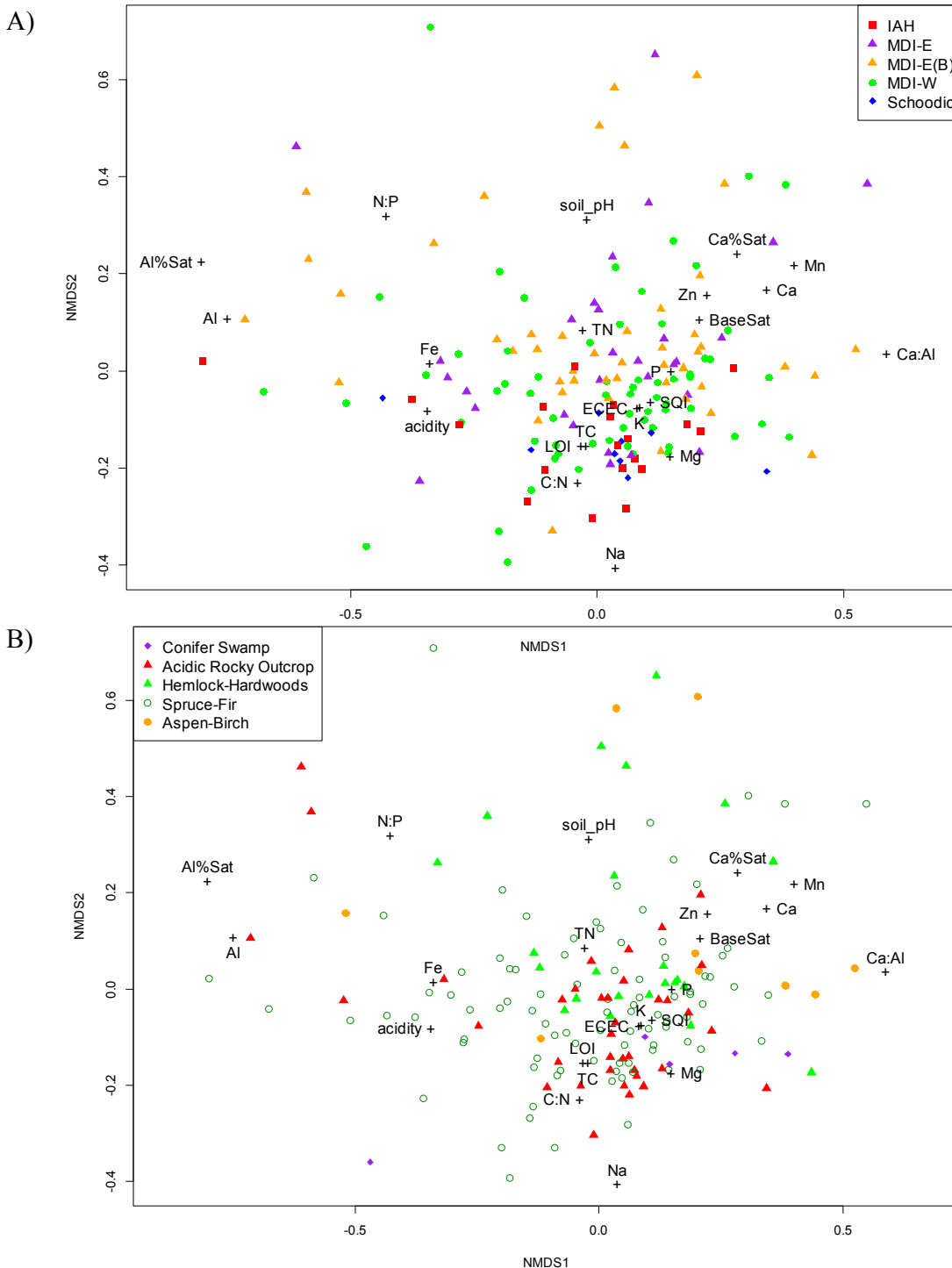


Figure 21. NMDS ordination on O horizon samples collected in ACAD between 2007 and 2010 grouped by A) Park subunit, and B) Forest type. Park subunit abbreviations are IAH for Isle au Haut, MDI-E for plots on the eastern side of Mount Desert Island (MDI), MDI-E (B) for plots on MDI-E that were burned in 1947, MDI-W for plots on western MDI, and Schoodic for plots on the Schoodic Peninsula. Colored symbols represent plots. The weighted average score for each soil chemistry variables is plotted as a black "+".

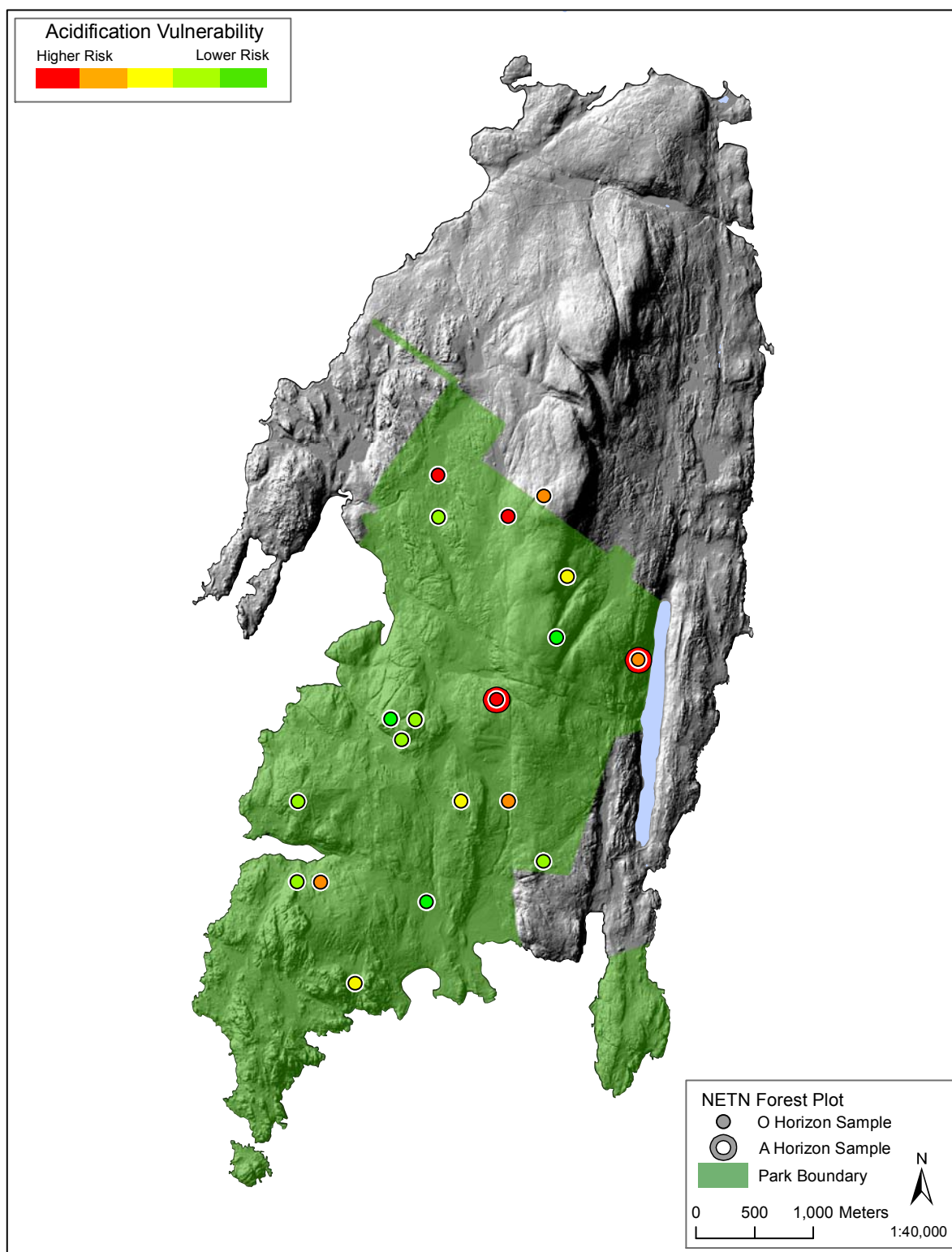


Figure 22. Acidification vulnerability of O and A horizons by forest plot on Isle au Haut.

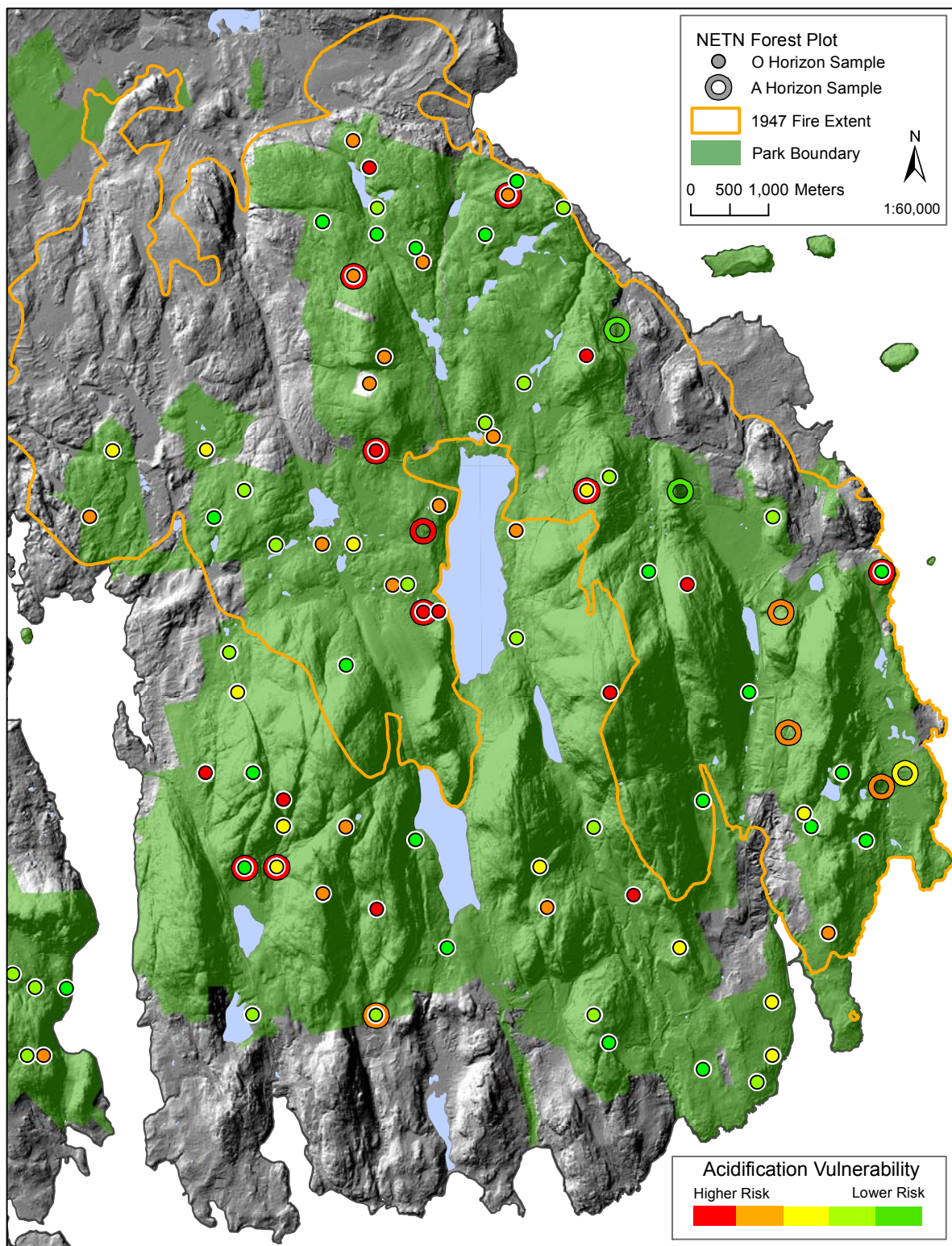


Figure 23. Acidification vulnerability of O and A horizons by forest plot on eastern Mount Desert Island.

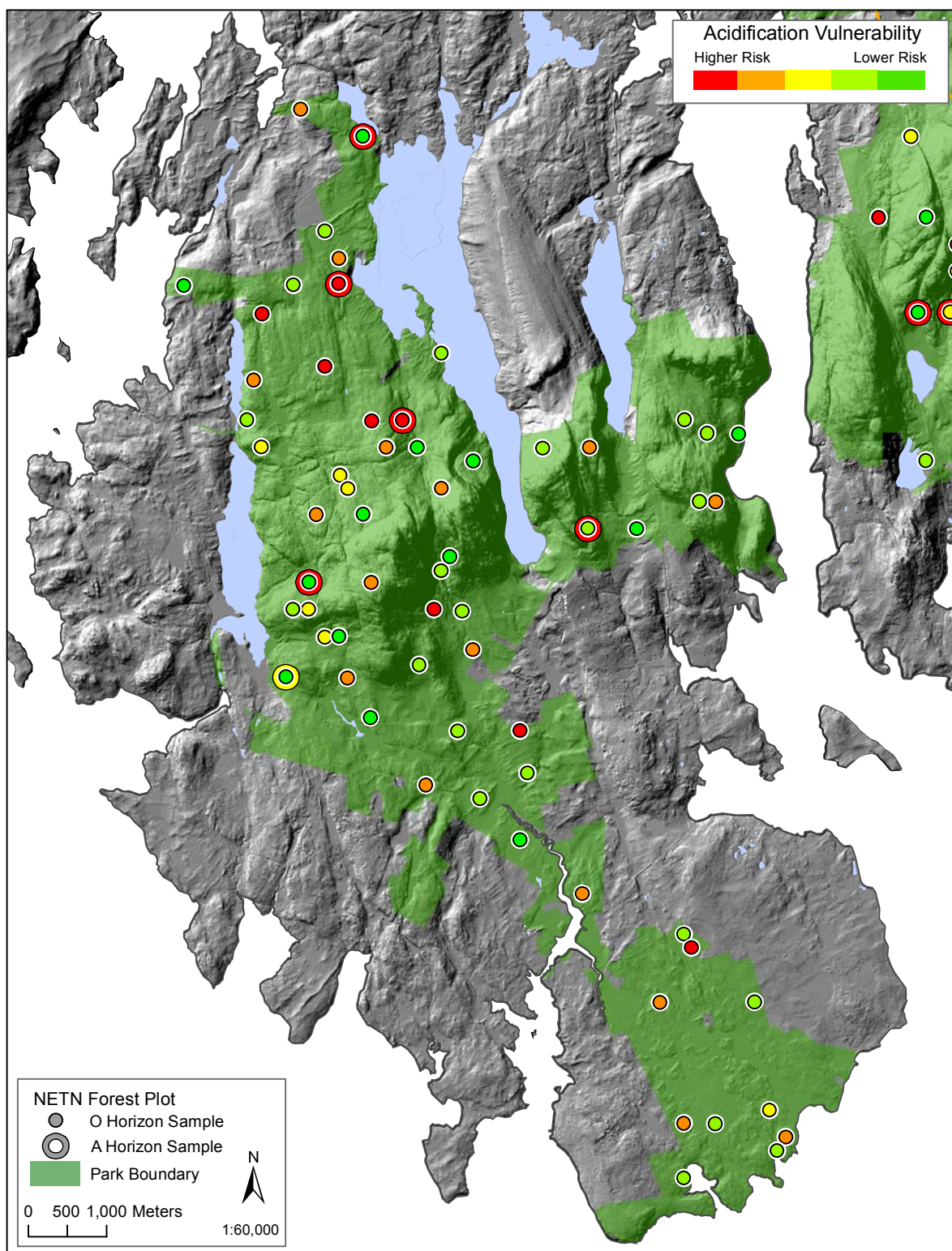


Figure 24. Acidification vulnerability of O and A horizons by forest plot on western Mount Desert Island.

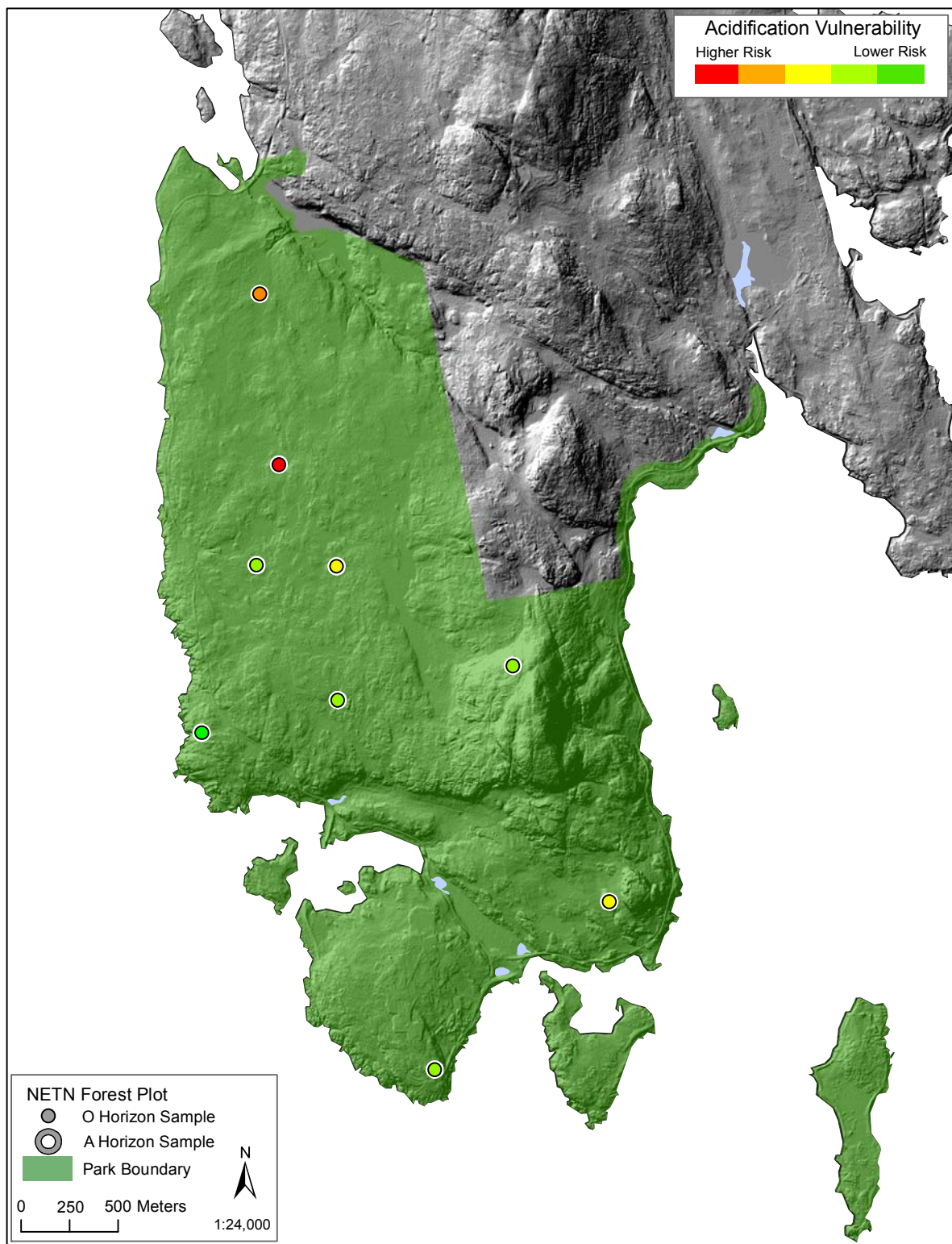


Figure 25. Acidification vulnerability of O horizon by forest plot on Schoodic Peninsula.

Conclusions

NETN is still in the early stages of forest soil monitoring, and our analyses are limited by a lack of temporal data. After multiple rounds of plot surveys are completed, it will be important to compare tree growth and mortality rates to our assessments of acidification vulnerability and risk of N saturation to validate and refine our conclusions. It will also be useful to examine potential relationships between atmospheric deposition and indicators of forest health, and determine how these variables respond to changes in deposition rates and climate. Based on current analyses, forest soils in ACAD are considerably more vulnerable to soil acidification than remaining parks in NETN and are robust to N saturation.

Tree Ring Analyses

Background

Knowing the stand ages of forest plots is an important piece of information that is often lacking in NETN parks. While tree size (diameter at breast height) is often used as a surrogate for tree age, including for our structural stage ecological integrity metric, tree size is also highly influenced by site condition, position in the canopy, past disturbances, and species. It is therefore difficult to know how well tree size can predict tree age, particularly in areas like ACAD where tree growth rates can be very slow on poor sites.

On eastern Mount Desert Island (MDI), we can assume that most forests within the boundary of the 1947 fire originated after the fire and are around 60 years old. While we know that the vast majority of forests in ACAD are second-growth, little else is known about stand age in the rest of the park. Without historic information or aerial photos, the only way to determine the age of a forest stand is to core trees and count the annual rings in the cores.

In 2013, NETN began coring trees in ACAD to better understand age dynamics of forest stands in the park, and to see how age is correlated with indicators of forest structure, composition and function. While the ultimate goal is to have a stand age for every forest plot in NETN, limited resources require us to implement this for a small number of plots at a time. In 2013, we collected cores for all of the plots we sampled on western MDI (n=16 plots). We will continue to collect cores from western MDI plots until all plots in this section have been sampled (n=65), and will expand this project to other units in ACAD and other NETN parks as resources allow.

The objectives of this work are to estimate the stand age for each forest plot on western MDI and determine the relationship between DBH and tree age. We are also interested in comparing stand ages with ecological integrity metrics that are expected to improve with older forests, including structural stage distribution, snag abundance and coarse woody debris (CWD).

Methods

Collecting and Processing Cores

At each plot, two co-dominant and two intermediate trees were cored outside of the plot, and within 10 m of the plot boundary. Trees were selected for coring that were representative (e.g., same species and similar size) of trees on the plot. Trees that were leaning or that had unusual swellings or wounds were not selected for coring.

At each tree, we recorded the tree species, crown class (co-dominant or intermediate), and measured the diameter at breast height (DBH). Tree cores were collected using increment borers that were located near DBH. This approach will not give the exact tree age, but rather the age at which the tree reached DBH.

Tree cores were stored in slotted plastic trays. After cores were collected, they were air dried for several days. Each core was then sanded with 200 grit sandpaper and cleaned with a moist paper towel. Tree rings on cores were counted using a 10x dissecting microscope. In situations where the core did not include the pith, we used a pith locator, designed by Applequist (1958), to approximate the number of missing rings to the pith. Tree cores were aged by at least two

observers for quality control/quality assurance. If ages differed by more than 5 years between two observers, cores were discarded from the analysis.

Data Analysis

To analyze the relationship between DBH and age, while accounting for multiple cores per plot and tree crown class, we used linear mixed effects models in the nlme package in R (Pinheiro et al. 2012, R Development Core Team 2011). DBH and crown class were fixed effects, and plot was treated as a random intercept.

Results and Discussion

The results of the mixed effects model indicate a significant positive relationship between tree DBH and age at DBH, but only for co-dominant trees ($p = 0.0005$, $df = 36$; Figure 20). Therefore, the larger the DBH of a co-dominant tree, the older it is likely to be. There is a lot of variability in the co-dominant data. For example, trees around 25 cm DBH range from 53 to 118 years old. This suggests that while there is a significant positive relationship, the DBH of a co-dominant tree is not the most accurate predictor of tree age and successional stage. As the model indicated, the relationship is less clear for intermediate trees. Intermediate trees in our sample only range from 10 to 25 cm DBH, and ages range from 35 to 126 years. With so much variation in age, including two red spruce trees of roughly the same size (16.5 cm) ranging from 35 to 113 years on different plots, tree DBH is clearly not a reliable predictor of age for intermediate trees.

Comparisons between co-dominant and intermediate tree ages within the same plot reveal interesting patterns. On plots with co-dominant trees less than 80 years old, intermediate trees tend to also be similar in age to the co-dominants, suggesting that canopy trees all originated around the same time period (Table 19).

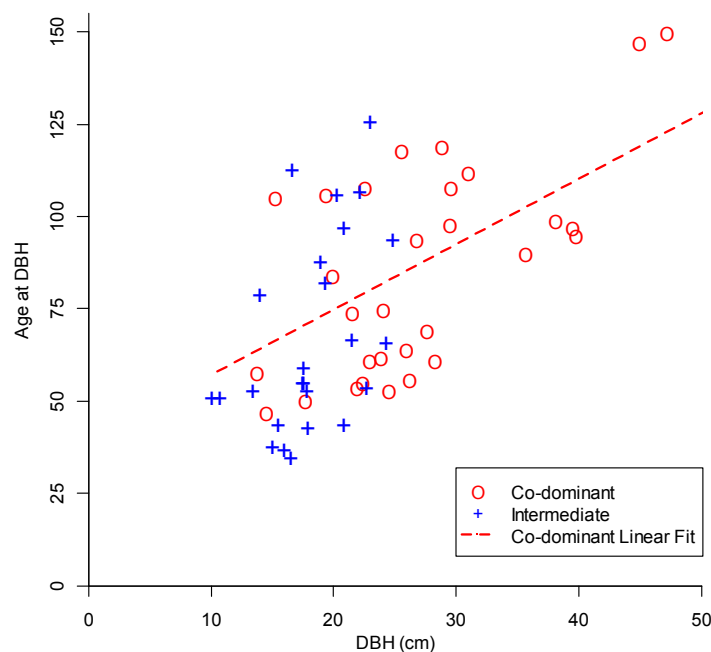


Figure 26. Scatterplot of tree DBH versus age at DBH by tree crown class. Linear fit is based on a linear mixed effects model.

This follows patterns expected for mature forests, which are generally composed of a single-aged cohort of trees. Co-dominant trees that are over 80 years old at DBH tend to be older than intermediate trees, indicating multiple events of canopy recruitment, and is suggestive of later successional forest.

In all but the four youngest plots, the approximate stand origin, based on the average age of co-dominant trees at DBH, is dated prior to when the parcel containing that plot was conveyed to the park (Table 19). The four youngest plots show as much as a 36-year lag between when the parcel was conveyed, and when cored trees first reached DBH. While the underlying cause is not certain, the time required for seedlings to establish and then reach DBH may partially explain the lag. Prior land use (e.g., grazing) may also be a factor. It will be interesting to see if these plots are isolated incidents, or if there are more plots like this in ACAD. Clues about the state of each parcel prior to becoming park may also be found in the historic deeds, and we may investigate this as time and resources allow.

Stand ages derived from tree cores mostly support the stage designated by our structural stage ecological integrity metric, although the ecological integrity metric may be a bit conservative about classifying late-successional plots (Table 20). The structural stage metric uses relative basal area of trees in pole (DBH: 10-19.9 cm), mature (DBH: 20-34.9cm) and large (DBH $\geq$ 35cm) size categories. Based on Davis 1966, the mean age for mature stands is 82 years, and the mean age for late-successional stands is 112 years. For plots with co-dominant trees over 80 years old and younger intermediate trees (e.g., plots 135 and 159), it may be more appropriate to designate these plots as late-succession instead of mature.

A few plots were completely misclassified by the structural stage metric. For example, plot 156, which was designated as late-succession by the structural stage metric, was determined to be 56 years old based on tree cores. Plot 154, which had tree cores dating over 100 years, was rated as pole by our ecological integrity metric. As more plots have stand ages, it will be important to adjust the structural stage metric, so that plots are correctly classified more consistently. Other metrics, such as stand height, DBH distributions, and overall basal area may help refine the structural stage metric.

Late-successional structure varied greatly across the plots, and there were no clear patterns with stand age. For example, we did not detect CWD on plot 143, which had the highest stand age of 149 (multiple CWD pieces were present on the plot, but they did not fall along the CWD transects). It is important to note that the ecologically important forest features of snags and coarse woody debris are pretty rare features in the landscape, even in older forests, and we need to assess across multiple plots, rather than individual plots to understand patterns of late-successional structure and stand age. With stand ages for 16 plots, and only a few plots over 100 years, we cannot form conclusions until older plots are better represented in the dataset.

Conclusions

Results from the first year of coring revealed valuable information about forest stands on western MDI. Coring more plots on western MDI will continue to expand our understanding of forest age dynamics on western MDI, and will help refine our ecological integrity metrics. As time permits, it will also be important to expand this project to all units of ACAD to increase sample size and examine if patterns hold up across the park.

Table 19. Co-dominant and Intermediate tree DBH and age at DBH data. Stand origin was calculated from the average age of co-dominant trees on each plot.

Plot	Co-dominant Trees		Intermediate Trees		Stand Origin	Year Conveyed
	DBH	Age at DBH	DBH	Age at DBH		
135	33.3	101	19.0	72	1912	1930
136	22.8	75	21.5	67	1938	1953
137	26.4	57	17.5	55	1956	1937
138	23.4	62	16.5	52	1952	1958
139	26.3	79	14.0	79	1934	1937
140	25.5	118	23.0	126	1895	1937
141	29.5	108	22.3	86	1905	1956
143	46.0	149	18.9	88	1865	1924
147	29.0	92	19.4	44	1922	1936
154	17.3	106	22.1	88	1908	1936
155	14.1	47	10.4	51	1966	1930
156	24.2	56	15.8	37	1958	1937
159	39.6	96	20.1	55	1917	1958
160	19.7	52	15.6	53	1961	1939
162	26.0	103	N/A	N/A	1910	1937
168	28.2	94	18.7	105	1919	1938

Table 20. Average co-dominant tree age for each plot compared with ecological integrity metrics. Plot 155 is a woodland, and not rated for structural stage or CWD metric. M-L stands for medium to large, and refers to trees ≥ 30 cm DBH. Late-successional is abbreviated as Late Succ. Significant Concern is abbreviated as Sign. Conc., and is not rated at the plot level for snag abundance.

Plot	Age	Structural Stage	CWD Volume	CWD Ratio	CWD Rating	Snags/ha	M-L Snags/ha	M-L Live/ha
155	47	N/A	0.00	N/A	N/A	0.00	0.00	0.00
160	52	Pole	28.02	0.10	Caution	133.33	0.00	0.00
156	56	Late Succ.	30.07	0.17	Good	266.66	0.00	88.89
137	57	Mature	12.01	0.05	Sign. Conc.	400.00	44.44	44.44
138	62	Mature	10.65	0.37	Sign. Conc.	311.11	0.00	44.44
136	75	Mature	5.93	0.04	Sign. Conc.	222.22	0.00	0.00
139	79	Mature	91.96	0.35	Good	133.33	0.00	177.78
147	92	Mature	40.14	0.28	Good	444.44	133.33	44.44
168	94	Mature	29.92	0.28	Good	311.11	0.00	0.00
159	96	Mature	70.59	0.24	Good	88.89	44.44	133.33
135	101	Mature	120.45	0.53	Good	311.11	44.44	177.78
162	103	Mature	33.42	0.12	Caution	577.77	0.00	0.00
154	106	Pole	0.00	0.00	Sign. Conc.	266.66	0.00	0.00
141	108	Mature	50.04	0.17	Good	133.33	0.00	133.33
140	118	Late Succ.	35.46	0.17	Good	177.78	44.44	133.33
143	149	Late Succ.	0.00	0.00	Sign. Conc.	222.22	0.00	222.22

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Appendix A. Landscape Context maps for NETN parks.

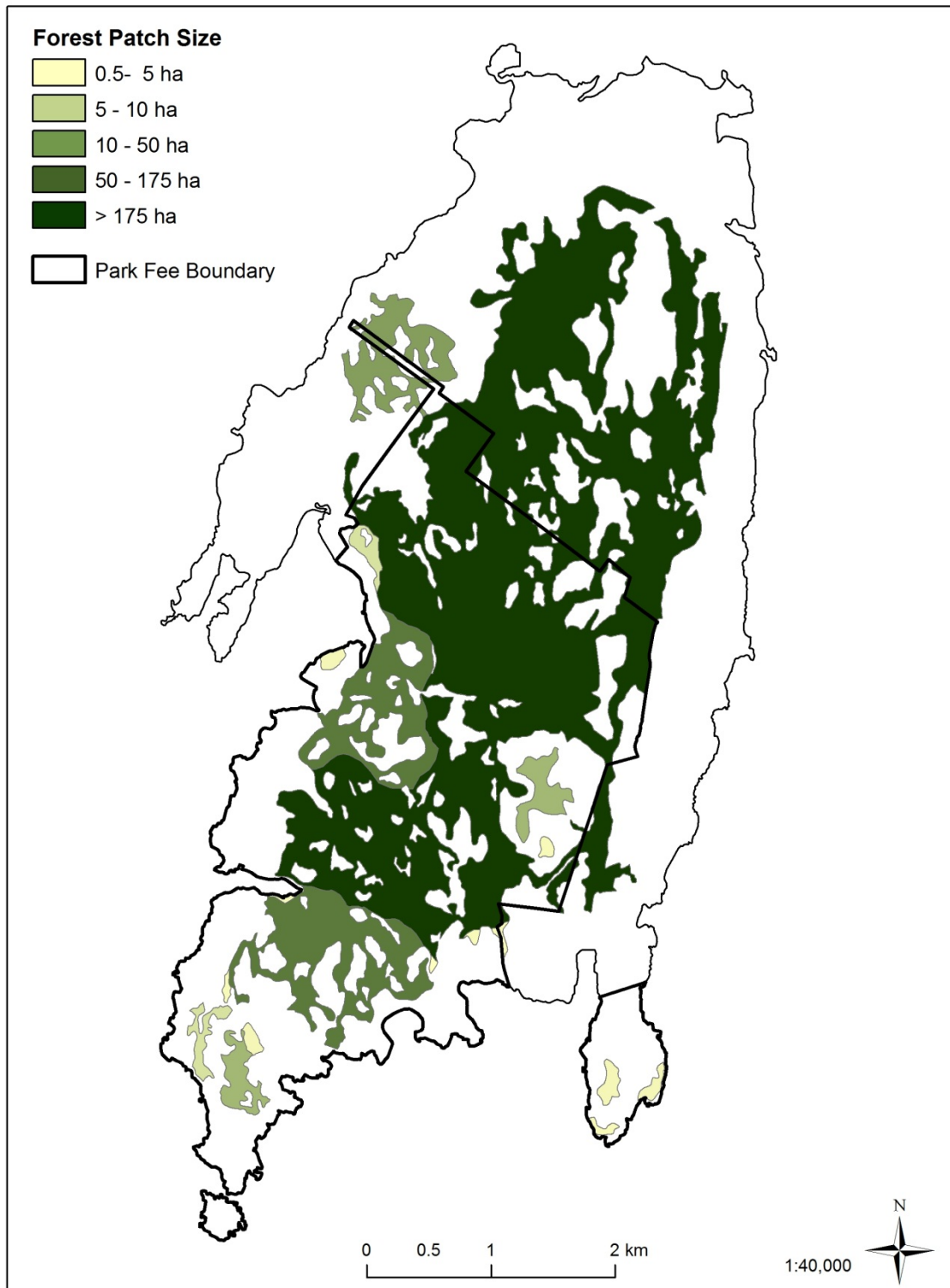


Figure A.1. Forest patches delineated on Isle au Haut in Acadia National Park.

Appendix A. Landscape Context maps for NETN parks (continued).

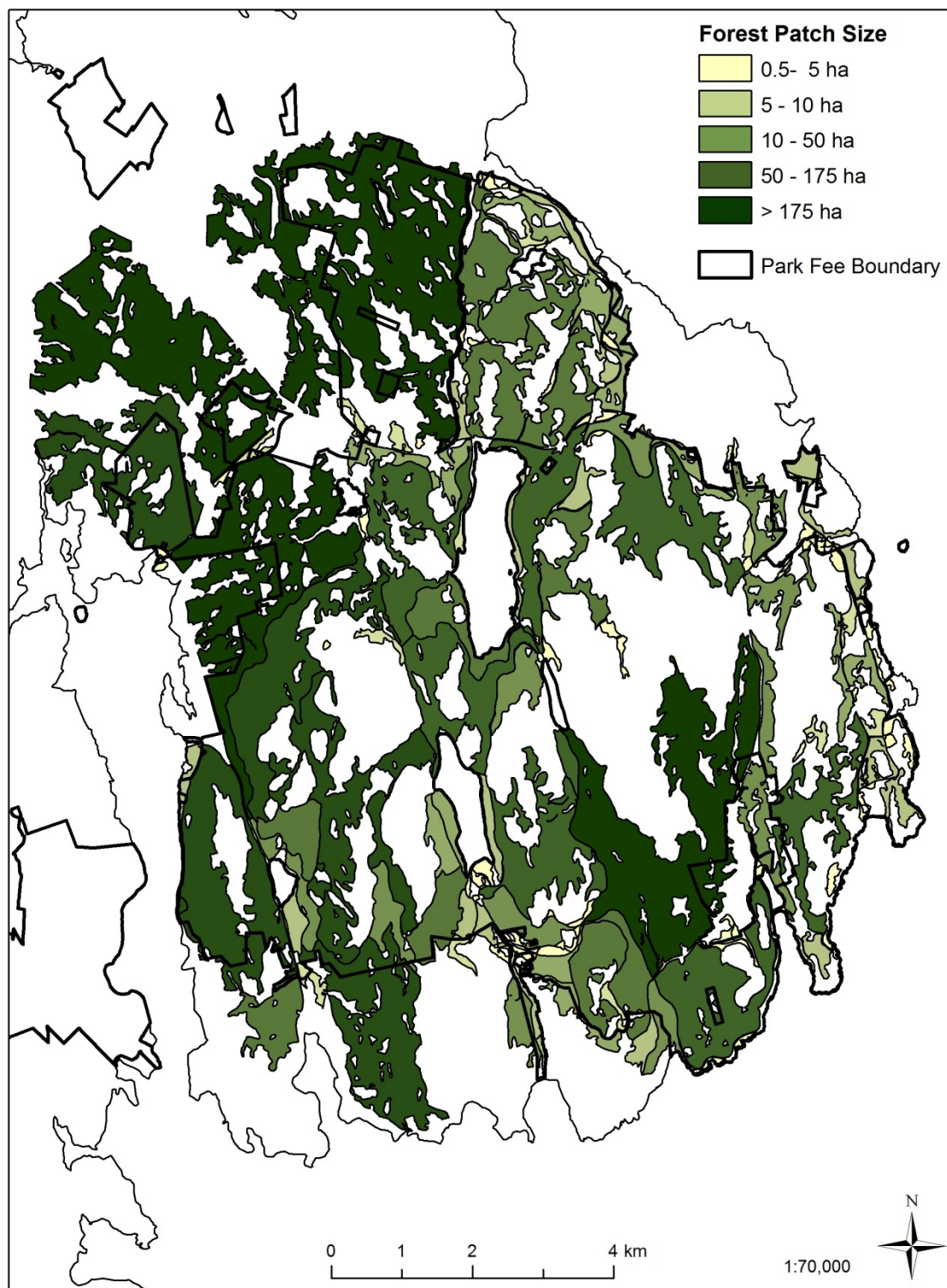


Figure A.2. Forest patches delineated for the eastern section of Mount Desert Island in Acadia National Park.

Appendix A. Landscape Context maps for NETN parks (continued).

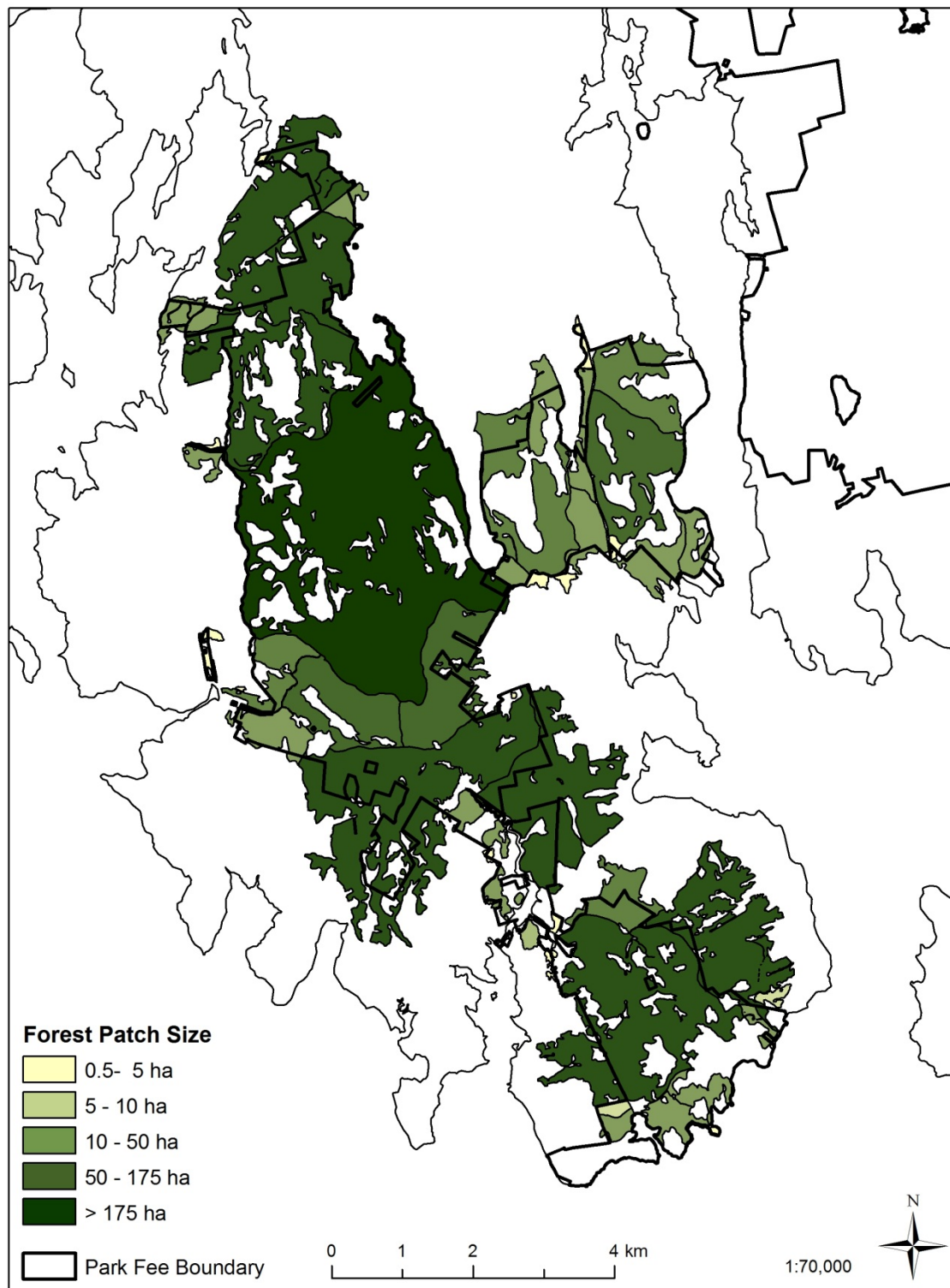


Figure A.3. Forest patches delineated for the western section of Mount Desert Island in Acadia National Park.

Appendix A. Landscape Context maps for NETN parks (continued).

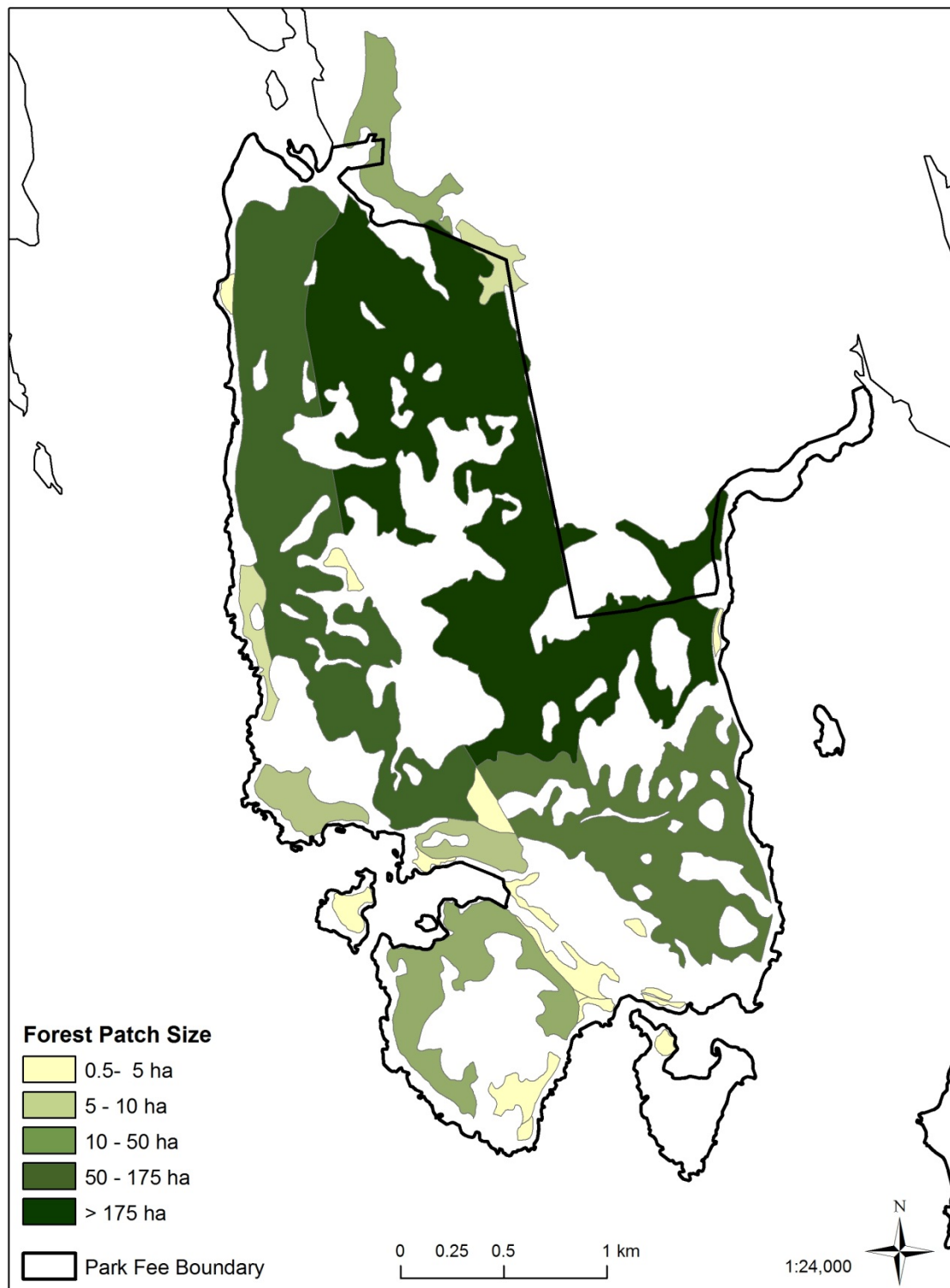


Figure A.4. Forest patches delineated for Schoodic Peninsula in Acadia National Park

Appendix B. Key indicator invasive exotic plant species of forest, woodland and successional ecosystems in the northeastern U.S. that were used in the NETN Ecological Integrity Scorecard.

Latin name	Common name	Habitat
<i>Acer platanoides</i>	Norway maple	Open & forest habitats
<i>Ailanthus altissima</i>	tree-of-heaven	Edges & successional habitats
<i>Alliaria petiolata</i>	garlic mustard	Open & forest habitats
<i>Berberis thunbergii</i>	Japanese barberry	Open & forest habitats
<i>Berberis vulgaris</i>	European barberry	Open & wooded habitats
<i>Cardamine impatiens</i>	narrowleaf bittercress	Open & wooded habitats
<i>Celastrus orbiculatus</i>	oriental bittersweet	Edges & successional habitats
<i>Cynanchum louiseae</i>	black swallow-wort	Open, successional & wooded habitats
<i>Cynanchum rossicum</i>	European swallow-wort	Open, successional & wooded habitats
<i>Euonymus alata</i>	winged burning bush	Open, successional & wooded habitats
<i>Frangula alnus</i>	glossy buckthorn	Wetland & successional habitats
<i>Ligustrum</i> spp. (<i>obtusifolium</i> , <i>vulgare</i>)	privet	Successional & forest habitats
<i>Lonicera japonica</i>	Japanese honeysuckle	Edges & successional habitats
<i>Lonicera</i> spp. (<i>morrowii</i> , <i>tatarica</i> , x <i>bella</i>)	exotic honeysuckles	Open & successional habitats
<i>Luzula luzuloides</i>	forest woodrush	Open & wooded habitats
<i>Microstegium vimineum</i>	Japanese stiltgrass	Open & forest habitats
<i>Polygonum caespitosum</i>	oriental ladythumb	Open & successional habitats
<i>Polygonum cuspidatum</i>	Japanese knotweed	Riparian, open & successional habitats
<i>Rhamnus cathartica</i>	common buckthorn	Riparian, open & successional habitats
<i>Rhodotypos scandens</i>	black jetbead	Open, successional & wooded habitats
<i>Rosa multiflora</i>	multiflora rose	Open & successional habitats
<i>Rubus phoenicolasius</i>	wineberry	Open, successional & wooded habitats

Appendix C. Herbaceous indicator species of deer browse pressure in northeastern U.S. forests.

Latin name	Common Name	Deer Preference	Citations
<i>Ageratina altissima</i> var. <i>altissima</i> (formerly <i>Eupatorium rugosum</i>)	white snakeroot	Avoided	Augustine and Jordan 1998
<i>Aralia nudicaulis</i>	wild sarsaparilla	Browsed	Balگوoyen and Waller 1995
<i>Eurybia divaricata</i>	white wood aster	Browsed	Williams et al. 2000
<i>Carex</i> spp.	sedge	Avoided	Williams et al. 2000, Augustine and deCalesta 2003, Horsley et al. 2003
<i>Clintonia borealis</i>	blue bead lily	Browsed	Balگوoyen and Waller 1995
<i>Dennstaedtia punctilobula</i>	hayscented fern	Avoided	Horsley et al. 2003
<i>Maianthemum (Smilacina)</i> spp. (<i>racemosum</i> , <i>stellatum</i>)	Canada mayflower and false Solomon's seal	Browsed	Fletcher et al. 2001, Augustine and deCalesta 2003
<i>Polygonatum</i> spp. (<i>biflorum</i> , <i>pubescens</i>)	smooth Solomon's seal	Browsed	Augustine and deCalesta 2003, Webster et al. 2005, Kraft et al. 2004
<i>Sanguinaria canadensis</i>	bloodroot	Browsed	Augustine and deCalesta 2003
<i>Thelypteris noveboracensis</i>	New York fern	Avoided	Williams et al. 2000, Horsley et al. 2003
<i>Trillium</i> spp. (<i>cernum</i> , <i>erectum</i> , <i>grandiflorum</i> , <i>undulatum</i>)	trillium	Browsed	Anderson 1994, Augustine and Frelich 1998, Rooney 2001, Augustine and deCalesta 2003, Webster et al. 2005
<i>Uvularia</i> spp. (<i>grandiflora</i> , <i>perfoliata</i> , <i>sessifolia</i>)	bellwort	Browsed	Fletcher et al. 2001, Augustine and deCalesta 2003, Webster et al. 2005

Appendix D. Soil chemistry boxplots of ACAD park units.

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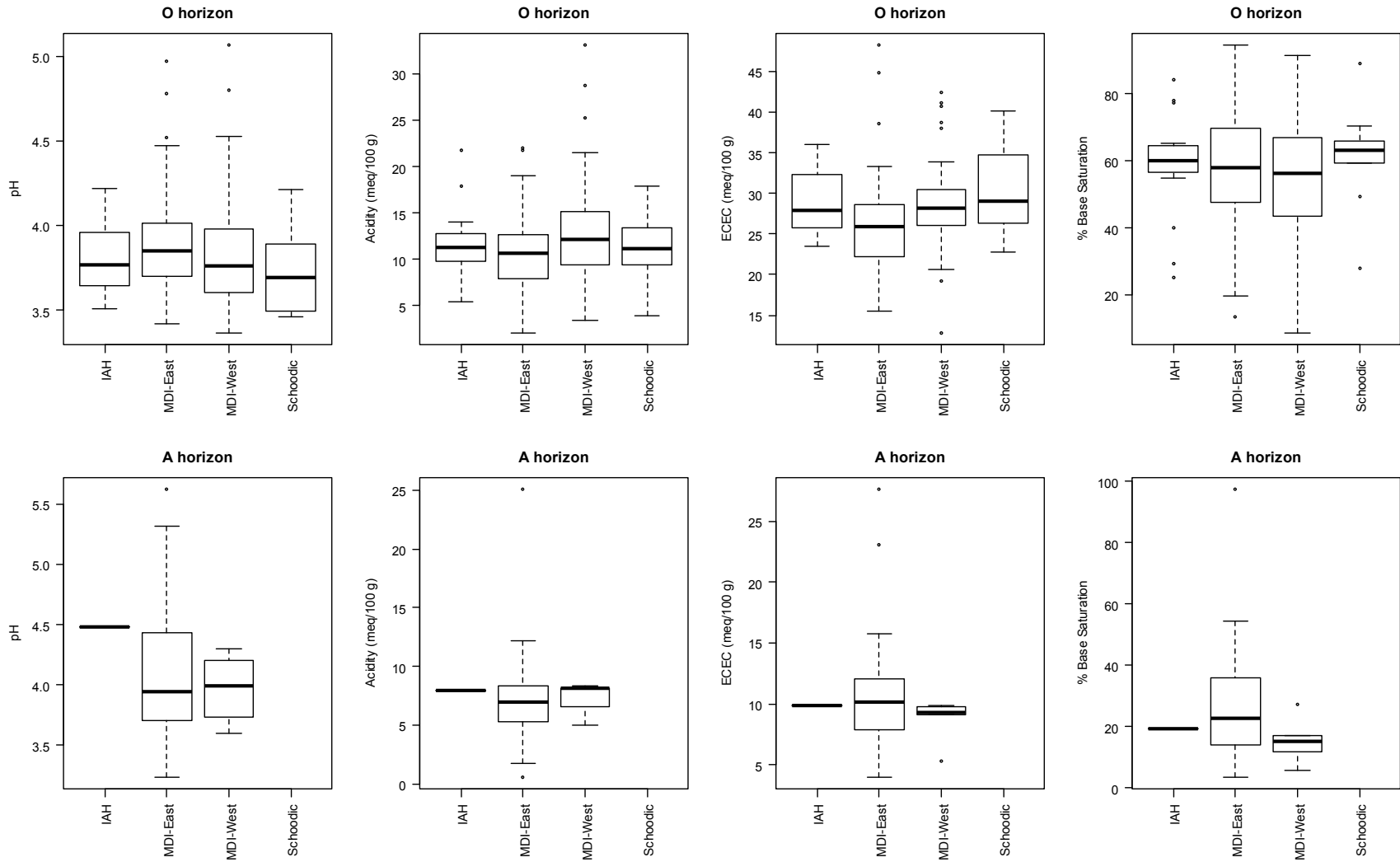


Figure D.1. Soil parameters for each park unit by horizon for samples collected from 2009 to 2012.

Appendix D Soil chemistry boxplots of ACAD park units (continued).

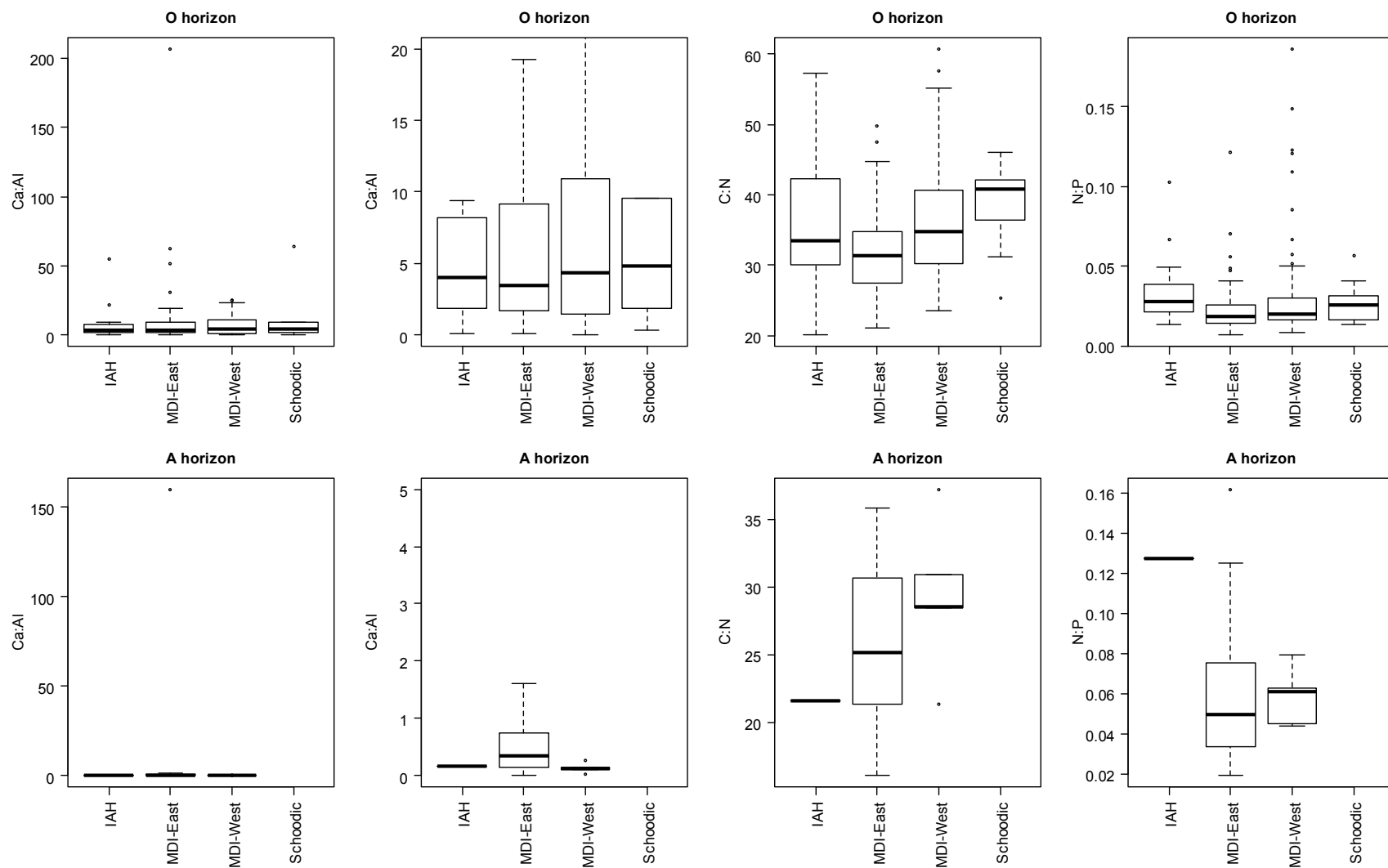


Figure D.2. Ca:Al, C:N, and N:P ratios for each park unit by horizon for samples collected from 2009 to 2012. The second column of Ca:Al shows the lower range of ratio values for the parks.

Appendix D. Soil chemistry boxplots of ACAD park units (continued).

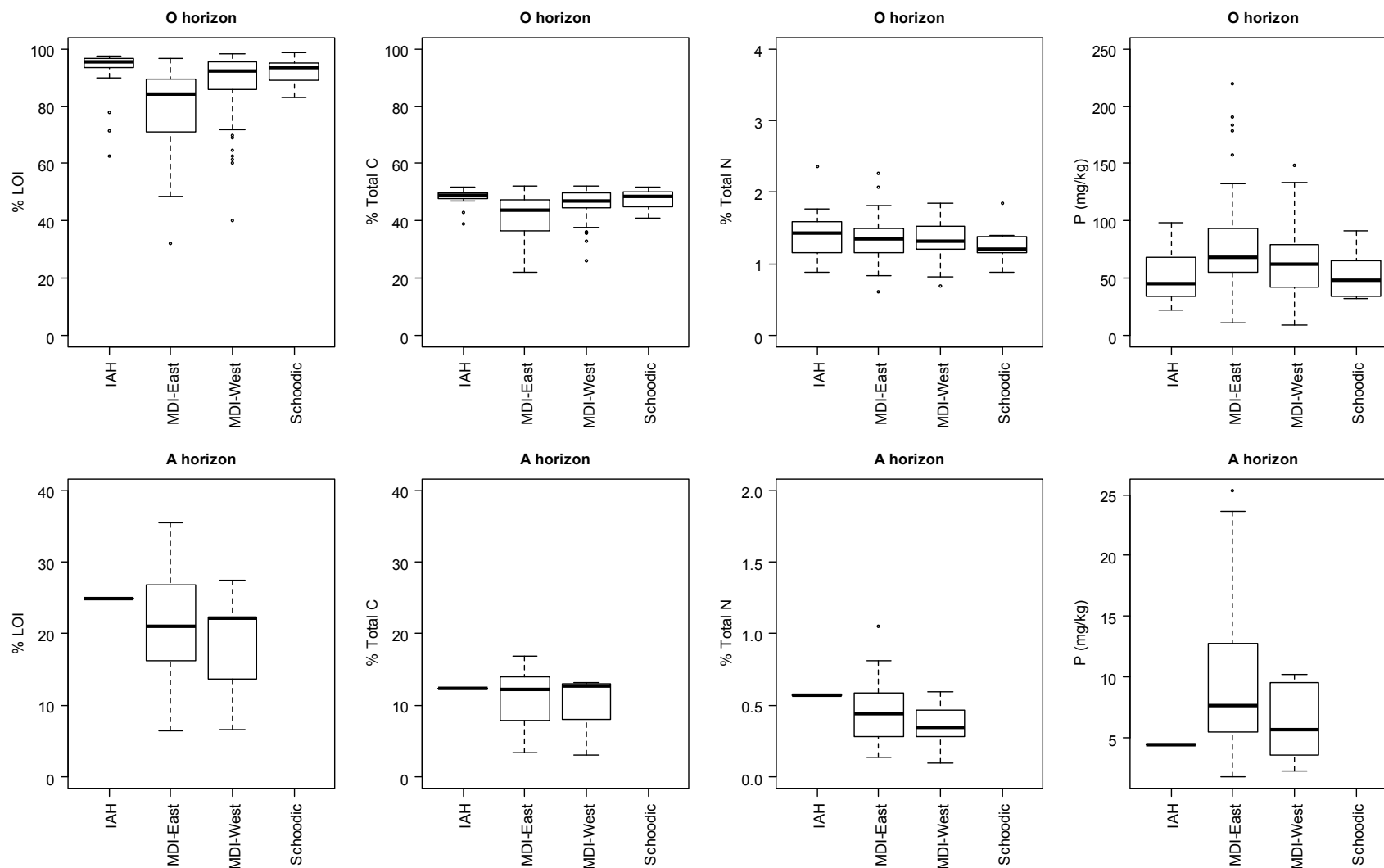


Figure D.3. C, N and P parameters for each park unit by horizon for samples collected from 2009 to 2012.

Appendix D. Soil chemistry boxplots of ACAD park units (continued).

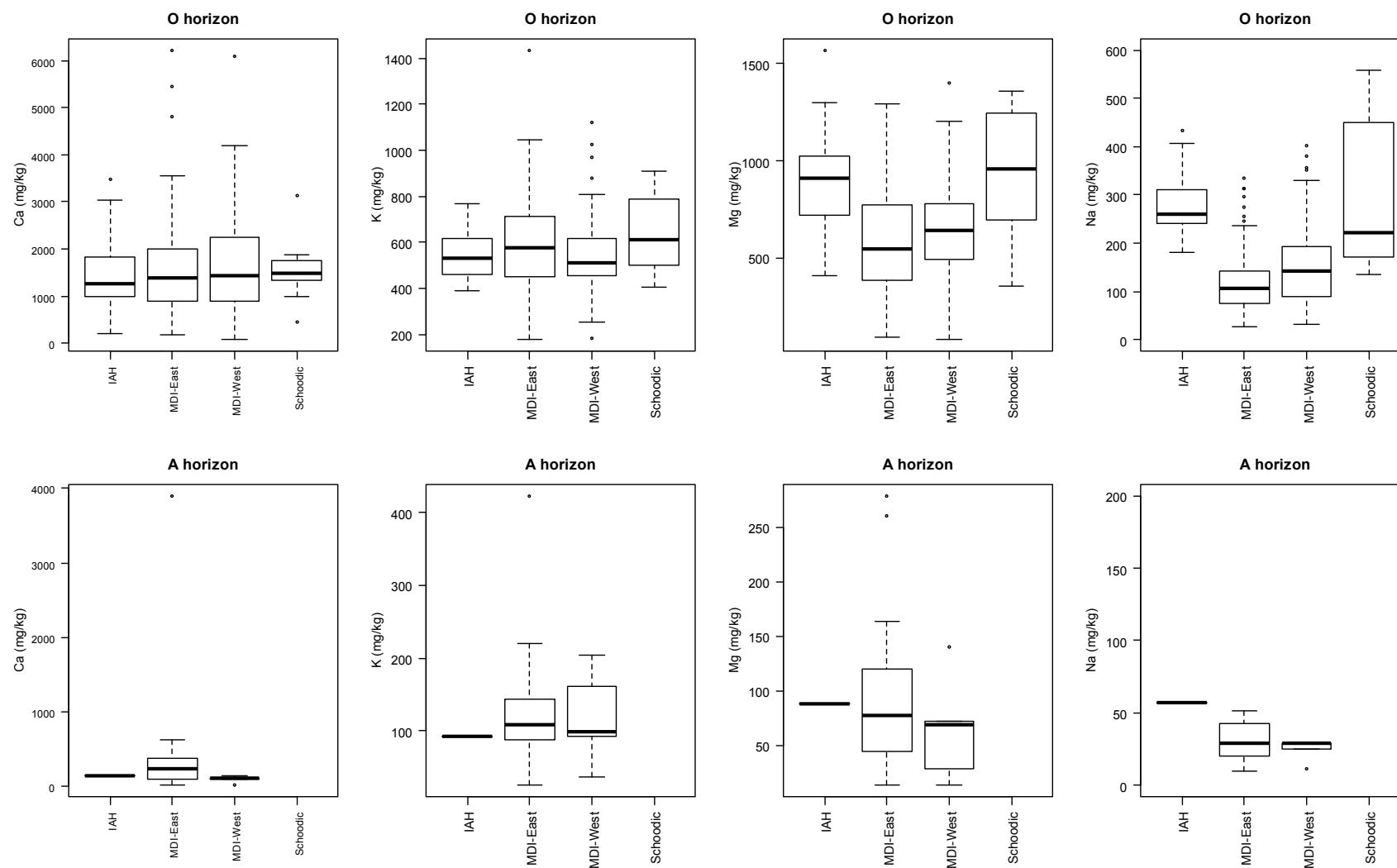


Figure D.4. Base cations for each park unit by horizon for samples collected from 2009 to 2012.

Appendix D. Soil chemistry boxplots of ACAD park units (continued).

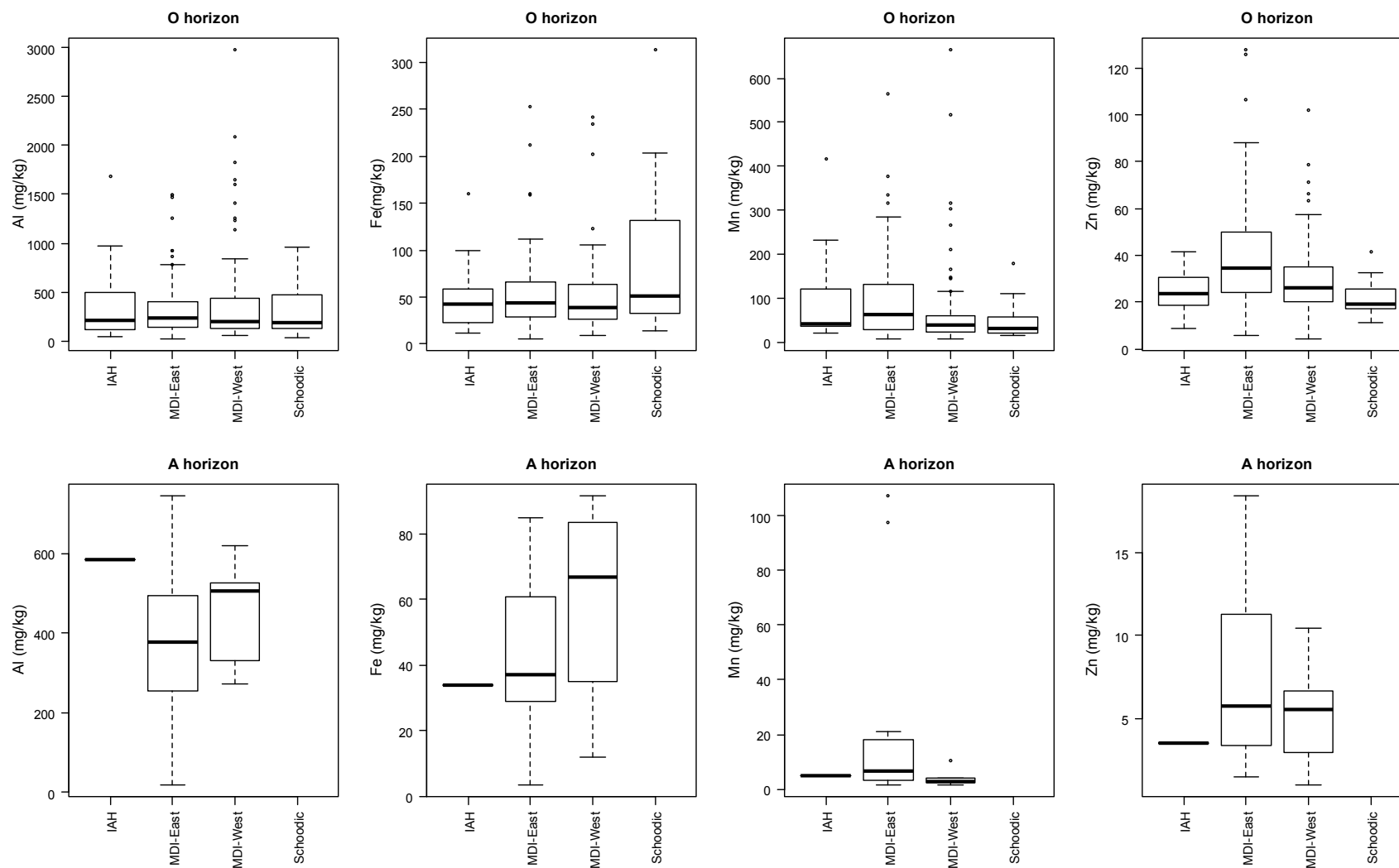


Figure D.5. Acid cations for each park unit by horizon for samples collected from 2009 to 2012.

Appendix D. Soil chemistry boxplots of ACAD park units (continued).

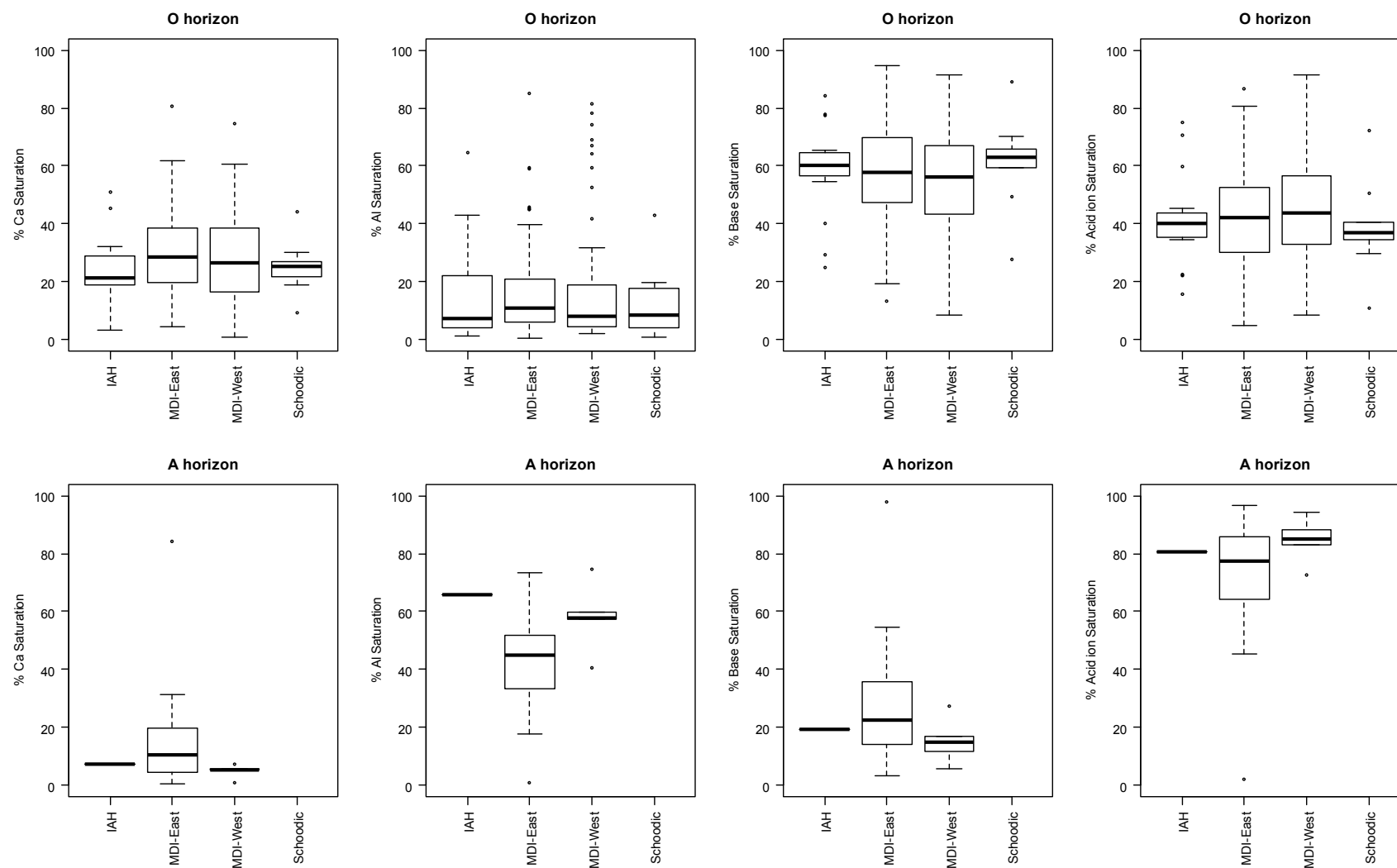


Figure D.6. Percent saturation metrics for each park unit by horizon for samples collected from 2009 to 2012. Acid ions include Al, Fe, Mn and Zn.

Appendix E. Supplemental ordination plots and figures used to interpret A horizon NMDS results.

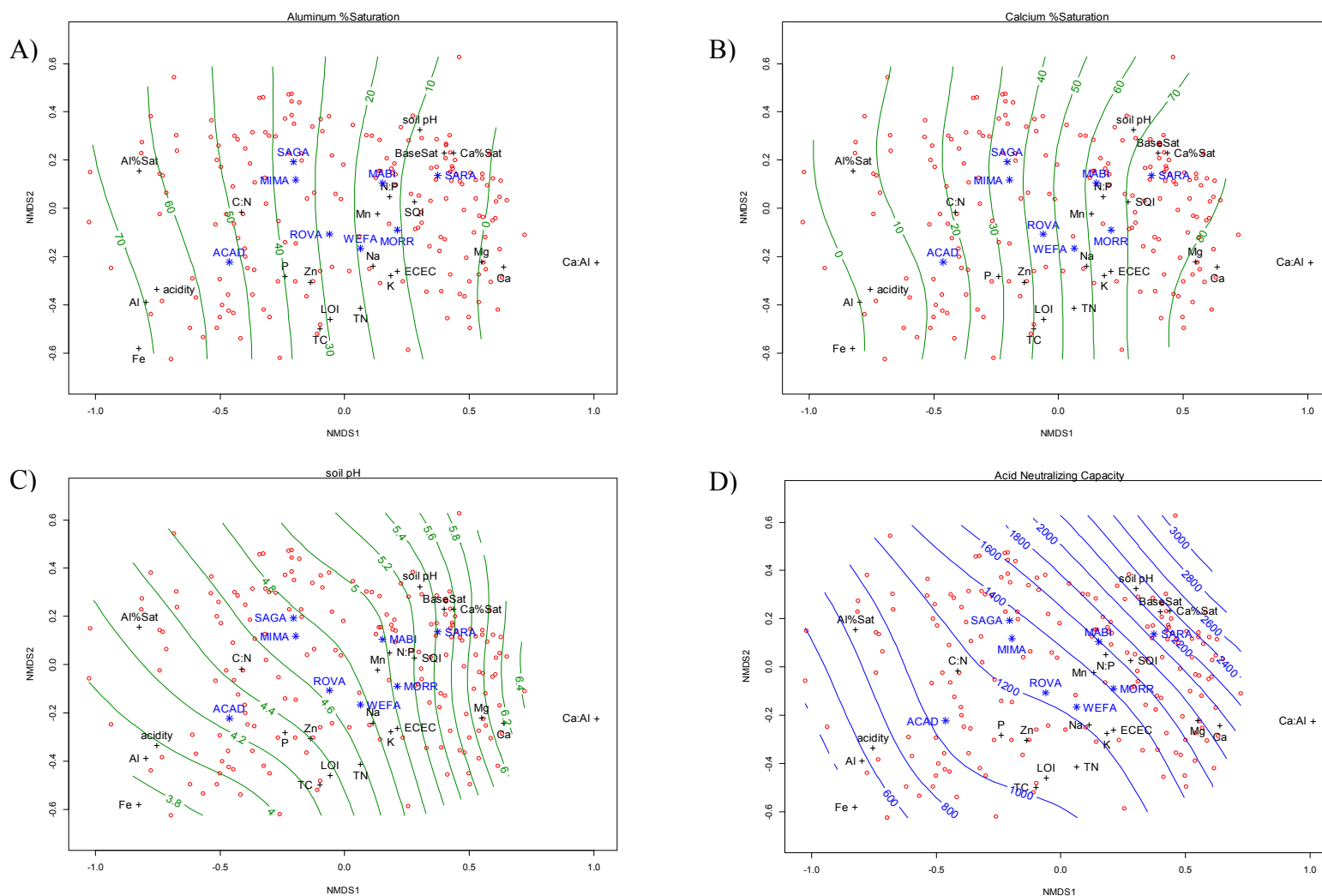


Figure E.1. Surface fitting contours for A) Al % Saturation, B) Ca % Saturation, C) soil pH, and D) water Acid Neutralizing Capacity (meq/L). Green contours are variables that were part of the soil chemical dataset for the NMDS ordination, and are expressed in their original units. Blue contours are environmental variables that did not influence the NMDS ordination.

Appendix E. Supplemental ordination plots and figures used to interpret A horizon NMDS results (continued).

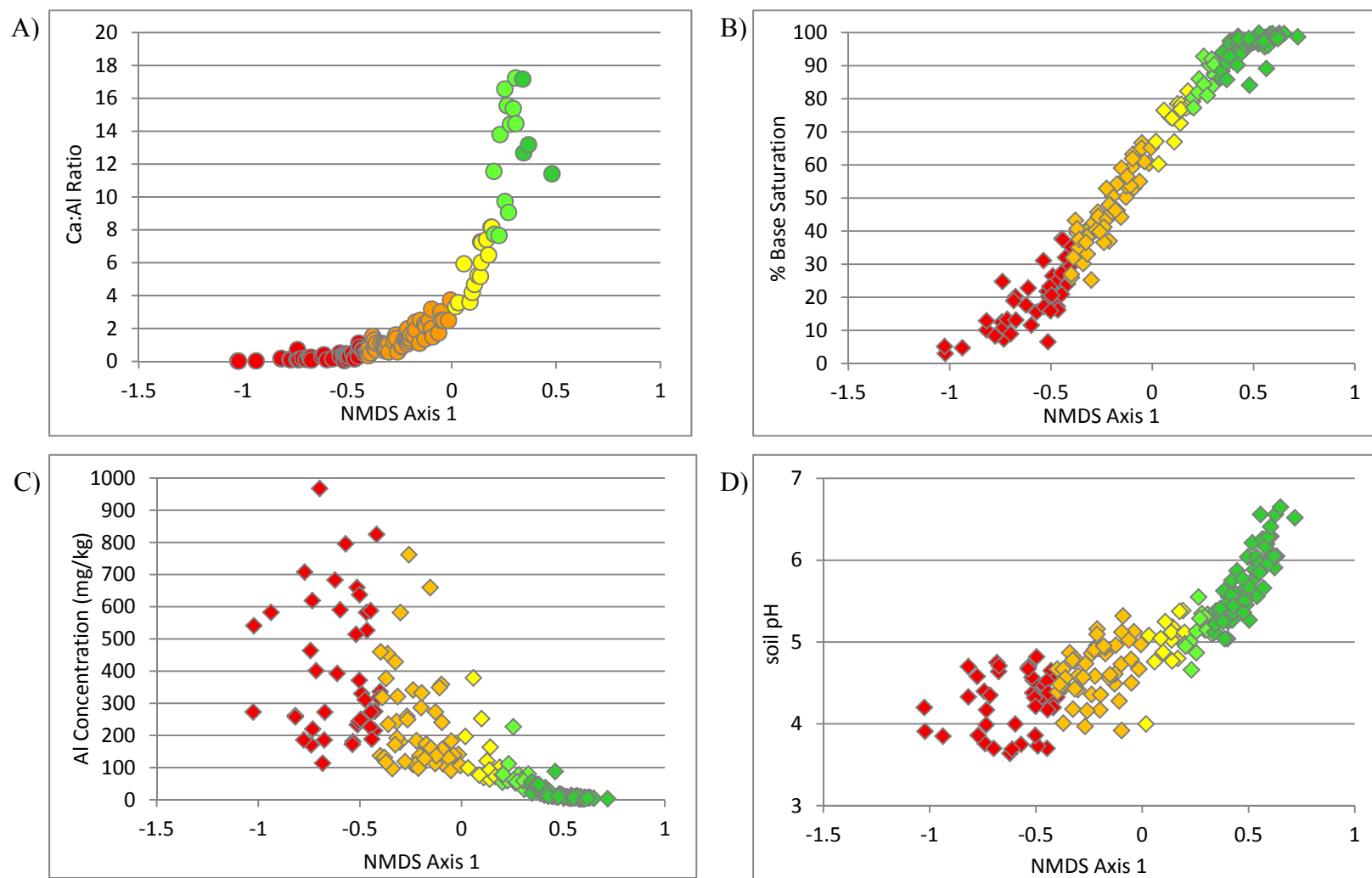


Figure E.2. Scatterplots of NMDS Axis 1 scores versus A) Ca:Al ratio, B) % Base Saturation, C) Al concentration (mg/kg), and D) soil pH. Point colors represent the bins used to assign relative acidification risk of plots in ACAD based on O horizon samples. Note that the Y axis in Figure A was cut off at 20 to view patterns in the lower range of this ratio.

Appendix E. Supplemental ordination plots and figures used to interpret A horizon NMDS results (continued).

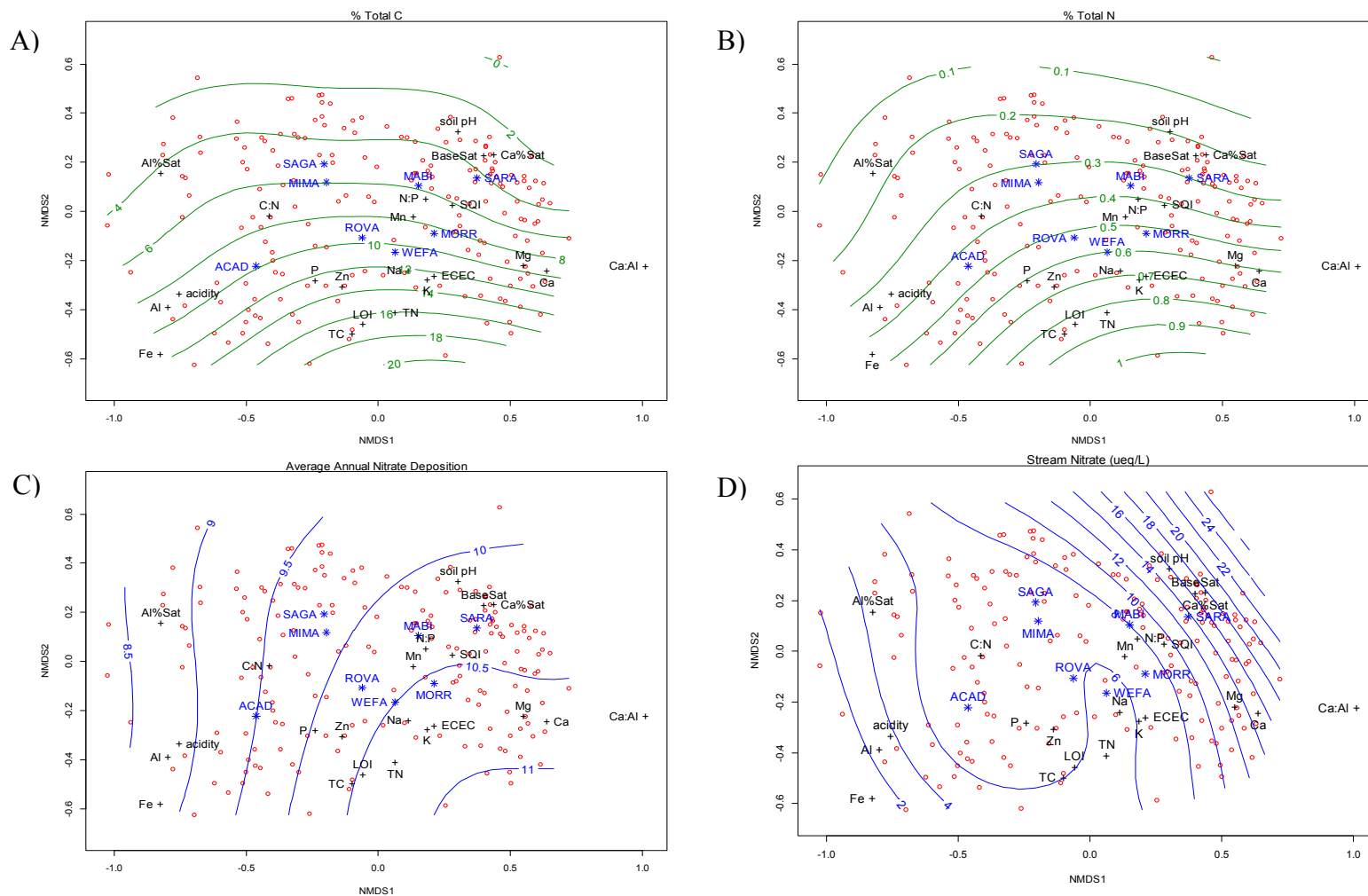


Figure E.3. Surface fitting contours for A) % Total C, B) % Total N, C) Average Annual N deposition, and D) Stream NO_3^- concentration ($\mu\text{eq/L}$). Green contours are variables that were part of the soil chemical dataset for the NMDS ordination, and are expressed in their original units. Blue contours are environmental variables that did not influence the NMDS ordination.

Appendix E. Supplemental ordination plots and figures used to interpret A horizon NMDS results (continued).

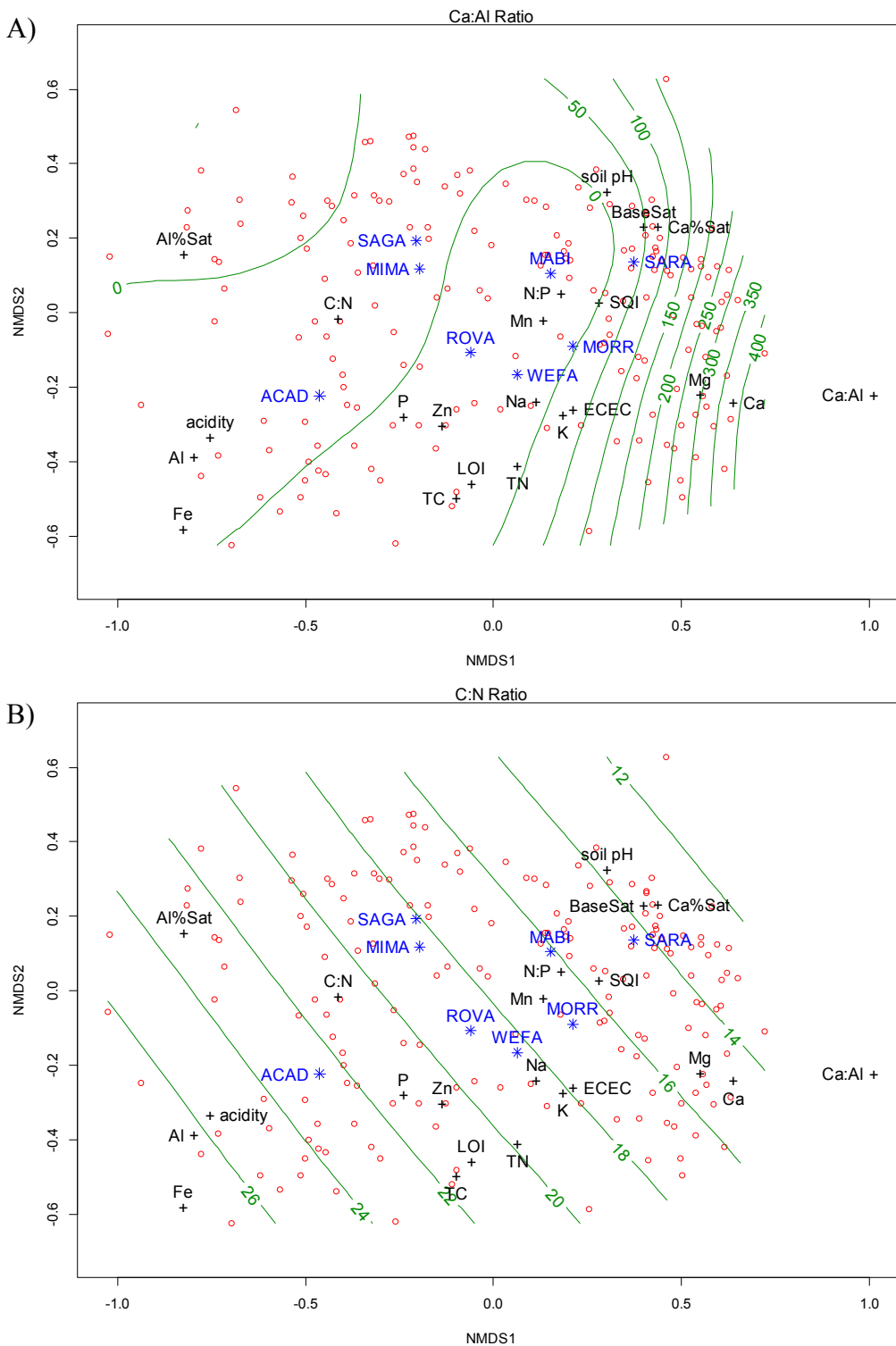


Figure E.4. Surface fitting contours for A) Ca:Al ratio and B) C:N ratio.

Appendix F. Ordination plots and figures used to interpret NMDS results for O horizon analyses in ACAD.

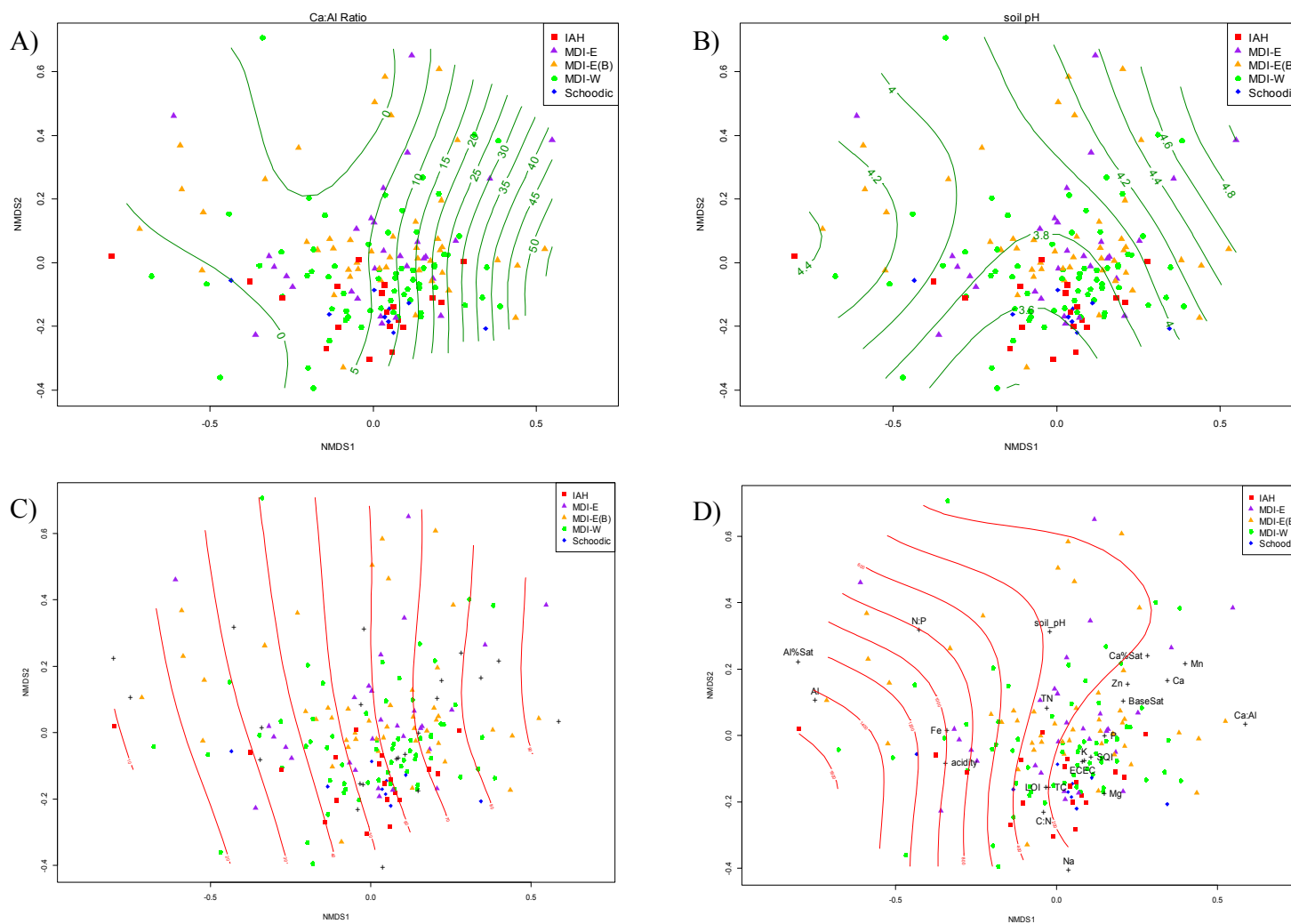


Figure F.1. Surface fitting contours for A) Ca:Al ratio, B) soil pH, C) % Base Saturation, and D) Al concentration (mg/kg). Green contours are variables that were part of the soil chemical dataset for the NMDS ordination, and are expressed in their original units. Red contours represent environmental variables that did not influence the NMDS ordination.

Appendix F. Ordination plots and figures used to interpret NMDS results for O horizon analyses in ACAD (continued).

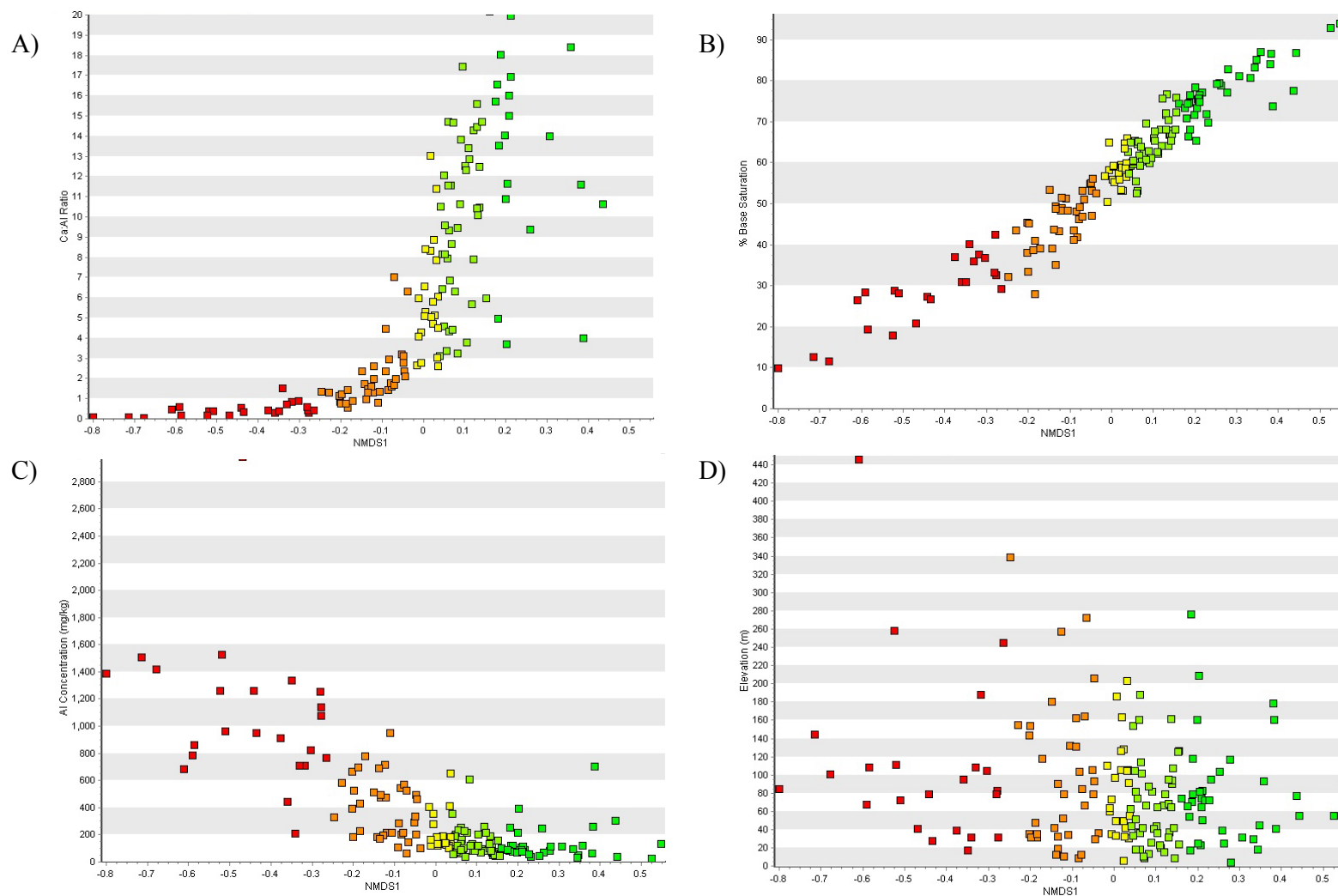


Figure F.2. Scatterplots of NMDS Axis 1 scores versus A) Ca:Al ratio, B) % Base Saturation, C) Al concentration (mg/kg), and D) Elevation. Point colors represent the bins used to assign relative acidification risk of plots in ACAD based on O horizon samples. Note that the Y axis in Figure A was cut off at 20 to view patterns in the lower range of this ratio.

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

NPS 123/123913, February 2014

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