2020

# Wildland Fire Disturbance - Recovery Dynamics in Upland Forests at Acadia National Park, Maine

Jessica E. Charpentier

Department of Environmental Studies

DISSERTATION COMMITTEE PAGE

The undersigned have examined the dissertation titled: Wildland Fire Disturbance - Recovery Dynamics in Upland Forests at Acadia National Park, Maine

presented by Jessica E. Charpentier, candidate for the degree of Doctor of Philosophy in Environmental Studies and hereby certify that it is accepted. *

Committee Chair: Peter A. Palmiotto, D.F.
Chair, Department of Environmental Studies, Antioch University New England

Committee Member: William A. Patterson III, Ph.D.
Emeritus Professor of Forestry, University of Massachusetts

Committee Member: Rachel K. Thiet, Ph.D.
Professor of Environmental Studies, Antioch University New England

Defense Date: December 4, 2019

Date Approved by All Committee Members: December 4, 2019

Date Deposited: May 15, 2020

*Signatures are on file with the Registrar's Office at Antioch University New England.

**Wildland Fire Disturbance - Recovery Dynamics in Upland Forests at**

**Acadia National Park, Maine**

**by**

**Jessica E. Charpentier**

A Dissertation submitted in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in

Environmental Studies

at

Antioch University New England

Keene, New Hampshire

2020

ACKNOWLEDGEMENTS

Thank you to Dr. Peter Palmiotto for his immense insight and support that made this dissertation possible. Thank you to Dr. Bill Patterson for his fire ecology expertise and long-term dataset for which I was able to build upon. Thank you to Dr. Rachel Thiet for her encouragement and guidance throughout the process. Thank you to Dr. Lisabeth Willey for her data analysis assistance. Thank you to the National Park Service for their financial support and participation. Especially thanks to Anthony Davis and Andrew Ruth at the Northeast Region - Fire Management Program who were instrumental in assisting with the research. I am grateful to the Antioch University New England community for the support, including my fellow students, the Environmental Studies Department faculty, and the campus' staff and administration. I would also like to extend my heartfelt appreciation to my family. To my parents, with more appreciation than I can express in words, thank you for a lifetime of support. Thank you to my brother, Jason, my best friend who always gives the perspective of patience and strength. And thank you to my dear friends who recharged my energy and kept me going - especially Jen, Emily, Amy, and Lindsay.

ABSTRACT

The overall goal of this study was to evaluate whether coastal Maine (USA) forests are resilient to changing climate and fire regimes. The occurrence of a catastrophic wildfire at Acadia National Park (ANP) in 1947 provided a unique opportunity to examine the impacts of wildfire on forest dynamics in upland communities of coastal spruce-fir and northern hardwood forests of the Maine coast. This study, conducted 68 years after the stand-replacing 1947 Bar Harbor Fire, builds on studies by W.A. Patterson conducted in 1980 and 1992-1994, 33 and 45-47 years after the fire. There were two lines of investigation in this study: vegetation change following a large-scale, stand-replacing wildfire; and an assessment of wildfire risk following a long period with no major disturbance.

In 2016 I quantified and characterized stand and site characteristics including: basal area and stem density of woody species; aboveground biomass and necromass of trees, saplings, and shrubs; dead downed woody fuel loads; duff depth; fuel height; soil depth to bedrock, and canopy closure for 23 stands throughout ANP. To evaluate long-term trends in post-fire recovery, I compared 2016 forest composition, structure and fuel loading data with that in 1980 and 1992-94. I mapped current wildfire risk to aid managers in identifying where mitigation practices would be most effective in reducing fire risk. I used an ArcGIS model that extends field data of current fuel conditions and spatially portrays wildfire risk across the landscape. Mixed effects models were used to determine the best remotely sensed numeric biomass data as a predictor of biomass and necromass measured on the ground.

Widespread regeneration of red spruce following the initial establishment of aspen and birch suggests that forests of ANP are resilient to wildfire. Stands that did not burn in 1947 remain as mature-to-overmature spruce and fir. Biomass and necromass is continuting to accumulate. Fuel loads are generally high to very high outside the 1947 fire boundary. Within the fire boundary, fuel loads are primarily low to moderate, with small areas of high to very high risk due to topography (e.g., steep versus shallow slopes, north versus south aspect) and unique species composition (e.g., maturing pitch pine/heath communities). After 70 years, replacement of aspen and birch by spruce and fir in many stands suggests potentially increasing wildfire risk within the 1947 fire boundary.

Mount Desert Island has and will continue to experience a marked increase in human development and visitation, thereby increasing the likelihood of human-caused ignitions. This, coupled with increasing fuel loads, may significantly increase the likelihood of wildfire occurrence. An uncertain climate future may exacerbate potential wildfire risk. Should climate warm substantially, spruce-fir stands may break up prematurely – significantly increasing dead, downed fuel for a period of time. Fire management programs should plan to operate strategically and efficiently to meet this challenge.

*Keywords: fire ecology, disturbance ecology, forest succession, resilience*

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# Chapter 1: Introduction

Ecological resilience is a property of ecosystems that relates to their ability to reorganize after a disturbance. Resilient ecosystems can reorganize on their own while undergoing change or disturbance (Holling 1973; Gauthier 2009; Gunderson et al. 2009). Resilience also relates to how quickly the ecosystem can return to original conditions after a disturbance. Even if the condition of a forest at a specific site 100 years after a disturbance is not an exact replica of a 100-year-old stand on the same site before the disturbance, the main stand characteristics (e.g., stand structure, dominant species, type of habitats created) and processes (e.g., water and nutrient cycles, energy flow) would essentially be the same.

In recent decades, promoting resilience has been a widespread goal of forest management, as the increasing pressure of large- and small-scale disturbances is pushing many forests toward and over resilience thresholds (Heinselman 1973, 1981b; Mladenoff et al. 1993; Robertson et al. 1993; Fleming 1996; Vogt et al. 1997; Paine et al. 1998; Dale et al. 2000; Johnstone & Chapin 2003; Hooper et al. 2005; Schulte & Mladenoff 2005; Hayhoe et al. 2007a; Seidl et al. 2009; Rogers et al. 2011; Seidl et al. 2011a; Hart et al. 2019). The consequences of increased temperatures, extreme droughts, catastrophic wildfires, and widespread insect outbreaks demonstrate that resilience thresholds can be exceeded and that major ecological transformations can result.

The results of paleoecological reconstructions from two upland sites at Acadia National Park (ANP) (i.e., Hadlock watershed and Cadillac watershed) suggest that climate warming could significantly alter forest types now dominant in northeastern North America (Patterson 2006; Schauffler et al. 2007; Fisichelli et al. 2013). Past shifts from conifer to deciduous (or the reverse – deciduous to conifer) have been associated with major fire events (Davis 1962, 1967; Patterson et al. 1983; Patterson & Backman 1988; Drake & Patterson 1994; Patterson 2006; Schauffler et al. 2007; Marlon 2015). Although ANP data suggest that catastrophic, large fires historically occurred infrequently (Davis 1962, 1967; Patterson et al. 1983; Patterson & Backman 1988; Drake & Patterson 1994; Patterson 2006; Schauffler et al. 2007; Marlon 2015), the potential exists for high-intensity fires in the future that could impact forest resilience.

In 1947, a high-intensity fire burned the eastern portion of the Mount Desert Island (MDI) location of ANP. The purpose of this research was to examine the long-term effects of the 1947 fire on plant species composition and structure in forest stands first sampled by Patterson et al. in 1980, resampled by Patterson in 1992-1994, and in this study in 2016. More than a third of the sampled stands burned in the 1947 fire. Based on the initial sampling, Patterson et al. (1983) provided a description of the fire regimes of Acadian spruce-fir and associated cover types. My research continues the stand-level documentation of vegetation and fuel load changes during a period characterized by a long period of climate warming. Remeasuring vegetation and fuels over a 68-year period following the most recent fire disaster in Maine provides the NPS, the Maine Forest Service, and conservation organizations in northern New England and adjacent areas of northeastern New York and the maritime provinces of Canada with valuable insights into the resiliency of spruce-fir/northern hardwood forest types and how they recover from severe wildfire.

The conditions and history of disturbance at ANP generate numerous ecological questions that, when answered, enhance our understanding of forest ecology in the region. The following specific questions were asked to determine whether the forests of ANP are resilient to wildfire:

*Study Questions*

1) Did the forests of ANP retain its characteristics post-wildfire (e.g., the maintenance of diverse forest compositions and age class distributions) so that is may be called resilient?

2) Did species composition and structure (i.e., live/dead basal area ($ft^2$/acre); standing woody fuel (live/dead in T/acre) of trees, saplings and shrubs; downed woody fuel (T/acre); mean diameter (in) of trees; average stem density of trees and saplings (live/dead); and average duff depth and fuel height (in)) change in a predictable manner (Egler 1954) in both unburned stands and stands burned in the 1947 fire (Davis 1962, 1967; Patterson et al. 1983)?

3) Did live conifer fuels (i.e., live conifer basal area ($ft^2$ /acre), live conifer biomass (T/acre) of trees, saplings and shrubs, and canopy density (%)) increase in stands burned in 1947 compared to unburned stands (density dependent thinning) across the three sampling periods?

4) In the understory of burned stands, did dead fuel load (i.e., dead standing biomass (T/acre) of saplings and shrubs, and downed fuel loads [sound and rotten (T/acre), fuel height (in.)] increase across the three sampling periods while the dead fuel load in unburned stands decrease as they are likely breaking-up?

In this dissertation, I describe patterns of forest recovery and change following the large 1947 wildland fire in ANP on MDI, Isle au Haut, and Schoodic Peninsula. I also describe patterns of stands which did not burn in the 1947 fire. Chapter 2 provides the historical context of fire regimes and climate and discusses forest disturbance-recovery dynamics (i.e., resilience) in the northeast United States (US). Chapter 3 outlines the study methods, including a description of the study area, the research completed by Patterson et al. (1983), and the data collection and analysis methodology used. The data collection and calculations presented in Chapter 3 follow closely those of Patterson et al. (1983) for the purpose of making direct temporal comparisons of forest composition and structure at the stand level.

Chapter 4 presents the results of the study, a field data-based analysis of the 1947 fire effects on long-term vegetation and fuel dynamics in ANP. Long-term successional trends and current conditions of forest stand composition and structure are discussed with quantitative analysis at both the individual stand level and larger forest-cover-type scale. The findings, management implications, and future research needs are outlined at the end of Chapter 4.

Chapter 5, which uses the data set generated in Chapter 4, describes a Geographic Information Systems (GIS) spatial model that quantifies current wildfire risk across ANP and the larger MDI landscape. I examined remotely sensed numeric biomass data and field-based data to find the most accurate predictor of biomass measured on-the-ground. I used the most accurate predictor to develop the wildfire risk maps. This chapter is presented as submitted to the National Park Service after revisions based on their peer review process. Comparing current forest data to past forest conditions provides a clear illustration of how the forest communities have developed and how they may change in the future.

## Chapter 2: Literature Review: Forest Resiliency, Fire and Climate

*Disturbance, recovery, and ecosystem services*

Disturbances, both natural and anthropogenic, are a dominant consideration in forest management because of the consequences for community composition, biodiversity, natural resources, ecosystem processes, and aesthetics (Turner 1987; Mladenoff et al. 1993; Robertson et al. 1993; Fleming 1996). Paine et al. (1998) argue that in a world of ever more pervasive anthropogenic impacts on forest ecosystems coupled with the increasing certainty of climate change, compounded disturbances and unwanted ecological consequences will become more common. Understanding these disturbance interactions will be the basis of forest management decisions in the 21st century. The health of the boreal and temperate forests is presently under threat given the stressors associated with climate change, and thus the long-term provisioning of vital ecosystem services is at risk (Gauthier et al. 2015; Dey et al. 2019; Hisano et al. 2019).

The maintenance of ecosystem services from forests depends on the preservation of forest health, which is threatened by the speed and amplitude of changes in climate (Dale et al. 2000, 2001; Iverson et al. 2004; Plummer et al. 2006; Hayhoe et al. 2007a, b; Campbell et al. 2009; Dukes et al. 2009; Flannigan et al. 2009; Seidl et al. 2011b; Brose et al. 2013; Duveneck et al. 2014; USGCRP 2018; IPCC 2019), and thus disturbance regimes projected for these northern latitudes (Brotak & Reifsnyder 1977; Manabe et al. 1981; Paine et al. 1998; Fischlin et al. 2007; Huntington et al. 2009; Mohan et al. 2009; Allen et al. 2010; Turner 2010; Buma & Wessman 2011; D’Amato et al. 2011; Brown & Johnstone 2012; Adams 2013; Amraoui et al. 2013; Keenan 2015; Appenzeller 2015; Millar & Stephenson 2015; IPCC 2019). Considering the importance of the potential impacts these changes may have and the extent over which they may take place, it is imperative that adaptive actions be taken to maintain the health of the forest or to enhance its contribution to climate change mitigation (Paine et al. 1998; Gauthier et al. 2015; Dey et al. 2019; Hisano et al. 2019). The challenge is determining when the frequency, spatial extent, and strength of stresses and disturbances exceed the natural range of variability and affect the trajectory of vegetation recovery at the regional to landscape scale (Ayres & Lombardero 2000; Dale et al. 2000; Iverson et al. 2004; Campbell et al. 2009; Frelich & Reich 2010; Churma et al. 2011; Brown & Johnstone 2012; Trumbore et al. 2015).

Disturbances and ecosystem recovery are important mechanisms for the maintenance of the vegetation mosaic at different spatial and temporal scales (Heinselman 1973; Shugart et al. 1992; Gimingham & Johnson 1993; Attiwill 1994; Dale et al. 2001; Jayen et al. 2006; Dey et al. 2019). Climate change will accelerate the frequency and/or increase the magnitude of disturbances (Dale et al. 2000, 2001; Iverson et al. 2004; Plummer et al. 2006; Hayhoe et al. 2007a, b; Campbell et al. 2009; Dukes et al. 2009; Flannigan et al. 2009; Seidl et al. 2011b; Brose et al. 2013; Duveneck et al. 2014; USGCRP 2018; IPCC 2019). Understanding and quantifying an ecosystem's resilience to disturbance is increasingly important for forest management (Seidl et al. 2009; Rogers et al. 2011; Seidl et al. 2011a), because changes expected in the climate system have the potential to change species composition, structure, and function (Heinselman 1973; 1981b; Mladenoff et al. 1993; Robertson et al. 1993; Paine et al. 1998; Fleming 1996; Dale et al. 2000; Johnstone & Chapin 2003; Schulte & Mladenoff 2005; Hayhoe et al. 2007a; Seidl et al. 2011a), or even change biome boundaries at the landscape scale (Ayres & Lombardero 2000; Dale et al. 2000; Iverson et al. 2004; Campbell et al. 2009; Frelich & Reich 2010; Churma et al. 2011; Brown & Johnstone 2012). Forests of the northeastern US are formed by their land use and disturbance history, and each disturbance affects forests differently. Some cause large-scale tree mortality, whereas others alter the community structure and composition without causing massive mortality (e.g., ground fires). Intensifying disturbance regimes are expected to be among the most detrimental impacts of climate change on ecosystem services (Dale et al. 2001; Bale et al. 2002; Netherer & Schopf 2010; Linder et al. 2010; Turner 2010; Hart et al. 2019).

Forests provide an important ecosystem service as a sink for atmospheric carbon dioxide ($CO_2$) and are estimated to absorb about half of the $CO_2$ currently released by human activities (Schimel et al. 2001; Dilling et al. 2003; Dore et al. 2008), thus playing a critical role in the global carbon (C) cycle (Goodale et al. 2002; Foster & Aber 2004; Hundiburg et al. 2009; Gauthier et al. 2015; Dey et al. 2019). Forest ecosystems store large quantities of C in biomass and other organic matter (Birdsey & Heath 1995). Climate warming has the potential to change northeastern US forests from C sinks (since the abandonment of agricultural activities in the 1800s) to sources in the late 21st century due to increases in disturbance (Tang et al. 2014). Disturbance history, together with forest age, forest type, and climate are important sources of variation in the amount and rate at which forests store C (Law et al. 2001, 2003; Humphreys et al. 2005; Gough et al. 2007; Dore et al. 2008; Hundiburg et al. 2009). Gough et al. (2007) found that recently disturbed ecosystems were strong sources of C to the atmosphere for up to four decades following stand-replacing disturbance. With the removal of aboveground biomass as a result of fire, the amount of $CO_2$ taken up by photosynthesis is significantly reduced while the remaining belowground biomass and slash decompose, releasing $CO_2$ (Humphreys et al. 2005; Dore et al. 2008). Forest disturbances such as fire and insect damage may add to the pool of

$CO_2$ in the atmosphere, while growing forests may reduce atmospheric $CO_2$ through increases in biomass and organic matter accumulation (Birdsey & Health 1995; Dore et al. 2008; Raymond & McKenzie 2013).

Net primary productivity (i.e., the uptake of C by ecosystems) increases rapidly in young forests through maximum canopy closure (Gower et al. 1996; Ryan et al. 2004; Raymond & McKenzie 2013). Forests continue to take up carbon from the atmosphere even past the point at which they reach maturity (Luyssaert et al. 2008; Keith et al. 2009; Keeton et al. 2011; Chen et al. 2014; Gunn et al. 2014). By measuring aboveground biomass accumulation, researchers have identified that carbon storage in trees increases continuously because the overall leaf area increases as trees grow, enabling older forests to assimilate more carbon from the atmosphere than young forests, thus storing vast quantities of carbon very late into stand development (Luyssaert et al. 2008; Keeton et al. 2011; Chen et al. 2014). Modeling by Euskirchen et al. (2002) suggests that repeated stand-replacing disturbances may prevent forests from reaching maximum C storage capacity. Their results show that stands which were disturbed by fire twice stored on average 45% less C annually than those experiencing similar fire only once (Euskirchen et al. 2002). The mechanism for this reduction in C storage was a long-lasting decrease in site quality (i.e., soil characteristics and growing conditions) that persisted. Gough et al. (2007) found that fire reduced site quality in northern US temperate and southern Canadian boreal forests, thereby slowing forest growth and limiting forest C storage rates.

Tree species vary in their capacity to sequester and store carbon because the density of wood varies by species. Hardwood trees have a higher density than softwood trees. If hardwoods are the dominant, mature cover type can increase carbon storage in a forest

(Catanzaro & D'Amato 2019). In addition, young forests have the capacity to store less C due to smaller diameter trees (Birdsey & Heath 1995; Bonan 2008). Older forests (i.e., large-diameter trees) store the maximum amount of aboveground carbon (D'Amato et al. 2019). Therefore, the combination of species and age class distributions across a landscape produces a wide range of carbon storage capacities (Birdsey & Heath 1995).

Natural disturbance and anthropogenic land-use changed the abundance and distribution of tree species across large areas of the northeastern US following European settlement (Foster & Aber 2004; Nowacki & Abrams 2008; Jantz. et al. 2016; Dey et al. 2019). This change in abundance and distribution of tree species has led to forests becoming increasingly mesophytic (Nowacki & Abrams 2008) and demonstrating a relatively low tolerance to drought (Gustafson & Sturtevant 2013; Varner et al. 2016; Lienard et al. 2016; Rogers et al. 2017). Species distributions are tied, in part, to temperature and drought thresholds (Worrall et al. 2013; Siefert et al. 2015; Jantz et al. 2016). As temperatures increase, trees will be subject to greater evaporative demand from a warmer atmosphere. When combined with periods of drought, this may increase mortality of vulnerable species such as red spruce and balsam fir (Hamburg & Cogbill 1988; Beckage & Ellingwood 2008; Gavin et al. 2008; Pontius et al. 2016; Rogers et al. 2017). Increased mortality of species such as red spruce and balsam fir may increase fire risk by increasing dead standing fuel of the more flammable species in the region (Abrahamson 2018).

Different types of disturbances have been an essential part of the dynamics of the northeastern US forested landscape with disturbance events that affect several square meters to millions of hectares (Hayhoe et al. 2007b). The combination of large and small-scale disturbances historically has shaped the biodiversity of the forests through the maintenance of a high landscape-level diversity of stands varying in size, age, structure, and composition (Heinselman 1973; Frelich & Lorimer 1991; He & Mladenoff 1999). All species have evolved in the presence of disturbance, and thus in a sense rely on the recurrence pattern of the disturbance. Consequently, disturbances within typical range, even at the extreme of that range as defined by large, infrequent disturbances, usually result in little long-term change to the system's fundamental characteristics (Paine et al. 1998). While intermediate levels of disturbance may maximize species diversity (Huston 1979, 1994; Luken et al. 1992; Wilson 1994), in the face of climate change, compounded disturbances (i.e., multiple perturbations, in the same location, separated by less time than is required for recovery) are expected. This can create disturbances leading to irreversible degradation of the ecosystem or cause a sharp shift to an alternative state (Vogt et al. 1997; Van Nes & Scheffer 2004; Hisano et al. 2019).

Recent evidence suggests that the successive occurrence of two types of natural disturbance – insect infestation and fire – in the same stand within a few years may lead to the collapse of tree regeneration and the inability of the forest to return to its predisturbance state (Payette et al. 2000). If the interaction results in a simple severity increase (e.g., two hurricanes which combine to destroy many trees), the cumulative effect may be equivalent to treating the disturbance combination as one large, infrequent disturbance (Turner et al. 1998). However, if the first disturbance alters the characteristics (e.g., composition and age class distribution) of the forest, the combination of successive disturbances may be a catastrophic disturbance, likely to cause unexpected results and potential non-linear ecosystem behavior as resistance and/or resilience mechanisms are exceeded (Paine et al. 1998; Buma & Wessman 2011; Cavard et al.

2019). In areas already affected by successive natural disturbances, the addition of other disturbances, such as climate variability, pests and pathogens, and anthropogenic perturbations (e.g., forest harvesting) can transform whole ecosystems (Paine et al. 1998; Payette et al. 2001; De Grandpré et al. 2019). For example, Buma and Wessman (2011) found that all subalpine forest sites in northern Colorado, which include both natural and anthropogenic disturbances, experienced catastrophic, stand-replacing fire as the last disturbance in the sequence that started with a severe windstorm.

Fires in the northeastern US region are relatively infrequent high-intensity crown fires initiating secondary successional processes (Heinselman 1981a, b; Patterson et al. 1987). Fire occurrence, area burned, and severity are projected to increase considerably in the future (Gauthier et al. 2015; Lesmeister et al. 2019). A short fire return interval can disrupt strategies selected to ensure the presence of post-fire propagules (e.g., serotiny), and fire-adapted species, such as spruce, may be at risk for post-fire regeneration failure with fire return intervals of less than 50 years (Johnstone & Chapin 2006; Brown & Johnstone 2012). For example, black spruce stands are suffering from naturally recurrent insect and fire disturbances (Payette & Delwaide 2003; Messaoud et al. 2019), and research has shown that successive disturbances can considerably reduce the number of seed-bearers, lending to the collapse of post-fire regeneration and a shift to heathland in this system (Payette et al. 2000; Payette & Delwaide 2003). Similarly, Simard and Payette (2005) found through stand reconstructions of the southern boreal forest in eastern Canada that closed-canopy spruce forests suffered from weak post-fire regeneration after successive disturbances. Their sites included stands that had been disturbed by consecutive fire events, insect outbreak followed shortly by fire, and the combination of timber harvesting, insect outbreak, and fire. Their results showed a successional trajectory far from expected for northern latitude forests influenced by a single disturbance and found that multiple disturbances resulted in the formation of a divergent community (e.g., spruce to heath). Boiffin and Munson (2013) found critically low black spruce regeneration after major fire activity caused by extreme fire weather in almost all their stands, leading to a decrease in stand density and a shift of species dominance from black spruce to jack pine. The sites were unfavorable to black spruce germination and survival in the context of warm and dry weather that prevailed in post-fire summers. These studies support the reality that ecosystems with low species diversity (e.g., spruce stands) may be most sensitive to climatic extremes and the resulting change is disturbance regimes (Tilman 1996), and ecosystems with low productivity (e.g., spruce stands) require a considerable amount of time to recover from perturbations (Moore et al. 1993; Huston 1994). During large fire years, high proportions of the landscape are subjected to the interaction of fire regime and weather that create unsuitable conditions for spruce regeneration. Hence northern latitude and mixed-forest vegetation are vulnerable to change at the broad scale. Therefore, the frequency of major fire years could have a significant influence on the rate of vegetation response to climate change in the Maine's spruce-fir and mixed-conifer forests (Boiffin & Munson 2013; Barton et al. 2012; Barton & Keeton 2018).

On the other hand, some scientists have used pollen abundance and fire return interval analysis to suggest there is no relationship between vegetation composition and fire frequency in northeastern US forests (Carcaillet et al. 2010). Carcaillet et al. (2010) found fire return intervals have no significant (or a delayed) impact on pollen data, for species diversity and successional trajectories. They conclude northeastern US forests appear resilient to changes in fire regimes. Jayen et al. (2006) found that fire severity was not statistically significant for predicting regeneration success and concluded that little change in stand composition occurs after fire in stands dominated by black spruce. These studies suggest the vegetative resilience under an increase in fire frequency and intensity in the northeastern US, associated with global warming, would not result in significant changes in the vegetation composition (Jayen et al. 2006; Carcaillet et al. 2010). However, Jayen et al. (2006) and Carcaillet et al. (2010) were looking at single fire events, rather than the effects of increased frequency and/or intensity of consecutive disturbances. Post-fire weather (e.g., high temperatures and low water retention) may also be a driver in determining post-fire regeneration success or failure causing shifts in species dominance. Post-fire weather was not considered in these studies. It should be noted that if weather was considered, and temperatures were high and soil moisture low during post-disturbance seedling establishment, a shift from black spruce to jack pine would likely have been observed. This is because jack pine stands usually dominate in dry, low-nutrient soils, whereas cold, moderate- to poorly drained sites favor black spruce (Boiffin & Munson 2013; Abrahamson 2018). Further, the disturbance effects investigated here were within the normal range of variability needed for regeneration success, mechanisms which have evolved over millennia. Black spruce ecosystems are widespread across the northern latitude boreal and near-boreal forest because of physiological adaptations that allowed these communities to thrive in fire-prone areas.

The inconsistent information regarding transformation of communities following fire is linked to the notion of resilience thresholds (Hooper et al. 2005; Hart et al. 2019). If the resilience thresholds of the dominant species are exceeded, then we can expect to see shifts in post-disturbance regeneration. Although northern latitude forests appear resilient to a single fire after 150 to 200 years of succession (e.g., Bergeron 2000; Jayen et al. 2006; Carcaillet et al. 2010), the abrupt change in mean fire return interval from, for example 100 to 500 years, or vice versa, could have consequences for the composition of communities by suppressing or facilitating tree growth, or transforming one ecosystem into another (Le Goff et al. 2005; Jasinski & Payette 2005). Ecosystems can react to modification of the fire regime by large- or small-scale changes in plant composition. At the landscape level, vegetation may or may not appear resilient depending on the size of the study area, the frequency and intensity of the disturbance regime, and post-disturbance weather patterns (Bergeron & Dansereau 1993; Bergeron 1998). The magnitude of fire frequency and intensity change is crucial for determining whether communities will be resilient or change abruptly or even gradually.

While disturbance events in a warming climate may make it more difficult to maintain existing communities, they provide opportunities for land managers to affect patterns of succession in ways that may help maintain ecosystem services (Jantz et al. 2016; Dey et al. 2019). In areas with high potential impact from climate change such as ANP (Tang et al. 2014; Star et al. 2015), NPS managers may be faced with situations where community composition could change rapidly as canopy trees die, creating gaps exposed to current climate conditions that may favor establishment or growth of different plant communities (Dale et al. 2001; Jantz et al. 2016). Given the potential for fire to cause large changes in forest ecosystems, and the likely increasing frequency and intensity of disturbances, it is important to investigate how forests respond to catastrophic disturbance (e.g., high-intensity wildfire) and the characteristics of altered forested ecosystems. Fire management capacity may be overwhelmed in the future, but planning can be adapted to changing fire regimes (Star et al. 2015; Trumbore et al. 2015; Hart et al. 2019). This should be a consideration in defining forest management goals and implementing forest management strategies at ANP in the context of the future climate.

*Historical context of fire regimes and climate*

Fire regimes are considered agents of change because the effects of changes in fire regimes on the structure and composition of forest stands are much more immediate than the direct effects on the distribution, extinction, or migration of species. In this era of rapid climate change, understanding past and predicting future fire activity are scientific challenges that are central to the development of sustainable forest management practices and policies. Efforts to develop a better understanding of the role of fire in northeastern US forests (e.g., spruce/fir) that have historically supported infrequent, high-intensity fires should be emphasized because these environments are susceptible to catastrophic large fires and their devastating effects (Lorimer 1977; Fahey & Reiners 1981; Schulte & Mladenoff 2005; Huntington et al. 2009; Rustad 2012; Brose et al. 2013; Miller 2019).

As fire regimes change, the balance in species composition shifts according to fire ecology traits that developed over an evolutionary timescale. Stocks et al. (1998) expects more extreme fire weather due to a projected increase in fire weather severity across the boreal biome with the most dramatic increases in southern Canada and southern Russia. Another study focusing on the Canadian boreal forest predicts that areas of maximum fire danger risk will double by 2050 (Malevsky-Malevich et al. 2008). However, while some studies in the northeastern US point to an increase in fire frequency (Clark 1988; Stocks et al. 1998; Hart et al. 2019) or size of area burned (Flannigan & van Wagner 1991; Wotton & Flannigan 1993; Bergeron et al. 2001; Hisano et al. 2019), simulations based on mathematical models predict that the Fire Weather Index (FWI) will increase in central and eastern North America but will be lower in northeast North America (Flannigan et al. 1998, 2001). Flannigan et al. (1998) found that the FWI in northern latitude forests will change in the future, but the changes are spatially dependent. Their study found FWI values may decrease in eastern Canada, western Canada, and most of northern Europe; increases are expected in southern Sweden and Finland and throughout central Canada.

The fire regime of northern latitude forests, fire frequency in particular, are believed to be primarily controlled by large-scale climate processes (Foster 1983a, 1983b, 1985; Baker 1995; Bessie & Johnson 1995; Pyne et al. 1996; Turner et al. 1998; Dale et al. 2001; Podur & Martell 2009; Balshi et al. 2009, Flannigan et al. 2009). Modern forest composition and fire return interval of the northeastern US results in infrequent, high-intensity fires and have strongly influenced long-term structure, composition, and function of forest ecosystems (Heinselman 1981b; Green 1982; Laing 1993; Drake & Patterson 1994; Schauffler et al. 2007). Therefore, long-term vegetation dynamics in northern latitude forests are controlled by changes in fire regimes. An understanding of the historical context of fire regimes and climate is necessary as it is the framework for understanding forest dynamics and sheds light on the interpretation of the results of this study.

Sedimentary analyses of fire records and vegetation histories for ANP suggest that historically (i.e., since European settlement: ca. 1760 AD – present), catastrophic large fires have occurred and might be expected to continue to occur at 200-250-year intervals (Figure 1; Patterson 2006). In northeastern US forest types such as those of ANP, an increase in fire frequency and area burned appears to be associated with warmer and drier weather (Bessie & Johnson 1995; Mohan et al. 2009; Marlon 2015; Miller 2019). While New England is experiencing an accelerating wetting trend, and thus we can expect less fire overall due to an increase in effective moisture, there is increased potential for large fires in the event of episodic drought given increases in temperature, biomass (from land-use changes and increased temperatures and $CO_2$) and fuels (from fire suppression, drought stress, insect, disease, and pathogens) (Marlon 2015; Miller 2019). Episodic drought conditions, which most often occur in the summer months (Marlon 2015; Patterson 2018), may yield high-severity fires due to high fuel buildup since the last burn, low decomposition rates, and/or a lack of mechanical thinning (Oliver et al. 1997; Moore 1981).

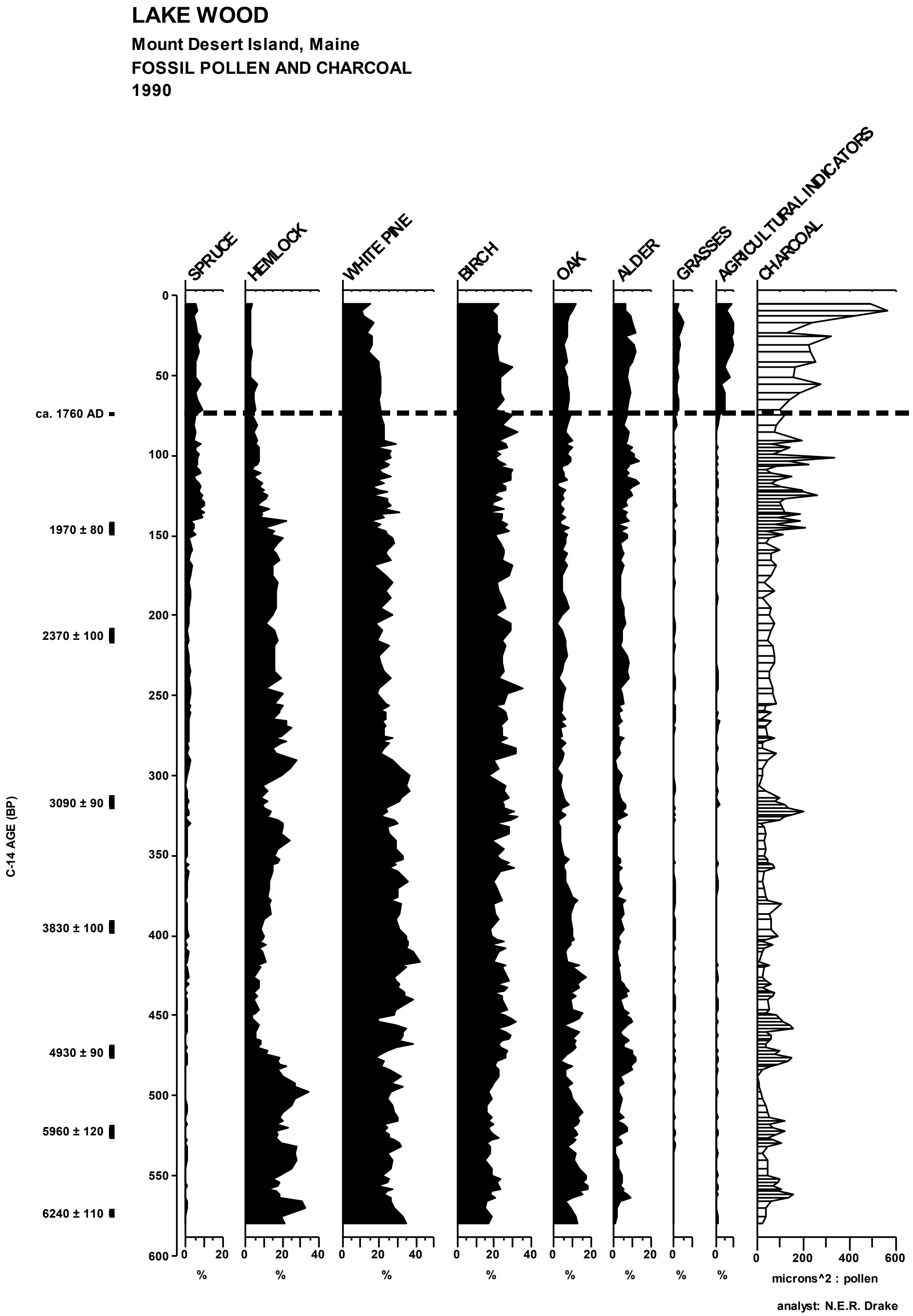



Figure 1. Six thousand (6,000) year record of fire and vegetation change in Acadia National Park, Maine (Patterson 2006).

Drought is an important driver of change in stand composition because it directly affects fuel loading and flammability by increasing tree mortality and lowering moisture content, and indirectly through compounding effects from insect epidemics and diseases that alter forest conditions (Allen et al. 2010). Mortality events may be extensive if drought periods become more frequent or more severe, as has been both predicted (Gao 2012) and observed over the past few decades (Li et al. 2011; Jantz et al. 2016). Interactions between multiple disturbances are increasing the frequency, extent, and severity of atypically large fires and longer fire seasons (Brotak & Reifsnyder 1977; Manabe et al. 1981; Fischlin et al. 2007; Huntington et al. 2009; Allen et al. 2010; Buma & Wessman 2011; Adams 2013; Amraoui et al. 2013; Keenan 2015; Appenzeller 2015; IPCC 2019), and could lead to permanent shifts in ecosystem function (Paine et al. 1998; Mohan et al. 2009; Turner 2010; D'Amato et al. 2011; Brown & Johnstone 2012; Millar & Stephenson 2015).

While large devastating wildfires are uncommon in the northeastern US they do occur. For example, in the summer of 1908, more than 300,000 acres (121,405 ha) burned in the Adirondacks of New York, 142,000 acres (57,465 ha) burned in Maine, and 16,000 acres (6,475 ha) burned in Vermont (Long 2016). New Hampshire's worst fire year was 1903 when 84,000 acres (33,994 ha) burned. In the 1940s, large landscape-scale wildfires fed on fuel left by the September 1938 hurricane occurred across the northeastern US region (Long 2016). In 1941, the largest post-hurricane wildfire, the Marlow-Stoddard fire in New Hampshire, burned 27,000 acres (10,927 ha) during the last three days of April before a May 1 precipitation event extinguished it (Long 2016).

An even worse wildfire season came in October 1947 when a prolonged drought gave way to wildfires which burned an additional 20,000 acres (8,094 ha) across New Hampshire (Long 2016), and more than 212,000 acres (96,000 ha) in Maine, of which more than 17,000 acres (7,730 ha) burned on MDI (Drake & Patterson 1994; Herberger & Patterson 1998; Patterson 2006). The precipitation on MDI for the entire month of October was 0.02 inches, the lowest on record (Drake & Patterson 1994). This created extreme drought conditions which, along with strong winds and a cold dry front, resulted in a catastrophic wildfire that burned for over a week (October 17-27, 1947) (Butler 2014). The fire claimed three lives, burned one-third of the town of Bar Harbor destroying 237 homes and the Jackson Laboratory, and caused an estimated $23,000,000 in damage (in 1947 dollars) (Herberger & Patterson 1998). The fire burned nearly 30% of the land area of the largest island off the coast of Maine, and nearly 20% of NPS land on the island, during a period when the Keetch-Byram Drought index (KBDI) was estimated to have exceeded 500 (Table 1) (Patterson et al. 1983).

Table 1. KBDI is the accepted index that categorizes drought levels specifically for fire potential assessment. The index ranges from 0-800. Source: http://www.wfas.net.

| **Index** | **Description:** |
|---|---|
| 0-200 | Soil moisture and large class fuel moistures are high and do not contribute much to fire intensity. Typical of spring dormant season following winter precipitation |
| 200-400 | Typical of late spring, early growing season. Lower litter and duff layers are drying and beginning to contribute to fire intensity |
| 400-600 | Typical of late summer, early fall. Lower litter and duff layers actively contribute to fire intensity and will burn actively |
| 600-800 | Often associated with more severe drought with increased wildfire occurrence. Intense, deep burning fires with significant downwind spotting can be expected. Live fuels can also be expected to burn actively at these levels |

The fire regime in the northeastern US during the past two millennia is typical of Heinselman's (1973, 1981a, 1981b) Fire Regime 6 (very long-return interval crown fires). During the prehistoric period (i.e., prior to European settlement), paleoecological studies and historical records suggest that several similar high-intensity fires burned on MDI (Wein & Moore 1977; Fahey & Reiners 1981; Wein et al. 1987; Patterson et al. 1983, 1984, 1987; Patterson & Backman 1988; Drake & Patterson 1994; Clark & Patterson 1997), and at intervals of approximately 250 years since spruce-fir replaced northern hardwood/hemlock forests with a cooling climate approximately 2,000 years ago (Drake & Patterson 1994; Clark & Patterson 1997; Schauffler et al. 2007). For the several millennia prior to approximately 2,000 years ago, back to more than 7,000 years ago when the climate of the northeastern US was warmer and presumably more moist, northern hardwood/hemlock forests dominated the MDI landscape and fires were rare – occurring only in conjunction with rapid declines in hemlock importance. This forest composition and fire return interval (250-1000 years) results in high intensity fires that affect long-term vegetation change (Patterson et al. 1983).

Model predictions of future fire activity in the northeastern US are largely in agreement and suggest that annual burned area and fire occurrence will increase by the end of the 21st century, and trends will be statistically detectable by the mid-21st century (McKenzie et al. 2004; Girardin & Mudelsee 2008; Gauthier et al. 2015; Miller 2019). In Maine, we may expect the mid-Holocene hemlock-northern hardwood forest fire regime (1,000 year return intervals) with less fire in the future (Marlon 2015), but the transition to less fire might be preceded by catastrophic decline of mature spruce-fir and the associated increased fire hazard (Patterson, personal communication). Many conifer stands on portions of MDI not burned in 1947 are now approaching 150 years of age, so a major fire could occur during the next century as forests become over-mature and fuel loads increase. On land that burned in 1947, fire might be delayed 100 years, but only where deciduous tree species (aspen/birch or longer-lived maple/beech) remain. Under severe drought conditions (e.g., 1947), even deciduous forests of ANP will burn with great severity (i.e., with consumption of most soil organic matter) (Patterson et al. 1983).

## Chapter 3: Methods: Measuring Forest Composition, Structure and Fuel Loads

*Site description*

Acadia National Park (ANP) lies in Hancock and Knox Counties on the eastern coast of Maine (Figure 2). Today the park encompasses approximately 35,000 acres (14,000 ha) in three primary units: Mount Desert Island (MDI) – 30,000 acres (12,000 ha); Isle au Haut – 3,000 acres (1,200 ha); and Schoodic Peninsula – 2,000 acres (800 ha). This study focuses primarily on the ANP stands on MDI because the 1947 fire burned much of the eastern half of the island. Several stands elsewhere on MDI burned in the 19th century, and most stands on Isle au Haut and Schoodic Peninsula also burned in the 19th century (Table 2, Figure 3).

Mount Desert Island is approximately 108 square miles (281 $km^2$) in land area, of which about 47 square miles (122 $km^2$) or 47% is owned by NPS. The island lies between 44° 13' and 44° 27' North Latitude and between 63° 20' and 68° 26' West Longitude. Elevations range from sea level to 1530 ft (466 m) at the top of Cadillac Mountain. Mount Desert Island is located in a region of cool moist climate. Temperatures at Bar Harbor range from a record low of -9°F (-22°C) in winter to a record high of 106°F (41°C) in summer with a mean annual temperature of 46.5°F (8°C). Average annual precipitation is approximately 48.5 inches (123 cm), with annual snowfall averaging 59 inches (1.5 m). Due to the proximity of MDI to the Atlantic Ocean and the location of ANP with respect to southerly to southwesterly winds, fog near the coast is common (Patterson et al. 1983).

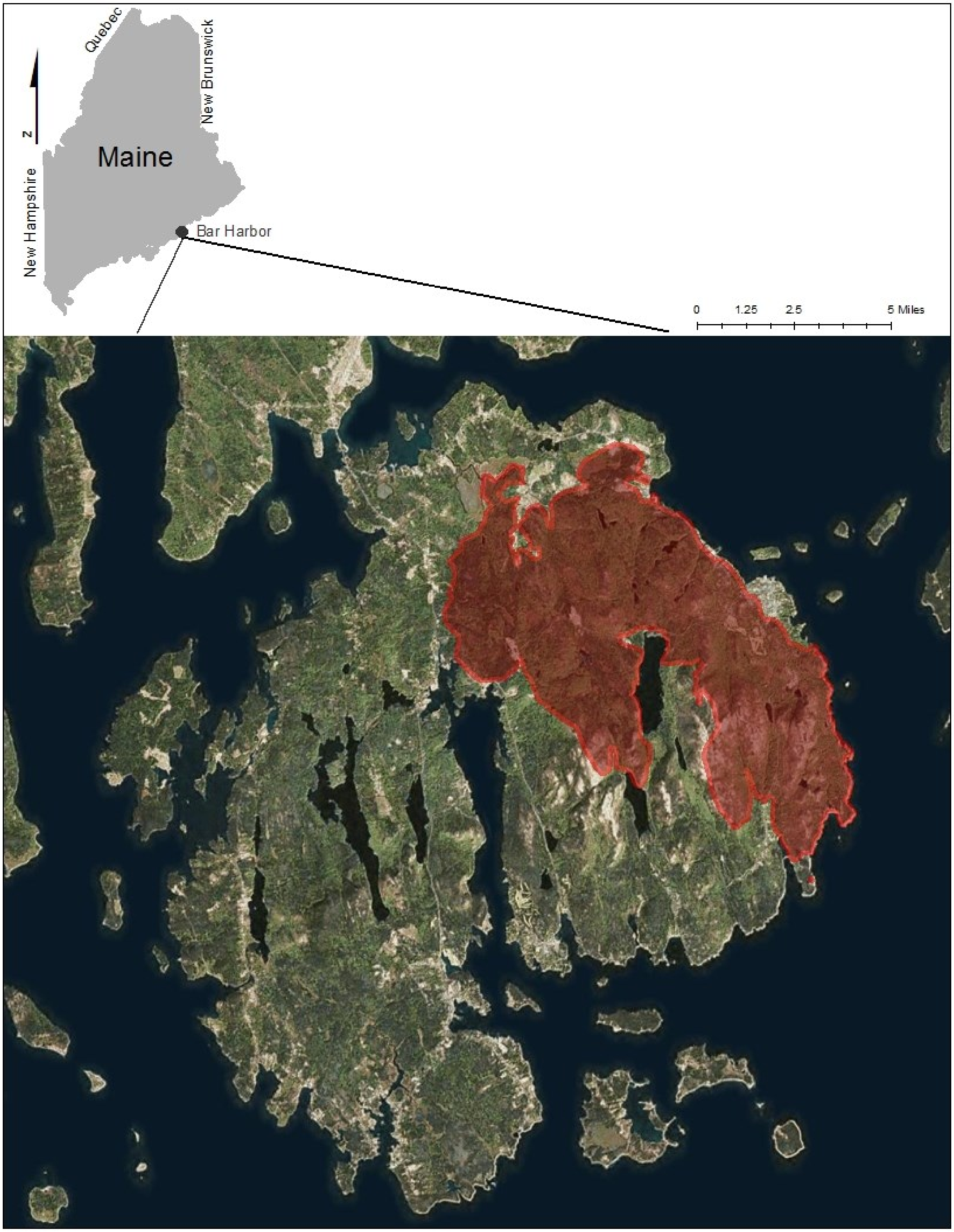


Figure 2. Study site location: Mount Desert Island, Maine. Area burned in the 1947 Bar Harbor Fire is shown in the red shaded section. GIS coverage of the fire extent is courtesy of the NPS at ANP.

Acadia National Park includes many primary and secondary roads, as well as 45 miles of carriage paths (Harbor 2019) that provided access to sample stands and have the potential to serve as access roads or fire breaks in the event of a wildfire. There is also an extensive trail system, with over 120 miles of trails (Harbor 2019), which provides non-motorized access to the Park lands.

Table 2. Stand ID, stand type, burn date(s) and stand initiation for sample stands in ANP (from Patterson et al. 1983).

| Stand ID | Stand Type | Burn Date(s) | Initiation Date |
|---|---|---|---|
| AC01 | spruce-fir | | 1850 |
| AC02 | Northern hardwoods | 1901, 1948 | 1870-1910 |
| AC03 | Northern hardwoods | 1948 | 1870-1900 |
| AC04 | spruce-fir | 1864 | 1840-1890 |
| AC05 | Northern white cedar | 1864, 1889 | 1889 |
| AC06 | Northern white cedar | | 1840-1900 |
| AC07 | red oak | 1948 | 1901 |
| AC08 | pitch pine | 1948 | 1948 |
| AC09 | birch-aspen | 1948 | 1948 |
| AC11 | mixed conifer | 1780, 1820 | 1820-1825 |
| AC12 | spruce-fir | | 1860-1870 |
| AC13 | spruce-fir | | 1860-1915 |
| AC15 | mixed conifer | | 1840-1865/1890-1910 |
| AC16 | mixed hardwood - conifer | fire scars, no date | 1840-1910 |
| AC17 | spruce-fir | fire scars, no date | 1780-1845 |
| AC18 | spruce-fir | | 1830-1840/1890-1900 |
| AC19 | mixed hardwood - conifer | 1880 | 1830-1880 |
| AC20 | spruce-fir | 1855, 1910 | 1855-1910 |
| AC22 | spruce-fir | | 1820-1860 |
| AC23 | mixed hardwood - conifer | 1880 | 1830-1920 |
| AC24 | pitch pine | 1860, 1885 | 1885 |
| AC25 | birch-aspen | 1948 | 1948 |
| AC26 | spruce-fir | unknown | unknown |

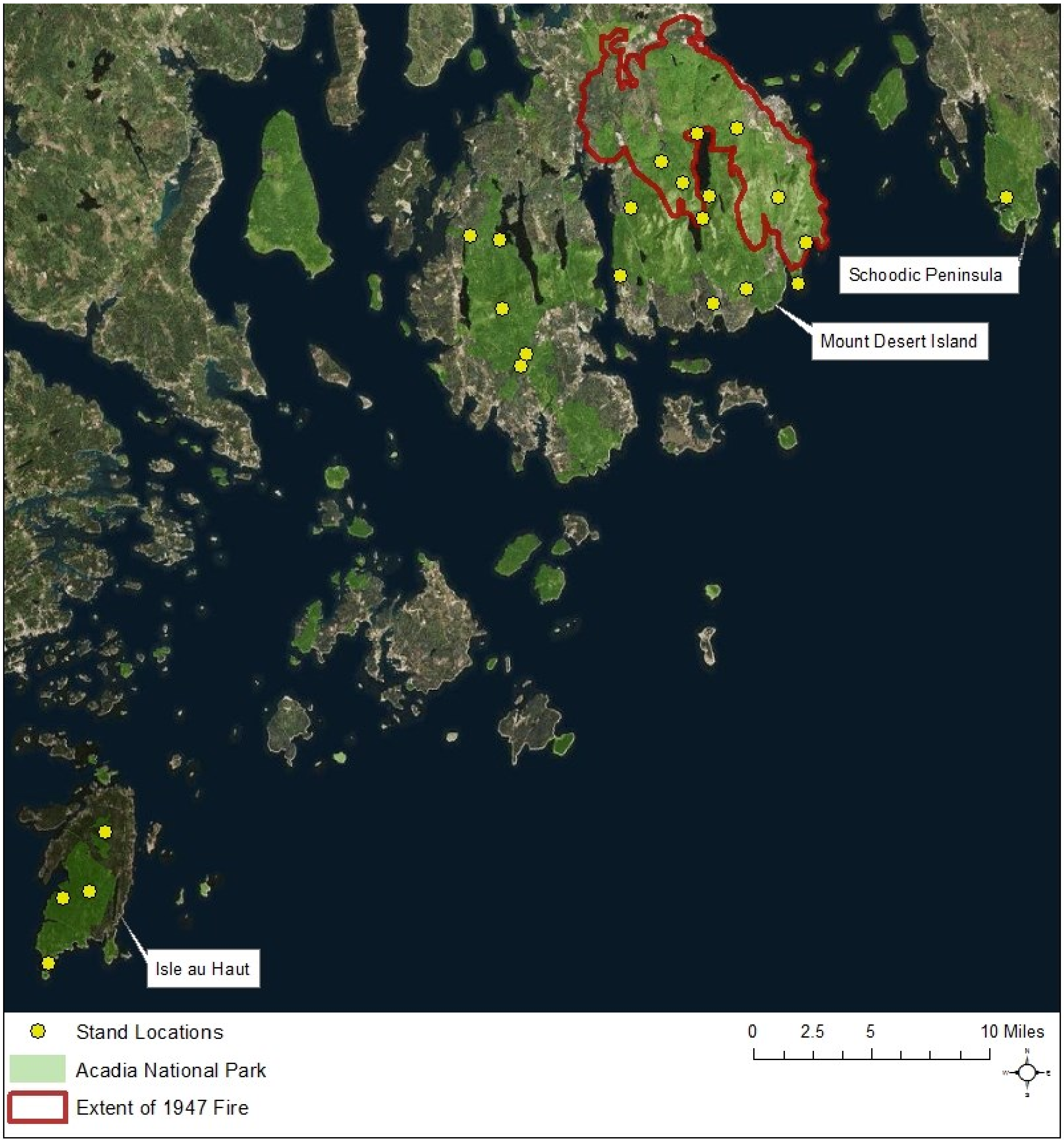



Figure 3. Map showing the stand sample locations for 2016 data collection, the extent of the 1947 fire, and Acadia National Park. GIS coverage of park boundaries and fire extent are courtesy of NPS at ANP.

Soils of ANP have been mapped as part of the Maine Coastal Inventory (Maine State Planning Office 1977) and the USDA Natural Resources Conservation Service (Soil Survey Staff 2019). General soils maps for MDI, Schoodic Peninsula, and Isle au Haut show the distribution of approximately 20 soil types occurring within ANP (Table 3). The most common soil type is the Schoodic-Rock outcrop-Lyman complex on 0-65% slope, which comprises approximately 41.5% of land area in the Park, followed by the Lyman-Tunbridge-Schoodic complex on 0-35% slope, which comprises approximately 15.1% of land area in the Park and the Naskeag-Schoodic complex on 0-8% slope, which comprises approximate 10% of land area in the Park (Table 3). Fire can alter both the physical and chemical characteristics of soils and in so doing affect water regimes and vegetative regrowth (Wells 1978; Ahlgren 1974; Brown & Davis 1973; Viro 1974; Patterson et al. 1983). Potential fire hazard damage is defined as the potential hazard of damage to soil nutrients, physical, and biotic characteristics from fire (Wells 1979). Potential fire hazard damage is measured at moderate fireline intensities (116-520 btu's/sec/ft), which provide the heat necessary to remove the duff layer and consume soil organic matter in the surface layer (Soil Survey Staff 2019). There are three categories of potential fire hazard damage: Low: little negative impact to the soil characteristics is expected; medium: negative impacts to the soil characteristics may occur; high: negative impacts to the soil characteristics are expected (Soil Survey Staff 2019). Approximately 33.5% of the soils in the Park have a high potential for damage, 43.5% have moderate potential damage, and 4.7% have low potential damage (Table 3).

Table 3. Soils of Acadia National Park including soil name,% slope, rockiness, potential fire hazard damage, acres and% cover. Low potential fire hazard damage is defined: little negative impact to the soil characteristics is expected; medium: negative impacts to the soil characteristics may occur; high: negative impacts to the soil characteristics are expected. Only soils comprised of at least one% land area are shown. Source: Soil survey staff, 2019.

| Soil Name, Percent Slope, Rockyness | Potential Fire Hazard Damage | Acres | Percent (%) |
|---|---|---|---|
| Schoodic-Rock outcrop-Lyman complex, 15 to 60 percent slopes | High-Moderate | 8816 | 17.2% |
| Schoodic-Rock outcrop complex, 0 to 15 percent slopes | Moderate | 4937 | 9.6% |
| Schoodic-Rock outcrop complex, 15 to 65 percent slopes | High | 3583 | 7.0% |
| Naskeag-Schoodic complex, 0 to 8 percent slopes, very stony | Low-Moderate | 3466 | 6.8% |
| Schoodic-Rock outcrop-Naskeag complex, rolling | Moderate-Low | 3335 | 6.5% |
| Lyman-Tunbridge complex, 0 to 15 percent slopes, very stony | Moderate-Moderate | 3031 | 5.9% |
| Naskeag-Schoodic-Lyman complex, 0 to 8 percent slopes, rocky | Low-High-Moderate | 1728 | 3.4% |
| Lyman-Schoodic-Rock outcrop complex, 15 to 35 percent slopes, very stony | Moderate-High | 1705 | 3.3% |
| Lyman-Tunbridge-Schoodic complex, 8 to 15 percent slopes, very stony | Moderate-Moderate-High | 1607 | 3.1% |
| Lyman-Schoodic complex, 15 to 35 percent slopes, rocky | Moderate-High | 1418 | 2.8% |
| Hermon-Colton-Rock outcrop complex, 3 to 15 percent slopes, very stony | Moderate-Low | 1360 | 2.7% |
| Wonsqueak and Bucksport mucks, 0 to 2 percent slopes | Low | 1302 | 2.5% |
| Rock outcrop-Lyman complex, 3 to 15 percent slopes | Moderate | 1224 | 2.4% |
| Lyman-Brayton outcrop-Turnbridge complex, 8-15 percent slopes | Moderate-Moderate | 977 | 1.9% |
| Lyman-Brayton variant-Rock outcrop complex, 0 to 8 percent slopes | Moderate-Low | 818 | 1.6% |
| Hermon and Monadnock soils, 8 to 15 percent slopes, very stony | Moderate-Low | 610 | 1.2% |
| Monadnock-Hermon-Peru complex, 8 to 45 percent slopes, extremely bouldery | Low-Moderate-Moderate | 601 | 1.2% |
| Wonsqueak, Bucksport, and Sebago soils | Low | 581 | 1.1% |
| Hermon-Monadnock-Peru complex, 8 to 15 percent slopes, very stony | Moderate-Low | 550 | 1.1% |
| Lamoine silt loam, 3 to 8 percent slopes | Low | 536 | 1.0% |
| **Total** | | **51275** | **100.0%** |

Moore and Taylor (1927) describe the early 20th century vegetation of ANP, when maturing spruce-fir forests dominated the landscape. Today, red spruce (*Picea rubens* Sarg.) is the dominant species and occurs throughout the Park in areas not burned in 1947. Balsam fir (*Abies balsamea* (L.) Mill) is common in the Park but less so than red spruce. Other coniferous species occur as scattered stands or individual trees throughout the Park. These include jack pine (*Pinus banksiana* Lamb.), red pine (*P. resinosa* Ait.), pitch pine (*P. rigida* Mill.), eastern white pine (*P. strobus* L.), and northern white cedar (*Thuja occidentalis* L.). Pitch pine and jack pine occur primarily in single species stands. White spruce (*Picea glauca* (Moench) Voss) forms nearly pure stands along the immediate ocean shore (Moore & Taylor 1927) and as scattered individuals inland. Eastern hemlock (*Tsuga canadensis* (L.) Carr) is uncommon today but was abundant during the mid-Holocene when climate was warmer (Drake & Patterson 1994; Schauffler et al. 2007). Northern hardwoods such as beech (*Fagus grandifolia* Ehrh.), yellow birch (*Betula alleghaniensis* Britton), red oak (*Quercus rubra* L.), and sugar maple (*Acer saccharum* Marsh.) are also less common today than during the Hypsithermal Period (ca. 8,000-4,000 years bp). These northern hardwood species, except for red oak, are known throughout the northeastern US as comprising the least flammable major forest type. Other deciduous forest tree species including aspen (*Populus tremuloides* Michx., *P. grandidentata* Michx.), paper birch (*Betula papyrifera* Marshall) and gray birch (*B. populifolia* Marshall) are abundant in areas burned in 1947. Their abundance in areas burned in the late 1800s and early 1900s was declining in 1980 in favor of spruce-fir stands (Patterson et al. 1983; Schauffler et al. 2007). Today, red oak is more abundant than jack pine and about as abundant as pitch pine on MDI.

Jack pine is most abundant on Schoodic Peninsula and absent from Isle au Haut. The reverse is true for pitch pine.

*Historic forest research at ANP*

Various techniques exist to reconstruct fire histories (Kent 2014). In ANP, fire histories were reconstructed using forest structure data and dendrochronology (both fire scar and tree age/growth data) (Patterson et al.1983) and sediment cores via charcoal and pollen analysis (Davis 1962, 1967; Patterson & Backman 1988; Patterson 2006; Shauffler et al. 2007) (Table 3). Soil probing for charcoal found evidence of fire at almost all plots sampled (Patterson et al. 1983).

In 1980, vegetation maps interpreted from 1979 aerial photos were used to identify 26 homogeneous, approximately 5-to-10 acre (2- to 4 ha) stands, and to establish plots representing the primary forest cover types within ANP (Table 4). At least one stand of each of the major upland vegetation types is included (e.g., mature spruce-fir vs. spruce-fir burned in 1947 but now supporting aspen-birch with a conifer understory) (Table 3).

Characteristics of sample stands, including fire histories, are described in Patterson et al. (1983). In 1980 and 1992-1994, variable-and fixed-radius plots for trees, saplings, shrubs, and fuels were examined in at least 20 plots per stand to characterize the forest composition, structure, and fuel load (Patterson et al. 1983; Patterson 1996). The number of transects and plots within each stand depended upon the size and shape of the stand. Within each stand, sample plots for standing live and dead, and dead downed woody fuels were located on a grid with plots at two-chain intervals along transects running two chains apart. Maps drawn in the field for plots sampled in 1980 and 1992-1994 show the general layout of transects. Locations were marked on the 1:50,000 Park USGS map, and GPS waypoint locations were archived with the NPS (Figure 3). Three of the original 26 stands were not resampled in 1992-1994 because they were no longer forested due to development (no longer ANP land). This historical research provides the current regional knowledge of fire in ANP.

*2016 Data collection*

To further the understanding of forest recovery following the 1947 fire, the same 23 stands from 1992-1994 were resampled in 2016 using methods and plot layout consistent with those used in 1980 and 1992-1994 (Patterson et al. 1983), as described below. Aerial photographs from 1979 used to delineate and select 1980 sample stands were scanned and georeferenced in ESRI ArcGIS to locate 1992-94 and 2016 plots within the original stand boundaries. Stands representative of eight major forest types in ANP were examined (Table 4).

Table 4. Number of sample stands by community type at ANP in 2016.

| **Stand Type** | **Number of Stands** |
|---|---|
| spruce/fir | 8 |
| mixed hardwood/conifer | 3 |
| northern hardwood | 2 |
| birch-aspen | 2 |
| cedar | 2 |
| pitch pine | 2 |
| mixed-conifer | 2 |
| red oak | 1 |
| old growth spruce | 1 |
| **TOTAL** | **23** |

In 2016, at least 20 plots were sampled per stand, with as many as 30 when the size and shape of the stand allowed. Within each stand, sampling points for standing live and dead, and dead downed woody fuels were located on a grid with points at two-chain intervals along transects running two chains apart. Locations were marked on the 1:50,000 Park USGS map, and GPS waypoint locations were archived with the National Park Service (NPS).

Tree species composition was characterized at each point using a ten-factor angle gauge (Cruz-All). Diameters at 4.5 ft (1.4 m) above the ground (dbh) to the nearest 0.1 in. (0.25 cm) were recorded by species for stems ≥ 1 in. (2.54 cm) in diameter. Saplings >4.5 ft (1.4 m) tall and ≤ 1 in. (2.54 cm) dbh were tallied by species in 0.1 in. (0.254 cm) size classes in the 0.01-acre (0.004 ha) radius plots (Figure 4). The number of shrubs and tree seedlings ≤ 4.5 ft (1.4 m) tall were recorded by 1 ft (0.3 m) height classes in 0.001-acre (0.0004 ha) radius plots (Figure 4).

To sample dead, downed woody fuels, a transect was established in a randomly determined direction originating at each point (Figure 4). Along each transect fuel parameters were surveyed using the planar-intercept method (Brown 1974). Sampling transects were 50 ft (15.2 m) long for 1000-hr fuels [>3 in. (7.6 cm) in diameter], 12-ft (3.7 m) long for 100-hr fuels [1-to-3 in. (2.5-to-7.6 cm) diameter], and 6 ft (1.8 m) long for 10- and 1-hr fuels [<1 in. (2.5 cm) and 0.25 in. (0.64 cm) in diameter] (Figure 4). Fuel up to 4.5 ft (1.4 m) in height was counted if it intersected the plane and was measured at its maximum height. Duff depth (in.) and fuel height (in.) was measured at 15 ft (4.57 m) and 30 ft (9.15 m) along the fuel transect.

In addition to quantitative data, qualitative data in the form of photographs were taken in 2016. These photos were taken in the same location as photos taken following the fire in late 1947 and in 1983 (Barnicle 1984). This chronosequence of photographs provides a strong visual image to complement the quantitative data (Appendix 1). Photos for 1947 and 1948 are not included due to copyright law, but they do exist and are stored in NPS archives at ANP. Further, data spanning a 68- year period were augmented by sampling stands of greater age, most of which regenerated after 19th century fires.

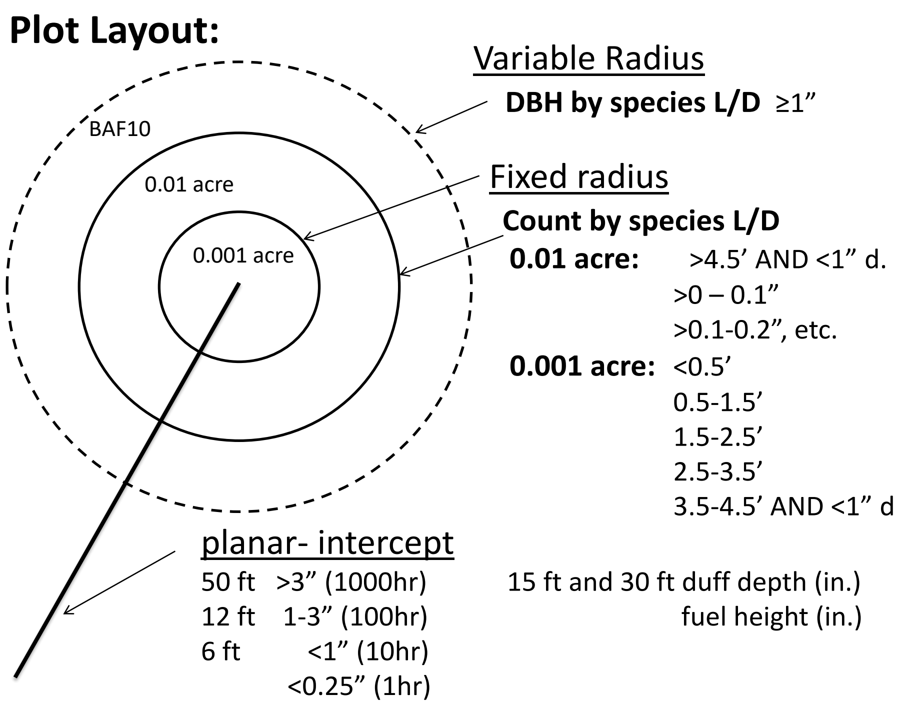



Figure 4. Plot layout for the vegetation and fuels inventory in ANP. L/D = live or dead.

*Data analysis*

Data for 1980 and 1992-1994 were reentered for 14 of 26 stands in 1980 and 17 of 24 stands in 1993. Reentered values were used in analysis and are reported in the tables in Chapter 4 for the respective stands. For the stands in which data were not reentered, data are from Patterson et al. (1983) and unpublished reports. There was a small difference in the values of the past reports and reentered calculated means. In 1980, the average difference across all species for live basal area was 4.1% (±4.6) and 24.4% (±73.5) for dead. In 1992-1994, the average difference across all species for live basal area was 2.3% (±2.9) and 21.0% (± 45.3) for dead.

Differences were likely due to the difficulty in reading old field data sheets, and in some cases missing field data from historic records.

Stand data is represented by averages of the plot data. Basal area per tree ($ft.^2$) was calculated as:

[((PI()*(diameter$^2$))/4)*0.006944]

Where: PI() is the constant = 3.14, area = PI * diameter$^2$/4, and 0.006944 is the unit conversion factor for square feet.

Trees per acre were calculated as follows:

[basal area factor 10/basal area per tree ($ft.^2$)]

Stem density values were calculated as:

[sum of trees per acre by live/dead]

Confidence intervals (95%) were calculated and trends were examined over time. Where the word 'significant' is used in the results section it refers to confidence intervals which do not overlap. Significant tests were not used due to sample size. The calculated standard error of the mean for most variables was greater than 10%. A standard error of the mean greater than 10% suggests the variation in the data is large and therefore statistical tests would not yield results that could be interpreted with confidence.

Biomass estimates in tons (T per acre of individual stems for trees, saplings, and shrubs) were calculated using regression equations and specific gravity coefficients from Young et al. (1980):

[(Ln weight = A+B (Ln DBH or Ln Height)]

Where: Ln= natural logarithm to the base e, A= dry weight aboveground specific gravity coefficient from Young et al. (1980), B= dry weight aboveground specific gravity coefficient from Young et al. (1980), DBH= diameter measured in inches (in.) at 4.5' above ground, Height= total tree height measured in feet (ft). For shrub biomass midpoints of the height classes were used (i.e., 0.25, 1, 2, 3, 4). Methods for shrub calculations has the potential to inflate biomass of individual plots with high stem counts.

Calculated biomass was converted to stand mass densities by averaging plot sums. Specific gravities for most species were obtained from the U.S. Forest Products Laboratory (Brown 1974). Specific gravity estimates for jack pine and pitch pine were found in Whittaker and Woodwell (1968), Alban (1978), and Ledig et al. (1975). Where specific gravity estimates did not exist for a species, respective general hardwood or softwood weights from Tritton and Hornbeck (1982) were used. Calculations of downed fuel loads by approximate timelag class at the plot level followed Brown (1974). Average secants were taken from Brown (1974). The constants $d^2$ (squared average-quadratic-mean diameters for slash and non-slash ground fuels) and s (average slope correction factor) are presented in Patterson et al. (1983). Outliers were not removed because they represented areas of particularly high or low biomass accumulation.

# Chapter 4: Acadia National Park Forest Condition and Long-Term Successional Trends

Quantitative analyses of each stands' species composition and structure across the three sampling periods, the methods which were described in Chapter 3, are presented by community type in the *Results* section below. I use a comparative approach to describe the change or lack thereof that occurred in each stand across the 1980 to 2016 time period. Data presented for each stand include species composition, basal area, stem density of trees and saplings, mean diameter of live trees, and fuel loads (i.e., biomass) of trees, saplings and shrubs. The chronosequence of photos supporting the quantitative analysis is in Appendix 1. Photos for 1947 and 1948 are not included due to copyright law, but they are stored in NPS archives at ANP. From the results the trajectory in species composition and structure of each stand is discussed to assess the conditions of each community type at the end of the stand description for single stand community types or after all stands are described where multiple stands are sampled. The overall resilience of the forests of ANP is also discussed.

Stand locations where sampling occurred, and the extent of the 1947 fire is shown in Figure 5. Stands burned in 1947 include: AC09 (Otter Creek), AC25 (Cadillac Mountain), AC02 (Connors Nubble), AC07 (Gilmore Meadow), and AC08 (Sand Beach) (Table 5, Figure 5). Stand ages were determined using historical records, forest structure, and dendrochronology (both fire scar and stand origin data) (Patterson et al. 1983). Over half the stands were mature spruce-fir (9) or mixed hardwood conifer (4), with most of the other stands representative of forest types established following the 1947 fire (Table 5).

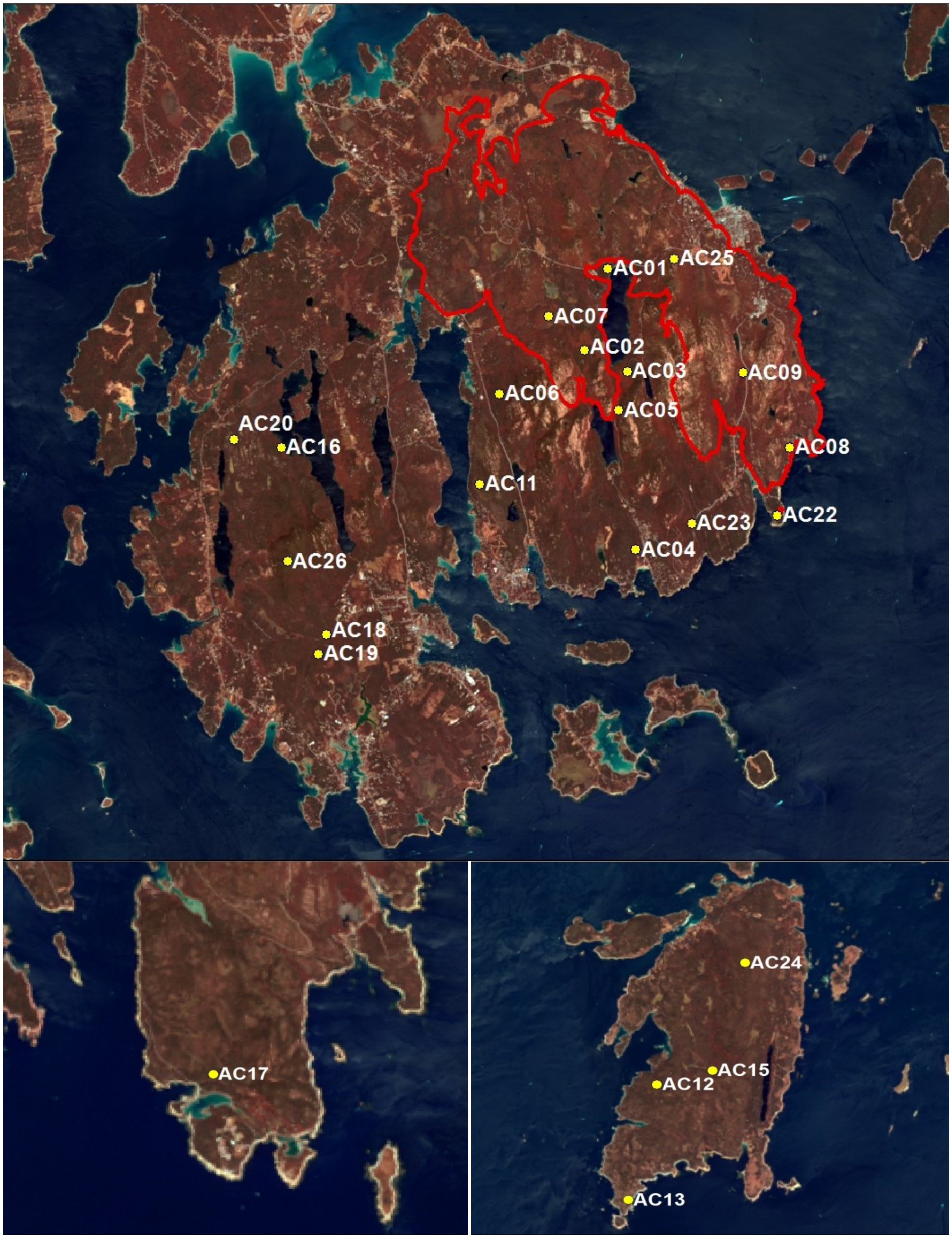



Figure 5. Map showing the sample stand locations for Mount Desert Island (top), Schoodic Peninsula (bottom left) and Isle au Haut (bottom right). The extent of the 1947 fire is outlined in red. Background: Landsat 8 satellite imagery acquired August 23, 2016.

Table 5. Cover type, species composition (≥5ft$^2$/acre of live basal area in 2016), age and physical characteristics of Acadia National Park sample stands. Species are listed in order of largest to smallest ecological importance. * indicates stands burned in 1947.

| Cover Type in 1980 | Stand | Species Composition in 2016 (≥ 5ft$^2$/acre of live basal area) | Age (years) | % Slope | Aspect |
|---|---|---|---|---|---|
| red oak | *AC07 | red oak, red maple | 69 | 10 | NNE-SSE |
| mixed conifer | AC11 | red spruce, white pine, northern white cedar, pitch pine, red maple, red pine | 186-196 | 20 | SW |
| birch-aspen | *AC09 | bigtooth aspen, American beech, red oak, red maple, red spruce, trembling aspen, paper birch, striped maple | 69 | 20 | W-NW |
| | *AC25 | red spruce, red oak | 69 | 20 | N |
| northern hardwoods | *AC02 | American beech, bigtooth aspen, red spruce, red maple, sugar maple, paper birch | 69 | 10 | N |
| | AC03 | American beech, sugar maple, red spruce, hemlock, paper birch, yellow birch, white ash, red maple, striped maple | 116-146 | 15 | NNW |
| pitch pine | *AC08 | red spruce, pitch pine | 69 | 15 | E |
| | AC24 | pitch pine, red spruce | 103 | 10 | NW |
| northern white cedar | AC05 | northern white cedar, paper birch, red spruce, yellow birch, striped maple | 126 | 60-80 | W |
| | AC06 | northern white cedar, white pine, red maple, red spruce, paper birch | 156 | 5 | NW |
| mixed hardwood-conifer | AC23 | red spruce, red maple, balsam fir, bigtooth aspen, yellow birch, white pine, sugar maple, white ash | 96-186 | 10-20 | E-ESE |
| | AC16 | red spruce, red maple, yellow birch, paper birch, hemlock | 106-116 | 10 | NE |
| | AC15 | red spruce, northern white cedar, red maple, yellow birch | 106-176 | 15-30 | WSW |
| | AC19 | red spruce, red maple, northern white cedar, balsam fir | 136 | 10-50 | SSW-SW |
| spruce-fir | AC20 | red spruce, white pine, red maple | 106 | <10 | variable |
| | AC13 | red spruce, balsam fir | 101-156 | 10 | ENE-SW |
| | AC18 | red spruce, northern white cedar, red maple | 116-186 | 20-45 | SSW-W |
| | AC04 | red spruce, red maple, white pine, balsam fir, yellow birch | 126-176 | 15 | E |
| | AC17 | red spruce, balsam fir | 136-171 | 25 | SW |
| | AC12 | red spruce | 146-156 | 10 | W |
| | AC22 | red spruce, balsam fir | 156 | <10 | variable |
| | AC01 | red spruce, hemlock, white pine, balsam fir, red maple | 166 | 15 | ESE |
| | AC26 | red spruce, balsam fir | >186 | 30 | NNE |

## Results

*Red Oak* (n=1)

Gilmore Meadow (AC07) – Red Oak

The following is a description of measured changes at Gilmore Meadow over the 36-year period from 1980 to 2016. Basal areas of red oak, white pine, and red spruce increased between 1980 and 2016. Live basal area of red oak increased by 19%. Total live basal area increased between 1980 and 1993 from 84.5 to 95.3 $ft^2$/acre and shows an increasing trend with 98.0 $ft^2$/acre in 2016 (Table 6). Total dead basal area shows an increasing trend with 2.0 $ft^2$/acre in 1980 and 10.5 $ft^2$/acre in 2016 (Table 6). Density data for trees and saplings are missing for the historic sampling periods (Table 8, Table 9). Mean diameter data are also missing for the historic sampling periods (Table 7).

Biomass (T/acre) of live trees decreased from 91.1 in 1980 to 61.9 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 0.5 in 1980 to 5.5 in 2016 (Table 10). Biomass (T/acre) of shrubs increased from less than 0.04 live and 0.1 dead in 1980 to 0.1 live and no measurable dead in 2016 (Table 10). Total average downed woody fuel load increased from 3.3 T/acre in 1980 to 3.6 T/acre in 2016 (Table 11). Duff depth (in.) increased from 1.6 in 1980 to 3.1 in 2016 (Table 11). Fuel height (in.) decreased from 3.9 in 1980 to 1.1 in 2016 (Table 11).

Table 6. Average basal area (X̄ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC07 (Gilmore Meadow) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 2016 data.

| AC07- Gilmore Meadow | | | | | |
|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | **2016** | | |
| *No. Points Sampled:* | *20* | *30* | *20* | 95% CI | |
| Live | X̄ BA | X̄ BA | X̄ BA ± SE | Lower | Upper |
| Red Oak | 67.5 | 60.7 | 80.0 ± 6.1 | 68.0 | 92.0 |
| Red Maple | 9.0 | 23.3 | 9.0 ± 1.2 | 6.6 | 11.4 |
| White Pine | 1.5 | 0.7 | 4.0 ± 1.5 | 1.0 | 7.0 |
| Red Spruce | 0 | 0.3 | 2.5 ± 1.4 | 0 | 5.3 |
| American Beech | 0 | 0 | 1.0 ± 1.0 | 0 | 3.0 |
| White Ash | 0 | 0.7 | 0.5 ± 0.5 | 0 | 1.5 |
| Striped Maple | 0 | 0 | 0.5 ± 0.5 | 0 | 1.5 |
| Bigtooth Aspen | 2.5 | 7.0 | 0.5 ± 0.5 | 0 | 1.5 |
| No. White Cedar | 0 | 0.7 | 0 | | |
| Paper Birch | 1.5 | 1.3 | 0 | | |
| Gray Birch | 2.5 | 0.3 | 0 | | |
| White Spruce | 0 | 0.3 | 0 | | |
| **Totals:** | **84.5** | **95.3** | **98.0 ± 4.9** | **88.3** | **107.7** |
| Dead | | | | | |
| Red Oak | 2.0 | 1.3 | 3.5 ± 1.5 | 0.6 | 6.4 |
| Bigtooth Aspen | 0 | 1.0 | 3.0 ± 1.5 | 0.1 | 5.9 |
| Paper Birch | 0 | 0 | 1.5 ± 1.1 | 0 | 3.6 |
| Red Maple | 0 | 0.7 | 1.0 ± 0.7 | 0 | 2.4 |
| Red Spruce | 0 | 0 | 0.5 ± 0.5 | 0 | 1.5 |
| Balsam Fir | 0 | 0 | 0.5 ± 0.5 | 0 | 1.5 |
| American Beech | 0 | 0 | 0.5 ± 0.5 | 0 | 1.5 |
| White Pine | 0 | 0.3 | 0 | | |
| White Ash | 0 | 0.3 | 0 | | |
| Gray Birch | 0 | 1.0 | 0 | | |
| **Totals:** | **2.0** | **4.6** | **10.5 ± 2.0** | **6.6** | **14.4** |

Table 7. Mean diameter ($\bar{X}$ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC07 (Gilmore Meadow) in 2016. Data missing for 1980/1992-1994. Confidence Intervals are for 2016 data.

| AC07 Species | 1980 | 1993 | 2016 $\bar{X}$ Dia. ± SE | 95% CI Lower | Upper |
|---|---|---|---|---|---|
| Red Oak | x | x | 11.7 ± 0.9 | 10.0 | 13.5 |
| White Pine | x | x | 10.8 ± 1.8 | 7.3 | 14.3 |
| Paper Birch | x | x | 10.0 ± 0.4 | 9.3 | 10.7 |
| White Ash | x | x | 8.9 ± 0 | 8.9 | 8.9 |
| Bigtooth Aspen | x | x | 8.3 ± 0.3 | 7.6 | 8.9 |
| American Beech | x | x | 7.8 ± 1.0 | 5.8 | 9.8 |
| Balsam Fir | x | x | 5.7 ± 0 | 5.7 | 5.7 |
| Red Maple | x | x | 5.3 ± 0.7 | 3.9 | 6.7 |
| Red Spruce | x | x | 5.2 ± 0.6 | 3.9 | 6.5 |
| Striped Maple | x | x | 1.9 ± 0 | 1.9 | 1.9 |
| **Overall Average** | | | **10.7 ± 1.0** | **8.6** | **12.7** |

*Red Oak Summary*

The Gilmore Meadow red oak stand is continuing to accumulate biomass as is expected for this 70-year-old post-1947 fire stand. Fuel loading and fire hazard remains low. Given the few conifer saplings in the understory it is possible that in time this stand could transition to a mixed-hardwood conifer stand like those described below. It is possible given 2016 sapling abundance that striped maple, white pine, red spruce, and balsam fir could establish in the face of dense understory competition. Red oak and red maple would, however, likely remain an important component due to the well-drained soils and acidic (Lyman-Tunbridge-Schoodic complex, 8 to 15% slopes, very stony; Soil Survey Staff, 2019).

Table 8. Average density ($\bar{X}$ Den) of live and dead trees (stems per acre) for all stands sampled in 1980, 1992-1994 and 2016 with standard error and 95% confidence intervals. 'X' denotes data missing.

| | | 1980 | | | 1993 | | | 2016 | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | 95% CI | | | 95% CI | | | 95% CI | |
| Stand | | $\bar{X}$Den ± SE | Lower | Upper | $\bar{X}$Den ± SE | Lower | Upper | $\bar{X}$Den ± SE | Lower | Upper |
| **AC01** | Live | 424.9 ± 91.5 | 245.6 | 604.2 | 380.4 ± 75.7 | 232.1 | 528.7 | 482.3 ± 156.4 | 175.8 | 788.8 |
| | Dead | 47.1 ± 21.8 | 4.3 | 89.9 | 50.6 ± 17.1 | 17.0 | 84.2 | 116.8 ± 68.0 | 0 | 250.0 |
| **AC02** | Live | 1412.9 ± 226.3 | 969.3 | 1856.5 | 898.1 ± 167.0 | 570.8 | 1225.4 | 760.7 ± 124.4 | 516.8 | 1004.6 |
| | Dead | 149.8 ± 81.4 | 0 | 309.3 | 164.9 ± 45.2 | 76.2 | 253.6 | 122.1 ± 26.3 | 70.5 | 173.7 |
| **AC03** | Live | 950.6 ± 169.4 | 618.5 | 1282.7 | 939.6 ± 108.6 | 726.8 | 1152.4 | 425.4 ± 55.9 | 315.9 | 534.9 |
| | Dead | 44.9 ± 27.3 | 0 | 98.5 | 88.0 ± 38.4 | 12.7 | 163.3 | 14.2 ± 4.2 | 5.9 | 22.5 |
| **AC04** | Live | 706.8 ± 185.0 | 344.1 | 1069.5 | 753.0 ± 218.7 | 324.4 | 1181.6 | 862.3 ± 284.0 | 305.6 | 1419.0 |
| | Dead | 34.0 ± 10.5 | 13.4 | 54.6 | 55.4 ± 21.3 | 13.6 | 97.2 | 65.9 ± 26.0 | 15.0 | 116.8 |
| **AC05** | Live | 872.4 ± 139.2 | 599.6 | 1145.2 | 897.2 ± 151.5 | 600.2 | 1194.2 | 347.7 ± 54.9 | 240.1 | 455.3 |
| | Dead | 15.2 ± 10.0 | 0 | 34.7 | 62.2 ± 32.8 | 0 | 126.4 | 36.8 ± 7.3 | 22.6 | 51.0 |
| **AC06** | Live | 746.7 ± 93.2 | 564.0 | 929.4 | 477.2 ± 58.4 | 362.8 | 591.6 | 449.3 ± 99.8 | 253.6 | 645.0 |
| | Dead | 146.9 ± 28.5 | 91.0 | 202.8 | 100.8 ± 40.5 | 21.5 | 180.1 | 65.1 ± 18.5 | 28.9 | 101.3 |
| **AC07** | Live | X | X | | X | X | | 362.6 ± 67.6 | 230.2 | 495.0 |
| | Dead | X | X | | X | X | | 56.5 ± 13.1 | 30.7 | 82.3 |
| **AC08** | Live | X | X | | X | X | | 291.2 ± 217.9 | 0 | 718.3 |
| | Dead | X | X | | X | X | | 68.9 ± 55.5 | 0 | 177.6 |
| **AC09** | Live | X | X | | X | X | | 558.3 ± 80.9 | 399.8 | 716.8 |
| | Dead | X | X | | X | X | | 38.9 ± 16.2 | 7.2 | 70.6 |
| **AC11** | Live | X | X | | X | X | | 426.8 ± 78.1 | 273.8 | 579.8 |
| | Dead | X | X | | X | X | | 79.1 ± 19.1 | 41.7 | 116.5 |
| **AC12** | Live | 705.1 ± 0 | 705.1 | 705.1 | 442.5 ± 51.4 | 341.8 | 543.2 | 321.5 ± 60.3 | 203.3 | 439.7 |
| | Dead | 258.7 ± 77.2 | 107.3 | 410.1 | 69.1 ± 16.3 | 37.1 | 101.1 | 57.3 ± 22.1 | 14.0 | 100.6 |
| **AC13** | Live | 520.4 ± 71.4 | 380.5 | 660.3 | 716.4 ± 154.0 | 414.6 | 1018.2 | 117.5 ± 24.3 | 69.9 | 165.1 |
| | Dead | 552.6 ± 208.6 | 143.7 | 961.5 | 300.7 ± 69.5 | 164.6 | 436.8 | 101.7 ± 26.5 | 49.7 | 153.7 |

Table 8 Continued. Average density ($\bar{X}$ Den) of live and dead trees (stems per acre) for all stands sampled in 1980, 1992-1994 and 2016 with standard error and 95% confidence intervals. 'X' denotes data missing.

| | | 1980 | | | 1993 | | | 2016 | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | 95% CI | | | 95% CI | | | 95% CI | |
| Stand | | $\bar{X}$Den ± SE | Lower | Upper | $\bar{X}$Den ± SE | Lower | Upper | $\bar{X}$Den ± SE | Lower | Upper |
| **AC15** | Live | X | X | | X | X | | 488.3 ± 88.4 | 315.0 | 661.6 |
| | Dead | X | X | | X | X | | 62.8 ± 13.8 | 35.8 | 89.8 |
| **AC16** | Live | X | X | | 1128.4 ± 116.8 | 899.5 | 1357.3 | 346.9 ± 65.3 | 218.9 | 474.9 |
| | Dead | X | X | | 248.6 ± 68.4 | 114.6 | 382.6 | 73.4 ± 24.2 | 25.9 | 120.9 |
| **AC17** | Live | 241.4 ± 100.6 | 44.2 | 438.6 | 850.0 ± 164.0 | 528.6 | 1171.4 | 847.4 ± 327.8 | 205.0 | 1489.8 |
| | Dead | 54.7 ± 24.2 | 7.2 | 102.2 | 408.9 ± 202.2 | 12.6 | 805.2 | 112.8 ± 34.0 | 46.2 | 179.4 |
| **AC18** | Live | 511.5 ± 48.5 | 416.5 | 606.5 | 639.7 ± 110.6 | 423.0 | 856.4 | 288.0 ± 47.0 | 195.9 | 380.1 |
| | Dead | 210.1 ± 63.1 | 86.4 | 333.8 | 145.5 ± 56.1 | 35.5 | 255.5 | 79.9 ± 25.5 | 30.0 | 129.8 |
| **AC19** | Live | X | X | | 659.4 ± 142.0 | 381.1 | 937.7 | 640.6 ± 124.1 | 397.4 | 883.8 |
| | Dead | X | X | | 76.7 ± 21.8 | 34.0 | 119.4 | 195.7 ± 101.4 | 0 | 394.4 |
| **AC20** | Live | X | X | | 767.8 ± 224.5 | 327.8 | 1207.8 | 342.0 ± 61.3 | 221.9 | 462.1 |
| | Dead | X | X | | 72.3 ± 24.1 | 25.1 | 119.5 | 85.9 ± 31.6 | 24.0 | 147.8 |
| **AC22** | Live | 261.3 ± 65.5 | 133.0 | 389.6 | 1252.6 ± 262.6 | 737.8 | 1767.4 | 1086.7 ± 266.0 | 565.2 | 1608.2 |
| | Dead | 114.7 ± 35.1 | 46.0 | 183.4 | 88.1 ± 36.4 | 16.7 | 159.5 | 187.8 ± 87.3 | 16.6 | 359.0 |
| **AC23** | Live | 730.7 ± 89.1 | 556.0 | 905.4 | 805.3 ± 112.6 | 584.6 | 1026.0 | 353.0 ± 62.8 | 229.9 | 476.1 |
| | Dead | 77.9 ± 21.0 | 36.8 | 119.0 | 205.0 ± 53.0 | 101.0 | 309.0 | 72.5 ± 21.3 | 30.7 | 114.3 |
| **AC24** | Live | 567.8 ± 80.6 | 409.8 | 725.8 | 547.4 ± 73.4 | 403.5 | 691.3 | 45.1 ± 6.1 | 33.1 | 57.1 |
| | Dead | 71.6 ± 40.2 | 0 | 150.5 | 64.8 ± 25.0 | 15.9 | 113.7 | 3.8 ± 1.1 | 1.6 | 6.0 |
| **AC25** | Live | X | X | | X | X | | 325.5 ± 82.7 | 163.3 | 487.7 |
| | Dead | X | X | | X | X | | 5.2 ± 5.2 | 0 | 15.4 |
| **AC26** | Live | 1168.1 ± 614.5 | 0 | 2372.4 | 1234.6 ± 269.7 | 706.1 | 1763.1 | 711.9 ± 210.1 | 300.1 | 1123.7 |
| | Dead | 109.7 ± 38.9 | 33.4 | 186.0 | 339.1 ± 188.8 | 0 | 709.1 | 178.1 ± 39.2 | 101.2 | 255.0 |

Table 9. Average density (X̄ Den) of live and dead saplings (stems per acre) for all stands sampled in 1992-1994 and 2016 with standard error and 95% confidence intervals. 'X' denotes data missing. Sapling data were not collected in 1980.

| 1993 | | | | | 2016 | | |
|---|---|---|---|---|---|---|---|
| | | | 95% CI | | | 95% CI | |
| Stand | | X̄Den ± SE | Lower | Upper | X̄Den ± SE | Lower | Upper |
| **AC01** | Live | 296.2 ± 66.9 | 165.0 | 427.4 | 295.0 ± 91.9 | 114.9 | 475.1 |
| | Dead | 73.1 ± 32.1 | 10.1 | 136.1 | 35.0 ± 18.2 | 0 | 70.6 |
| **AC02** | Live | 334.3 ± 107.1 | 124.5 | 544.1 | 165.0 ± 53.0 | 61.2 | 268.8 |
| | Dead | 91.4 ± 26.4 | 39.7 | 143.1 | 80.0 ± 43.9 | 0 | 166.0 |
| **AC03** | Live | 394.7 ± 45.6 | 305.4 | 484.0 | 345.0 ± 67.8 | 212.0 | 478.0 |
| | Dead | 86.8 ± 26.4 | 35.0 | 138.6 | 40.0 ± 16.9 | 7.0 | 73.0 |
| **AC04** | Live | 568.2 ± 105.6 | 361.3 | 775.1 | 381.0 ± 146.6 | 93.6 | 668.4 |
| | Dead | 309.1 ± 211.3 | 0 | 723.2 | 333.3 ± 166.1 | 7.8 | 658.8 |
| **AC05** | Live | 176.0 ± 50.1 | 77.8 | 274.2 | 255.0 ± 63.9 | 129.8 | 380.2 |
| | Dead | 44.0 ± 16.4 | 11.8 | 76.2 | 20.0 ± 11.7 | 0 | 42.9 |
| **AC06** | Live | X | X | | 520.0 ± 105.5 | 313.1 | 726.9 |
| | Dead | X | X | | 20.0 ± 15.6 | 0 | 50.5 |
| **AC07** | Live | X | X | | 225.0 ± 54.2 | 118.7 | 331.3 |
| | Dead | X | X | | 20.0 ± 9.2 | 2.0 | 38.0 |
| **AC08** | Live | X | X | | 205.0 ± 76.9 | 54.2 | 355.8 |
| | Dead | X | X | | 95.0 ± 35.2 | 26.1 | 163.9 |
| **AC09** | Live | X | X | | 130.0 ± 44.2 | 43.4 | 216.6 |
| | Dead | X | X | | 45.0 ± 17.0 | 11.7 | 78.3 |
| **AC11** | Live | X | X | | 160.0 ± 55.4 | 51.4 | 268.6 |
| | Dead | X | X | | 80.0 ± 34.5 | 12.4 | 147.6 |
| **AC12** | Live | 70.0 ± 45.4 | 0 | 158.9 | 430.0 ± 139.8 | 156.1 | 703.9 |
| | Dead | 10.0 ± 2.4 | 5.3 | 14.7 | 25.0 ± 25.0 | 0 | 74.0 |
| **AC13** | Live | X | X | | 810.0 ± 196.9 | 424.0 | 1196.0 |
| | Dead | X | X | | 15.0 ± 10.9 | 0 | 36.4 |

Table 9 Continued. Average density (X̄ Den) of live and dead saplings (stems per acre) for all stands sampled in 1992-1994 and 2016 with standard error and 95% confidence intervals. 'X' denotes data missing. Sapling data were not collected in 1980.

| | | 1993 | | | 2016 | | |
|---|---|---|---|---|---|---|---|
| | | | 95% CI | | | 95% CI | |
| Stand | | X̄Den ± SE | Lower | Upper | X̄Den ± SE | Lower | Upper |
| **AC15** | Live | X | X | | 330.0 ± 123.3 | 88.3 | 571.7 |
| | Dead | X | X | | 10.0 ± 6.9 | 0 | 23.5 |
| **AC16** | Live | 425.6 ± 87.6 | 253.9 | 597.3 | 355.0 ± 76.9 | 204.2 | 505.8 |
| | Dead | 41.9 ± 13.4 | 15.6 | 68.2 | 30.0 ± 16.4 | 0 | 62.1 |
| **AC17** | Live | 204.5 ± 81.3 | 45.2 | 363.8 | 820.0 ± 261.1 | 308.2 | 1331.8 |
| | Dead | 90.0 ± 58.1 | 0 | 203.8 | 110.0 ± 38.3 | 34.9 | 185.1 |
| **AC18** | Live | X | X | | 835.0 ± 202.1 | 438.9 | 1231.1 |
| | Dead | X | X | | 70.0 ± 26.6 | 17.9 | 122.1 |
| **AC19** | Live | X | X | | 753.3 ± 288.4 | 188.0 | 1318.6 |
| | Dead | X | X | | 186.7 ± 71.1 | 47.3 | 326.1 |
| **AC20** | Live | 422.7 ± 86.1 | 253.9 | 591.5 | 335.0 ± 71.9 | 194.1 | 475.9 |
| | Dead | 63.6 ± 33.1 | 0 | 128.4 | 0 ± 0 | 0 | 0 |
| **AC22** | Live | 419.2 ± 139.5 | 145.9 | 692.5 | 705.0 ± 302.6 | 111.9 | 1298.1 |
| | Dead | 396.2 ± 152.6 | 97.2 | 695.2 | 425.0 ± 163.2 | 105.2 | 744.8 |
| **AC23** | Live | 325.5 ± 47.3 | 232.8 | 418.2 | 180.0 ± 40.1 | 101.3 | 258.7 |
| | Dead | 150.9 ± 36.0 | 80.3 | 221.5 | 20.0 ± 9.2 | 2.0 | 38.0 |
| **AC24** | Live | X | X | | 185.0 ± 51.9 | 83.2 | 286.8 |
| | Dead | X | X | | 0 ± 0 | 0 | 0 |
| **AC25** | Live | X | X | | 415.0 ± 221.0 | 0 | 848.2 |
| | Dead | X | X | | 55.0 ± 29.4 | 0 | 112.7 |
| **AC26** | Live | 333.3 ± 82.6 | 171.4 | 495.2 | 1645.0 ± 422.4 | 817.0 | 2473.0 |
| | Dead | 120.0 ± 78.2 | 0 | 273.2 | 40.0 ± 15.2 | 10.2 | 69.8 |

Table 10. Fuel loading in 1980 and 2016 for standing live and dead woody stems. Sapling data were not measured in 1980. * indicates stands burned in 1947.

| | 1980 Standing Woody Fuel (T/acre) | | | | 2016 Standing Woody Fuel (T/acre) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Live | | Dead | | Live | | | Dead | | |
| Stand | Trees | Shrubs | Trees | Shrubs | Trees ± SE | Saplings ± SE | Shrubs ± SE | Trees ± SE | Saplings ± SE | Shrubs ± SE |
| AC01 | 91.0 | 0.2 | 8.1 | 0 | 77.3 ± 5.7 | 0.1 ± 0 | 0.6 ± 0.4 | 9.6 ± 1.8 | 0 ± 0 | 0 ± 0 |
| *AC02 | 57.0 | 0 | 2.5 | 0.1 | 66.0 ± 4.6 | 0.1 ± 0 | 0.5 ± 0.3 | 9.5 ± 1.7 | 0 ± 0 | 0 ± 0 |
| AC03 | 59.4 | 0 | 3.6 | 0 | 73.3 ± 5.1 | 0.1 ± 0 | 0 ± 0 | 6.1 ± 1.5 | 0 ± 0 | 0 ± 0 |
| AC04 | 88.3 | 8.9 | 8.1 | 0 | 78.1 ± 3.7 | 0.2 ± 0.1 | 0.6 ± 0.3 | 7.9 ± 1.9 | 0.1 ± 0.1 | 0 ± 0 |
| AC05 | 49.1 | 0 | 1.6 | 0 | 43.0 ± 3.5 | 0.1 ± 0 | 0.6 ± 0.5 | 6.7 ± 1.1 | 0 ± 0 | 0 ± 0 |
| AC06 | 100.5 | 0 | 10.2 | 0 | 71.9 ± 6.5 | 0.2 ± 0 | 1.1 ± 0.4 | 12.8 ± 1.7 | 0 ± 0 | 0 ± 0 |
| *AC07 | 91.1 | 0 | 0.5 | 0.1 | 61.9 ± 3.7 | 0.1 ± 0 | 0.1 ± 0 | 5.5 ± 1.1 | 0 ± 0 | 0 ± 0 |
| *AC08 | 7.2 | 0.2 | 0 | 0 | 21.6 ± 3.0 | 0 ± 0 | 0.9 ± 0.3 | 0.3 ± 0.2 | 0 ± 0 | 0 ± 0 |
| *AC09 | 34.3 | 0 | 0.7 | 0 | 72.6 ± 5.3 | 0.1 ± 0 | 0 ± 0 | 2.8 ± 0.9 | 0 ± 0 | 0 ± 0 |
| AC11 | 100.0 | 0.2 | 8.7 | 0 | 54.1 ± 4.1 | 0.1 ± 0 | 0.4 ± 0.2 | 17.6 ± 2.1 | 0 ± 0 | 0 ± 0 |
| AC12 | 60.9 | 0.2 | 12.1 | 0 | 45.7 ± 4.3 | 0.1 ± 0 | 2.1 ± 1.1 | 6.1 ± 1.3 | 0 ± 0 | 0.1 ± 0.1 |
| AC13 | 66.5 | 0.1 | 17.1 | 0 | 18.4 ± 4.0 | 0.3 ± 0.1 | 0.6 ± 0.2 | 8.3 ± 1.9 | 0 ± 0 | 0 ± 0 |
| AC15 | 76.3 | 0.1 | 10.4 | 0 | 65.0 ± 5.4 | 0.1 ± 0 | 8.7 ± 5.0 | 13.0 ± 2.4 | 0 ± 0 | 0 ± 0 |
| AC16 | 81.2 | 0.3 | 7.0 | 0 | 73.0 ± 5.0 | 0.1 ± 0 | 0.6 ± 0.2 | 16.1 ± 2.2 | 0 ± 0 | 0 ± 0 |
| AC17 | 96.9 | 0.3 | 10.6 | 0 | 76.0 ± 4.6 | 0.5 ± 0.1 | 13.0 ± 7.7 | 15.4 ± 2.6 | 0.1 ± 0 | 0 ± 0 |
| AC18 | 90.7 | 0.2 | 8.2 | 0 | 59.9 ± 3.6 | 0.3 ± 0.1 | 16.9 ± 4.8 | 7.7 ± 1.1 | 0 ± 0 | 0 ± 0 |
| AC19 | 75.8 | 0.1 | 5.0 | 0 | 65.7 ± 4.0 | 0.4 ± 0.1 | 2.9 ± 1.1 | 9.5 ± 2.1 | 0.1 ± 0 | 0 ± 0 |
| AC20 | 24.6 | 1.1 | 2.3 | 0 | 38.6 ± 2.7 | 0.1 ± 0 | 0.5 ± 0.1 | 4.8 ± 1.5 | 0 ± 0 | 0 ± 0 |
| AC22 | 17.5 | 0 | 9.5 | 0.1 | 35.6 ± 5.4 | 0.2 ± 0.1 | 0.8 ± 0.5 | 3.6 ± 1.1 | 0.2 ± 0.1 | 0.2 ± 0.2 |
| AC23 | 82.5 | 0.1 | 6.0 | 0 | 73.8 ± 3.7 | 0.1 ± 0 | 0.1 ± 0 | 12.5 ± 2.3 | 0 ± 0 | 0 ± 0 |
| AC24 | 48.7 | 2.0 | 2.2 | 0.3 | 35.7 ± 2.9 | 0 ± 0 | 3.7 ± 0.5 | 1.8 ± 0.5 | 0 ± 0 | 0 ± 0 |
| *AC25 | 0.8 | 0.6 | 0 | 0.2 | 12.9 ± 2.2 | 0.1 ± 0 | 0.4 ± 0.1 | 0.1 ± 0.1 | 0 ± 0 | 0 ± 0 |
| AC26 | 12.5 | 0 | 7.2 | 0 | 48.6 ± 5.4 | 0.6 ± 0.2 | 1.0 ± 0.3 | 16.5 ± 2.2 | 0 ± 0 | 0.3 ± 0.2 |

Table 11. Average duff depth, fuel height, depth to bedrock, canopy density and downed woody fuel load for stands sampled in 1980, 1993 and 2016. 'X' denotes data not collected.

| | AC01 Eagle Lake (NW) spruce-fir | | | AC02 Connors Nubble northern hardwoods | | | AC03 Eagle Lake (S) northern hardwoods | | | AC04 Stanley Brook spruce-fir | | AC05 Pemetic Mtn. northern white cedar | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sample Year: | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1993 | 2016 | 1980 | 1993 | 2016 |
| No. Points Sampled: | 21 | 26 | 20 | 22 | 35 | 20 | 21 | 38 | 20 | 22 | 21 | 20 | 25 | 20 |
| average duff depth (in.) | 2.6 | 3.6 | 2.5 | 2.0 | 0.1 | 3.4 | 1.0 | 0.5 | 2.2 | 5.5 | 4.5 | 1.5 | 1.4 | 1.2 |
| average fuel height (in.) | 6.5 | 7.3 | 3.7 | 6.0 | 3.7 | 1.5 | 2.8 | 5.2 | 1.5 | 6.4 | 1.1 | 2.8 | 2.9 | 2.2 |
| average depth to bedrock (in.) | X | X | 9.2 | X | X | 17.4 | X | X | 12.2 | X | 10.1 | X | X | 2.2 |
| average canopy density (%) | X | X | 92.0 | X | X | 93.0 | X | X | 96.0 | X | 88.0 | X | X | 87.0 |
| Size Class (T/acre) | | | | | | | | | | | | | | |
| 1-hr | 2.2 | 1.8 | 0.4 | 1.1 | 0.8 | 0.4 | 0.8 | 1.0 | 0.4 | 0.7 | 0.6 | 0.9 | 0.6 | 0.4 |
| 10-hr | 2.1 | 1.6 | 0.1 | 3.6 | 2.2 | 0.2 | 2.1 | 2.4 | 0.1 | 0.6 | 0.1 | 0.6 | 0.4 | 0.1 |
| 100-hr | 1.3 | 0.8 | 2.0 | 1.3 | 2.0 | 1.8 | 2.6 | 2.2 | 1.2 | 0.7 | 0.9 | 1.1 | 0.6 | 1.4 |
| 1000-hr (sound) | 3.9 | 2.6 | 8.6 | 1.1 | 0.3 | 0.8 | 3.2 | 1.7 | 3.3 | 2.3 | 2.8 | 0.8 | 1.5 | 2.9 |
| 1000-hr (rotten) | 3.5 | 2.5 | 2.0 | 2.5 | 1.5 | 1.3 | 3.1 | 2.7 | 3.0 | 3.7 | 4.0 | 3.7 | 2.9 | 1.8 |
| Total (T/acre) | 12.9 | 9.3 | 13.1 | 9.7 | 6.7 | 4.3 | 11.9 | 10.0 | 8.0 | 8.0 | 8.5 | 7.2 | 6.1 | 6.6 |

Table 11 Continued. Average duff depth, fuel height, depth to bedrock, canopy density and downed woody fuel load for stands sampled in 1980, 1993 and 2016. 'X' denotes data not collected.

| | AC06 Sargent Mtn. northern white cedar | | | AC07 Gilmore Meadow red oak | | | AC08 Sand Beach pitch pine | | | AC09 Otter Creek birch-aspen | | AC11 Norumbega Mtn. (SW) mixed conifer | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sample Year: | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1993 | 2016 | 1980 | 1993 | 2016 |
| No. Points Sampled: | 21 | 20 | 20 | 20 | 30 | 20 | 20 | 24 | 20 | 24 | 20 | 21 | 34 | 20 |
| average duff depth (in.) | 5.2 | 4.8 | 3.7 | 1.6 | 1.0 | 3.0 | 0.9 | 1.9 | 3.6 | 0.7 | 2.0 | 5.9 | 3.4 | 5.0 |
| average fuel height (in.) | 3.3 | 4.4 | 5.9 | 3.9 | 3.4 | 1.1 | 1.5 | 2.7 | 0.8 | 3.6 | 1.7 | 5.3 | 5.3 | 2.5 |
| average depth to bedrock (in.) | X | X | 11.0 | X | X | 7.9 | X | X | 5.4 | X | 11.1 | X | X | 8.7 |
| average canopy density (%) | X | X | 84.0 | X | X | 86.0 | X | X | 42.0 | X | 92.0 | X | X | 83.0 |
| Size Class (T/acre) | | | | | | | | | | | | | | |
| 1-hr | 0.5 | 0.8 | 0.4 | 0.3 | 0.9 | 0.3 | 0.3 | 0.3 | 0.5 | 0.9 | 0.3 | 0.8 | 0.4 | 0.7 |
| 10-hr | 0.8 | 1.4 | 0.1 | 1.7 | 1.7 | 0.1 | 0.5 | 0.6 | 0.1 | 2.1 | 0.1 | 1.3 | 0.8 | 0.1 |
| 100-hr | 1.4 | 1.1 | 1.4 | 0.5 | 1.5 | 1.7 | 0.6 | 0.9 | 1.2 | 2.0 | 1.3 | 1.7 | 1.3 | 1.2 |
| 1000-hr (sound) | 4.4 | 6.1 | 12.3 | 0.5 | 0.1 | 0.1 | 6.5 | 1.2 | 0.1 | 0.4 | 1.0 | 4.5 | 5.1 | 3.6 |
| 1000-hr (rotten) | 1.7 | 1.5 | 1.6 | 0.3 | 0.1 | 1.4 | 3.8 | 1.3 | 2.2 | 2.2 | 1.1 | 2.6 | 2.2 | 3.4 |
| Total (T/acre) | 8.8 | 10.9 | 15.8 | 3.3 | 4.4 | 3.6 | 11.6 | 4.2 | 4.0 | 7.6 | 3.8 | 10.8 | 9.7 | 9.0 |

Table 11 Continued. Average duff depth, fuel height, depth to bedrock, canopy density and downed woody fuel load for stands sampled in 1980, 1993 and 2016. 'X' denotes data not collected.

| | AC12 Deep Cove (IaH) spruce-fir | | | AC13 Western Head (IaH) spruce-fir | | | AC15 Jerusalem Mtn. (IaH) mixed hardwood-conifer | | | AC16 Long Pond mixed hardwood-conifer | | AC17 Schoodic Peninsula (W) spruce-fir | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sample Year: | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1993 | 2016 | 1980 | 1993 | 2016 |
| No. Points Sampled: | 20 | 20 | 20 | 26 | 21 | 20 | 28 | 28 | 20 | 43 | 20 | 22 | 22 | 20 |
| ave. duff depth (in.) | 5.2 | 7.2 | 7.7 | 6.2 | 7.3 | 4.4 | 4.0 | 5.0 | 5.9 | 3.4 | 4.9 | 4.2 | 5.4 | 7.3 |
| ave. fuel height (in.) | 6.5 | 5.6 | 4.3 | 4.0 | 5.7 | 10.0 | 3.6 | 5.0 | 5.2 | 3.9 | 1.0 | 5.9 | 5.7 | 1.7 |
| average depth to bedrock (in.) | X | X | 8.6 | X | X | 8.1 | X | X | 7.4 | X | 8.8 | X | X | 10.0 |
| average canopy density (%) | X | X | 74.0 | X | X | 53.0 | X | X | 81.0 | X | 92.0 | X | X | 82.0 |
| Size Class (T/acre) | | | | | | | | | | | | | | |
| 1-hr | 1.8 | 2.9 | 1.2 | 1.6 | 1.1 | 1.9 | 1.0 | 2.2 | 0.7 | 1.2 | 0.7 | 1.5 | 1.8 | 0.9 |
| 10-hr | 0.4 | 2.0 | 0.2 | 1.7 | 2.1 | 0.4 | 1.1 | 0.9 | 0.2 | 1.0 | 0.2 | 0.9 | 2.1 | 0.1 |
| 100-hr | 1.9 | 2.3 | 1.4 | 2.0 | 1.2 | 7.7 | 1.3 | 1.2 | 2.3 | 2.9 | 1.6 | 1.0 | 1.4 | 1.6 |
| 1000-hr (sound) | 4.8 | 2.0 | 4.2 | 3.9 | 3.5 | 12.2 | 4.5 | 3.1 | 4.3 | 3.4 | 4.2 | 3.3 | 5.8 | 9.3 |
| 1000-hr (rotten) | 3.7 | 5.9 | 5.8 | 2.5 | 1.5 | 5.8 | 3.6 | 4.3 | 4.8 | 3.4 | 5.0 | 2.4 | 2.3 | 8.0 |
| Total (T/acre) | 12.5 | 15.1 | 12.8 | 11.6 | 9.3 | 28.0 | 11.4 | 11.7 | 12.3 | 11.9 | 11.5 | 9.1 | 13.3 | 19.9 |

Table 11 Continued. Average duff depth, fuel height, depth to bedrock, canopy density and downed woody fuel load for stands sampled in 1980, 1993 and 2016. 'X' denotes data not collected.

| | AC18 (upper) Western Mtn. spruce-fir | | | AC19 (lower) Western Mtn. mixed hardwood-conifer | | | AC20 Hodgdon Pond spruce-fir | | | AC22 Otter Point spruce-fir | | AC23 Day Mtn. mixed hardwood-conifer | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sample Year: | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1993 | 2016 | 1980 | 1993 | 2016 |
| No. Points Sampled: | 23 | 36 | 20 | 20 | 20 | 30 | 23 | 24 | 20 | 26 | 20 | 25 | 55 | 20 |
| ave. duff depth (in.) | 5.1 | 7.8 | 6.6 | 2.6 | 4.8 | 5.5 | 3.2 | 3.6 | 4.4 | 2.9 | 4.6 | 2.4 | 3.3 | 2.5 |
| ave. fuel height (in.) | 2.4 | 5.0 | 3.6 | 3.7 | 5.2 | 2.5 | 3.0 | 3.8 | 4.2 | 9.3 | 2.8 | 2.9 | 4.8 | 4.6 |
| average depth to bedrock (in.) | X | X | 10.2 | X | X | 12.6 | X | X | 8.7 | X | 6.7 | X | X | 12.7 |
| average canopy density (%) | X | X | 85.0 | X | X | 89.0 | X | X | 74.0 | X | 67.0 | X | X | 88.0 |
| Size Class (T/acre) | | | | | | | | | | | | | | |
| 1-hr | 1.9 | 3.4 | 0.8 | 1.0 | 1.4 | 0.7 | 0.5 | 1.0 | 0.7 | 0.7 | 1.1 | 0.6 | 1.1 | 0.6 |
| 10-hr | 0.4 | 1.4 | 0.2 | 0.5 | 0.9 | 0.1 | 0.6 | 0.7 | 0.1 | 1.6 | 0.2 | 1.0 | 2.0 | 0.1 |
| 100-hr | 2.4 | 1.8 | 1.7 | 1.2 | 0.6 | 1.3 | 0.5 | 0.8 | 0.2 | 2.1 | 1.8 | 1.7 | 1.9 | 2.3 |
| 1000-hr (sound) | 0.6 | 1.9 | 8.2 | 2.9 | 2.2 | 3.8 | 0.7 | 0.8 | 1.4 | 6.1 | 1.7 | 1.0 | 2.9 | 3.7 |
| 1000-hr (rotten) | 1.4 | 3.6 | 2.0 | 0.9 | 3.4 | 1.4 | 0.7 | 1.3 | 0.1 | 18.3 | 4.0 | 1.2 | 2.3 | 5.8 |
| Total (T/acre) | 6.6 | 12.0 | 12.9 | 6.6 | 8.5 | 7.3 | 3.0 | 4.4 | 2.4 | 28.7 | 8.7 | 5.5 | 10.2 | 12.4 |

Table 11 Continued. Average duff depth, fuel height, depth to bedrock, canopy density and downed woody fuel load for stands sampled in 1980, 1993 and 2016. 'X' denotes data not collected.

| | AC24 Champlain Mtn. (IaH) pitch pine | | | AC25 Cadillac Mtn. North Face spruce-fir | | | AC26 Bernard Mtn. blown-down spruce-fir | | |
|---|---|---|---|---|---|---|---|---|---|
| Sample Year: | 1980 | 1993 | 2016 | 1980 | 1993 | 2016 | 1981 | 1993 | 2016 |
| No. Points Sampled: | 20 | 20 | 20 | 20 | 22 | 20 | 15 | 15 | 20 |
| ave. duff depth (in.) | 1.7 | 2.7 | 5.1 | 0.1 | 1.3 | 3.5 | 5.2 | 6.3 | 5.3 |
| ave. fuel height (in.) | 2.3 | 4.9 | 2.8 | 1.5 | 3.1 | 1.1 | 35.1 | 8.9 | 2.9 |
| average depth to bedrock (in.) | x | x | 5.4 | x | x | 5.2 | x | x | 10.8 |
| average canopy density (%) | x | x | 0.6 | x | x | 0.4 | x | x | 0.8 |
| Size Class (Tons/Acre) | | | | | | | | | |
| 1-hr | 0.3 | 0.2 | 0.5 | 0.2 | 0.3 | 0.2 | 0.7 | 1.5 | 1.0 |
| 10-hr | 0.6 | 0.5 | 0.1 | 0.8 | 1.2 | 0.1 | 2.9 | 3.6 | 0.2 |
| 100-hr | 0.1 | 0.3 | 0.6 | 0.7 | 0.6 | 0.9 | 3.0 | 2.7 | 2.7 |
| 1000-hr (sound) | 0.0 | 0.1 | 0.1 | 0.4 | 0.6 | 0.1 | 0.0 | 8.9 | 5.6 |
| 1000-hr (rotten) | 0.1 | 0.0 | 0.0 | 1.1 | 1.2 | 0.2 | 26.9 | 8.6 | 4.4 |
| Total (Tons/Acre) | 1.1 | 1.2 | 1.3 | 3.2 | 3.8 | 1.5 | 33.5 | 25.3 | 13.9 |

*Mixed Conifer* (n=1)

Norumbega Mountain (Southwest) (AC11) – Mixed Conifer

This is the only stand where we did not find any 1992 rebar during the 2016 sampling. Given historic hand drawn maps and stand description, together with features located on-the-ground and on-site stand composition, we were confident we were in the correct location. However, there was considerably less (~82%) northern white cedar present in 2016 than I expected, and I did not find it accounted for in the dead or downed fuel load. It should be with caution that any conclusions be drawn from the long-term dataset of this stand.

Basal area of red spruce, white pine, northern white cedar, and red pine decreased significantly, while that of pitch pine increased between 1980 and 2016. Live basal area of northern white cedar decreased by 82% and red pine by 85%. Total live basal area decreased between 1980 and 1993 from 217.8 to 207.6 $ft^2$/acre and shows a decreasing trend with 127.0 $ft^2$/acre in 2016 (Table 12). Total dead basal area shows an increasing trend with 22.4 $ft^2$/acre in 1980 and 38.5 $ft^2$/acre in 2016 (Table 12). There was a significant increase in dead standing red pine over the 36-year period and is a common species die-off occurrence across the Park. Data for density of trees and saplings is missing from historic records and thus missing to report (Table 8 and Table 9). Mean diameters were also missing from historic records (Table 13).

Biomass (T/acre) of live trees decreased from 100.0 in 1980 to 54.1 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 8.7 in 1980 to 17.6 in 2016 (Table 10). Biomass (T/acre) of shrubs increased from 0.2 live and no measurable dead in 1980 to 0.4 live and less than 0.02 dead in 2016 (Table 10). Total downed woody fuel load decreased from 10.8 T/acre in

1980 to 9.0 T/acre in 2016 (Table 11). Duff depth (in.) decreased from 5.9 in 1980 to 5.0 in 2016 (Table 11). Fuel height (in.) decreased from 5.3 in 1980 to 2.5 in 2016 (Table 11).

Table 12. Average basal area (X̄ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC11 (Norumbega Mountain (Southwest)) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 2016 data.

| AC11-Norumbega Mountain (Southwest) | | | | | |
|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | **2016** | | |
| *No. Points Sampled:* | *21* | *34* | *20* | 95% CI | |
| Live | X̄ BA | X̄ BA | X̄ BA ± SE | Lower | Upper |
| Red Spruce | 98.6 | 78.2 | 80.0 ± 8.7 | 62.9 | 97.1 |
| White Pine | 29.0 | 29.1 | 19.5 ± 3.9 | 11.9 | 27.1 |
| No. White Cedar | 44.3 | 43.5 | 8.0 ± 4.6 | 0 | 16.9 |
| Pitch Pine | 0.0 | 0.6 | 6.5 ± 3.4 | 0 | 13.1 |
| Red Maple | 5.7 | 9.1 | 6.5 ± 2.3 | 1.9 | 11.1 |
| Red Pine | 36.2 | 44.4 | 5.5 ± 1.9 | 1.9 | 9.1 |
| Red Oak | 0.5 | 0.9 | 1.0 ± 1.0 | 0 | 3.0 |
| Hemlock | 1.0 | 0.3 | 0 | | |
| Paper Birch | 1.0 | 0.6 | 0 | | |
| Balsam Fir | 0.5 | 0.9 | 0 | | |
| Yellow Birch | 0.5 | 0 | 0 | | |
| Bigtooth Aspen | 0.5 | 0 | 0 | | |
| **Totals:** | **217.8** | **207.6** | **127.0 ± 8.7** | **109.9** | **144.1** |
| Dead | | | | | |
| Red Pine | 1.0 | 0.9 | 26.5 ± 4.3 | 18.0 | 35.0 |
| Red Spruce | 9.5 | 9.1 | 7.0 ± 2.4 | 2.3 | 11.7 |
| No. White Cedar | 5.2 | 9.1 | 3.0 ± 2.1 | 0 | 7.0 |
| White Pine | 5.7 | 5.0 | 1.5 ± 0.8 | 0 | 3.1 |
| Red Oak | 0 | 0 | 0.5 ± 0.5 | 0 | 1.5 |
| Paper Birch | 0.5 | 0.6 | 0 | | |
| Bigtooth Aspen | 0.5 | 0 | 0 | | |
| **Totals:** | **22.4** | **24.7** | **38.5 ± 4.6** | **29.5** | **47.5** |

Table 13. Mean diameter ($\bar{X}$ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC11 (Norumbega Mountain (Southwest)) in 2016. Data missing for 1980/1992-1994. Confidence Intervals are for 2016 data.

| **AC11** | **1980** | **1993** | **2016** | | 95% CI | |
|---|---|---|---|---|---|---|
| Species | | | $\bar{X}$ Dia. | ± SE | Lower | Upper |
| White Pine | x | x | 15.3 | ± 1.0 | 13.5 | 17.2 |
| Red Pine | x | x | 11.7 | ± 0.8 | 10.1 | 13.2 |
| Red Oak | x | x | 11.4 | ± 0.2 | 11.1 | 11.7 |
| Pitch Pine | x | x | 10.7 | ± 0.5 | 9.8 | 11.5 |
| Red Spruce | x | x | 10.2 | ± 1.0 | 8.3 | 12.0 |
| No. White Cedar | x | x | 9.3 | ± 0.6 | 8.2 | 10.5 |
| Red Maple | x | x | 8.2 | ± 0.7 | 7.0 | 9.5 |
| **Overall Average** | | | **11.0** | **± 1.0** | **9.1** | **12.9** |

*Mixed Conifer Summary*

Barring stand replacing disturbance and assuming a predictable successional trajectory without disturbance I expect this stand to remain as mixed conifer though with a lesser component of red pine, and with less certainty, northern white cedar. It is likely that red pine will be gone with no prospect for returning in the near term. Without a fire to regenerate red and white pine, the other conifer species and hardwoods will eventually dominate. Downed woody fuel accumulation is moderate (compared to other stands sampled) but could under the right conditions support a high-intensity fire. Photo series 7 and 21 are examples of this change over time (Appendix 1).

*Birch-Aspen* (n=2)

Otter Creek (AC09) – Birch-Aspen

Stand basal area is continuing to increase in response to post-fire growth from 98.4 in 1980 to 109.9 $ft^2$/acre in 1993 and 124.0 $ft^2$/acre in 2016 (Table 14). Paper birch decreased from 21.6 to 6 $ft^2$/acre (down 72%). The decrease in paper birch was more than made up for by increases in several other hardwoods (e.g., red maple, American beech, trembling aspen, red oak) (Table 14). Red spruce was absent in the understory in 1980, but by 1993 was beginning to grow into measurable size classes (Patterson 1996). Today red spruce basal area has increased to 9.5 $ft^2$/acre. Bigtooth aspen is undergoing density dependent thinning. Total dead basal area increased from 2.7 $ft^2$/acre in 1980 to 6.0 $ft^2$/acre in 2016 (Table 14).

Biomass (T/acre) of trees increased from 34.3 live and 0.7 dead in 1980 to 72.6 and 2.8 respectively in 2016 (Table 10). Biomass (T/acre) of shrubs remained absent in 1980 and 2016 (Table 10). Total downed woody fuel load decreased from 5.6 T/acre in 1980 to 3.8 T/acre in 2016 (Table 11). Duff depth (in.) increased overall from 1.2 in 1980 to 2.0 in 2016 (Table 11). Fuel height (in.) decreased from 2.6 in 1980 to 1.7 in 2016 (Table 11). Fuel loading remains low in this young stand.

Table 14. Average basal area (X̄ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC09 (Otter Creek) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 2016 data.

| AC09-Otter Creek | | | | | | |
|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | **2016** | | | |
| *No. Points Sampled:* | *20* | *24* | *20* | | 95% CI | |
| Live | X̄ BA | X̄ BA | X̄ BA ± | SE | Lower | Upper |
| Bigtooth Aspen | 57.4 | 39.6 | 50.5 ± | 10.7 | 29.5 | 71.5 |
| American Beech | 1.6 | 7.1 | 17.0 ± | 6.4 | 4.5 | 29.5 |
| Red Oak | 3.7 | 10.0 | 16.0 ± | 5.8 | 4.7 | 27.3 |
| Red Maple | 4.7 | 10.8 | 10.0 ± | 2.5 | 5.1 | 14.9 |
| Red Spruce | 0 | 2.9 | 9.5 ± | 4.8 | 0.1 | 18.9 |
| Trembling Aspen | 1.6 | 3.3 | 6.5 ± | 4.8 | 0 | 16.0 |
| Paper Birch | 21.6 | 22.1 | 6.0 ± | 2.1 | 1.9 | 10.1 |
| Striped Maple | 5.8 | 10.4 | 5.5 ± | 2.5 | 0.7 | 10.3 |
| White Ash | 1.6 | 2.9 | 3.0 ± | 2.5 | 0 | 8.0 |
| White Pine | 0 | 0.8 | 0 | | | |
| Gray Birch | 0.4 | 0 | 0 | | | |
| **Totals:** | **98.4** | **109.9** | **124.0 ±** | **8.8** | **106.7** | **141.3** |
| Dead | | | | | | |
| Bigtooth Aspen | 1.1 | 2.5 | 2.0 ± | 0.9 | 0.2 | 3.8 |
| Paper Birch | 1.1 | 3.8 | 1.0 ± | 0.7 | 0 | 2.4 |
| Red Maple | 0 | 0.4 | 1.0 ± | 1.0 | 0 | 3.0 |
| Striped Maple | 0 | 0 | 1.0 ± | 1.0 | 0 | 3.0 |
| White Ash | 0 | 0 | 0.5 ± | 0.5 | 0 | 1.5 |
| Trembling Aspen | 0 | 0 | 0.5 ± | 0.5 | 0 | 1.5 |
| American Beech | 0 | 0.4 | 0 | | | |
| Red Oak | 0.5 | 0 | 0 | | | |
| **Totals:** | **2.7** | **7.1** | **6.0 ±** | **1.8** | **2.4** | **9.6** |

Table 15. Mean diameter ($\bar{X}$ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC09 (Otter Creek) in 2016. Data missing for 1980/1992-1994. Confidence Intervals are for 2016 data.

| AC09 | 1980 | 1993 | 2016 | | 95% CI | |
|---|---|---|---|---|---|---|
| Species | | | $\bar{X}$ Dia. | ± SE | Lower | Upper |
| Bigtooth Aspen | x | x | 11.5 | ± 0.5 | 10.4 | 12.5 |
| Red Oak | x | x | 10.2 | ± 0.6 | 9.1 | 11.4 |
| Trembling Aspen | x | x | 9.6 | ± 0.5 | 8.7 | 10.5 |
| Red Maple | x | x | 8.6 | ± 0.7 | 7.1 | 10.1 |
| White Ash | x | x | 7.9 | ± 0.6 | 6.8 | 9.1 |
| American Beech | x | x | 7.7 | ± 0.6 | 6.6 | 8.9 |
| Paper Birch | x | x | 5.8 | ± 0.4 | 5.1 | 6.6 |
| Striped Maple | x | x | 4.1 | ± 0.2 | 3.7 | 4.6 |
| Red Spruce | x | x | 3.8 | ± 0.2 | 3.4 | 4.2 |
| **Overall Average** | | | **9.2** | **± 0.8** | **7.6** | **10.7** |

Cadillac Mountain (AC25) – Birch-Aspen

Basal areas of red spruce, red oak, white spruce, white pine, pitch pine, and paper birch increased, whereas that of gray birch, bigtooth, and trembling aspen decreased. Red spruce were absent as trees in 1980 but were the most common in 1993 and 2016. Total stand basal area (live) increased significantly between 1980 and 2016 from 2.5 to 34.0 ft$^2$/acre (Table 16). Dead basal area changed little (Table 16).

This young stand is continuing to accumulate biomass, but fuel loading remains low. Biomass (T/acre) of all tree species combined increased from 0.8 live and no measurable dead in 1980 to 12.9 live and 0.1 dead in 2016 (Table 10). Biomass (T/acre) of shrubs decreased from 0.6 live and 0.2 dead in 1980 to 0.4 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load decreased from 3.2 T/acre in 1980 to 1.5 T/acre in 2016 (Table 11). Duff depth (in.) increased considerably from 0.1 in 1980 to 3.5 in 2016 (Table 11). Fuel height (in.) decreased from 1.5 in 1980 to 1.2 in 2016 (Table 11).

Table 16. Average basal area ($\bar{X}$ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC25 (Cadillac Mountain) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 2016 data.

| AC25-Cadillac Mountain | | | | | |
|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | **2016** | | |
| *No. Points Sampled:* | *20* | *22* | *20* | 95% CI | |
| Live | $\bar{X}$ BA | $\bar{X}$ BA | $\bar{X}$ BA ± SE | Lower | Upper |
| Red Spruce | 0 | 12.1 | 15.5 ± 4.1 | 7.4 | 23.6 |
| Red Oak | 0.5 | 3.4 | 6.5 ± 2.5 | 1.5 | 11.5 |
| White Spruce | 0 | 1.4 | 3.5 ± 1.8 | 0 | 7.0 |
| White Pine | 0 | 0.2 | 3.0 ± 1.3 | 0.5 | 5.5 |
| Pitch Pine | 0 | 0 | 3.0 ± 2.5 | 0 | 8.0 |
| Paper Birch | 0 | 1.4 | 1.5 ± 1.5 | 0 | 4.4 |
| Red Pine | 0 | 0.5 | 0.5 ± 0.5 | 0 | 1.5 |
| Gray Birch | 1.0 | 2.1 | 0.5 ± 0.5 | 0 | 1.5 |
| Balsam Fir | 0 | 1.6 | 0 | | |
| Bigtooth Aspen | 0.5 | 1.8 | 0 | | |
| Trembling Aspen | 0.5 | 0.9 | 0 | | |
| **Totals:** | **2.5** | **25.4** | **34.0 ± 5.7** | **22.9** | **45.1** |
| Dead | | | | | |
| Balsam Fir | 0 | 0 | 0.5 ± 0.5 | 0 | 1.5 |
| Red Oak | 0 | 0.2 | 0 | | |
| **Totals:** | **0** | **0.2** | **0.5 ± 0.5** | **-0.5** | **1.5** |

Table 17. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC25 (Cadillac Mountain) in 2016. Data missing for 1980/1992-1994. Confidence Intervals are for 2016 data.

| **AC25** | **1980** | **1993** | **2016** | 95% CI | |
|---|---|---|---|---|---|
| Species | | | X̄ Dia. ± SE | Lower | Upper |
| Red Pine | x | x | 9.4 ± 0 | 9.4 | 9.4 |
| White Pine | x | x | 7.7 ± 0.8 | 6.1 | 9.3 |
| Red Oak | x | x | 7.5 ± 0.5 | 6.5 | 8.4 |
| Red Spruce | x | x | 5.8 ± 0.4 | 4.9 | 6.6 |
| White Spruce | x | x | 5.1 ± 0.6 | 4.0 | 6.2 |
| Pitch Pine | x | x | 4.5 ± 0.2 | 4.0 | 4.9 |
| Balsam Fir | x | x | 4.2 ± 0 | 4.2 | 4.2 |
| Paper Birch | x | x | 3.0 ± 0 | 2.9 | 3.0 |
| Grey Birch | x | x | 1.3 ± 0 | 1.3 | 1.3 |
| **Overall Average** | | | **5.7 ± 0.6** | **4.6** | **6.9** |

*Birch – Aspen Summary*

Barring stand replacing disturbance and assuming a predictable successional trajectory without disturbance I might expect these birch-aspen stands to continue to accumulate biomass and transition from the post-1947 fire birch-aspen community type of the 1980s to a later successional conifer dominant community type with a more closed canopy. Given the abundance of the suppressed red spruce in the understory, I expect a strong component of red spruce will grow into the canopy in the next several decades. Today fuel loading and fire hazard remains low. We may expect that when the even aged overstory becomes over-mature and the trees die the downed woody fuel loading could accumulate rapidly in the short term. Since aspen and birch wood decompose quickly this accumulation of downed fuel load would only be within the decade after the mortality of these species. The flammability of the stand may be further increased by the dense red spruce regeneration. Photo series 13, 14, 18, and 28 are examples of this change over time (Appendix 1).

*Northern Hardwoods* (n=2)

Conners Nubble (AC02) – Northern Hardwoods

Basal area of bigtooth aspen, red spruce, and red maple increased significantly, while that of paper birch decreased significantly. The increase of live basal area of Bigtooth aspen is especially notable. Total live basal area increased between 1980 and 2016 from 100.6 to 115.5 $ft^2$/acre (Table 18). Total dead basal area shows a significant increasing trend with 5.5 $ft^2$/acre in 1980 and 18.5 $ft^2$/acre in 2016 (Table 18). Density of live trees decreased from 1412.9 in 1980 to 760.7 stems per acre in 2016 (Table 8). Density of dead trees also decreased from 149.8 in 1980 to 122.1 stems per acre in 2016 (Table 8). Density of live saplings decreased measuring at 334.3 in 1993 and 165.0 stems per acre in 2016 (Table 9). Density of dead saplings also decreased from 91.4 in 1993 to 80.0 stems per acre in 2016 (Table 9). Mean diameters increased from 6.4 (in.) in 1980 to 8.1 (in.) in 2016 (Table 19).

Biomass (T/acre) of trees increased from 12.5 live and 7.2 dead in 1980 to 48.6 and 16.5 respectively in 2016 (Table 10). There was no biomass of shrubs measured in 1980, but we measured an increase in shrubs to 0.5 T/acre in 2016 (Table 10). Total downed woody fuel load decreased from 9.7 T/acre in 1980 to 4.3 T/acre in 2016 (Table 11). Duff depth (in.) increased overall from 2.0 in 1980 to 3.4 in 2016 (Table 11). Fuel height (in.) decreased from 6.0 in 1980 to 1.5 in 2016 (Table 11).

Table 18. Average basal area (X̄ BA) of live and dead trees (ft²/acre) ± standard error with 95% confidence intervals for stand AC02 (Connors Nubble) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC02- Connors Nubble | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | | **1993** | | | | **2016** | | | |
| *No. Points Sampled:* | *22* | | 95% CI | | *35* | | 95% CI | | *20* | | 95% CI | |
| Live | X̄ BA ± | SE | Lower | Upper | X̄ BA ± | SE | Lower | Upper | X̄ BA ± | SE | Lower | Upper |
| American Beech | 53.2 ± | 6.9 | 39.7 | 66.7 | 65.7 ± | 6.6 | 52.8 | 78.6 | 45.5 ± | 8.3 | 29.2 | 61.8 |
| Bigtooth Aspen | 3.2 ± | 2.3 | 0 | 7.8 | 6.4 ± | 2.8 | 0.8 | 12.0 | 24.0 ± | 7.9 | 8.6 | 39.4 |
| Red Spruce | 0 | | | | 1.1 ± | 0.8 | 0 | 2.7 | 18.5 ± | 5.3 | 8.0 | 29.0 |
| Red Maple | 0.9 ± | 0.6 | 0 | 2.1 | 4.3 ± | 1.6 | 1.2 | 7.4 | 10.0 ± | 2.3 | 5.5 | 14.5 |
| Sugar Maple | 19.1 ± | 4.8 | 9.6 | 28.6 | 13.2 ± | 2.5 | 8.3 | 18.1 | 7.5 ± | 2.5 | 2.6 | 12.4 |
| Paper Birch | 17.3 ± | 3.6 | 10.2 | 24.4 | 18.6 ± | 3.3 | 12.2 | 25.0 | 6.0 ± | 1.7 | 2.7 | 9.3 |
| Hop hornbeam | 0 | | | | 0 | | | | 1.5 ± | 1.1 | 0 | 3.6 |
| Yellow Birch | 0.9 ± | 0.6 | 0 | 2.1 | 1.1 ± | 0.6 | 0 | 2.2 | 1.0 ± | 0.7 | 0 | 2.4 |
| White Pine | 0 | | | | 0 | | | | 0.5 ± | 0.5 | 0 | 1.5 |
| White Ash | 0.5 ± | 0.4 | 0 | 1.4 | 0 | | | | 0.5 ± | 0.5 | 0 | 1.5 |
| Striped Maple | 2.7 ± | 1.2 | 0.4 | 5.0 | 6.8 ± | 2.2 | 2.5 | 11.1 | 0.5 ± | 0.5 | 0 | 1.5 |
| Hemlock | 0.5 ± | 0.4 | 0 | 1.4 | 0.4 ± | 0.4 | 0 | 1.1 | 0 | | | |
| Gray Birch | 1.8 ± | 1.4 | 0 | 4.6 | 0 | | | | 0 | | | |
| Trembling Aspen | 0.5 ± | 0.4 | 0 | 1.4 | 0.7 ± | 0.7 | 0 | 2.1 | 0 | | | |
| **Totals:** | **100.6 ±** | **6.2** | **88.4** | **112.8** | **118.3 ±** | **6.5** | **105.5** | **131.1** | **115.5 ±** | **8.5** | **98.8** | **132.2** |
| Dead | | | | | | | | | | | | |
| White Pine | 0 | | | | 0 | | | | 7.5 ± | 2.0 | 3.5 | 11.5 |
| American Beech | 2.7 ± | 1.3 | 0.1 | 5.3 | 2.9 ± | 1.2 | 0.5 | 5.3 | 5.0 ± | 2.0 | 1.1 | 8.9 |
| Bigtooth Aspen | 0 | | | | 0.4 ± | 0.4 | 0 | 1.1 | 3.5 ± | 1.5 | 0.6 | 6.4 |
| Red Spruce | 0 | | | | 0 | | | | 2.0 ± | 1.2 | 0 | 4.3 |
| Sugar Maple | 1.4 ± | 0.7 | 0 | 2.9 | 0.7 ± | 0.5 | 0 | 1.7 | 0.5 ± | 0.5 | 0 | 1.5 |
| Yellow Birch | 0 | | | | 0 | | | | 0 | | | |
| Striped Maple | 0 | | | | 1.1 ± | 0.6 | 0 | 2.2 | 0 | | | |
| Paper Birch | 0.9 ± | 0.6 | 0 | 2.1 | 5.4 ± | 1.7 | 2.0 | 8.8 | 0 | | | |
| Red Maple | 0 | | | | 0.4 ± | 0.4 | 0 | 1.1 | 0 | | | |
| Gray Birch | 0.5 ± | 0.4 | 0 | 1.4 | 0 | | | | 0 | | | |
| **Totals:** | **5.5 ±** | **1.8** | **1.9** | **9.1** | **10.9 ±** | **2.1** | **6.9** | **14.9** | **18.5 ±** | **3.3** | **12.1** | **24.9** |

Table 19. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC02 (Connors Nubble) in 1980, 1992-1994 and 2016.

| AC02 | 1980 | | 95% CI | | 1993 | | 95% CI | | 2016 | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper |
| White Pine | | | | | | | | | 14.7 | ± 0 | 14.7 | 14.7 |
| Bigtooth Aspen | 6.7 | ± 0.4 | 6.0 | 7.5 | 9.7 | ± 0.6 | 8.6 | 10.8 | 12.0 | ± 0.8 | 10.5 | 13.5 |
| Sugar Maple | 9.4 | ± 0.7 | 8.0 | 10.7 | 9.2 | ± 0.8 | 7.7 | 10.7 | 11.7 | ± 1.2 | 9.4 | 14.0 |
| Red Maple | 11.0 | ± 0.7 | 9.6 | 12.4 | 8.1 | ± 0.7 | 6.8 | 9.4 | 9.9 | ± 0.7 | 8.6 | 11.2 |
| Striped Maple | 3.4 | ± 0.1 | 3.1 | 3.6 | 4.3 | ± 0.3 | 3.7 | 4.8 | 7.5 | ± 0 | 7.5 | 7.5 |
| American Beech | 5.4 | ± 0.7 | 3.9 | 6.8 | 6.0 | ± 0.6 | 4.8 | 7.1 | 7.1 | ± 0.7 | 5.8 | 8.5 |
| White Ash | 7.9 | ± 0 | 7.9 | 7.9 | | | | | 7.0 | ± 0 | 7.0 | 7.0 |
| Yellow Birch | 12.9 | ± 0.2 | 12.4 | 13.4 | 13.8 | ± 1.4 | 10.9 | 16.6 | 6.1 | ± 0 | 6.0 | 6.1 |
| Paper Birch | 6.4 | ± 0.7 | 4.9 | 7.8 | 5.1 | ± 0.5 | 4.2 | 6.1 | 6.0 | ± 0.4 | 5.1 | 6.9 |
| Red Spruce | | | | | 4.5 | ± 0.4 | 3.7 | 5.4 | 4.9 | ± 0.5 | 4.0 | 5.8 |
| Hophornbeam | | | | | | | | | 2.9 | ± 0.6 | 1.8 | 4.0 |
| Grey Birch | 2.8 | ± 0.2 | 2.4 | 3.2 | | | | | | | | |
| Trembling Aspen | 6.0 | ± 0 | 6.0 | 6.0 | 8.7 | ± 0.3 | 8.0 | 9.3 | | | | |
| Hemlock | 20.4 | ± 0 | 20.4 | 20.4 | 17.7 | ± 0 | 17.7 | 17.7 | | | | |
| **Overall Average** | **6.4** | **± 0.8** | **4.8** | **8.0** | **6.4** | **± 0.7** | **5.1** | **7.7** | **8.1** | **± 0.9** | **6.4** | **9.9** |

Eagle Lake (South) (AC03) – Northern Hardwoods

Basal area of American beech decreased significantly while that of red spruce, hemlock, and yellow birch increased significantly between 1980 and 2016. Live basal area of American beech decreased by 45% due to chronic beech-bark-disease. The increase in hemlock is important as it shows the stand heading in the direction we might expect with a warming climate although hemlock wooly adelgid may be a factor in the future. Total live basal area increased between 1980 and 2016 from 114.9 to 118.0 ft$^2$/acre (Table 20). Total dead basal area barely changed with 7.2 ft$^2$/acre in 1980 and 9.0 ft$^2$/acre in 2016 (Table 20). Density of live trees decreased significantly from 950.6 in 1980 to 425.4 stems per acre in 2016 (Table 8). Density of dead trees also decreased significantly from 44.9 in 1980 to 14.2 stems per acre in 2016 (Table 8). Density of live saplings decreased with 394.7 in 1993 and 345.0 stems per acre in 2016 (Table 9). Density of dead saplings decreased from 86.8 in 1993 to 40.0 stems per acre in 2016 (Table 9). Mean diameters increased from 9.3 (in) in 1980 to 12.3 (in) in 2016 (Table 21).

Biomass (T/acre) of trees increased from 59.4 live and 3.6 dead in 1980 to 73.3 live and 6.1 dead in 2016 (Table 10). There were no shrubs at this stand in 1980 or 2016 (Table 10). Total downed woody fuel load decreased from 11.9 T/acre in 1980 to 8.0 T/acre in 2016 (Table 11). Duff depth (in.) increased from 1.0 in 1980 to 2.2 in 2016 (Table 11). Fuel height (in.) decreased from 2.8 in 1980 to 1.5 in 2016 (Table 11).

Table 20. Average basal area ($\bar{X}$ BA) of live and dead trees ($ft^2$/acre) ± standard error with 95% confidence intervals for stand AC03 (Eagle Lake (South)) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC03- Eagle Lake (South) | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | | | **1993** | | | | | **2016** | | | | |
| *No. Points Sampled:* | *21* | | | 95% CI | | *38* | | | 95% CI | | *20* | | | 95% CI | |
| Live | $\bar{X}$ BA | ± | SE | Lower | Upper | $\bar{X}$ BA | ± | SE | Lower | Upper | $\bar{X}$ BA | ± | SE | Lower | Upper |
| American Beech | 42.4 | ± | 7.0 | 28.6 | 56.2 | 38.4 | ± | 4.0 | 30.6 | 46.2 | 23.5 | ± | 3.9 | 15.9 | 31.1 |
| Sugar Maple | 29.5 | ± | 4.1 | 21.5 | 37.5 | 20.8 | ± | 2.7 | 15.5 | 26.1 | 21.5 | ± | 3.0 | 15.6 | 27.4 |
| Red Spruce | 2.4 | ± | 1.2 | 0.1 | 4.7 | 9.7 | ± | 3.5 | 2.8 | 16.6 | 19.0 | ± | 5.0 | 9.3 | 28.7 |
| Hemlock | 1.9 | ± | 0.9 | 0.2 | 3.6 | 5.8 | ± | 1.7 | 2.5 | 9.1 | 13.0 | ± | 3.9 | 5.3 | 20.7 |
| Paper Birch | 17.6 | ± | 4.6 | 8.6 | 26.6 | 18.2 | ± | 3.2 | 12.0 | 24.4 | 10.0 | ± | 3.2 | 3.6 | 16.4 |
| Yellow Birch | 0.5 | ± | 0.5 | 0 | 1.4 | 2.9 | ± | 0.9 | 1.1 | 4.7 | 9.5 | ± | 3.7 | 2.3 | 16.7 |
| White Ash | 6.7 | ± | 2.7 | 1.4 | 12.0 | 5.8 | ± | 1.6 | 2.6 | 9.0 | 9.5 | ± | 3.5 | 2.6 | 16.4 |
| Red Maple | 1.9 | ± | 0.9 | 0.2 | 3.6 | 4.7 | ± | 1.6 | 1.5 | 7.9 | 5.5 | ± | 2.9 | 0 | 11.1 |
| Striped Maple | 3.8 | ± | 1.7 | 0.4 | 7.2 | 9.5 | ± | 2.2 | 5.3 | 13.7 | 5.0 | ± | 1.5 | 0 | 8.0 |
| Bigtooth Aspen | 6.2 | ± | 5.2 | 0 | 16.5 | 1.6 | ± | 1.2 | 0 | 3.9 | 1.5 | ± | 1.1 | 0 | 3.6 |
| Unknown | 0 | | | | | 0.3 | ± | 0.3 | 0 | 0.8 | 0 | | | | |
| Shadbush | 1.0 | ± | 0.7 | 0 | 2.3 | 0.3 | ± | 0.3 | 0 | 0.8 | 0 | | | | |
| Hophornbeam | 1.0 | ± | 0.7 | 0 | 2.3 | 0.3 | ± | 0.3 | 0 | 0.8 | 0 | | | | |
| Ironwood | | | | | | 0 | | | | | 0 | | | | |
| **Totals:** | **114.9** | **±** | **6.4** | **102.5** | **127.3** | **118.3** | **±** | **5.1** | **108.3** | **128.3** | **118.0** | **±** | **8.3** | **101.7** | **134.3** |
| Dead | | | | | | | | | | | | | | | |
| American Beech | 1.4 | ± | 1.0 | 0 | 3.5 | 4.5 | ± | 1.2 | 2.1 | 6.9 | 3.0 | ± | 1.3 | 0.5 | 5.5 |
| White Pine | 0 | | | | | 0 | | | | | 2.0 | ± | 0.9 | 0.2 | 3.8 |
| Pitch Pine | 0 | | | | | 0 | | | | | 1.5 | ± | 1.1 | 0 | 3.6 |
| Bigtooth Aspen | 0.5 | ± | 0.5 | 0 | 1.4 | 0 | | | | | 1.0 | ± | 0.7 | 0 | 2.4 |
| Red Spruce | 0.5 | ± | 0.5 | 0 | 1.4 | 0.3 | ± | 0.3 | 0 | 0.8 | 0.5 | ± | 0.5 | 0 | 1.5 |
| Balsam Fir | 0 | | | | | 0 | | | | | 0.5 | ± | 0.5 | 0 | 1.5 |
| Sugar Maple | 1.0 | ± | 0.7 | 0 | 2.3 | 2.1 | ± | 1.2 | 0 | 4.5 | 0.5 | ± | 0.5 | 0 | 1.5 |
| Yellow Birch | 0 | | | | | 0 | ± | 0 | 0 | 0 | 0 | | | | |
| White Ash | 0 | | | | | 1.3 | ± | 0.7 | 0 | 2.6 | 0 | | | | |
| Striped Maple | 0.5 | ± | 0.5 | 0 | 1.4 | 0.3 | ± | 0.3 | 0 | 0.8 | 0 | | | | |
| Red Maple | 0.5 | ± | 0.5 | 0 | 1.4 | 0 | | | | | 0 | | | | |
| Paper Birch | 1.4 | ± | 0.8 | 0 | 2.9 | 2.9 | ± | 1.0 | 1.0 | 4.8 | 0 | | | | |
| Gray Birch | 1.4 | ± | 1.0 | 0 | 3.5 | 0 | | | | | 0 | | | | |
| **Totals:** | **7.2** | **±** | **2.1** | **3.1** | **11.3** | **11.4** | **±** | **2.5** | **6.5** | **16.3** | **9.0** | **±** | **2.2** | **4.7** | **13.3** |

Table 21. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC03 (Eagle Lake (South)) in 1980, 1992-1994 and 2016.

| AC03 | 1980 | 95% CI | | 1993 | 95% CI | | 2016 | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper |
| Hemlock | 24.9 ± 0.8 | 23.4 | 26.4 | 17.3 ± 1.1 | 15.2 | 19.3 | 22.4 ± 1.7 | 19.0 | 25.8 |
| Bigtooth Aspen | 14.0 ± 0.5 | 13.0 | 15.1 | 16.7 ± 0.4 | 15.9 | 17.5 | 19.3 ± 1.1 | 17.2 | 21.3 |
| Red Maple | 8.6 ± 0.6 | 7.3 | 9.8 | 8.6 ± 0.6 | 7.5 | 9.7 | 13.1 ± 1.0 | 11.3 | 15.0 |
| Yellow Birch | 6.7 ± 0 | 6.7 | 6.7 | 12.2 ± 1.0 | 10.2 | 14.3 | 13.0 ± 0.8 | 11.5 | 14.5 |
| White Ash | 12.1 ± 0.8 | 10.6 | 13.7 | 12.8 ± 0.7 | 11.5 | 14.1 | 12.9 ± 0.5 | 11.9 | 13.8 |
| Red Spruce | 11.5 ± 1.7 | 8.1 | 14.8 | 9.7 ± 1.0 | 7.6 | 11.7 | 12.3 ± 1.4 | 9.6 | 15.0 |
| Paper Birch | 10.4 ± 0.5 | 9.4 | 11.4 | 10.8 ± 0.5 | 9.9 | 11.7 | 11.7 ± 0.4 | 10.9 | 12.6 |
| Sugar Maple | 7.6 ± 0.7 | 6.1 | 9.0 | 9.0 ± 0.6 | 7.7 | 10.3 | 9.9 ± 0.8 | 8.4 | 11.4 |
| American Beech | 9.0 ± 1.0 | 7.1 | 10.8 | 8.3 ± 0.7 | 6.9 | 9.6 | 9.4 ± 1.0 | 7.3 | 11.4 |
| Striped Maple | 3.6 ± 0.4 | 2.8 | 4.4 | 4.3 ± 0.3 | 3.6 | 4.9 | 4.5 ± 0.4 | 3.8 | 5.2 |
| Shadbush | 7.6 ± 0 | 7.5 | 7.6 | 8.4 ± 0 | 8.4 | 8.4 | | | |
| Grey Birch | 6.3 ± 0.1 | 6.0 | 6.5 | | | | | | |
| Hophornbeam | 7.5 ± 0.9 | 5.7 | 9.3 | 6.4 ± 0 | 6.4 | 6.4 | | | |
| **Overall Average** | **9.3 ± 1.0** | **7.4** | **11.3** | **9.5 ± 0.8** | **7.9** | **11.1** | **12.3 ± 1.4** | **9.6** | **15.0** |

*Northern Hardwoods Summary*

Barring stand replacing disturbance and assuming a predictable successional trajectory without disturbance I expect these two stands to remain as northern hardwoods, with Eagle Lake (AC03) having and increased component of red spruce and hemlock. This suggests that the Eagle Lake stand may transition to a northern hardwood-mixed conifer stand like AC16 on the west shore of Long Pond or AC23 on Day Mountain. Today Eagle Lake is more similar in species composition to the northern hardwood-mixed conifer stands than to Conners Nubble (AC02). The data suggest that at one time these stands were likely dominated by beech, sugar maple, aspen, and birch. These two stands are continuing to accumulate biomass, although fuel loading and fire hazard remain low. Photo series 4, 5, 9, 15, 16, 23, and 26 are examples of this change over time (Appendix 1).

*Pitch Pine* (n=2)

Sand Beach (AC08) – Pitch Pine

The Sand Beach stand accumulated basal area of red spruce and pitch pine between 1980 and 2016. Live basal area of red spruce increased 12-fold and pitch pine by 86%. Total live basal area doubled between 1980 and 1993 (20.5 to 45.1 $ft^2$/acre), and by an additional 63% in 2016 (73.5 $ft^2$/acre) compared to 1993 (Table 22). Total dead basal area shows an increasing trend with no measurable in 1980 and 1.0 $ft^2$/acre in 2016 (Table 22).

Biomass (T/acre) of trees increased from 7.2 live and no measurable dead in 1980 to 21.6 live and 0.3 dead in 2016 (Table 10). This is a three-fold increase in live tree biomass across the sampling period. Biomass (T/acre) of shrubs increased from 0.2 live and less than 0.02 dead in 1980 to 0.9 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load decreased from 11.6 T/acre in 1980 to 4.1 T/acre in 2016 (Table 11), likely due to decay of logs left behind from the 1947 fire (Patterson 1996). Duff depth (in.) more than tripled from 1.0 in 1980 to 3.6 in 2016 (Table 11). Fuel height (in.) decreased from 1.5 in 1980 to 0.8 in 2016 (Table 11).

Table 22. Average basal area (X̄ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC08 (Sand Beach) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 2016 data.

| AC08- Sand Beach | | | | | | | |
|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | **2016** | | | | |
| *No. Points Sampled:* | *20* | *24* | *20* | | | 95% CI | |
| Live | X̄ BA | X̄ BA | X̄ BA | STDEV ± | SE | Lower | Upper |
| Red Spruce | 3.0 | 12.1 | 38.5 | 35.0 ± | 7.8 | 23.2 | 53.8 |
| Pitch Pine | 17.5 | 27.1 | 32.5 | 31.9 ± | 7.1 | 18.5 | 46.5 |
| Red Maple | 0 | 1.7 | 1.5 | 4.9 ± | 1.1 | 0 | 3.6 |
| Paper Birch | 0 | 1.3 | 0.5 | 2.2 ± | 0.5 | 0 | 1.5 |
| Red Pine | 0 | 0 | 0.5 | 2.2 ± | 0.5 | 0 | 1.5 |
| Bigtooth Aspen | 0 | 0.4 | 0 | | | | |
| Trembling Aspen | 0 | 0.4 | 0 | | | | |
| White Spruce | 0 | 0.8 | 0 | | | | |
| Shadbush | 0 | 1.3 | 0 | | | | |
| **Totals:** | **20.5** | **45.1** | **73.5** | **43.4 ±** | **9.7** | **54.5** | **92.5** |
| Dead | | | | | | | |
| Red Spruce | 0 | 0 | 1.0 | 3.1 ± | 0.7 | 0 | 2.4 |
| Pitch Pine | 0 | 0.8 | 0 | | | | |
| Red Maple | 0 | 0.4 | 0 | | | | |
| Bigtooth Aspen | 0 | 0.8 | 0 | | | | |
| **Totals:** | **0** | **2.0** | **1.0** | **3.1 ±** | **0.7** | **0** | **2.4** |

Table 23. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC08 (Sand Beach) in 2016. Data missing for 1980/1992-1994. Confidence Intervals are for 2016 data.

| **AC08** | **1980** | **1993** | **2016** | | 95% CI | |
|---|---|---|---|---|---|---|
| Species | | | X̄ Dia. ± | SE | Lower | Upper |
| Red Pine | x | x | 12.3 ± | 0 | 12.3 | 12.3 |
| Pitch Pine | x | x | 7.0 ± | 0.4 | 6.1 | 7.8 |
| Red Maple | x | x | 4.8 ± | 0.3 | 4.3 | 5.3 |
| Red Spruce | x | x | 4.1 ± | 0.3 | 3.4 | 4.8 |
| Paper Birch | x | x | 2.2 ± | 0 | 2.2 | 2.2 |
| **Overall Average** | | | **5.4 ±** | **0.5** | **4.4** | **6.4** |

Champlain Mountain (IaH) (AC24) – Pitch Pine

Total live basal area decreased between 1980 and 2016 from 101.6 to 85.5 $ft^2$/acre (Table 24). Total dead basal area decreased with 5.8 $ft^2$/acre in 1980 and 5.0 $ft^2$/acre in 2016 (Table 24). Live basal area of pitch pine decreased by 26%, while that of red spruce increased by 90% between 1980 and 2016. Density of live trees decreased significantly from 567.8 in 1980 to 45.1 stems per acre in 2016 (Table 8), resulting in an open canopy savannah-like-woodland blanked with shrubs. Density of dead trees also decreased significantly from 71.6 in 1980 to 3.8 stems per acre in 2016 (Table 8). Density of saplings is missing for the historic sampling periods (Table 9). Mean diameters increased from 7.1 (in.) in 1980 to 23.4 (in.) in 2016 (Table 25).

Biomass (T/acre) of trees decreased from 48.7 live and 2.2 dead in 1980 to 35.7 live and 1.8 dead in 2016 (Table 10). Biomass (T/acre) of shrubs increased from 2.1 live and 0.3 dead in 1980 to 3.7 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load did not change measuring 1.1 T/acre in 1980 to 1.3 T/acre in 2016 (Table 11). Duff depth (in.) increased from 1.7 in 1980 to 5.1 in 2016 (Table 11). Fuel height (in.) increased from 2.3 in 1980 to 2.8 in 2016 (Table 11).

Table 24. Average basal area (X̄ BA) of live and dead trees ($ft^2$/acre) ± standard error with 95% confidence intervals for stand AC24 (Champlain Mountain) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC24-Champlain Mountain (IaH) | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | | | **1993** | | | | | **2016** | | | | |
| *No. Points Sampled:* | *20* | | | 95% CI | | *20* | | | 95% CI | | *20* | | | 95% CI | |
| Live | X̄ BA | ± | SE | Lower | Upper | X̄ BA | ± | SE | Lower | Upper | X̄ BA | ± | SE | Lower | Upper |
| Pitch Pine | 91.1 | ± | 9.0 | 73.4 | 108.8 | 94.5 | ± | 9.2 | 76.5 | 112.5 | 67.5 | ± | 5.3 | 57.2 | 77.8 |
| Red Spruce | 8.4 | ± | 2.1 | 4.4 | 12.4 | 5.5 | ± | 1.5 | 2.5 | 8.5 | 16.0 | ± | 3.1 | 9.9 | 22.1 |
| Red Maple | 1.6 | ± | 1.1 | 0 | 3.8 | 4.0 | ± | 2.5 | 0 | 8.8 | 2.0 | ± | 1.6 | 0 | 5.1 |
| Bigtooth Aspen | 0.5 | ± | 0.5 | 0 | 1.5 | | | | | | 0 | | | | |
| **Totals:** | **101.6** | **±** | **9.8** | **82.3** | **120.9** | **104.0** | **±** | **9.4** | **85.5** | **122.5** | **85.5** | **±** | **6.0** | **73.7** | **97.3** |
| Dead | | | | | | | | | | | | | | | |
| Pitch Pine | 5.3 | ± | 2.9 | 0 | 11.0 | 4.5 | ± | 1.7 | 1.2 | 7.8 | 4.5 | ± | 1.1 | 2.3 | 6.7 |
| Red Spruce | 0.5 | ± | 0.5 | 0 | 1.5 | 0 | | | | | 0.5 | ± | 0.5 | 0 | 1.5 |
| **Totals:** | **5.8** | **±** | **2.9** | **0.1** | **11.5** | **4.5** | **±** | **1.7** | **1.2** | **7.8** | **5.0** | **±** | **1.1** | **2.8** | **7.2** |

Table 25. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC24 (Champlain Mountain) in 1980, 1992-1994 and 2016.

| **AC24** | **1980** | | | 95% CI | | **1993** | | | 95% CI | | **2016** | | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. | ± | SE | Lower | Upper | X̄ Dia. | ± | SE | Lower | Upper | X̄ Dia. | ± | SE | Lower | Upper |
| Red Maple | 3.4 | ± | 0.3 | 2.7 | 4.0 | 4.6 | ± | 0.2 | 4.3 | 4.9 | 25.9 | ± | 1.7 | 22.5 | 29.3 |
| Red Spruce | 9.0 | ± | 0.6 | 7.8 | 10.2 | 10.4 | ± | 0.8 | 8.9 | 12.0 | 25.7 | ± | 2.8 | 20.2 | 31.1 |
| Pitch Pine | 7.0 | ± | 0.6 | 5.9 | 8.1 | 7.4 | ± | 0.6 | 6.2 | 8.5 | 22.8 | ± | 1.3 | 20.3 | 25.2 |
| Bigtooth Aspen | 12.3 | ± | 0 | 12.3 | 12.3 | | | | | | | | | | |
| **Overall Average** | **7.1** | **±** | **0.6** | **6.0** | **8.3** | **7.4** | **±** | **0.6** | **6.2** | **8.6** | **23.4** | **±** | **1.7** | **20.1** | **26.6** |

*Pitch Pine Summary*

Sand Beach (AC08) is a young post-1947 fire stand that is accumulating biomass. Barring stand replacing disturbance and assuming a predictable successional trajectory without disturbance I expect the more tolerant red spruce, which is increasing, will eventually overtake and shade out the pitch pine in the absence of fire. However, climate warming may obviate this trend. Without fire (natural or prescribed), it is possible this stand will move towards conditions with a greater red spruce importance like AC24, which has remained undisturbed for over 100 years.

In the absence of fire, pitch pine on Champlain Mountain (AC24) may be outcompeted by red spruce as the 36-year change analysis which the increase in red spruce and decrease in pitch pine suggests. As litter accumulates pitch pine seedlings cannot establish. In the absence of fire, seedlings of other species will establish, replacing the pitch pine as they die. Today this stand has a dense, continuous cover of huckleberry, sheep laurel, and bay berry which would support fire (Patterson 1996). The fuel loading and fire hazard of these stands is high. The species present have physiological characteristics which promote and support fire. A fire of this nature would, however, promote the perpetuation of the pitch pine. Photo series 6, 8, 10, 11, 17, 20, 24, and 25 are examples of this change over time (Appendix 1).

*Northern White Cedar* (n=2)

<u>Pemetic Mountain (AC05) – Northern White Cedar</u>

Basal area of northern white cedar decreased by 32%. Paper birch decreased significantly across the sampling period. Basal area of red spruce and yellow birch increased by 45% and three-fold respectively between 1980 and 2016. Total live basal area decreased between 1980 and 1993 from 148.5 to 116.4 $ft^2$/acre, and to 113.0 $ft^2$/acre in 2016 (Table 26). Total dead basal area shows a significant increasing trend with 3.5 $ft^2$/acre in 1980 and 12.5 $ft^2$/acre in 2016 (Table 26). Density of live trees significantly decreased from 872.4 in 1980 to 347.7 stems per acre in 2016 (Table 8). Density of dead trees decreased from 15.2 in 1980 to 36.8 stems per acre in 2016 (Table 8). Density of live saplings increased from 176.0 in 1993 and 255.0 stems per acre in 2016 (Table 9). Density of dead saplings decreased from 44.0 in 1993 to 20.0 stems per acre in 2016 (Table 9). Mean diameters for the stand as a whole increased from 7.5 (in.) in 1980 to 10.2 (in.) in 2016 (Table 27).

Biomass (T/acre) of live trees decreased from 49.0 in 1980 to 43.0 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 1.6 in 1980 to 6.7 in 2016 (Table 10). Biomass (T/acre) of shrubs increased from less than 0.04 live and less than 0.02 dead in 1980 to 0.6 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load decreased from 7.2 T/acre in 1980 to 6.6 T/acre in 2016 (Table 11). Duff depth (in.) decreased from 1.5 in 1980 to 1.2 in 2016 (Table 11). Fuel height (in.) decreased from 2.8 in 1980 to 2.2 in 2016 (Table 11).

Table 26. Average basal area ($\bar{X}$ BA) of live and dead trees ($ft^2$/acre) ± standard error with 95% confidence intervals for stand AC05 (Pemetic Mountain) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC05-Pemetic Mountain | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | | **1993** | | | | **2016** | | | |
| *No. Points Sampled:* | *20* | | 95% CI | | *25* | | 95% CI | | *20* | | 95% CI | |
| Live | $\bar{X}$ BA ± | SE | Lower | Upper | $\bar{X}$ BA ± | SE | Lower | Upper | $\bar{X}$ BA ± | SE | Lower | Upper |
| No. White Cedar | 90.5 ± | 12.3 | 66.3 | 114.7 | 70.0 ± | 11.5 | 47.5 | 92.5 | 61.5 ± | 11.9 | 38.3 | 84.7 |
| Paper Birch | 34.0 ± | 4.9 | 24.3 | 43.7 | 25.6 ± | 5.5 | 14.8 | 36.4 | 16.0 ± | 4.0 | 8.2 | 23.8 |
| Red Spruce | 10.0 ± | 2.1 | 6.0 | 14.0 | 11.6 ± | 2.6 | 6.6 | 16.6 | 14.5 ± | 3.9 | 6.9 | 22.1 |
| Yellow Birch | 2.5 ± | 1.6 | 0 | 5.7 | 1.6 ± | 0.7 | 0.1 | 3.1 | 7.0 ± | 2.3 | 2.5 | 11.5 |
| Striped Maple | 4.0 ± | 1.7 | 0.7 | 7.3 | 5.2 ± | 2.8 | 0 | 10.8 | 5.5 ± | 2.2 | 1.1 | 9.9 |
| White Ash | 0.5 ± | 0.5 | 0 | 1.5 | 0.8 ± | 0.6 | 0 | 1.9 | 3.0 ± | 1.8 | 0 | 6.5 |
| Sugar Maple | 0.5 ± | 0.5 | 0 | 1.5 | 0 | | | | 2.5 ± | 1.4 | 0 | 5.3 |
| White Pine | 0 | | | | 0 | | | | 1.0 ± | 0.7 | 0 | 2.4 |
| Bigtooth Aspen | 2.5 ± | 1.2 | 0.1 | 4.9 | 1.2 ± | 0.9 | 0 | 2.9 | 1.0 ± | 1.0 | 0 | 3.0 |
| Hemlock | 2.5 ± | 1.4 | 0 | 5.3 | 0 | | | | 0.5 ± | 0.5 | 0 | 1.5 |
| Shadbush | 0.5 ± | 0.5 | 0 | 1.5 | 0 | | | | 0.5 ± | 0.5 | 0 | 1.5 |
| Balsam Fir | 0.5 ± | 0.5 | 0 | 1.5 | 0 | | | | 0 | | | |
| Gray Birch | 0 | | | | 0.4 ± | 0.4 | 0 | 1.2 | 0 | | | |
| Trembling Aspen | 0.5 ± | 0.5 | 0 | 1.5 | 0 | | | | 0 | | | |
| **Totals:** | **148.5 ±** | **11.2** | **126.5** | **170.5** | **116.4 ±** | **10.7** | **95.5** | **137.3** | **113.0 ±** | **10.7** | **92.0** | **134.0** |
| Dead | | | | | | | | | | | | |
| Paper Birch | 2.0 ± | 1.2 | 0 | 4.3 | 2.4 ± | 0.9 | 0.7 | 4.1 | 8.0 ± | 1.7 | 4.6 | 11.4 |
| No. White Cedar | 0 | | | | 1.6 ± | 1.2 | 0 | 4.0 | 2.0 ± | 1.6 | 0 | 5.1 |
| Red Spruce | 0 | | | | 1.2 ± | 0.7 | 0 | 2.5 | 0.5 ± | 0.5 | 0 | 1.5 |
| Hemlock | 0 | | | | 0 | | | | 0.5 ± | 0.5 | 0 | 1.5 |
| Yellow Birch | 0 | | | | 0 | | | | 0.5 ± | 0.5 | 0 | 1.5 |
| Sugar Maple | 0 | | | | 0 | | | | 0.5 ± | 0.5 | 0 | 1.5 |
| Striped Maple | 0 | | | | 1.6 ± | 0.9 | 0 | 3.4 | 0.5 ± | 0.5 | 0 | 1.5 |
| Balsam Fir | 1.5 ± | 1.1 | 0 | 3.6 | 0 | | | | 0 | | | |
| Bigtooth Aspen | 0 | | | | 0.4 ± | 0.4 | 0 | 1.2 | 0 | | | |
| **Totals:** | **3.5 ±** | **2.0** | **0** | **7.4** | **7.2 ±** | **1.9** | **3.5** | **10.9** | **12.5 ±** | **2.0** | **8.5** | **16.5** |

Table 27. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC05 (Pemetic Mountain) in 1980, 1992-1994 and 2016.

| AC05 | 1980 | | 95% CI | | 1993 | | 95% CI | | 2016 | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper |
| Hemlock | 10.2 | ± 1.2 | 7.9 | 12.6 | | | | | 18.2 | ± 1.3 | 15.7 | 20.7 |
| White Ash | 6.0 | ± 0 | 6.0 | 6.0 | 13.0 | ± 0.3 | 12.5 | 13.5 | 13.7 | ± 0.5 | 12.8 | 14.6 |
| Sugar Maple | 8.7 | ± 0 | 8.7 | 8.7 | | | | | 12.9 | ± 0.8 | 11.3 | 14.5 |
| White Pine | | | | | | | | | 12.6 | ± 2.5 | 7.6 | 17.5 |
| No. White Cedar | 7.4 | ± 0.6 | 6.2 | 8.6 | 8.4 | ± 0.7 | 7.1 | 9.7 | 10.9 | ± 0.7 | 9.5 | 12.3 |
| Bigtooth Aspen | 12.7 | ± 0.8 | 11.1 | 14.3 | 9.9 | ± 1.0 | 8.0 | 11.8 | 10.8 | ± 0.6 | 9.7 | 12.0 |
| Red Spruce | 8.9 | ± 0.7 | 7.5 | 10.3 | 9.5 | ± 0.8 | 8.0 | 11.0 | 9.8 | ± 0.9 | 8.0 | 11.6 |
| Yellow Birch | 5.8 | ± 0.7 | 4.3 | 7.2 | 7.1 | ± 0.4 | 6.3 | 7.9 | 9.8 | ± 0.5 | 8.8 | 10.7 |
| Paper Birch | 7.4 | ± 0.6 | 6.2 | 8.6 | 6.1 | ± 0.5 | 5.1 | 7.1 | 9.0 | ± 0.6 | 7.9 | 10.2 |
| Shadbush | 4.1 | ± 0 | 4.1 | 4.1 | | | | | 5.7 | ± 0 | 5.7 | 5.7 |
| Striped Maple | 4.1 | ± 0.3 | 3.6 | 4.7 | 3.7 | ± 0.3 | 3.1 | 4.3 | 3.9 | ± 0.3 | 3.4 | 4.4 |
| Balsam Fir | 8.9 | ± 1.1 | 6.6 | 11.1 | | | | | | | | |
| Grey Birch | | | | | 3.7 | ± 0 | 3.7 | 3.7 | | | | |
| Trembling Aspen | 4.7 | ± 0 | 4.7 | 4.7 | | | | | | | | |
| **Overall Average** | **7.5** | **± 0.7** | **6.2** | **8.8** | **7.7** | **± 0.7** | **6.4** | **9.1** | **10.2** | **± 0.8** | **8.6** | **11.8** |

Sargent Mountain (AC06) – Northern White Cedar

Basal areas of northern white cedar, red maple, and paper birch decreased, while those of white pine and red spruce increased between 1980 and 2016. Northern white decreased significantly. Total live basal area decreased significantly between 1980 and 1993 from 262.7 to 172.5 ft$^2$/acre and shows a decreasing trend with 166.0 ft$^2$/acre in 2016 (Table 28). Total dead basal area has remained stable with 35.0 ft$^2$/acre in 1980 and 33.0 ft$^2$/acre in 2016 (Table 28). Density of live trees decreased from 746.7 in 1980 to 449.3 stems per acre in 2016 (Table 8). Density of dead trees decreased significantly from 146.9 in 1980 to 65.1 stems per acre in 2016 (Table 8). Data for density of saplings is missing from historic records. Mean diameters (in.) increased from 12.0 in 1980 to 14.4 in 2016 (Table 29).

Biomass (T/acre) of live trees decreased from 100.5 in 1980 to 71.9 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 10.2 in 1980 to 12.8 in 2016 (Table 10). Biomass (T/acre) of shrubs increased from less than 0.04 live and no measurable dead in 1980 to 1.1 live and less than 0.04 dead in 2016 (Table 10). Mature trees continue to be lost to blowdown as evidenced in the increase in heavy downed-woody fuel load. Total downed woody fuel load increased from 8.8 T/acre in 1980 to 15.8 T/acre in 2016 (Table 11). This is an 80% increase in downed woody fuel load over the 36-year period from 1980 to 2016. Duff depth (in.) decreased from 5.2 in 1980 to 3.7 in 2016 (Table 11). Fuel height (in.) increased from 3.3 in 1980 to 5.9 in 2016 (Table 11).

Table 28. Average basal area (X̄ BA) of live and dead trees ($ft^2$/acre) ± standard error with 95% confidence intervals for stand AC06 (Sargent Mountain) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC06- Sargent Mountain | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | | | **1993** | | | | | **2016** | | | | |
| *No. Points Sampled:* | *21* | | | 95% CI | | *20* | | | 95% CI | | *20* | | | 95% CI | |
| Live | X̄ BA | ± | SE | Lower | Upper | X̄ BA | ± | SE | Lower | Upper | X̄ BA | ± | SE | Lower | Upper |
| No. White Cedar | 151.8 | ± | 12.9 | 126.6 | 177.0 | 80.5 | ± | 13.4 | 54.2 | 106.8 | 64.5 | ± | 10.5 | 43.9 | 85.1 |
| White Pine | 22.7 | ± | 5.7 | 11.6 | 33.8 | 16.0 | ± | 4.4 | 7.4 | 24.6 | 29.0 | ± | 8.1 | 13.1 | 44.9 |
| Red Maple | 42.7 | ± | 7.7 | 27.6 | 57.8 | 24.5 | ± | 6.0 | 12.7 | 36.3 | 27.0 | ± | 5.0 | 17.2 | 36.8 |
| Red Spruce | 19.1 | ± | 4.7 | 9.9 | 28.3 | 20.5 | ± | 4.3 | 12.0 | 29.0 | 26.5 | ± | 5.4 | 15.9 | 37.1 |
| Paper Birch | 10.0 | ± | 2.4 | 5.4 | 14.6 | 9.5 | ± | 2.9 | 3.9 | 15.1 | 5.0 | ± | 1.5 | 2.0 | 8.0 |
| Red Pine | 0.9 | ± | 0.9 | 0 | 2.7 | 2.5 | ± | 1.4 | 0 | 5.3 | 4.0 | ± | 2.0 | 0.1 | 7.9 |
| Red Oak | 1.8 | ± | 1.1 | 0 | 3.9 | 4.0 | ± | 2.0 | 0.1 | 7.9 | 3.5 | ± | 1.5 | 0.6 | 6.4 |
| Balsam Fir | 3.2 | ± | 2.0 | 0 | 7.2 | 1.0 | ± | 0.7 | 0 | 2.4 | 2.5 | ± | 1.2 | 0.1 | 4.9 |
| Bigtooth Aspen | 0.5 | ± | 0.4 | 0 | 1.4 | 4.0 | ± | 3.0 | 0 | 9.9 | 2.0 | ± | 1.6 | 0 | 5.1 |
| White Ash | 5.0 | ± | 2.6 | 0 | 10.1 | 3.0 | ± | 1.3 | 0.5 | 5.5 | 1.5 | ± | 1.1 | 0 | 3.6 |
| Hemlock | 1.4 | ± | 1.0 | 0 | 3.4 | 1.5 | ± | 0.8 | 0 | 3.1 | 0.5 | ± | 0.5 | 0 | 1.5 |
| Yellow Birch | 1.8 | ± | 1.1 | 0 | 3.9 | 1.0 | ± | 1.0 | 0 | 3.0 | 0 | | | | |
| Sugar Maple | 0 | | | | | 2.5 | ± | 1.2 | 0.1 | 4.9 | 0 | | | | |
| Striped Maple | 0 | | | | | 0.5 | ± | 0.5 | 0 | 1.5 | 0 | | | | |
| Trembling Aspen | 0 | | | | | 1.0 | ± | 0.7 | 0 | 2.4 | 0 | | | | |
| White Spruce | 0.9 | ± | 0.6 | 0 | 2.1 | 0 | | | | | 0 | | | | |
| Shadbush | 0.9 | ± | 0.6 | 0 | 2.1 | 0 | | | | | 0 | | | | |
| Mountain Ash | 0 | | | | | 0.5 | ± | 0.5 | 0 | 1.5 | 0 | | | | |
| **Totals:** | **262.7** | **±** | **17.4** | **228.6** | **296.8** | **172.5** | **±** | **12.5** | **148.1** | **196.9** | **166.0** | **±** | **12.7** | **141.1** | **190.9** |
| Dead | | | | | | | | | | | | | | | |
| No. White Cedar | 26.8 | ± | 5.6 | 15.8 | 37.8 | 20.5 | ± | 6.8 | 7.1 | 33.9 | 18.0 | ± | 5.7 | 6.8 | 29.2 |
| Red Spruce | 3.2 | ± | 1.5 | 0.2 | 6.2 | 5.0 | ± | 1.9 | 1.4 | 8.6 | 4.0 | ± | 1.5 | 1.0 | 7.0 |
| Paper Birch | 0.9 | ± | 0.6 | 0 | 2.1 | 0.5 | ± | 0.5 | 0 | 1.5 | 4.0 | ± | 1.1 | 1.8 | 6.2 |
| Red Maple | 0.9 | ± | 0.9 | 0 | 2.7 | 0 | | | | | 4.0 | ± | 1.8 | 0.4 | 7.6 |
| White Pine | 0.9 | ± | 0.6 | 0 | 2.1 | 1.0 | ± | 0.7 | 0 | 2.4 | 2.0 | ± | 0.9 | 0.2 | 3.8 |
| Balsam Fir | 2.3 | ± | 1.1 | 0.1 | 4.5 | 0 | | | | | 0.5 | ± | 0.5 | 0 | 1.5 |
| Red Pine | 0 | | | | | 0 | | | | | 0.5 | ± | 0.5 | 0 | 1.5 |
| Yellow Birch | 0 | | | | | 0.5 | ± | 0.5 | 0 | 1.5 | 0 | | | | |
| **Totals:** | **35.0** | **±** | **6.0** | **23.3** | **46.7** | **27.5** | **±** | **7.0** | **13.9** | **41.1** | **33.0** | **±** | **5.6** | **22.1** | **43.9** |

Table 29. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC06 (Sargent Mountain) in 1980, 1992-1994 and 2016.

| AC06 | 1980 | 95% CI | | 1993 | 95% CI | | 2016 | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper |
| Hemlock | 10.8 ± 1.2 | 8.5 | 13.1 | 21.5 ± 1.2 | 19.2 | 23.8 | 30.1 ± 0 | 30.1 | 30.1 |
| White Pine | 22.3 ± 1.3 | 19.7 | 24.9 | 24.3 ± 1.5 | 21.3 | 27.3 | 26.2 ± 1.7 | 22.9 | 29.6 |
| Bigtooth Aspen | 9.7 ± 0 | 9.7 | 9.7 | 15.5 ± 0.5 | 14.4 | 16.6 | 18.5 ± 0.5 | 17.4 | 19.5 |
| Red Pine | 15.7 ± 0.5 | 14.7 | 16.6 | 15.6 ± 1.1 | 13.5 | 17.7 | 16.3 ± 1.1 | 14.1 | 18.5 |
| Red Oak | 11.3 ± 0.4 | 10.6 | 12.1 | 14.2 ± 1.2 | 11.8 | 16.6 | 16.1 ± 1.2 | 13.8 | 18.5 |
| No. White Cedar | 11.2 ± 0.9 | 9.4 | 13.1 | 10.9 ± 0.8 | 9.2 | 12.6 | 12.6 ± 0.8 | 10.9 | 14.2 |
| Red Maple | 10.5 ± 0.9 | 8.8 | 12.3 | 9.4 ± 0.9 | 7.6 | 11.1 | 11.6 ± 1.0 | 9.7 | 13.5 |
| Red Spruce | 12.6 ± 1.0 | 10.7 | 14.5 | 11.1 ± 1.1 | 8.9 | 13.3 | 11.2 ± 1.5 | 8.3 | 14.0 |
| Paper Birch | 11.4 ± 0.8 | 9.8 | 13.1 | 9.3 ± 0.7 | 7.9 | 10.6 | 10.9 ± 0.7 | 9.7 | 12.2 |
| White Ash | 9.9 ± 1.6 | 6.7 | 13.1 | 12.8 ± 1.9 | 9.1 | 16.5 | 10.1 ± 1.3 | 7.5 | 12.7 |
| Balsam Fir | 5.0 ± 0.3 | 4.3 | 5.6 | 3.7 ± 0.2 | 3.2 | 4.1 | 6.3 ± 0.5 | 5.2 | 7.3 |
| Striped Maple | | | | 2.4 ± 0 | 2.4 | 2.4 | | | |
| Sugar Maple | | | | 7.5 ± 0.9 | 5.7 | 9.2 | | | |
| Shadbush | 6.8 ± 0.8 | 5.1 | 8.4 | | | | | | |
| Yellow Birch | 9.2 ± 0.9 | 7.3 | 11.0 | 9.5 ± 0.6 | 8.3 | 10.6 | | | |
| White Spruce | 12.8 ± 0.4 | 12.1 | 13.5 | | | | | | |
| Trembling Aspen | | | | 5.5 ± 0.6 | 4.3 | 6.6 | | | |
| Mountain Ash | | | | 7.3 ± 0 | 7.3 | 7.3 | | | |
| **Overall Average** | **12.0 ± 1.2** | **9.6** | **14.3** | **12.0 ± 1.3** | **9.4** | **14.6** | **14.4 ± 1.7** | **11.2** | **17.6** |

*Northern White Cedar Summary*

Barring stand replacing disturbance and assuming a predictable successional trajectory without disturbance I expect Pemetic Mountain (AC05) to remain as northern white cedar. This stand has remained relatively stable. There was some difficulty relocating the exact stand in 1993, and the changes observed are more likely due to the 1993 sample plots being in a somewhat different location than that of 1980 rather than actual decline in species (Patterson 1996). This is supported by the fact that dead standing cedar and downed woody fuel load did not increase substantially across the sample period. Given the steep slopes of this stand, I expect it to remain as is with an open canopy, or it is possible it could move to conditions more like those of Sargent Mountain which is approximately 30 years older than that of Pemetic Mountain. It is unlikely, however, that this stand will accumulate the duff and depth to bedrock as Sargent Mountain given the steep slopes, large boulders, and rocky substrate. The fire hazard of this stand is very high primarily due to the steep slopes and flammable nature of species present.

The Sargent Mountain (AC06) stand is breaking up as evidenced in the increase in downed fuel load. It is possible given the changes in basal area observed over the 36-year period from 1980 to 2016 that this stand could change to a mixed conifer stand in the future. We might expect to see red spruce as the dominant cover type with some component of white pine and northern white cedar in the canopy, such as that of the older mixed conifer stand located on Norumbega Mountain (AC11). It should be noted that this stand was also hard to relocate in 1993 so results described here could be misleading, and it is possible that comparisons should only be made between 1993 and 2016 given the uncertainties in the location of the plots in 1980 (Patterson 1996). However, the overall trends remain the same across the 36-year sample period. Fuel loading and fire hazard are high in both stands and could support high-intensity fire under the right conditions.

*Mixed Hardwood-Conifer* (n=4)

Day Mountain (AC23) – Mixed Hardwood-Conifer

Basal area of balsam fir increased by 44%, while bigtooth aspen decreased by 57% and paper birch decreased significantly between 1980 and 2016. Striped maple has almost completely dropped out of the stand with a decrease in basal area of 6.8 $ft^2$/acre in 1980 and 0.5 $ft^2$/acre in 2016. Total live basal area decreased between 1980 and 1993 from 170.8 to 139.7 $ft^2$/acre, and shows a decreasing trend with 131.0 $ft^2$/acre in 2016 (Table 30). Total dead basal area shows an increasing trend with 13.6 $ft^2$/acre in 1980 and 21.5 $ft^2$/acre in 2016 (Table 30). Density of live trees decreased significantly from 730.7 in 1980 to 353.0 stems per acre in 2016 (Table 8). Density of dead trees decreased from 77.9 in 1980 to 72.5 stems per acre in 2016 (Table 8). Density of live saplings decreased significantly and measured at 325.5 in 1993 and 180.0 stems per acre in 2016 (Table 9). Average stem density of dead saplings also decreased significantly from 150.9 in 1993 to 20.0 stems per acre in 2016 (Table 9). This is expected with succession as stem diameters increase density decreases. Mean diameters (in.) increased overall from 11.0 in 1980 to 13.5 in 2016 (Table 31).

Mature trees continue to be lost to blowdown as evidenced in the increase in heavy downed-woody fuel load. Biomass (T/acre) of live trees decreased from 82.5 in 1980 to 73.8 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 6.0 in 1980 to 12.5 in 2016 (Table 10). Biomass (T/acre) of shrubs remained stable from 0.1 live and less than 0.01 dead in 1980 to 0.1 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load increased by 127% from 5.5 T/acre in 1980 to 12.5 T/acre in 2016 (Table 11). Duff depth (in.)

remained stable from 2.4 in 1980 to 2.5 in 2016 (Table 11). Fuel height (in.) increased from 2.9 in 1980 to 4.6 in 2016 (Table 11).

Table 30. Average basal area ($\bar{X}$ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC23 (Day Mountain) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC23-Day Mountain | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *25* | 95% CI | | *55* | 95% CI | | *20* | 95% CI | |
| Live | $\bar{X}$ BA ± SE | Lower | Upper | $\bar{X}$ BA ± SE | Lower | Upper | $\bar{X}$ BA SE | Lower | Upper |
| Red Spruce | 44.8 ± 7.0 | 31.2 | 58.4 | 38.2 ± 5.1 | 28.2 | 48.2 | 45.0 ± 8.7 | 28.0 | 62.0 |
| Red Maple | 39.2 ± 7.1 | 25.2 | 53.2 | 26.5 ± 3.3 | 19.9 | 33.1 | 20.0 ± 3.2 | 13.6 | 26.4 |
| Balsam Fir | 9.2 ± 2.4 | 4.5 | 13.9 | 11.8 ± 2.3 | 7.3 | 16.3 | 14.0 ± 4.3 | 5.5 | 22.5 |
| Bigtooth Aspen | 28.8 ± 4.6 | 19.7 | 37.9 | 11.5 ± 2.4 | 6.8 | 16.2 | 12.5 ± 3.9 | 4.9 | 20.1 |
| Yellow Birch | 10.4 ± 2.7 | 5.1 | 15.7 | 10.5 ± 2.4 | 5.8 | 15.2 | 10.0 ± 3.2 | 3.8 | 16.2 |
| White Pine | 2.8 ± 1.5 | 0 | 5.7 | 5.1 ± 1.4 | 2.4 | 7.8 | 7.5 ± 2.4 | 2.8 | 12.2 |
| Sugar Maple | 4.0 ± 2.2 | 0 | 8.4 | 2.5 ± 1.3 | 0 | 5.1 | 7.0 ± 3.5 | 0.2 | 13.8 |
| White Ash | 2.0 ± 1.3 | 0 | 4.5 | 2.0 ± 0.8 | 0.4 | 3.6 | 6.0 ± 2.5 | 1.2 | 10.8 |
| No. White Cedar | 5.2 ± 2.1 | 1.1 | 9.3 | 3.8 ± 1.3 | 1.3 | 6.3 | 3.0 ± 2.5 | 0 | 8.0 |
| Paper Birch | 13.2 ± 3.9 | 5.6 | 20.8 | 14.2 ± 2.1 | 10.1 | 18.3 | 2.0 ± 0.9 | 0.2 | 3.8 |
| Hemlock | 2.8 ± 1.2 | 0.4 | 5.2 | 1.1 ± 0.7 | 0 | 2.4 | 1.5 ± 1.1 | 0 | 3.6 |
| Trembling Aspen | 0 | | | 0.4 ± 0.4 | 0 | 1.1 | 1.0 ± 1.0 | 0 | 3.0 |
| American Beech | 0.4 ± 0.4 | 0 | 1.2 | 0.5 ± 0.4 | 0 | 1.3 | 0.5 ± 0.5 | 0 | 1.5 |
| Striped Maple | 6.8 ± 2.7 | 1.5 | 12.1 | 10.5 ± 1.8 | 7.0 | 14.0 | 0.5 ± 0.5 | 0 | 1.5 |
| Shadbush | 0.8 ± 0.6 | 0 | 1.9 | 0.7 ± 0.4 | 0 | 1.4 | 0.5 ± 0.5 | 0 | 1.5 |
| Red Pine | 0.4 ± 0.4 | 0 | 1.2 | 0.2 ± 0.2 | 0 | 0.5 | 0 | | |
| White Spruce | 0 | | | 0.2 ± 0.2 | 0 | 0.5 | 0 | | |
| **Totals:** | **170.8 ± 10.1** | **151.0** | **190.6** | **139.7 ± 5.7** | **128.6** | **150.8** | **131.0 ± 7.5** | **116.4** | **145.6** |
| Dead | | | | | | | | | |
| Red Spruce | 1.6 ± 0.7 | 0.1 | 3.1 | 1.6 ± 0.7 | 0.3 | 2.9 | 5.0 ± 1.5 | 2.0 | 8.0 |
| Paper Birch | 1.6 ± 0.7 | 0.1 | 3.1 | 5.3 ± 1.3 | 2.7 | 7.9 | 4.5 ± 1.5 | 1.5 | 7.5 |
| Balsam Fir | 1.6 ± 0.7 | 0.1 | 3.1 | 3.3 ± 1.1 | 1.1 | 5.5 | 4.5 ± 2.1 | 0.4 | 8.6 |
| Bigtooth Aspen | 2.0 ± 0.8 | 0.4 | 3.6 | 1.1 ± 0.5 | 0.1 | 2.1 | 2.5 ± 1.4 | 0 | 5.3 |
| Yellow Birch | 1.2 ± 0.7 | 0 | 2.5 | 0.2 ± 0.2 | 0 | 0.5 | 2.0 ± 1.6 | 0 | 5.1 |
| Red Maple | 0.8 ± 0.6 | 0 | 1.9 | 1.3 ± 0.5 | 0.3 | 2.3 | 1.5 ± 1.1 | 0 | 3.6 |
| American Beech | 0 | | | 0.4 ± 0.3 | 0 | 0.9 | 1.0 ± 1.0 | 0 | 3.0 |
| No. White Cedar | 1.2 ± 0.9 | 0 | 2.9 | 0.2 ± 0.2 | 0 | 0.5 | 0.5 ± 0.5 | 0 | 1.5 |
| White Ash | 0 | | | 0.2 ± 0.2 | 0 | 0.5 | 0 | | |
| Striped Maple | 3.2 ± 1.5 | 0.3 | 6.1 | 1.8 ± 0.6 | 0.7 | 2.9 | 0 | | |
| White Spruce | 0 | | | 0.2 ± 0.2 | 0 | 0.5 | 0 | | |
| Grey Birch | 0.4 ± 0.4 | 0 | 1.2 | 0 | | | 0 | | |
| White Pine | 0 | | | 0.2 ± 0.2 | 0 | 0.5 | 0 | | |
| Trembling Aspen | 0 | | | 0 | | | 0 | | |
| **Totals:** | **13.6 ± 2.5** | **8.7** | **18.5** | **15.8 ± 1.8** | **12.3** | **19.3** | **21.5 ± 3.9** | **13.8** | **29.2** |

Table 31. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC23 (Day Mountain) in 1980, 1992-1994 and 2016.

| AC23 | 1980 | 95% CI | | 1993 | 95% CI | | 2016 | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper |
| White Pine | 30.4 ± 1.6 | 27.3 | 33.6 | 26.6 ± 1.1 | 24.3 | 28.8 | 29.9 ± 2.4 | 25.2 | 34.6 |
| Hemlock | 16.3 ± 1.4 | 13.5 | 19.1 | 17.9 ± 0.9 | 16.1 | 19.6 | 24.5 ± 2.0 | 20.6 | 28.5 |
| Bigtooth Aspen | 12.9 ± 0.5 | 11.9 | 13.9 | 14.4 ± 0.4 | 13.5 | 15.2 | 17.7 ± 0.9 | 16.0 | 19.3 |
| Sugar Maple | 14.8 ± 1.0 | 12.8 | 16.8 | 14.5 ± 0.7 | 13.1 | 16.0 | 15.9 ± 0.9 | 14.2 | 17.6 |
| Red Spruce | 11.4 ± 1.2 | 9.1 | 13.7 | 12.7 ± 0.9 | 11.0 | 14.4 | 13.6 ± 1.5 | 10.7 | 16.5 |
| Red Maple | 9.8 ± 0.8 | 8.1 | 11.4 | 10.5 ± 0.6 | 9.3 | 11.7 | 13.1 ± 0.8 | 11.5 | 14.7 |
| No. White Cedar | 11.0 ± 0.8 | 9.4 | 12.6 | 11.4 ± 0.5 | 10.4 | 12.4 | 12.1 ± 0.9 | 10.3 | 13.9 |
| Paper Birch | 9.8 ± 0.7 | 8.4 | 11.1 | 11.7 ± 0.6 | 10.6 | 12.9 | 12.0 ± 0.8 | 10.4 | 13.6 |
| White Ash | 12.8 ± 1.3 | 10.1 | 15.4 | 10.4 ± 0.5 | 9.3 | 11.4 | 11.4 ± 1.2 | 9.1 | 13.7 |
| Yellow Birch | 13.1 ± 0.8 | 11.5 | 14.6 | 11.2 ± 0.8 | 9.7 | 12.7 | 11.2 ± 0.7 | 9.9 | 12.5 |
| Shadbush | 3.2 ± 0.3 | 2.6 | 3.7 | 5.9 ± 0.3 | 5.4 | 6.4 | 9.3 ± 0 | 9.3 | 9.3 |
| Trembling Aspen | | | | 12.8 ± 0.3 | 12.2 | 13.3 | 8.8 ± 0.7 | 7.4 | 10.2 |
| American Beech | 5.0 ± 0 | 5.0 | 5.0 | 5.4 ± 0.3 | 4.8 | 6.0 | 6.6 ± 0.6 | 5.4 | 7.8 |
| Balsam Fir | 4.3 ± 0.7 | 3.0 | 5.6 | 4.0 ± 0.2 | 3.5 | 4.5 | 6.3 ± 0.5 | 5.2 | 7.3 |
| Striped Maple | 6.1 ± 0.6 | 5.0 | 7.2 | 4.2 ± 0.2 | 3.7 | 4.6 | 3.4 ± 0 | 3.4 | 3.4 |
| Grey Birch | 9.3 ± 0 | 9.3 | 9.3 | | | | | | |
| White Spruce | | | | 24.8 ± 0.2 | 24.5 | 25.1 | | | |
| Red Pine | 21.3 ± 0 | 21.3 | 21.3 | 22.5 ± 0 | 22.5 | 22.5 | | | |
| **Overall Average** | **11.0 ± 1.1** | **8.8** | **13.2** | **11.1 ± 0.9** | **9.4** | **12.9** | **13.5 ± 1.6** | **10.4** | **16.7** |

Long Pond (AC16) – Mixed Hardwood-Conifer

Basal areas of red maple, paper birch, sugar maple, bigtooth aspen, and balsam fir decreased, while those of red spruce and hemlock increased between 1980 and 2016. Live basal area of red spruce increased by 96%. Total live basal area decreased between 1980 and 1993 from 170.6 to 160.7 $ft^2$/acre and shows a decreasing trend with 136.0 $ft^2$/acre in 2016 (Table 32). Total dead basal area shows an increasing trend with 15.6 $ft^2$/acre in 1980 and 25.0 $ft^2$/acre in 2016 (Table 32). Density of live trees decreased significantly from 1128.4 in 1993 to 346.9 stems per acre in 2016 (Table 8). Density of dead trees also decreased from 248.6 in 1993 to 73.4 stems per acre in 2016 (Table 8). Density of live saplings decreased and measured at

425.6 in 1993 and 355.0 stems per acre in 2016 (Table 9). Density of dead saplings also decreased from 73.1 in 1993 to 35.0 stems per acre in 2016 (Table 9). Mean diameters (in.) increased from 9.1 in 1980 to 12.6 in 2016 (Table 33).

This stand is continuing to accumulate biomass, while mature canopy trees continue to be lost to blowdown as evidenced in the increase in heavy downed-woody fuel load. Biomass (T/acre) of trees increased from 12.5 live and 7.2 dead in 1980 to 48.6 live and 16.5 dead in 2016 (Table 10). Biomass (T/acre) of shrubs increased from 0.3 live and no measurable dead in 1980 to 0.6 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load increased from 7.2 T/acre in 1980 to 11.6 T/acre in 2016 (Table 11). Duff depth (in.) increased overall from 3.4 in 1980 to 4.9 in 2016 (Table 11). Fuel height (in.) decreased from 2.8 in 1980 to 1.0 in 2016 (Table 11).

Table 32. Average basal area ($\bar{X}$ BA) of live and dead trees ($ft^2$/acre) ± standard error with 95% confidence intervals for stand AC16 (Long Pond) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 1993 and 2016 data.

| AC16-Long Pond | | | | | | | |
|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *22* | *43* | 95% CI | | *20* | 95% CI | |
| Live | $\bar{X}$ BA | $\bar{X}$ BA ± SE | Lower | Upper | $\bar{X}$ BA ± SE | Lower | Upper |
| Red Spruce | 36.7 | 84.9 ± 6.3 | 72.6 | 97.2 | 72.0 ± 8.9 | 54.6 | 89.4 |
| Red Maple | 46.7 | 35.1 ± 3.9 | 27.4 | 42.8 | 25.0 ± 5.0 | 15.2 | 34.8 |
| Yellow Birch | 10.0 | 4.0 ± 1.2 | 1.6 | 6.4 | 9.5 ± 4.2 | 1.3 | 17.7 |
| Paper Birch | 37.6 | 17.9 ± 3.5 | 11.1 | 24.7 | 9.0 ± 2.9 | 3.3 | 14.7 |
| Hemlock | 2.9 | 1.9 ± 1.0 | 0 | 3.9 | 6.0 ± 2.8 | 0.6 | 11.4 |
| Sugar Maple | 5.2 | 0.7 ± 0.5 | 0 | 1.7 | 3.5 ± 2.5 | 0 | 8.5 |
| Bigtooth Aspen | 11.0 | 1.4 ± 1.4 | 0 | 4.1 | 3.5 ± 2.2 | 0 | 7.8 |
| Balsam Fir | 6.2 | 3.0 ± 1.1 | 0.9 | 5.1 | 2.5 ± 1.4 | 0 | 5.3 |
| White Pine | 1.4 | 3.0 ± 1.0 | 1.0 | 5.0 | 2.0 ± 0.9 | 0.2 | 3.8 |
| No. White Cedar | 1.0 | 5.1 ± 1.8 | 1.6 | 8.6 | 1.0 ± 0.7 | 0 | 2.4 |
| American Beech | 3.3 | 0.5 ± 0.3 | 0 | 1.1 | 1.0 ± 1.0 | 0 | 3.0 |
| Striped Maple | 3.3 | 0.9 ± 0.6 | 0 | 2.0 | 0.5 ± 0.5 | 0 | 1.5 |
| Shadbush | 1.0 | 0.2 ± 0.2 | 0 | 0.6 | 0.5 ± 0.5 | 0 | 1.5 |
| Red Pine | 0.5 | 0 | | | 0 | | |
| White Ash | 0.5 | 1.9 ± 0.9 | 0.1 | 3.7 | 0 | | |
| Black Ash | 3.3 | 0 | | | 0 | | |
| Trembling Aspen | 0 | 0.2 ± 0.2 | 0 | 0.6 | 0 | | |
| **Totals:** | **170.6** | **160.7 ± 6.9** | **147.2** | **174.2** | **136.0 ± 9.3** | **117.8** | **154.2** |
| Dead | | | | | | | |
| Paper Birch | 1.4 | 7.7 ± 1.9 | 3.9 | 11.5 | 9.5 ± 2.3 | 4.9 | 14.1 |
| Red Spruce | 5.5 | 4.7 ± 1.4 | 2.0 | 7.4 | 7.5 ± 2.0 | 3.5 | 11.5 |
| Red Maple | 3.2 | 3.0 ± 0.9 | 1.3 | 4.7 | 3.0 ± 1.8 | 0 | 6.5 |
| American Beech | 0 | 0.9 ± 0.4 | 0 | 1.8 | 2.0 ± 1.4 | 0 | 4.7 |
| Bigtooth Aspen | 0.5 | 0.2 ± 0.2 | 0 | 0.6 | 1.0 ± 1.0 | 0 | 3.0 |
| No. White Cedar | 0 | 1.9 ± 1.0 | 0 | 3.8 | 0.5 ± 0.5 | 0 | 1.5 |
| Yellow Birch | 1.8 | 2.1 ± 1.9 | 0 | 5.8 | 0.5 ± 0.5 | 0 | 1.5 |
| Sugar Maple | 0 | 0 | | | 0.5 ± 0.5 | 0 | 1.5 |
| White Ash | 0.9 | 0 | | | 0.5 ± 0.5 | 0 | 1.5 |
| Balsam Fir | 2.3 | 1.6 ± 0.7 | 0.2 | 3.0 | 0 | | |
| Striped Maple | 0 | 0.2 ± 0.2 | 0 | 0.6 | 0 | | |
| **Totals:** | **15.6** | **22.3 ± 3.5** | **15.5** | **29.1** | **25.0 ± 3.0** | **19.2** | **30.8** |

Table 33. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC16 (Long Pond) in 1992-1994 and 2016. Data missing for 1980.

| AC16 | 1980 | 1993 | | 95% CI | | 2016 | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper |
| White Pine | x | 19.8 | ± 2.7 | 14.6 | 25.1 | 26.0 | ± 1.2 | 23.5 | 28.4 |
| Hemlock | x | 21.3 | ± 0.8 | 19.7 | 22.9 | 23.6 | ± 1.4 | 20.8 | 26.4 |
| Bigtooth Aspen | x | 15.3 | ± 0.3 | 14.7 | 15.9 | 18.1 | ± 0.5 | 17.2 | 19.1 |
| American Beech | x | 9.9 | ± 0.6 | 8.6 | 11.1 | 16.7 | ± 1.1 | 14.5 | 18.8 |
| No. White Cedar | x | 10.6 | ± 1.2 | 8.3 | 12.8 | 16.5 | ± 1.2 | 14.2 | 18.8 |
| Red Maple | x | 8.4 | ± 0.7 | 7.0 | 9.9 | 12.9 | ± 1.0 | 10.9 | 15.0 |
| Paper Birch | x | 9.6 | ± 0.6 | 8.4 | 10.8 | 11.9 | ± 0.6 | 10.6 | 13.1 |
| Red Spruce | x | 8.5 | ± 1.2 | 6.2 | 10.8 | 11.6 | ± 1.0 | 9.7 | 13.5 |
| Yellow Birch | x | 9.9 | ± 1.1 | 7.7 | 12.1 | 11.1 | ± 0.9 | 9.4 | 12.8 |
| White Ash | x | 9.5 | ± 0.8 | 7.9 | 11.0 | 10.6 | ± 0 | 10.6 | 10.6 |
| Sugar Maple | x | 15.4 | ± 1.1 | 13.2 | 17.6 | 8.2 | ± 0.7 | 6.8 | 9.6 |
| Balsam Fir | x | 5.8 | ± 0.5 | 4.8 | 6.8 | 8.1 | ± 0.5 | 7.2 | 9.0 |
| Striped Maple | x | 4.6 | ± 0.3 | 4.1 | 5.1 | 8.1 | ± 0 | 8.1 | 8.1 |
| Shadbush | x | 4.3 | ± 0 | 4.3 | 4.3 | 6.4 | ± 0 | 6.4 | 6.4 |
| Trembling Aspen | x | 9.8 | ± 0 | 9.8 | 9.8 | | | | |
| **Overall Average** | | **9.1** | **± 1.1** | **6.9** | **11.3** | **12.6** | **± 1.2** | **10.4** | **14.9** |

Jerusalem Mountain (IaH) (AC15) – Mixed Hardwood-Conifer

At Jerusalem Mountain, for the period of 1980 to 2016, live basal area of red spruce decreased by 13%, northern white cedar by 23%, and yellow birch by 48%. Live basal area of dead red spruce increased by 147% from 1980 to 2016. Total live basal area decreased between 1980 and 1993 from 181.8 to 162.9 $ft^2$/acre, and to 151.5 $ft^2$/acre in 2016 (Table 34). Total dead basal area shows an increasing trend with 21.9 $ft^2$/acre in 1980 and 30.5 $ft^2$/acre in 2016 (Table 34). Density of trees and saplings is missing for the historic sampling periods (Table 8, Table 9). Mean diameter data are also missing for the historic sampling period (Table 35).

Biomass (T/acre) of live trees decreased from 76.3 in 1980 to 65.0 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 10.4 in 1980 to 13.0 in 2016 (Table 10). Biomass

(T/acre) of shrubs increased 87-fold from 0.1 in 1980 to 8.7 in 2016 (Table 10). Total downed woody fuel load increased from 11.4 T/acre in 1980 to 12.3 T/acre in 2016 (Table 11). Duff depth (in.) increased overall from 4.0 in 1980 to 5.9 in 2016 (Table 11). Fuel height (in.) increased from 3.6 in 1980 to 5.2 in 2016 (Table 11).

Table 34. Average basal area (X̅ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC15 (Jerusalem Mountain) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 2016 data.

| AC15-Jerusalem Mtn. (IaH) | | | | | |
|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | **2016** | | |
| *No. Points Sampled:* | *28* | *28* | *20* | *95% CI* | |
| Live | X̅ BA | X̅ BA | X̅ BA ± SE | Lower | Upper |
| Red Spruce | 86.4 | 92.9 | 75.0 ± 6.2 | 62.9 | 87.1 |
| No. White Cedar | 50.4 | 38.9 | 39.0 ± 10.1 | 19.2 | 58.8 |
| Red Maple | 22.5 | 11.1 | 24.5 ± 6.3 | 12.1 | 36.9 |
| Yellow Birch | 15.4 | 13.7 | 8.0 ± 4.6 | 0 | 17.1 |
| White Pine | 3.2 | 4.4 | 1.5 ± 0.8 | 0 | 3.1 |
| Red Pine | 0 | 0 | 1.5 ± 0.8 | 0 | 3.1 |
| Paper Birch | 2.1 | 0.4 | 1.0 ± 1.0 | 0 | 3.0 |
| Striped Maple | 0 | 0 | 0.5 ± 0.5 | 0 | 1.5 |
| Shadbush | 0 | 0 | 0.5 ± 0.5 | 0 | 1.5 |
| Balsam Fir | 0.7 | 1.5 | 0 | | |
| Bigtooth Aspen | 0.4 | 0 | 0 | | |
| Trembling Aspen | 0.7 | 0 | 0 | | |
| **Totals:** | **181.8** | **162.9** | **151.5 ± 10.9** | **130.2** | **172.8** |
| Dead | | | | | |
| Red Spruce | 7.9 | 9.3 | 19.5 ± 4.8 | 10.0 | 29.0 |
| No. White Cedar | 5.4 | 4.4 | 8.0 ± 2.6 | 3.0 | 13.0 |
| Red Maple | 3.2 | 0.1 | 2.5 ± 1.2 | 0.1 | 4.9 |
| Yellow Birch | 1.4 | 0.7 | 0.5 ± 0.5 | 0 | 1.5 |
| White Pine | 0 | 0.4 | 0 | | |
| Pitch Pine | 0.7 | 0 | 0 | | |
| Balsam Fir | 1.8 | 1.9 | 0 | | |
| Red Pine | 0.4 | 0 | 0 | | |
| Bigtooth Aspen | 1.1 | 0 | 0 | | |
| Trembling Aspen | 0 | 1.1 | 0 | | |
| **Totals:** | **21.9** | **17.9** | **30.5 ± 5.3** | **20.2** | **40.8** |

Table 35. Mean diameter ($\bar{X}$ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC15 (Jerusalem Mountain) in 2016. Data missing for 1980/1992-1994. Confidence Intervals are for 2016 data.

| **AC15** | **1980** | **1993** | **2016** | | 95% CI | |
|---|---|---|---|---|---|---|
| Species | | | $\bar{X}$ Dia. | ± SE | Lower | Upper |
| White Pine | x | x | 19.8 | ± 0.6 | 18.6 | 21.0 |
| Red Pine | x | x | 17.1 | ± 0.4 | 16.2 | 18.0 |
| Red Spruce | x | x | 11.7 | ± 0.9 | 10.0 | 13.5 |
| Yellow Birch | x | x | 11.1 | ± 1.0 | 9.1 | 13.1 |
| Red Maple | x | x | 10.8 | ± 0.9 | 9.0 | 12.6 |
| No. White Cedar | x | x | 10.6 | ± 0.8 | 8.9 | 12.2 |
| Paper Birch | x | x | 9.5 | ± 0.2 | 9.2 | 9.8 |
| Striped Maple | x | x | 8.7 | ± 0 | 8.7 | 8.7 |
| Shadbush | x | x | 5.4 | ± 0 | 5.4 | 5.4 |
| **Overall Average** | | | **11.3** | **± 0.9** | **9.6** | **13.1** |

Western Mountain (lower) (AC19) – Mixed Hardwood-Conifer

Basal areas of red maple, northern white cedar, paper birch, white pine, and bigtooth aspen decreased, while those of red spruce and balsam fir increased between 1980 and 2016. Live basal area of red spruce increased by 69%. Total live basal area decreased between 1980 and 1993 from 158.5 to 155.5 ft$^2$/acre and shows a decreasing trend with 144.6 ft$^2$/acre in 2016 (Table 36). Total dead basal area has remained stable with 21.7 ft$^2$/acre in 1980 and 21.3 ft$^2$/acre in 2016 (Table 36). Density of live trees decreased from 659.4 in 1993 to 640.6 stems per acre in 2016 (Table 8). Density of dead trees increased from 76.7 in 1993 to 195.7 stems per acre in 2016 (Table 8). Density data for saplings is missing for the historic sampling periods (Table 9). Mean diameters (in.) increased from 10.9 in 1993 to 11.7 in 2016 (Table 37).

Biomass (T/acre) of live trees decreased from 75.8 in 1980 to 65.7 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 5.0 in 1980 to 9.5 in 2016 (Table 10). Biomass (T/acre) of shrubs increased from 0.1 live and less than 0.01 dead in 1980 to 2.9 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load increased from 6.6 T/acre in 1980 to 7.3 T/acre in 2016 (Table 11). Duff depth (in.) increased from 2.6 in 1980 to 5.5 in 2016 (Table 11). Fuel height (in.) decreased from 3.7 in 1980 to 2.5 in 2016 (Table 11).

Table 36. Average basal area (X̄ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC19 (Western Mountain (lower)) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 1993 and 2016 data.

| AC19-Western Mtn. (lower) | | | | | | | |
|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *20* | *20* | 95% CI | | *30* | 95% CI | |
| Live | X̄ BA | X̄ BA ± SE | Lower | Upper | X̄ BA ± SE | Lower | Upper |
| Red Spruce | 56.5 | 76.0 ± 6.8 | 62.7 | 89.3 | 95.7 ± 8.9 | 78.3 | 113.1 |
| Red Maple | 28.0 | 28.5 ± 4.9 | 18.9 | 38.1 | 19.0 ± 4.6 | 9.9 | 28.1 |
| No. White Cedar | 21.5 | 19.5 ± 8.9 | 2.1 | 36.9 | 11.7 ± 3.2 | 5.4 | 18.0 |
| Balsam Fir | 5.0 | 4.5 ± 1.9 | 0.9 | 8.1 | 7.3 ± 2.4 | 2.6 | 12.0 |
| Paper Birch | 13.0 | 7.0 ± 3.2 | 0.8 | 13.2 | 4.3 ± 1.8 | 0.7 | 7.9 |
| White Pine | 11.0 | 7.0 ± 2.5 | 2.0 | 12.0 | 3.0 ± 1.2 | 0.7 | 5.3 |
| Yellow Birch | 0.5 | 0 | | | 3.0 ± 1.3 | 0.4 | 5.6 |
| Bigtooth Aspen | 20.5 | 9.0 ± 3.4 | 2.3 | 15.7 | 0.3 ± 0.4 | 0 | 1.1 |
| Shadbush | 0.5 | 0.5 ± 0.5 | 0 | 1.5 | 0.3 ± 0.4 | 0 | 1.1 |
| Red Pine | 0 | 1.0 ± 1.0 | 0 | 3.0 | 0 | | |
| Sugar Maple | 0 | 1.5 ± 1.1 | 0 | 3.6 | 0 | | |
| Striped Maple | 1.0 | 0 | | | 0 | | |
| Red Oak | 1.0 | 1.0 ± 1.0 | 0 | 3.0 | 0 | | |
| **Totals:** | **158.5** | **155.5 ± 10.4** | **135.2** | **175.8** | **144.6 ± 8.7** | **127.6** | **161.6** |
| Dead | | | | | | | |
| Red Spruce | 3.3 | 1.5 ± 0.8 | 0 | 3.1 | 8.7 ± 3.1 | 2.7 | 14.7 |
| No. White Cedar | 2.5 | 3.5 ± 1.7 | 0.2 | 6.8 | 5.0 ± 2.7 | 0 | 10.3 |
| Paper Birch | 0.8 | 4.5 ± 1.9 | 0.9 | 8.1 | 2.7 ± 1.0 | 0.7 | 4.7 |
| Red Maple | 5.8 | 0 | | | 2.3 ± 1.1 | 0.1 | 4.5 |
| Balsam Fir | 4.2 | 2.0 ± 1.2 | 0 | 4.3 | 1.3 ± 0.8 | 0 | 2.8 |
| Yellow Birch | 0 | 0 | | | 1.0 ± 0.9 | 0 | 2.8 |
| Bigtooth Aspen | 2.5 | 2.5 ± 1.4 | 0 | 5.3 | 0.3 ± 0.4 | 0 | 1.1 |
| White Pine | 1.7 | 1.0 ± 0.7 | 0 | 2.4 | 0 | | |
| Striped Maple | 0.8 | 0 | | | 0 | | |
| **Totals:** | **21.7** | **15.0 ± 3.4** | **8.3** | **21.7** | **21.3 ± 4.4** | **12.7** | **29.9** |

Table 37. Mean diameter ($\bar{X}$ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC19 (Western Mountain (lower)) in 1992-1994 and 2016. Data missing for 1980.

| AC19 | 1980 | 1993 | | | 95% CI | | 2016 | | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | | $\bar{X}$ Dia. | ± | SE | Lower | Upper | $\bar{X}$ Dia. | ± | SE | Lower | Upper |
| White Pine | x | 18.6 | ± | 1.1 | 16.5 | 20.7 | 27.2 | ± | 0.8 | 25.6 | 28.8 |
| Bigtooth Aspen | x | 11.5 | ± | 0.7 | 10.0 | 13.0 | 15.5 | ± | 2.2 | 11.2 | 19.7 |
| Red Spruce | x | 12.3 | ± | 1.1 | 10.2 | 14.4 | 12.0 | ± | 1.0 | 10.0 | 14.0 |
| Red Maple | x | 8.5 | ± | 0.7 | 7.1 | 9.8 | 11.0 | ± | 0.7 | 9.7 | 12.3 |
| No. White Cedar | x | 9.4 | ± | 0.7 | 8.0 | 10.9 | 11.0 | ± | 0.5 | 10.0 | 12.0 |
| Paper Birch | x | 7.3 | ± | 0.5 | 6.4 | 8.2 | 10.4 | ± | 0.6 | 9.3 | 11.6 |
| Balsam Fir | x | 4.3 | ± | 0.4 | 3.6 | 5.1 | 7.3 | ± | 0.6 | 6.1 | 8.5 |
| Shadbush | x | 9.7 | ± | 0 | 9.7 | 9.7 | 4.1 | ± | 0 | 4.1 | 4.1 |
| Sugar Maple | x | 7.6 | ± | 0.6 | 6.4 | 8.8 | | | | | |
| Red Pine | x | 17.0 | ± | 0.5 | 16.1 | 17.9 | | | | | |
| Red Oak | x | 15.9 | ± | 0.1 | 15.7 | 16.1 | | | | | |
| **Overall Average** | | **10.9** | **±** | **1.1** | **8.7** | **13.0** | **11.7** | **±** | **1.0** | **9.7** | **13.6** |

*Mixed Hardwood-Conifer Summary*

Barring stand replacing disturbance and assuming a predictable successional trajectory I expect Day Mountain (AC23), Jerusalem Mountain (AC15), and Western Mountain (lower) (AC19) to remain as mature mixed hardwood-conifer. However, disturbance of some form is a part of all forested ecosystems on a time scale relevant to tree species ages. Day Mountain and Jerusalem Mountain arose from a fire that burned through the landscape around 1880 (Patterson et al. 1983), and we might expect that they are still in a recovery stage given the presence of pioneer species like bigtooth aspen, paper birch, and red pine. These mature stands are beginning to break up as evidenced in the increase in downed fuel load. Given the increase in red spruce at Day Mountain and Western Mountain (lower), these stands may transition to the spruce-fir community type, possibly in one to two decades or longer based on the stands data and abundance of conifers in the understory. As the hardwoods begin to die off, the suppressed spruce trees are released and grow into the canopy. Thus there may be a gradual transition from the initial mixture of red spruce, paper birch, and aspen seedlings, through birch-aspen stand, to spruce-hardwoods, and eventually a spruce stand (Patterson et al. 1983). It is likely that these stands will move towards conditions of the older sample stands in this study such as AC01 located on Eagle Lake (Northwest) or AC04 located near Stanley Brook. If a disturbance such as fire or substantial blowdown occurs these stands will likely regenerate as birch-aspen with a red spruce component, and eventually return to a mixed hardwood-conifer stand in the distant future.

At Long Pond (AC16) the data show this stand is transitioning to mature spruce. The data show an increase in red spruce basal area and decrease in hardwood species across the sampling periods, and an increase in red spruce abundance in the understory. This stand was categorized as aspen-birch in the 1980s but today it appears to be a maturing spruce-fir stand. Perhaps in time this stand will move towards conditions like the older spruce-fir stands in this study such as, AC01 located at Eagle Lake (Northwest) or AC17 located on Schoodic Peninsula described below. Fuel loading is high in all sample stands in the mixed hardwood-conifer community type. Fuel loading combined with the flammable nature of the species present could support a high-intensity fire under the right weather conditions. However, without a blowdown, there is a lack of fine fuels to support fire initiation – unless fire starts outside the stands and burns into them as occurred in 1947.

*Spruce-Fir* (n=9)

<u>Hodgdon Pond (AC20) – Spruce-Fir</u>

Basal areas of red spruce and white pine increased between 1980 and 2016. Live basal area of red spruce increased by 41%, and white pine increased by 349%. Total live basal area increased between 1980 and 1993 from 53.5 to 84.8 $ft^2$/acre and shows an increasing trend with 90.0 $ft^2$/acre in 2016 (Table 38). Total dead basal area shows an increasing trend with 6.3 $ft^2$/acre in 1980 and 12.0 $ft^2$/acre in 2016 (Table 38). Density of live trees decreased from 767.8 in 1993 to 342.0 stems per acre in 2016 (Table 8). Density of dead trees also increased from 72.3 in 1993 to 85.9 stems per acre in 2016 (Table 8). Density of live saplings decreased and measured at 422.7 in 1993 and 335.0 stems per acre in 2016 (Table 9). Density of dead saplings decreased from 63.6 in 1993 to 0 stems per acre in 2016 (Table 9). Mean diameters (in.) increased from 8.9 in 1980 to 10.7 in 2016 (Table 39).

Biomass (T/acre) of trees increased from 24.6 live and 2.3 dead in 1980 to 38.6 live and 4.8 dead in 2016 (Table 10). Biomass (T/acre) of shrubs decreased from 1.1 live and less than 0.01 dead in 1980 to 0.5 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load increased from 3.0 T/acre in 1980 to 2.4 T/acre in 2016 (Table 11). Duff depth (in.) increased from 3.2 in 1980 to 4.4 in 2016 (Table 11). Fuel height (in.) increased from 3.0 in 1980 to 4.2 in 2016 (Table 11).

Table 38. Average basal area (X̄ BA) of live and dead trees ($ft^2$/acre) ± standard error with 95% confidence intervals for stand AC20 (Hodgdon Pond) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016. Confidence Intervals are for 1993 and 2016 data.

| AC20-Hodgdon Pond | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | **1993** | | | | **2016** | | | |
| *No. Points Sampled:* | *23* | *24* | | 95% CI | | *20* | | 95% CI | |
| Live | X̄ BA | X̄ BA ± | SE | Lower | Upper | X̄ BA ± | SE | Lower | Upper |
| Red Spruce | 38.7 | 37.1 ± | 4.8 | 27.6 | 46.6 | 54.5 ± | 7.5 | 39.8 | 69.2 |
| White Pine | 3.9 | 9.2 ± | 2.4 | 4.5 | 13.9 | 17.5 ± | 3.2 | 11.3 | 23.7 |
| Red Maple | 7.9 | 8.8 ± | 3.9 | 1.1 | 16.5 | 7.0 ± | 3.5 | 0.2 | 13.8 |
| No. White Cedar | 1.0 | 11.7 ± | 5.9 | 0.2 | 23.2 | 4.5 ± | 3.5 | 0 | 11.4 |
| Balsam Fir | 0.5 | 13.8 ± | 3.3 | 7.2 | 20.4 | 3.0 ± | 1.3 | 0.5 | 5.5 |
| Pitch Pine | 0 | 0 | | | | 2.5 ± | 2.5 | 0 | 7.4 |
| Red Pine | 0 | 1.3 ± | 0.7 | 0 | 2.7 | 0.5 ± | 0.5 | 0 | 1.5 |
| Shadbush | 0 | 0 | | | | 0.5 ± | 0.5 | 0 | 1.5 |
| Paper Birch | 1.0 | 2.5 ± | 1.1 | 0.4 | 4.6 | 0 | | | |
| Bigtooth Aspen | 0.5 | 0.4 ± | 0.4 | 0 | 1.2 | 0 | | | |
| **Totals:** | **53.5** | **84.8 ±** | **12.1** | **61.0** | **108.6** | **90.0 ±** | **6.4** | **77.4** | **102.6** |
| Dead | | | | | | | | | |
| Red Spruce | 3.1 | 2.5 ± | 0.9 | 0.7 | 4.3 | 6.5 ± | 2.2 | 2.2 | 10.8 |
| Balsam Fir | 1.0 | 1.3 ± | 0.7 | 0 | 2.7 | 3.0 ± | 1.5 | 0.1 | 5.9 |
| White Pine | 0 | 0.4 ± | 0.4 | 0 | 1.2 | 1.0 ± | 0.7 | 0 | 2.4 |
| Red Pine | 0 | 0 | | | | 1.0 ± | 0.7 | 0 | 2.4 |
| Red Maple | 2.2 | 1.3 ± | 0.7 | 0 | 2.7 | 0.5 ± | 0.5 | 0 | 1.5 |
| No. White Cedar | 0 | 0.8 ± | 0.8 | 0 | 2.4 | 0 | | | |
| Paper Birch | 0 | 1.7 ± | 0.8 | 0.2 | 3.2 | 0 | | | |
| **Totals:** | **6.3** | **8.0 ±** | **1.5** | **5.1** | **10.9** | **12.0 ±** | **3.7** | **4.8** | **19.2** |

Table 39. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC20 (Hodgdon Pond) in 1992-1994 and 2016. Data missing for 1980.

| **AC20** | **1980** | **1993** | | 95% CI | | **2016** | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper |
| White Pine | x | 17.4 | ± 1.4 | 14.7 | 20.1 | 14.8 | ± 1.5 | 11.8 | 17.8 |
| Red Pine | x | 14.4 | ± 0.4 | 13.6 | 15.2 | 13.5 | ± 1.2 | 11.0 | 15.9 |
| No. White Cedar | x | 8.2 | ± 0.4 | 7.3 | 9.1 | 11.0 | ± 0.6 | 9.8 | 12.1 |
| Red Spruce | x | 9.2 | ± 1.1 | 7.1 | 11.3 | 10.4 | ± 1.1 | 8.3 | 12.5 |
| Red Maple | x | 7.0 | ± 0.4 | 6.1 | 7.8 | 8.7 | ± 0.5 | 7.8 | 9.6 |
| Pitch Pine | x | | | | | 7.7 | ± 1.0 | 5.8 | 9.7 |
| Balsam Fir | x | 4.4 | ± 0.4 | 3.7 | 5.1 | 5.2 | ± 0.3 | 4.5 | 5.8 |
| Shadbush | x | | | | | 5.0 | ± 0 | 5.0 | 5.0 |
| Paper Birch | x | 8.0 | ± 0.4 | 7.2 | 8.7 | | | | |
| Bigtooth Aspen | x | 13.1 | ± 0 | 13.1 | 13.1 | | | | |
| **Overall Average** | | **8.9** | **± 1.1** | **6.8** | **11.1** | **10.7** | **± 1.2** | **8.4** | **13.1** |

Western Head (IaH) (AC13) – Spruce-Fir

The Western Head stand continued to experience blowdown and over-mature break-up since the last sample period in 1993 to 2016. Most notably, there was a 73% significant reduction in red spruce live basal area, 77% decrease in live density, 82% decrease in dead density, and a 213% increase in 1000-hr sound size class with 3.9 T/acre in 1980 and 12.2 T/acre in 2016. Basal areas of red spruce, balsam fir, white spruce and paper birch decreased between 1980 and 2016. Total live basal area significantly decreased between 1980 and 2016 from 144.1 to 41.5 ft$^2$/acre (Table 40). Total dead basal area decreased by more than 50% with 45.4 ft$^2$/acre in 1980 and 20.0 ft$^2$/acre in 2016 (Table 40). Density of live trees significantly decreased from 520.4 in 1980 to 117.5 stems per acre in 2016 (Table 8). Density of dead trees also significantly decreased from 552.6 in 1980 to 101.7 stems per acre in 2016 (Table 8). Density of saplings was not measured prior to 2016, but in 2016 live stems per acre were 810.0 and 15.0

dead stems per acre (Table 9). Mean diameters (in.) increased from 7.7 in 1980 to 9.5 in 2016 (Table 41).

Mature trees continue to be lost to blowdown as evidenced in the increase in heavy downed-woody fuel load. Biomass (T/acre) of trees decreased from 66.5 live and 17.1 dead in 1980 to 18.4 live and 8.3 dead in 2016 (Table 10). Biomass (T/acre) of shrubs increased from 0.1 live and no measurable dead in 1980 to 0.6 live and less than 0.02 dead in 2016 (Table 10). Total downed woody fuel load increased from 11.6 T/acre in 1980 to 28.0 T/acre in 2016 (Table 11). Notably, the 1000-hr sound size class increased from 3.9 T/acre in 1980 to 12.2 T/acre in 2016 (Table 11). Duff depth (in.) decreased from 6.2 in 1980 to 4.4 in 2016 (Table 11). Fuel height (in.) increased from 4.0 in 1980 to 10.0 in 2016 (Table 11).

Table 40. Average basal area (X̄ BA) of live and dead trees ($ft^2$/acre) ± standard error with 95% confidence intervals for stand AC13 (Western Head) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC13-Western Head (IaH) | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | | **1993** | | | | **2016** | | | |
| *No. Points Sampled:* | *26* | | 95% CI | | *21* | | 95% CI | | *20* | | 95% CI | |
| Live | X̄ BA | ± SE | Lower | Upper | X̄ BA | ± SE | Lower | Upper | X̄ BA | ± SE | Lower | Upper |
| Red Spruce | 124.1 | ± 14.0 | 96.6 | 151.6 | 106.7 | ± 12.8 | 81.7 | 131.7 | 34.0 | ± 9.2 | 16.0 | 52.0 |
| Balsam Fir | 8.6 | ± 2.6 | 3.5 | 13.7 | 5.7 | ± 2.4 | 1.1 | 10.3 | 4.5 | ± 2.2 | 0.1 | 8.9 |
| White Spruce | 8.2 | ± 2.5 | 3.3 | 13.1 | 2.9 | ± 1.7 | 0 | 6.2 | 1.5 | ± 1.1 | 0 | 3.6 |
| Paper Birch | 2.7 | ± 1.0 | 0.8 | 4.6 | 1.0 | ± 0.7 | 0 | 2.3 | 1.0 | ± 0.7 | 0 | 2.4 |
| Red Maple | 0 | | | | 0 | | | | 0.5 | ± 0.5 | 0 | 1.5 |
| Pitch Pine | 0 | | | | 0 | | | | 0 | | | |
| Shadbush | 0 | | | | 0.5 | ± 0.5 | 0 | 1.4 | 0 | | | |
| Mountain Ash | 0.5 | ± 0.4 | 0 | 1.4 | 0 | | | | 0 | | | |
| **Totals:** | **144.1** | **± 12.8** | **119.0** | **169.2** | **116.8** | **± 12.1** | **93.1** | **140.5** | **41.5** | **± 8.8** | **24.2** | **58.8** |
| Dead | | | | | | | | | | | | |
| Red Spruce | 24.5 | ± 5.4 | 13.9 | 35.1 | 20.0 | ± 5.2 | 9.9 | 30.1 | 16.0 | ± 4.4 | 7.3 | 24.7 |
| Paper Birch | 0.5 | ± 0.4 | 0 | 1.4 | 1.0 | ± 0.7 | 0 | 2.3 | 2.0 | ± 1.6 | 0 | 5.1 |
| Balsam Fir | 14.5 | ± 4.9 | 5.0 | 24.0 | 10.5 | ± 4.1 | 2.5 | 18.5 | 1.5 | ± 0.8 | 0 | 3.1 |
| White Spruce | 5.9 | ± 3.8 | 0 | 13.4 | 0 | | | | 0.5 | ± 0.5 | 0 | 1.5 |
| Pitch Pine | 0 | | | | 0 | | | | 0 | | | |
| **Totals:** | **45.4** | **± 9.3** | **27.1** | **63.7** | **31.5** | **± 5.3** | **21.2** | **41.8** | **20.0** | **± 4.4** | **11.5** | **28.5** |

Table 41. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC13 (Western Head) in 1980, 1992-1994 and 2016.

| **AC13** | **1980** | | 95% CI | | **1993** | | 95% CI | | **2016** | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper |
| Red Maple | | | | | | | | | 12.8 | ± 0 | 12.8 | 12.8 |
| Red Spruce | 8.2 | ± 0.7 | 6.8 | 9.5 | 8.2 | ± 0.7 | 6.7 | 9.6 | 10.0 | ± 0.8 | 8.4 | 11.6 |
| Paper Birch | 5.0 | ± 0.4 | 4.2 | 5.9 | 6.6 | ± 1.0 | 4.7 | 8.5 | 7.7 | ± 0.2 | 7.3 | 8.2 |
| Balsam Fir | 5.0 | ± 0.5 | 4.0 | 5.9 | 5.6 | ± 0.4 | 4.8 | 6.5 | 7.5 | ± 0.3 | 7.0 | 8.1 |
| White Spruce | 7.8 | ± 0.9 | 6.0 | 9.6 | 8.5 | ± 0.7 | 7.1 | 9.8 | 5.9 | ± 0.8 | 4.4 | 7.4 |
| Shadbush | | | | | 5.8 | ± 0 | 5.8 | 5.8 | | | | |
| Mountain Ash | 14.0 | ± 0 | 14.0 | 14.0 | | | | | | | | |
| **Overall Average** | **7.7** | **± 0.7** | **6.3** | **9.1** | **7.9** | **± 0.7** | **6.4** | **9.3** | **9.5** | **± 0.8** | **7.9** | **11.1** |

Western Mountain (upper) (AC18) – Spruce-Fir

During 1980 to 2016 in Western Mountain (upper) stand, basal area of red spruce significantly decreased by 27%, while northern white cedar increased by 221% between 1980 and 2016. Total live basal area decreased between 1980 and 1993 from 190.5 to 174.8 $ft^2$/acre, and overall shows a significant decreasing trend with 143.5 $ft^2$/acre in 2016 (Table 42). Total dead basal area has remained stable with 19.1 $ft^2$/acre in 1980 and 19.0 $ft^2$/acre in 2016 (Table 42). Density of live trees significantly decreased from 511.5 in 1980 to 288.0 stems per acre in 2016 (Table 8). Density of dead trees also decreased from 210.1 in 1980 to 79.9 stems per acre in 2016 (Table 8). Mean diameters (in.) increased overall from 10.5 in 1980 to 12.2 in 2016 (Table 43).

Biomass (T/acre) of live trees decreased from 90.7 in 1980 to 59.9 in 2016 (Table 10). Biomass (T/acre) of dead trees also decreased from 8.2 in 1980 to 7.7 in 2016 (Table 10). Biomass (T/acre) of shrubs increased from 0.2 in 1980 to 16.9 in 2016 (Table 10). Mature trees continue to be lost to blowdown as evidenced in the increase in heavy downed woody fuel load. Total downed woody fuel load increased from 6.6 T/acre in 1980 to 12.9 T/acre in 2016 (Table 11). Duff depth (in.) increased from 5.1 in 1980 to 6.5 in 2016 (Table 11). Fuel height (in.) increased from 2.4 in 1980 to 3.5 in 2016 (Table 11).

Table 42. Average basal area (X̄ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC18 (Western Mountain (upper)) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC18-Western Mountain (upper) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *23* | 95% CI | | *36* | 95% CI | | *20* | 95% CI | |
| Live | X̄ BA ± SE | Lower | Upper | X̄ BA ± SE | Lower | Upper | X̄ BA ± SE | Lower | Upper |
| Red Spruce | 160.9 ± 10.7 | 139.9 | 181.9 | 133.9 ± 9.1 | 116.1 | 151.7 | 118.0 ± 8.6 | 101.2 | 134.8 |
| No. White Cedar | 3.9 ± 1.9 | 0.3 | 7.5 | 21.4 ± 4.6 | 12.4 | 30.4 | 12.5 ± 6.1 | 0.6 | 24.4 |
| Red Maple | 7.4 ± 2.4 | 2.7 | 12.1 | 5.6 ± 1.2 | 3.3 | 7.9 | 6.5 ± 2.2 | 2.2 | 10.8 |
| White Pine | 2.6 ± 1.1 | 0.4 | 4.8 | 5.6 ± 1.6 | 2.5 | 8.7 | 3.5 ± 1.1 | 1.4 | 5.6 |
| Yellow Birch | 0.9 ± 0.9 | 0 | 2.6 | 0 | | | 1.5 ± 0.8 | 0 | 3.1 |
| Paper Birch | 6.1 ± 2.0 | 2.3 | 9.9 | 2.2 ± 0.9 | 0.4 | 4.0 | 1.0 ± 1.0 | 0 | 3.0 |
| Balsam Fir | 5.7 ± 2.8 | 0.2 | 11.2 | 3.3 ± 1.3 | 0.7 | 5.9 | 0.5 ± 0.5 | 0 | 1.5 |
| Red Pine | 0 | | | 0.6 ± 0.4 | 0 | 1.4 | 0 | | |
| Bigtooth Aspen | 1.7 ± 1.0 | 0 | 3.7 | 2.2 ± 0.9 | 0.4 | 4.0 | 0 | | |
| Shadbush | 1.3 ± 0.7 | 0 | 2.7 | 0 | | | 0 | | |
| **Totals:** | **190.5 ± 9.9** | **171.0** | **210.0** | **174.8 ± 6.5** | **162.1** | **187.5** | **143.5 ± 9.3** | **125.3** | **161.7** |
| Dead | | | | | | | | | |
| Red Spruce | 8.7 ± 2.2 | 4.4 | 13.0 | 6.1 ± 2.1 | 2.0 | 10.2 | 11.0 ± 2.4 | 6.3 | 15.7 |
| No. White Cedar | 1.3 ± 0.7 | 0 | 2.7 | 1.4 ± 1.0 | 0 | 3.3 | 5.5 ± 2.0 | 1.6 | 9.4 |
| White Pine | 0 | | | 1.9 ± 1.2 | 0 | 4.2 | 1.0 ± 0.7 | 0 | 2.4 |
| Paper Birch | 2.2 ± 0.9 | 0.5 | 3.9 | 1.9 ± 0.8 | 0.4 | 3.4 | 0.5 ± 0.5 | 0 | 1.5 |
| Balsam Fir | 4.8 ± 2.3 | 0.2 | 9.4 | 0.3 ± 0.3 | 0 | 0.9 | 0.5 ± 0.5 | 0 | 1.5 |
| Yellow Birch | 0 | | | 0 | | | 0.5 ± 0.5 | 0 | 1.5 |
| Red Maple | 0.9 ± 0.6 | 0 | 2.1 | 0 | | | 0 | | |
| Shadbush | 0.4 ± 0.4 | 0 | 1.3 | 0 | | | 0 | | |
| Unknown | 0.4 ± 0.4 | 0 | 1.3 | 0 | | | 0 | | |
| Bigtooth Aspen | 0.4 ± 0.4 | 0 | 1.3 | 1.7 ± 0.8 | 0.2 | 3.2 | 0 | | |
| **Totals:** | **19.1 ± 2.9** | **13.3** | **24.9** | **13.3 ± 2.7** | **8.0** | **18.6** | **19.0 ± 3.2** | **12.8** | **25.2** |

Table 43. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC18 (Western Mountain (upper)) in 1980, 1992-1994 and 2016.

| AC18 | 1980 | | 95% CI | | 1993 | | 95% CI | | 2016 | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. ± | SE | Lower | Upper | X̄ Dia. ± | SE | Lower | Upper | X̄ Dia. ± | SE | Lower | Upper |
| White Pine | 14.2 ± | 0.5 | 13.2 | 15.1 | 17.1 ± | 0.8 | 15.5 | 18.7 | 20.6 ± | 0.8 | 19.0 | 22.2 |
| Red Spruce | 11.3 ± | 0.9 | 9.6 | 13.0 | 11.0 ± | 0.7 | 9.6 | 12.4 | 12.3 ± | 0.9 | 10.6 | 14.0 |
| Red Maple | 9.5 ± | 0.7 | 8.2 | 10.8 | 8.7 ± | 0.5 | 7.7 | 9.7 | 10.9 ± | 0.6 | 9.7 | 12.2 |
| No. White Cedar | 9.1 ± | 0.6 | 8.0 | 10.2 | 8.9 ± | 0.5 | 7.9 | 9.9 | 10.4 ± | 0.8 | 8.7 | 12.0 |
| Paper Birch | 5.7 ± | 0.3 | 5.1 | 6.2 | 6.1 ± | 0.3 | 5.6 | 6.6 | 10.4 ± | 0.7 | 9.0 | 11.7 |
| Balsam Fir | 4.3 ± | 0.3 | 3.6 | 5.0 | 5.3 ± | 0.3 | 4.8 | 5.8 | 10.1 ± | 1.3 | 7.6 | 12.5 |
| Yellow Birch | 8.9 ± | 0.2 | 8.4 | 9.4 | | | | | 9.7 ± | 0.3 | 9.0 | 10.4 |
| Shadbush | 4.6 ± | 0.3 | 4.0 | 5.2 | | | | | | | | |
| Red Pine | | | | | 13.0 ± | 0.4 | 12.2 | 13.8 | | | | |
| Bigtooth Aspen | 11.0 ± | 0.3 | 10.4 | 11.6 | 11.8 ± | 0.5 | 10.9 | 12.7 | | | | |
| **Overall Average** | **10.5 ±** | **0.9** | **8.7** | **12.3** | **10.7 ±** | **0.7** | **9.3** | **12.2** | **12.2 ±** | **0.9** | **10.4** | **14.0** |

Stanley Brook (AC04) – Spruce-Fir

Basal areas of red spruce, red maple, sugar maple, striped maple, northern white cedar, and paper birch decreased, while that of white pine and balsam fir increased between 1980 and 2016. Most notably, live basal area of red maple decreased by 24%, northern white cedar decreased by 94%, and paper birch significantly decreased by 100% (i.e., was absent in 2016). Total live basal area increased between 1980 and 1993 from 178.6 to 201.0 ft$^2$/acre, but shows a decreasing trend with 159.1 ft$^2$/acre in 2016 (Table 44). Total dead basal area has remained stable with 15.5 ft$^2$/acre in 1980 and 15.3 ft$^2$/acre in 2016 (Table 44). Density of live trees increased from 706.8 in 1980 to 862.3 stems per acre in 2016 (Table 8). Density of dead trees also increased from 34.0 in 1980 to 65.9 stems per acre in 2016 (Table 8). Density of live saplings decreased from 568.2 in 1993 to 381.0 stems per acre in 2016 (Table 9). Density of dead saplings increased from 309.1 in 1993 to 333.3 stems per acre in 2016 (Table 9). Mean diameters (in.) increased from 12.4 in 1980 to 13.0 in 2016 (Table 45).

Mature trees continue to be lost to blowdown as evidenced in the increase in the 1,000-hr downed-woody fuel load. Biomass (T/acre) of trees decreased from 88.3 live and 8.1 dead in 1980 to 78.1 live and 7.9 dead in 2016 (Table 10). There was a notable decrease in the biomass (T/acre) of shrubs from 8.9 live and less than 0.04 dead in 1980 to 0.6 live and no measurable dead in 2016 (Table 10). Total downed woody fuel load increased from 7.0 T/acre in 1980 to 8.5 T/acre in 2016 (Table 11). Duff depth (in.) decreased overall from 5.7 in 1980 to 4.5 in 2016 (Table 11). Fuel height (in.) decreased from 3.6 in 1980 to 1.1 in 2016 (Table 11).

Table 44. Average basal area ($\bar{X}$ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC04 (Stanley Brook) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC04-Stanley Brook | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *22* | 95% CI | | *22* | 95% CI | | *21* | 95% CI | |
| Live | $\bar{X}$ BA ± SE | Lower | Upper | $\bar{X}$ BA ± SE | Lower | Upper | $\bar{X}$ BA ± SE | Lower | Upper |
| Red Spruce | 115.9 ± 11.5 | 93.3 | 138.5 | 136.8 ± 7.3 | 122.6 | 151.0 | 114.8 ± 7.3 | 100.5 | 129.1 |
| Red Maple | 18.2 ± 4.8 | 8.8 | 27.6 | 17.3 ± 4.0 | 9.5 | 25.1 | 13.8 ± 2.7 | 8.5 | 19.1 |
| White Pine | 10.0 ± 3.0 | 4.1 | 15.9 | 11.8 ± 3.7 | 4.5 | 19.1 | 11.0 ± 3.1 | 5.0 | 17.0 |
| Balsam Fir | 5.0 ± 1.9 | 1.2 | 8.8 | 9.1 ± 3.7 | 1.9 | 16.3 | 7.6 ± 2.7 | 2.3 | 12.9 |
| Yellow Birch | 11.8 ± 4.7 | 2.6 | 21.0 | 15.9 ± 4.4 | 7.3 | 24.5 | 7.1 ± 3.3 | 0.6 | 13.6 |
| Sugar Maple | 3.6 ± 2.1 | 0 | 7.8 | 2.3 ± 1.5 | 0 | 5.2 | 3.3 ± 2.0 | 0 | 7.3 |
| Striped Maple | 2.3 ± 1.3 | 0 | 4.8 | 0.5 ± 0.4 | 0 | 1.4 | 1.0 ± 1.0 | 0 | 2.9 |
| No. White Cedar | 8.2 ± 4.2 | 0 | 16.5 | 2.7 ± 1.5 | 0 | 5.6 | 0.5 ± 0.5 | 0 | 1.5 |
| Hemlock | 0 | | | 0.5 ± 0.4 | 0 | 1.4 | 0 | | |
| Paper Birch | 3.6 ± 1.7 | 0.3 | 6.9 | 4.1 ± 1.8 | 0.5 | 7.7 | 0 | | |
| **Totals:** | **178.6 ± 9.6** | **159.8** | **197.4** | **201.0 ± 6.7** | **188.0** | **214.0** | **159.1 ± 5.3** | **148.6** | **169.6** |
| Dead | | | | | | | | | |
| Red Spruce | 9.5 ± 3.0 | 3.5 | 15.5 | 5.0 ± 1.8 | 1.4 | 8.6 | 7.1 ± 2.4 | 2.5 | 11.7 |
| Yellow Birch | 2.3 ± 1.1 | 0.1 | 4.5 | 1.4 ± 1.0 | 0 | 3.4 | 2.4 ± 1.4 | 0 | 5.1 |
| Balsam Fir | 0 | | | 1.4 ± 0.7 | 0 | 2.9 | 1.9 ± 1.1 | 0 | 4.1 |
| Red Maple | 0.9 ± 0.6 | 0 | 2.1 | 4.5 ± 1.9 | 0.7 | 8.3 | 1.9 ± 0.9 | 0.1 | 3.7 |
| Paper Birch | 0.9 ± 0.6 | 0 | 2.1 | 1.4 ± 1.4 | 0 | 4.1 | 1.0 ± 0.7 | 0 | 2.3 |
| White Pine | 0.5 ± 0.4 | 0 | 1.4 | 0 | | | 0.5 ± 0.5 | 0 | 1.5 |
| Sugar Maple | 0 | | | 0 | | | 0.5 ± 0.5 | 0 | 1.5 |
| No. White Cedar | 1.4 ± 1.0 | 0 | 3.4 | 0 | | | 0 | | |
| **Totals:** | **15.5 ± 3.6** | **8.5** | **22.5** | **13.7 ± 3.4** | **7.1** | **20.3** | **15.3 ± 3.5** | **8.4** | **22.2** |

Table 45. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC04 (Stanley Brook) in 1980, 1992-1994 and 2016.

| AC04 | 1980 | 95% CI | | 1993 | 95% CI | | 2016 | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper |
| White Pine | 21.0 ± 1.0 | 19.1 | 22.9 | 21.5 ± 0.6 | 20.2 | 22.7 | 19.9 ± 1.2 | 17.5 | 22.2 |
| Red Maple | 9.9 ± 0.8 | 8.4 | 11.4 | 11.2 ± 1.0 | 9.2 | 13.2 | 13.3 ± 1.0 | 11.3 | 15.3 |
| Red Spruce | 12.8 ± 1.1 | 10.7 | 14.9 | 13.2 ± 1.0 | 11.2 | 15.2 | 13.2 ± 1.2 | 10.7 | 15.6 |
| No. White Cedar | 10.3 ± 0.9 | 8.5 | 12.1 | 11.0 ± 0.5 | 10.0 | 12.0 | 13.1 ± 0.0 | 13.1 | 13.1 |
| Sugar Maple | 16.2 ± 0.8 | 14.6 | 17.8 | 11.8 ± 1.1 | 9.7 | 13.9 | 13.0 ± 1.2 | 10.7 | 15.3 |
| Yellow Birch | 11.7 ± 0.8 | 10.0 | 13.3 | 12.0 ± 0.9 | 10.3 | 13.7 | 12.6 ± 0.8 | 11.0 | 14.2 |
| Paper Birch | 10.6 ± 0.8 | 8.9 | 12.2 | 9.9 ± 0.5 | 9.0 | 10.8 | 10.9 ± 0.8 | 9.4 | 12.3 |
| Balsam Fir | 3.9 ± 0.3 | 3.3 | 4.5 | 3.4 ± 0.3 | 2.9 | 4.0 | 4.8 ± 0.5 | 3.9 | 5.7 |
| Striped Maple | 2.7 ± 0.5 | 1.8 | 3.6 | 1.8 ± 0 | 1.8 | 1.8 | 2.6 ± 0.1 | 2.5 | 2.7 |
| Hemlock | | | | 21.0 ± 0 | 21.0 | 21.0 | | | |
| **Overall Average** | **12.4 ± 1.1** | **10.2** | **14.7** | **12.7 ± 1.1** | **10.5** | **15.0** | **13.0 ± 1.3** | **10.5** | **15.6** |

Schoodic Peninsula (West) (AC17) – Spruce-Fir

Basal areas of red spruce significantly decreased, while that of balsam fir increased between 1980 and 2016. Live basal area of paper birch increased by 293% between 1980 and 1993, and then decreased by 61% between 1993 and 2016. Total live basal area decreased between 1980 and 1993 from 222.9 to 193.3 $ft^2$/acre and shows a decreasing trend with 170.0 $ft^2$/acre in 2016 (Table 46). Even with this decrease in basal area it still remains among one of the highest of all sample stands. Total dead basal area shows an increasing trend with 24.3 $ft^2$/acre in 1980 and 32.5 $ft^2$/acre in 2016 (Table 46). Red spruce is the main contributor to the increase in dead basal area. Density of live trees increased from 241.4 in 1980 to 847.4 stems per acre in 2016 (Table 8). Density of dead trees also increased from 54.7 in 1980 to 112.8 stems per acre in 2016 (Table 8). Density of live saplings increased and measured at 204.5 in 1993 and 820.0 stems per acre in 2016 (Table 9). Density of dead saplings increased from 90.0

in 1993 to 110.0 stems per acre in 2016 (Table 9). Mean diameters (in.) increased overall from 10.0 in 1980 to 11.0 in 2016 (Table 47).

Biomass (T/acre) of live trees decreased from 96.9 in 1980 to 76.0 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 10.6 in 1980 to 15.4 in 2016. Biomass (T/acre) of shrubs increased from 0.3 live and less than 0.01 dead in 1980 to 13.0 live and less than 0.02 dead in 2016 (Table 10). Mature trees continue to be lost to blowdown as evidenced in the increase in heavy downed-woody fuel load. Total downed woody fuel load increased from 9.1 T/acre in 1980 to 19.9 T/acre in 2016 (Table 11). Duff depth (in.) increased from 4.2 in 1980 to 7.3 in 2016 (Table 11). Fuel height (in.) decreased from 5.9 in 1980 to 1.7 in 2016 (Table 11).

Table 46. Average basal area (X̄ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC17 (Schoodic Peninsula (West)) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC17-Schoodic Peninsula (West) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *22* | 95% CI | | *22* | 95% CI | | *20* | 95% CI | |
| Live | X̄ BA ± SE | Lower | Upper | X̄ BA ± SE | Lower | Upper | X̄ BA ± SE | Lower | Upper |
| Red Spruce | 220.0 ± 18.5 | 183.7 | 256.3 | 163.6 ± 12.6 | 138.9 | 188.3 | 155.0 ± 13.3 | 129.0 | 181.0 |
| Balsam Fir | 0 | | | 16.4 ± 6.7 | 3.2 | 29.6 | 10.0 ± 3.5 | 3.2 | 16.8 |
| Paper Birch | 2.9 ± 2.9 | 0 | 8.5 | 11.4 ± 4.7 | 2.2 | 20.6 | 4.5 ± 2.0 | 0.6 | 8.4 |
| Yellow Birch | 0 | | | 0.5 ± 0.4 | 0 | 1.4 | 0.5 ± 0.5 | 0 | 1.5 |
| No. White Cedar | 0 | | | 0.9 ± 0.9 | 0 | 2.7 | 0 | | |
| White Spruce | 0 | | | 0.5 ± 0.4 | 0 | 1.4 | 0 | | |
| Unknown | 0 | | | | | | 0 | | |
| **Totals:** | **222.9 ± 17.4** | **188.7** | **257.1** | **193.3 ± 11.0** | **171.7** | **214.9** | **170.0 ± 10.9** | **148.7** | **191.3** |
| Dead | | | | | | | | | |
| Red Spruce | 18.6 ± 6.7 | 5.5 | 31.7 | 17.7 ± 3.9 | 10.0 | 25.4 | 26.0 ± 4.1 | 18.0 | 34.0 |
| Balsam Fir | 4.3 ± 6.8 | 0 | 17.6 | 5.5 ± 2.5 | 0.6 | 10.4 | 3.5 ± 1.7 | 0.2 | 6.8 |
| Paper Birch | 1.4 ± 1.4 | 0 | 4.2 | 3.2 ± 2.8 | 0 | 8.6 | 2.5 ± 1.2 | 0.1 | 4.9 |
| Yellow Birch | 0 | | | 0 | | | 0.5 ± 0.5 | 0 | 1.5 |
| No. White Cedar | 0 | | | 0.9 ± 0.9 | 0 | 2.7 | 0 | | |
| **Totals:** | **24.3 ± 7.2** | **10.2** | **38.4** | **27.3 ± 5.1** | **17.4** | **37.2** | **32.5 ± 4.7** | **23.2** | **41.8** |

Table 47. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC17 (Schoodic Peninsula (West)) in 1980, 1992-1994 and 2016.

| AC17 | 1980 | | 95% CI | | 1993 | | 95% CI | | 2016 | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper |
| Yellow Birch | | | | | 7.1 | ± 0 | 7.1 | 7.1 | 14.0 | ± 0.6 | 12.8 | 15.2 |
| Red Spruce | 10.1 | ± 1.5 | 7.2 | 13.1 | 10.4 | ± 0.9 | 8.6 | 12.2 | 11.4 | ± 1.0 | 9.4 | 13.3 |
| Paper Birch | 9.3 | ± 1.4 | 6.5 | 12.1 | 9.1 | ± 0.8 | 7.5 | 10.6 | 9.3 | ± 0.7 | 7.8 | 10.7 |
| Balsam Fir | 4.8 | ± 0.9 | 3.1 | 6.6 | 4.8 | ± 0.6 | 3.6 | 6.0 | 6.1 | ± 0.7 | 4.7 | 7.5 |
| White Spruce | | | | | 16.4 | ± 0 | 16.4 | 16.4 | | | | |
| No. White Cedar | | | | | 9.7 | ± 0.7 | 8.3 | 11.1 | | | | |
| **Overall Average** | **10.0** | **± 1.5** | **7.1** | **13.0** | **9.8** | **± 1.0** | **7.9** | **11.6** | **11.0** | **± 1.0** | **9.0** | **12.9** |

Deep Cove (IaH) (AC12) – Spruce-Fir

Live basal area of red spruce decreased by 26%. Total live basal area significantly decreased between 1980 and 2016 from 137.0 to 101.0 $ft^2$/acre (Table 48). Total dead basal area also shows a decreasing trend with 28.0 $ft^2$/acre in 1980 and 14.0 $ft^2$/acre in 2016 (Table 48). Density of live trees significantly decreased from 705.1 in 1980 to 321.5 stems per acre in 2016 (Table 8). Density of dead trees also significantly decreased from 258.7 in 1980 to 57.3 stems per acre in 2016 (Table 8). Density of live saplings increased from 70.0 in 1993 to 430.0 stems per acre in 2016 (Table 9). Density of dead saplings also decreased from 10.0 in 1993 to 25.0 stems per acre in 2016 (Table 9). Mean diameters (in.) increased from 9.0 in 1980 to 11.0 in 2016 (Table 49).

Biomass (T/acre) of trees decreased from 60.9 live and 12.1 dead in 1980 to 45.7 live and 6.1 dead in 2016 (Table 10). Biomass (T/acre) of shrubs increased from 0.2 live and less than 0.03 dead in 1980 to 2.1 live and 0.1 dead in 2016 (Table 10). Total downed woody fuel load increased from 12.5 T/acre in 1980 to 12.8 T/acre in 2016 (Table 11). Duff depth (in.)

decreased from 5.2 in 1980 to 7.7 in 2016 (Table 11). Fuel height (in.) decreased from 6.5 in 1980 to 4.3 in 2016 (Table 11).

Table 48. Average basal area (X̄ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC12 (Deep Cove) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC12 - Deep Cove (IaH) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *20* | 95% CI | | *20* | 95% CI | | *20* | 95% CI | |
| Live | X̄ BA ± SE | Lower | Upper | X̄ BA ± SE | Lower | Upper | X̄ BA ± SE | Lower | Upper |
| Red Spruce | 132.0 ± 7.7 | 116.8 | 147.2 | 118.0 ± 6.7 | 104.9 | 131.1 | 97.5 ± 8.9 | 80.1 | 114.9 |
| Red Maple | 1.0 ± 1.0 | 0 | 3.0 | 2.0 ± 1.6 | 0 | 5.1 | 2.5 ± 1.8 | 0 | 6.0 |
| Balsam Fir | 2.0 ± 1.2 | 0 | 4.3 | 0.5 ± 0.5 | 0 | 1.5 | 1.0 ± 0.7 | 0 | 2.4 |
| White Pine | 0 | | | 0.5 ± 0.5 | 0 | 1.5 | 0 | | |
| Paper Birch | 0 | | | 0.5 ± 0.5 | 0 | 1.5 | 0 | | |
| Bigtooth Aspen | 0.5 ± 0.5 | 0 | 1.5 | 0 | | | 0 | | |
| Shadbush | 1.5 ± 1.1 | 0 | 3.6 | 0.5 ± 0.5 | 0 | 1.5 | 0 | | |
| **Totals:** | **137.0 ± 7.3** | **122.8** | **151.2** | **122.0 ± 7.0** | **108.2** | **135.8** | **101.0 ± 8.8** | **83.8** | **118.2** |
| Dead | | | | | | | | | |
| Red Spruce | 20.5 ± 3.9 | 12.8 | 28.2 | 16.0 ± 3.7 | 8.8 | 23.2 | 14.0 ± 3.1 | 7.9 | 20.1 |
| Paper Birch | 0 | | | 1.0 ± 0.7 | 0 | 2.4 | 0 | | |
| Balsam Fir | 7.0 ± 3.2 | 0.6 | 13.4 | 2.0 ± 0.9 | 0.2 | 3.8 | 0 | | |
| Red Maple | 0.5 ± 0.5 | 0 | 1.5 | 1.0 ± 0.7 | 0 | 2.4 | 0 | | |
| **Totals:** | **28.0 ± 4.4** | **19.3** | **36.7** | **20.0 ± 3.8** | **12.5** | **27.5** | **14.0 ± 3.1** | **7.9** | **20.1** |

Table 49. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC12 (Deep Cove) in 1980, 1992-1994 and 2016.

| **AC12** | **1980** | 95% CI | | **1993** | 95% CI | | **2016** | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper |
| Red Maple | 15.5 ± 0.3 | 14.8 | 16.2 | 13.0 ± 1.0 | 11.0 | 14.9 | 16.3 ± 0.8 | 14.8 | 17.8 |
| Red Spruce | 9.1 ± 0.9 | 7.4 | 10.9 | 10.0 ± 0.8 | 8.4 | 11.6 | 10.9 ± 0.9 | 9.1 | 12.7 |
| Balsam Fir | 6.9 ± 1.0 | 5.0 | 8.8 | 5.6 ± 0.3 | 5.0 | 6.2 | 5.9 ± 0.3 | 5.4 | 6.4 |
| Shadbush | 6.7 ± 0.3 | 6.2 | 7.3 | 4.5 ± 0 | 4.5 | 4.5 | | | |
| Paper Birch | | | | 13.6 ± 1.2 | 11.2 | 15.9 | | | |
| White Pine | | | | 16.7 ± 0 | 16.7 | 16.7 | | | |
| Bigtooth Aspen | 6.1 ± 0 | 6.1 | 6.1 | | | | | | |
| **Overall Average** | **9.0 ± 0.9** | **7.3** | **10.8** | **10.1 ± 0.8** | **8.4** | **11.7** | **11.0 ± 0.9** | **9.2** | **12.8** |

Otter Point (AC22) – Spruce-Fir

Significant changes were observed at Otter Point over the sample period, as would be expected of a stand that had broken up during the period 1950 to 1980 and is now entering the stabilization phase. Basal area of red spruce significantly increased between 1980 and 2016 by 346% from 19.3 to 86.0 $ft^2$/acre. Balsam fir basal area significantly increased from 14.0 $ft^2$/acre in 1980 to 38.5 $ft^2$/acre in 1993 and has since significantly declined to 5.5 $ft^2$/acre. Total live basal area significantly increased between 1980 and 1993 from 38.7 to 108.6 $ft^2$/acre, and now shows a decreasing trend with 94.5 $ft^2$/acre in 2016 (Table 50). Total dead basal area decreased by 56% (Table 50). Density of live trees significantly increased from 261.3 in 1980 to 1086.7 stems per acre in 2016 (Table 8). Density of dead trees decreased from 114.7 in 1980 to 187.8 stems per acre in 2016 (Table 8). Density of live saplings increased from 419.2 in 1993 to 705.0 stems per acre in 2016 (Table 9). Density of dead saplings increased from 396.2 in 1993 and 425.0 stems per acre in 2016 (Table 9). Mean diameters (in.) decreased from 9.1 in 1980 to 7.1 in 2016 which is further evidence of the regrowth in this stand (Table 51).

Biomass (T/acre) of live trees doubled during 1980 to 2016 (Table 10). Biomass (T/acre) of dead trees decreased from 9.5 in 1980 to 3.6 in 2016 (Table 10). Biomass (T/acre) of shrubs increased from less than 0.02 live and 0.1 dead in 1980 to 0.8 live and 0.2 dead in 2016 (Table 10). Notably, total downed woody fuel load decreased from 64.4 T/acre in 1980 to 8.7 T/acre in 2016 (Table 11). Duff depth (in.) increased from 2.8 in 1980 to 4.6 in 2016 (Table 11). Fuel height (in.) decreased from 15.6 in 1980 to 2.8 in 2016 (Table 11).

Table 50. Average basal area (X̅ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC22 (Otter Point) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC22-Otter Point | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *15* | 95% CI | | *26* | 95% CI | | *20* | 95% CI | |
| Live | X̅ BA ± SE | Lower | Upper | X̅ BA ± SE | Lower | Upper | X̅ BA ± SE | Lower | Upper |
| Red Spruce | 19.3 ± 4.7 | 10.0 | 28.6 | 63.1 ± 11.2 | 41.2 | 85.0 | 86.0 ± 13.0 | 60.6 | 111.4 |
| Balsam Fir | 14.0 ± 5.0 | 4.3 | 23.7 | 38.5 ± 7.3 | 24.2 | 52.8 | 5.5 ± 3.5 | 0 | 12.4 |
| No. White Cedar | 0 | | | 0.4 ± 0.4 | 0 | 1.2 | 2.0 ± 1.2 | 0 | 4.3 |
| White Pine | 0 | | | 0 | | | 1.0 ± 1.0 | 0 | 3.0 |
| Paper Birch | 2.7 ± 1.5 | 0 | 5.7 | 5.0 ± 2.1 | 0.9 | 9.1 | 0 | | |
| Red Maple | 0 | | | 0.4 ± 0.4 | 0 | 1.2 | 0 | | |
| White Spruce | 2.7 ± 1.2 | 0.4 | 5.0 | 0 | | | 0 | | |
| Pin Cherry | 0 | | | 1.2 ± 0.8 | 0 | 2.9 | 0 | | |
| **Totals:** | **38.7 ± 5.3** | **28.2** | **49.2** | **108.6 ± 11.5** | **86.1** | **131.1** | **94.5 ± 13.4** | **68.2** | **120.8** |
| Dead | | | | | | | | | |
| Red Spruce | 15.3 ± 4.9 | 5.8 | 24.8 | 18.8 ± 4.2 | 10.5 | 27.1 | 6.5 ± 2.1 | 2.4 | 10.6 |
| Balsam Fir | 8.0 ± 2.6 | 2.9 | 13.1 | 1.2 ± 0.6 | 0 | 2.5 | 3.5 ± 1.5 | 0.6 | 6.4 |
| Paper Birch | 0 | | | 0.8 ± 0.8 | 0 | 2.3 | 0.5 ± 0.5 | 0 | 1.5 |
| **Totals:** | **23.3 ± 5.2** | **13.1** | **33.5** | **20.8 ± 4.1** | **12.7** | **28.9** | **10.5 ± 3.0** | **4.5** | **16.5** |

Table 51. Mean diameter (X̅ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC22 (Otter Point) in 1980, 1992-1994 and 2016.

| **AC22** | **1980** | 95% CI | | **1993** | 95% CI | | **2016** | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | X̅ Dia. ± SE | Lower | Upper | X̅ Dia. ± SE | Lower | Upper | X̅ Dia. ± SE | Lower | Upper |
| White Pine | | | | | | | 16.0 ± 0 | 15.9 | 16.1 |
| No. White Cedar | | | | 6.8 ± 0 | 6.8 | 6.8 | 10.1 ± 0.2 | 9.7 | 10.5 |
| Paper Birch | 7.8 ± 0.4 | 7.0 | 8.5 | 8.6 ± 0.5 | 7.7 | 9.5 | 7.2 ± 0 | 7.2 | 7.2 |
| Red Spruce | 11.8 ± 1.3 | 9.3 | 14.4 | 8.0 ± 0.9 | 6.1 | 9.8 | 7.0 ± 0.9 | 5.3 | 8.7 |
| Balsam Fir | 5.1 ± 0.5 | 4.2 | 6.1 | 6.1 ± 0.5 | 5.1 | 7.1 | 6.4 ± 0.5 | 5.5 | 7.4 |
| Red Maple | | | | 11.2 ± 0 | 11.2 | 11.2 | | | |
| White Spruce | 7.9 ± 1.4 | 5.2 | 10.5 | | | | | | |
| Pin Cherry | | | | 4.8 ± 0.3 | 4.2 | 5.3 | | | |
| **Overall Average** | **9.1 ± 1.3** | **6.5** | **11.7** | **7.4 ± 0.8** | **5.8** | **9.0** | **7.1 ± 0.9** | **5.4** | **8.8** |

Eagle Lake (Northwest) (AC01) – Spruce-Fir

This stand escaped the ravages of the 1947 fire and is in the late stages of converting from spruce-fir to mixed conifer-hardwood. Basal areas of red spruce, hemlock, red pine, and northern white cedar decreased, while those of most hardwoods (except for sugar maple) increased between 1980 and 2016. Live basal area of paper birch decreased by 65%. Total live basal area increased between 1980 and 1993 from 179.6 to 194.0 $ft^2$/acre, but declined to 156.5 $ft^2$/acre in 2016 (Table 52). Total dead basal area shows an increasing trend with 16.2 $ft^2$/acre in 1980 and 20.0 $ft^2$/acre in 2016 (Table 52). Density of live trees increased from 424.9 in 1980 to 482.3 stems per acre in 2016 (Table 8). Density of dead trees increased from 47.1 in 1980 to 116.8 stems per acre in 2016 (Table 8). Density of live saplings remained stable (296.2 in 1993 versus 295.0 stems per acre in 2016) (Table 9). Density of dead saplings decreased from 73.1 in 1993 to 35.0 stems per acre in 2016 (Table 9). Mean diameters (in.) increased from 13.9 in 1980 to 15.1 in 2016 (Table 53).

Mature trees continue to be lost to blowdown as evidenced in the increase in heavy downed-woody fuel load. Biomass (T/acre) of live trees decreased from 91.0 in 1980 to 77.3 in 2016 (Table 10). Biomass (T/acre) of dead trees increased from 8.1 in 1980 to 9.6 in 2016. Biomass (T/acre) of shrubs increased from 0.2 in 1980 to 0.6 in 2016 (Table 10). Total downed woody fuel load increased from 12.9 T/acre in 1980 to 13.1 T/acre in 2016 (Table 11). There was no change in duff depth (in.) (Table 11). Fuel height (in.) decreased from 6.5 in 1980 to 3.7 in 2016 (Table 11).

Table 52. Average basal area ($\bar{X}$ BA) of live and dead trees (ft$^2$/acre) ± standard error with 95% confidence intervals for stand AC01 (Eagle Lake (Northwest)) at Acadia National Park based on variable radius plot sampling in 1980, 1992-1994 and 2016.

| AC01-Eagle Lake (Northwest) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1980** | | | **1993** | | | **2016** | | |
| *No. Points Sampled:* | *21* | 95% CI | | *25* | 95% CI | | *20* | 95% CI | |
| Live | $\bar{X}$ BA ± SE | Lower | Upper | $\bar{X}$ BA ± SE | Lower | Upper | $\bar{X}$ BA ± SE | Lower | Upper |
| Red Spruce | 101.9 ± 11.9 | 78.5 | 125.3 | 102.4 ± 11.4 | 80.1 | 124.7 | 73.0 ± 7.2 | 58.9 | 87.1 |
| Hemlock | 45.7 ± 9.4 | 27.2 | 64.2 | 56.8 ± 9.6 | 38.1 | 75.5 | 35.0 ± 8.7 | 18.0 | 52.0 |
| White Pine | 12.9 ± 2.2 | 8.6 | 17.2 | 12.4 ± 2.7 | 7.1 | 17.7 | 23.0 ± 7.5 | 8.4 | 37.6 |
| Balsam Fir | 2.4 ± 1.2 | 0.1 | 4.7 | 5.2 ± 2.3 | 0.7 | 9.7 | 10.5 ± 4.0 | 2.7 | 18.3 |
| Red Maple | 0.5 ± 0.5 | 0 | 1.4 | 2.0 ± 1.0 | 0 | 4.0 | 5.5 ± 2.3 | 0.9 | 10.1 |
| No. White Cedar | 4.8 ± 1.9 | 1.1 | 8.5 | 4.4 ± 1.3 | 1.9 | 6.9 | 3.5 ± 1.7 | 0.2 | 6.8 |
| Paper Birch | 7.1 ± 2.2 | 2.8 | 11.4 | 4.4 ± 1.2 | 2.1 | 6.7 | 2.5 ± 1.0 | 0.6 | 4.4 |
| Yellow Birch | 0.5 ± 0.5 | 0 | 1.4 | 3.2 ± 1.4 | 0.5 | 5.9 | 1.5 ± 0.8 | 0 | 3.1 |
| White Ash | 0 | | | 1.6 ± 0.9 | 0 | 3.4 | 1.5 ± 1.1 | 0 | 3.6 |
| Trembling Aspen | 0 | | | 0 | | | 0.5 ± 0.5 | 0 | 1.5 |
| Red Pine | 1.4 ± 0.8 | 0 | 2.9 | 0.8 ± 0.8 | 0 | 2.4 | 0 | | |
| White Spruce | 0.5 ± 0.5 | 0 | 1.4 | 0.4 ± 0.4 | 0 | 1.2 | 0 | | |
| Shadbush | 0.5 ± 0.5 | 0 | 1.4 | 0 | | | 0 | | |
| Sugar Maple | 1.4 ± 1.4 | 0 | 4.2 | 0.4 ± 0.4 | 0 | 1.2 | 0 | | |
| **Totals:** | **179.6 ± 11.7** | **156.7** | **202.5** | **194.0 ± 8.8** | **176.8** | **211.2** | **156.5 ± 11.1** | **134.8** | **178.2** |
| Dead | | | | | | | | | |
| Red Spruce | 10.5 ± 3.1 | 4.4 | 16.6 | 13.2 ± 2.4 | 8.6 | 17.8 | 9.0 ± 2.6 | 3.9 | 14.1 |
| Paper Birch | 1.4 ± 0.8 | 0 | 2.9 | 1.2 ± 0.7 | 0 | 2.5 | 3.0 ± 1.8 | 0 | 6.5 |
| Balsam Fir | 1.9 ± 1.5 | 0 | 4.8 | 0.4 ± 0.4 | 0 | 1.2 | 3.0 ± 1.5 | 0.1 | 5.9 |
| No. White Cedar | 0 | | | 0.8 ± 0.6 | 0 | 1.9 | 2.0 ± 0.9 | 0.2 | 3.8 |
| White Pine | 1.9 ± 0.9 | 0.2 | 3.6 | 2.0 ± 0.8 | 0.4 | 3.6 | 1.5 ± 0.8 | 0 | 3.1 |
| Hemlock | 0 | | | 2.8 ± 1.4 | 0.1 | 5.5 | 1.0 ± 0.7 | 0 | 2.4 |
| Red Maple | 0 | | | 0 | | | 0.5 ± 0.5 | 0 | 1.5 |
| White Spruce | 0.5 ± 0.5 | 0 | 1.4 | 0 | | | 0 | | |
| Yellow Birch | 0 | | | 0 | | | 0 | | |
| **Totals:** | **16.2 ± 3.7** | **9.0** | **23.4** | **20.4 ± 3.1** | **14.2** | **26.6** | **20.0 ± 2.8** | **14.5** | **25.5** |

Table 53. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC01 (Eagle Lake (Northwest)) in 1980, 1992-1994 and 2016.

| AC01 | 1980 | 95% CI | | 1993 | 95% CI | | 2016 | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper | X̄ Dia. ± SE | Lower | Upper |
| Hemlock | 16.1 ± 1.1 | 13.9 | 18.2 | 16.5 ± 1.1 | 14.3 | 18.7 | 19.3 ± 1.6 | 16.2 | 22.4 |
| White Pine | 22.1 ± 1.4 | 19.4 | 24.9 | 22.9 ± 1.2 | 20.5 | 25.2 | 18.2 ± 2.1 | 14.2 | 22.3 |
| Yellow Birch | 17.0 ± 0 | 17.0 | 17.0 | 12.1 ± 0.8 | 10.5 | 13.6 | 15.9 ± 0.9 | 14.2 | 17.7 |
| Red Spruce | 12.5 ± 1.1 | 10.4 | 14.6 | 13.2 ± 0.9 | 11.4 | 15.0 | 14.2 ± 1.1 | 12.1 | 16.3 |
| Paper Birch | 10.7 ± 0.9 | 8.9 | 12.5 | 14.6 ± 0.6 | 13.4 | 15.9 | 13.2 ± 0.7 | 11.9 | 14.5 |
| No. White Cedar | 11.9 ± 1.0 | 10.0 | 13.8 | 12.0 ± 0.5 | 11.1 | 13.0 | 12.9 ± 1.3 | 10.4 | 15.5 |
| Red Maple | 12.0 ± 0 | 12.0 | 12.0 | 12.8 ± 0.9 | 11.1 | 14.5 | 12.5 ± 1.0 | 10.5 | 14.6 |
| White Ash | | | | 12.7 ± 1.3 | 10.2 | 15.2 | 11.8 ± 0.9 | 10.1 | 13.6 |
| Trembling Aspen | | | | | | | 10.6 ± 0 | 10.6 | 10.6 |
| Balsam Fir | 6.7 ± 0.8 | 5.1 | 8.3 | 4.5 ± 0.4 | 3.7 | 5.3 | 6.3 ± 0.7 | 4.9 | 7.6 |
| Sugar Maple | 23.7 ± 1.5 | 20.6 | 26.7 | 13.3 ± 0 | 13.3 | 13.3 | | | |
| Shadbush | 7.3 ± 0 | 7.3 | 7.3 | | | | | | |
| White Spruce | 10.8 ± 0.5 | 9.8 | 11.8 | 13.1 ± 0 | 13.1 | 13.1 | | | |
| Red Pine | 17.4 ± 0.4 | 16.6 | 18.3 | 20.4 ± 0.6 | 19.1 | 21.6 | | | |
| **Overall Average** | **13.9 ± 1.3** | **11.4** | **16.5** | **14.6 ± 1.2** | **12.3** | **16.8** | **15.1 ± 1.5** | **12.1** | **18.0** |

Bernard Mountain (AC26) – Spruce-Fir

Ronald B. Davis sampled this stand in the 1960s and characterized it as the only stand that he found that was 'virgin' old growth (i.e. had never been cut) (Davis 1962). When Patterson et al. (1983) first sampled the stand in 1981, it had suffered significant windthrow and was in the process of reestablishing. The Bernard Mountain stand has been regenerating since. This is an important stand as one example of how spruce-fir stands change in the absence of fire.

Live basal area of red spruce significantly increased 346% from 22.3 $ft^2$/acre in 1981 to 99.5 $ft^2$/acre in 2016. Dead basal area of red spruce increased by 84% from 19.3 $ft^2$/acre in 1981 to 35.5 $ft^2$/acre in 2016. Total live basal area significantly increased between 1981 and 1993 from 32.0 to 128.0 $ft^2$/acre, and in 2016 decreased to 110.0 $ft^2$/acre (Table 54). Total dead basal area significantly increased with 20.6 $ft^2$/acre in 1981 and 39.0 $ft^2$/acre in 2016 (Table 54). Density of live trees decreased from 1168.1 in 1981 to 711.9 stems per acre in 2016 (Table 8). Density of dead trees increased from 109.7 in 1981 to 178.1 stems per acre in 2016 (Table 8). Density of live saplings significantly increased by 394% measuring at 333.3 in 1993 and 1645.0 stems per acre in 2016 (Table 9). This is the densest stand of regeneration of all stands in this study. Density of dead saplings decreased by 67% from 120.0 in 1993 to 40.0 stems per acre in 2016 (Table 9). Mean diameters (in.) increased from 6.4 in 1981 to 10.6 in 2016 (Table 55).

Biomass (T/acre) of trees increased from 12.5 live and 7.2 dead in 1981 to 48.6 live and 16.5 dead in 2016 (Table 10). Biomass (T/acre) of shrubs increased from no measurable live and dead in 1981 to 1.0 live and 0.3 dead in 2016 (Table 10). Total downed woody fuel load decreased from 33.5 T/acre in 1981 to 13.9 T/acre in 2016 (Table 11). Duff depth (in.) increased from 5.2 in 1981 to 5.3 in 2016 (Table 11). Fuel height (in.) decreased from 35.1 in 1981 to 2.9 in 2016 (Table 11).

Table 54. Average basal area (X̄ BA) of live and dead trees ($ft^2$/acre) ± standard error with 95% confidence intervals for stand AC26 (Bernard Mountain) at Acadia National Park based on variable radius plot sampling in 1981, 1992-1994 and 2016.

| AC26-Bernard Mountain | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sample Year:** | **1981** | | | | **1993** | | | | **2016** | | | |
| *No. Points Sampled:* | *15* | | 95% CI | | *15* | | 95% CI | | *20* | | 95% CI | |
| Live | X̄ BA ± | SE | Lower | Upper | X̄ BA ± | SE | Lower | Upper | X̄ BA ± | SE | Lower | Upper |
| Red Spruce | 22.3 ± | 5.1 | 12.3 | 32.3 | 111.3 ± | 14.3 | 83.3 | 139.3 | 99.5 ± | 10.9 | 78.1 | 120.9 |
| Balsam Fir | 6.7 ± | 2.4 | 2.0 | 11.4 | 1.3 ± | 0.9 | 0 | 3.1 | 5.5 ± | 2.3 | 0.9 | 10.1 |
| Red Maple | 0.7 ± | 0.5 | 0 | 1.6 | 0 | | | | 2.5 ± | 1.2 | 0.1 | 4.9 |
| Paper Birch | 2.3 ± | 1.0 | 0.4 | 4.2 | 12.0 ± | 4.8 | 2.6 | 21.4 | 1.5 ± | 0.8 | 0 | 3.1 |
| No. White Cedar | 0 | | | | 0.7 ± | 0.7 | 0 | 2.0 | 1.0 ± | 1.0 | 0 | 3.0 |
| Yellow Birch | 0 | | | | 0.7 ± | 0.7 | 0 | 2.0 | 0 | | | |
| Striped Maple | 0 | | | | 0.7 ± | 0.7 | 0 | 2.0 | 0 | | | |
| Sugar Maple | 0 | | | | 1.3 ± | 1.3 | 0 | 3.9 | 0 | | | |
| **Totals:** | **32.0 ±** | **5.8** | **20.6** | **43.4** | **128.0 ±** | **16.5** | **95.7** | **160.3** | **110.0 ±** | **10.9** | **88.6** | **131.4** |
| Dead | | | | | | | | | | | | |
| Red Spruce | 19.3 ± | 3.0 | 13.4 | 25.2 | 17.3 ± | 5.6 | 6.4 | 28.2 | 35.5 ± | 5.3 | 25.1 | 45.9 |
| Balsam Fir | 0 | | | | 2.0 ± | 2.0 | 0 | 5.9 | 3.5 ± | 1.7 | 0.2 | 6.8 |
| Paper Birch | 1.3 ± | 1.0 | 0 | 3.3 | 11.3 ± | 5.5 | 0.5 | 22.1 | 0 | | | |
| **Totals:** | **20.6 ±** | **2.9** | **14.9** | **26.3** | **30.6 ±** | **6.9** | **17.0** | **44.2** | **39.0 ±** | **5.1** | **29.1** | **48.9** |

Table 55. Mean diameter (X̄ Dia.) of live trees (in.) ± standard error with 95% confidence intervals by species and totals for stand AC26 (Bernard Mountain) in 1981, 1992-1994 and 2016.

| **AC26** | **1981** | | 95% CI | | **1993** | | 95% CI | | **2016** | | 95% CI | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper | X̄ Dia. | ± SE | Lower | Upper |
| No. White Cedar | | | | | 9.8 | ± 0 | 9.8 | 9.8 | 14.0 | ± 0.3 | 13.5 | 14.6 |
| Red Spruce | 6.7 | ± 1.2 | 4.4 | 9.0 | 7.9 | ± 1.1 | 5.7 | 10.0 | 11.0 | ± 1.1 | 8.9 | 13.1 |
| Red Maple | 11.9 | ± 1.2 | 9.5 | 14.3 | | | | | 10.9 | ± 0.8 | 9.3 | 12.5 |
| Paper Birch | 7.2 | ± 0.7 | 5.8 | 8.7 | 8.6 | ± 1.1 | 6.4 | 10.7 | 6.4 | ± 1.1 | 4.3 | 8.5 |
| Balsam Fir | 3.6 | ± 0.4 | 2.8 | 4.4 | 5.2 | ± 0.9 | 3.4 | 7.0 | 5.5 | ± 0.7 | 4.2 | 6.8 |
| Striped Maple | | | | | 6.8 | ± 0 | 6.8 | 6.8 | | | | |
| Sugar Maple | | | | | 13.1 | ± 0.3 | 12.5 | 13.6 | | | | |
| Yellow Birch | | | | | 8.0 | ± 0 | 8.0 | 8.0 | | | | |
| **Overall Average** | **6.4** | **± 1.1** | **4.2** | **8.6** | **8.0** | **± 1.1** | **5.8** | **10.1** | **10.6** | **± 1.1** | **8.5** | **12.8** |

*Spruce-Fir Summary*

Barring stand replacing disturbance and assuming a predictable successional trajectory without disturbance I expect the Hodgdon Pond (AC20), Western Head (AC13), Western Mountain (upper) (AC18), Schoodic Peninsula (West) (AC17), Deep Cove (AC12), Otter Point (AC22), Eagle Lake (Northwest) (AC01), and Bernard Mountain (AC26) stands to continue to develop as the spruce-fir community type. Stanley Brook (AC04) is currently and will likely remain predominantly red spruce in the absence of disturbance. Fuel loading in all these spruce-fir stands is high to very high, including ladder fuels (i.e., well developed layer of advanced regeneration), and would support high-intensity fire under drought conditions.

At Hodgdon Pond (AC20) red spruce, white pine and red maple will likely continue to increase in basal area. With a decrease in live tree stem density (56%) this stand is likely beyond the stem exclusion stage. A decline in saplings and shrubs is also evident. Unlike some of the older spruce-fir stands, AC20 continues to see development of the duff layer. Perhaps in time this stand will move towards conditions similar to that of the older stands in this community type such as AC01 (Eagle Lake (NW))) or AC22 (Otter Point). This of course assumes no large stand replacing event such as a windstorm which are predicted to increase with changing climate.

Western Head (AC13) is a regenerating stand with a few remnant mature trees. This stand suffered from a major blowdown event(s). In 2016 it presented rapidly growing regeneration of spruce and fir. Due to the windthrow disturbance there was substantial decrease in live basal area and increase in downed fuel load. I expect to see this stand move in a direction from the initial mixture of red spruce, paper birch, and aspen before returning towards spruce-fir. It is unlikely this stand will become a uniformly closed canopy spruce-fir stand due to the exposure to the elements on Western Head.

On Western Mountain (upper) (AC18) shrubs and tree saplings will exploit canopy gaps. These canopy gaps will continue to open allowing for new growth in the understory. Increased red spruce importance should be expected.

Western Mountain (upper) (AC18), Stanley Brook, (AC04), Schoodic Peninsula (AC17), and Eagle Lake (Northwest) (AC01) are beginning to break up as evidenced by the increase in downed fuel load. Schoodic Peninsula (AC17) had dense shrub regeneration in canopy openings in 2016. Patterson et al. (1983) notes that there were two age groups present in the overstory. Many of the older trees (~171-236 years old in 2016) have fire scars, and charcoal is present in the duff (Patterson et al. 1983, 1996). The young age class in the canopy is ~136 years in 2016. It is likely that these trees grew up in an open canopy environment through which a fire spread (Patterson et al. 1983). Many of these older spruce-fir stands apparently grew up following large fires, or after a prolonged period of grazing - all of which followed late 18th and early 19th century logging and land clearing (Patterson et al. (1983). Patterson et al. (1983) further notes, "In these [spruce-fir] stands the natural fire cycle appears to be tied to the maturation cycle of the dominant trees. Downed woody fuel accumulations remain moderate until the canopy breaks up, at which time they increase rapidly, then remain high for 10 to 20 years."

Deep Cove (AC12) appears to be over-mature and breaking up as evidenced in the increase in heavy downed-woody fuel load; and the overall decrease in basal area, average fuel height and duff depth which suggests decomposition rather than accumulation. There is also an increase in shrub biomass suggesting the canopy may be allowing light to penetrate the forest floor. The exposure to the sea and wind on Isle au Haut makes it unlikely that this stand will develop into a closed canopy spruce stand.

The Otter Point (AC22) stand continues to release regeneration in the open spaces created from past blowdown event(s). Decomposition of the blown-down timber is evident in the 87% decrease in total average downed woody fuel load over the 36-year period 1980 to 2016. This is the densest stand studied. In the future this stand will likely move towards the more closed-canopy condition like prior to 1960, and then blowdown over a period of decades as it has before.

Bernard Mountain (AC26) which was the oldest stand is now the youngest, with old trees having been replaced following blowdown in the last half century. This stand is still recovering from blowdown events as evidenced in the dense regeneration and by the 59% decrease in heavy downed fuel load and 92% decrease in average fuel height. In time this stand may move towards closed canopy spruce-fir, like that of Otter Point (AC22).

These nine spruce-fir stands, which range in age from ~100 to >180 years old, illustrate that the forest of MDI is not likely to reach some stable condition, but will always, in the absence of fire, be churning though growth, maturation, break-up and regrowth stages over centuries-long cycles. Photo series 29 visually exhibits the longevity of this community type (Appendix 1).

## Discussion

*Post-fire vegetation development*

This post-1947 fire data set collected within ANP bounds provided a unique opportunity to examine the impacts of wildfire on upland forest communities of coastal Maine. Although the 1947 fire may be viewed as catastrophic by some measures – e.g., by resulting in the loss of a forest cover type over extensive areas and conversion from spruce-fir forests to northern hardwoods – the results suggest that stand development after wildfire varies and not all areas across the park experienced severe change. This is expected given the variability in fire intensity and burn severity that occurs during a burn across the landscape. Some areas burned severely while others did not, depending on site factors (e.g. slope, aspect, landscape position) and the fire environment. For example, in the Penobscot Bay area, the vegetation is comprised of a mixture of successional stages, the pattern of which is largely controlled by recent fire and land-use history (Patterson et al. 1983).

Based on this study, the forests of ANP still resemble, more or less, the previous descriptions (Patterson et al. 1983). Stands which did not burn in the 1947 fire were dominated primarily by spruce-fir and mixed hardwood-conifer community types. Canopies in the spruce-fir cover type are dense except for wind-thrown sites (e.g., AC13, AC22, and AC26). Patterson et al. (1983) notes areas where fuel accumulations were heavy due to blowdown. These concentrations tended to occur in pockets (e.g., in the vicinity of Otter Point, on Western Head, Isle au Haut, and Western Mountain). This held true in 2016 except for Otter Point (AC22), which grew to nearly closed-canopy spruce-fir and with dramatically less downed wood.

In 2016, after 68 years of post-fire vegetation development, burned stands were dominated by birch-aspen, northern hardwoods and in small isolated pockets the fire-dependent pitch pine. The 1947 fire favored regeneration of species that respond well to disturbance, e.g., bigtooth and quaking aspen, paper and gray birch, white pine, and red maple (Barton et al. 2012). Nearly 70 years after the fire, longer-lived, shade-tolerant species are becoming more important, exerting greater ecological influence (e.g., influence on the understory and nutrient cycling). Stands which were primarily birch and aspen in 1980 now contain a mixture of hardwoods and red spruce in the canopy. One birch-aspen stand has now transitioned to red spruce and red oak (AC25) (Table 5). Stand AC02 was a northern hardwood stand in 1980, and while still less than half the basal area as American beech, now has a strong component of red spruce. The one red oak stand (AC07) that burned remains red oak. With no fire since 1947, the MDI pitch pine stand (AC08) canopy is being replaced with red spruce which is now overtopping pitch pine.

Understory vegetation was influenced by overstory conditions. The dominant species regenerating across all stands is red spruce, and to a lesser extent balsam fir. This is true for both unburned and burned stands. In stands which did not burn in 1947, regeneration is uniformly red spruce with balsam fir as a secondary component. In stands burned in 1947, red spruce seedlings germinating after the fire now dominate the understory with lesser amounts of striped maple, paper birch, gray birch, and white pine. Analysis of stand structure gives evidence that spruce establishment starts early after fire from seeds released from canopy cones which were not consumed by the fire; protected by the higher needle moisture contents of young foliage at the top of tree canopies, as suggested by Sirois and Payette (1989) and W.A.

Patterson (personal communication). Some plant species, but not red spruce, produce seeds that can be stored in the soil for long periods and germinate when exposed to the warmer, moister, and higher nutrient content conditions that follow fire (Patterson et al. 1983). Other species have light seeds (e.g., aspen) or seeds that can be carried long distances by wind (e.g., birch) rapidly recolonizing burned soils (Patterson et al. 1983; Sirois & Payette 1989). This is evidenced in the successional sequence of the hardwood stands of ANP.

Blueberry, huckleberry, and sheep laurel are important components of the shrub layer in some forest types (e.g., pitch pine) across ANP, and their presence may increase stand flammability. The flammable nature of these ericaceous shrubs and extensive mats of lichens on the forest floor combine to increase fuel bed flammability (Patterson et al. 1983). The persistence of these shrubs may likewise depend upon recurring fire (Patterson et al. 1983), and thus in part, suggests that ANP has burned regularly in the past and will likely burn in the future.

In 2016, the 1947 burned stands still had less duff accumulation and vertical fuel structure than the unburned stands. Organic matter is often consumed in intense fires, impacting the depth for many years following a burn (Barnes et al. 1998). At many locations on MDI, the 1947 fire completely consumed the humus layers. This was a consequence of the very high drought condition at the time of the fire (Patterson et al. 1983).

Figure 6a and 6b are conceptual models of the stand development across ANP. The years in the model reflect precision based on one fire event. Stands which burned regenerated as aspen-birch, red oak, or northern hardwoods which persist for approximately 68-146 years. We then see a transition to mixed conifer-hardwood which remains as mixed conifer-hardwood based on species composition, although changes in dominance reflect succession. In the mixed hardwood-conifer stands sampled at ANP data show that stem density in this community is not stable which is expected with succession, as a general principle, as stem diameters increase density decreases with minimal changes in basal area. The ANP data show (e.g., Long Pond and Day Mountain) that the mixed hardwood-conifer (i.e., aspen, birch, beech, maple with some white pine and/or red spruce/balsam fir) community type persists for up to approximately 186 years and then transitions to either mixed conifer or spruce-fir which maintains as a community until another disturbance resets the sequence (Figure 6a). However, it can be expected that climate change will alter this model. A warming climate will likely accelerate the process and result in a change to a different community type than that of the past two centuries. During the next 100-200 years, climate is projected to warm substantially, and this will favor hardwoods over conifers as during the mid-Holocene warm period. Pitch pine, jack pine, and northern white cedar, which are present at ANP, do not follow the expected successional trajectory as shown in Figure 6a. Figure 6b shows these three community types which regenerate post-disturbance and self-perpetuate due to site factors (e.g., steep slopes, rocky shallow soils).

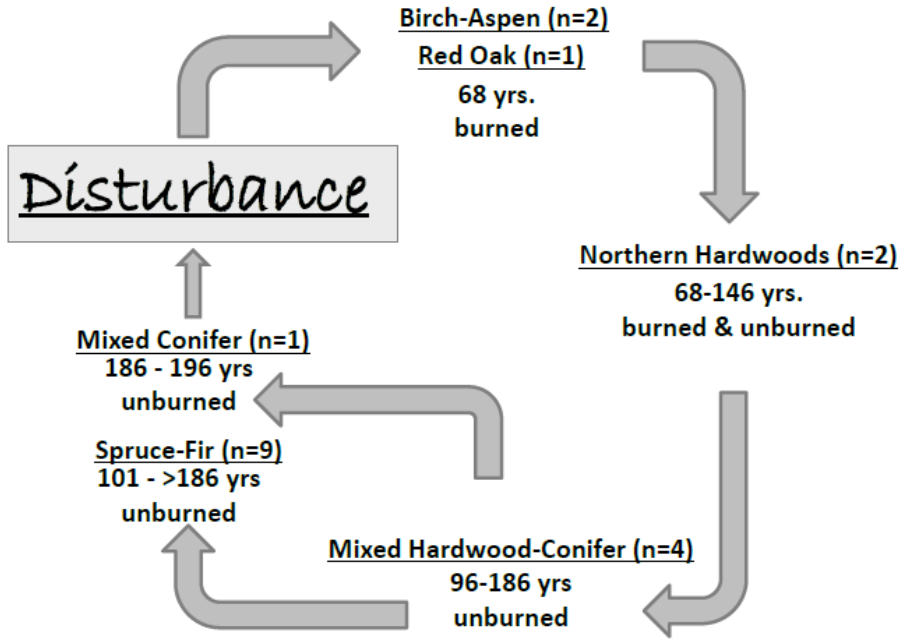



Figure 6a. Conceptual model of successional trajectories and timeline for stands of ANP. Years refers to stand age range for the community type based on the sample stands. Burned= burned in the 1947 fire; unburned= did not burn in the 1947 fire. n = number of stands sampled.

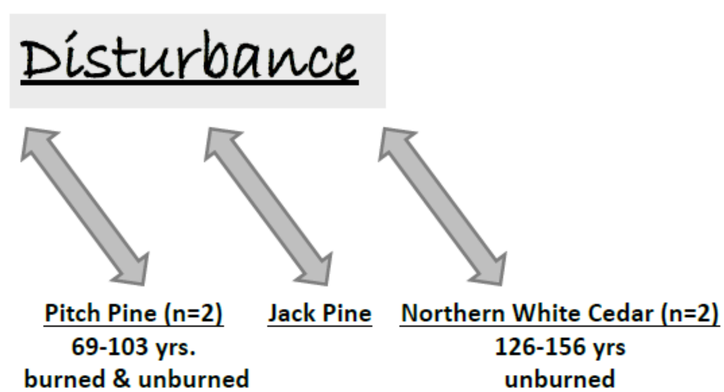



Figure 6b. Conceptual model of community types that do not follow typical successional sequence as shown in Figure 6a.

*Ecology and management*

In recent decades, promoting resilience has been a widespread goal of forest management, as the increasing pressure of large- and small-scale disturbances is pushing many forests toward and over resilience thresholds (Heinselman 1973, 1981b; Mladenoff et al. 1993; Robertson et al. 1993; Fleming 1996; Vogt et al. 1997; Paine et al. 1998; Dale et al. 2000; Johnstone & Chapin 2003; Hooper et al. 2005; Schulte & Mladenoff 2005; Hayhoe et al. 2007a; Seidl et al. 2009; Rogers et al. 2011; Seidl et al. 2011a; Hart et al. 2019). The consequences of increased temperatures, extreme droughts, catastrophic wildfire, and widespread insect outbreaks demonstrate that resilience thresholds can be exceeded and that major ecological transformations can result. Thresholds are crossed when forests convert to vegetation types without trees (e.g., heathlands) (Payette et al. 2000; Payette & Delwaide 2003; Simard & Payette 2005) and, as a result, lose valuable forest ecosystem services such as functioning as a net sink of atmospheric $CO_2$ (Schimel et al., 2001; Goodale et al. 2002; Dilling et al., 2003; Foster & Aber 2004; Gough et al. 2007; Dore et al. 2008; Hundiburg et al. 2009; Gauthier et al. 2015; Dey et al. 2019). These changes are increasingly challenging the main objectives of forest management, which are to provide ecosystem services sustainably to society and maintain the biological diversity of the forests (Jantz et al. 2016; Dey et al. 2019). Therefore, implementation of forest management must rely on a solid understanding of the main disturbance regimes' effects on forest ecosystems at various spatial and temporal scales (Dale et al. 2001; Jantz et al. 2016). In the so-called catastrophic 1947 wildfire of ANP resilience thresholds were not exceeded based on the data in this study and the previous two sample periods. The data show forest regeneration on an expected successional trajectory (Figure 6a) with diverse forest compositions and age class distributions, including continued ecosystem services provisions.

The maintenance of ecosystem services from forests depends on the preservation of forest health, which is threatened by the speed and amplitude of changes in climate (Dale et al. 2000, 2001; Iverson et al. 2004; Plummer et al. 2006; Hayhoe et al. 2007a, b; Campbell et al. 2009; Dukes et al. 2009; Flannigan et al. 2009; Seidl et al. 2011b; Brose et al. 2013; Duveneck et al. 2014; USGCRP 2018; IPCC 2019), and thus disturbance regimes projected for these northern latitudes (Brotak & Reifsnyder 1977; Manabe et al. 1981; Paine et al. 1998; Fischlin et al. 2007; Huntington et al. 2009; Mohan et al. 2009; Allen et al. 2010; Turner 2010; Buma & Wessman 2011; D'Amato et al. 2011; Brown & Johnstone 2012; Adams 2013; Amraoui et al. 2013; Keenan 2015; Appenzeller 2015; Millar & Stephenson 2015; IPCC 2019). Considering the importance of the potential impacts these changes may have and the extent over which they may take place, it is imperative that adaptive actions be taken to maintain the health of the forest or to enhance its contribution to climate change mitigation (Paine et al. 1998; Gauthier et al. 2015; Dey et al. 2019; Hisano et al. 2019). The challenge is determining when the frequency, spatial extent, and strength of stresses and disturbances exceed the natural range of variability and affect the trajectory of vegetation recovery at the regional to landscape scale (Ayres & Lombardero 2000; Dale et al. 2000; Iverson et al. 2004; Campbell et al. 2009; Frelich & Reich 2010; Churma et al. 2011; Brown & Johnstone 2012; Trumbore et al. 2015). For example, such as the 1947 fire on MDI, a large stand-replacing fire may have a human ignition source, but its intensity could reflect drought conditions and fuel-load buildup from previous disturbance or management decisions. Mount Desert Island has and will continue to experience a marked increase in human development and visitation, thereby increasing the likelihood of human-caused ignitions. This, coupled with increasing fuel loads, may significantly increase the likelihood of wildfire occurrence. An uncertain climate future may exacerbate potential wildfire risk. Should climate warm substantially, spruce-fir stands may breakup prematurely – significantly increasing dead, downed fuel for a period of time. Fire management capacity will be overwhelmed in the future, but silvicultural practices can be adapted to changing fire regimes (Star et al. 2015; Trumbore et al. 2015; Hart et al. 2019). This should be a consideration in defining forest management goals and implementing forest management strategies in the context of the future climate.

Adaptive management strategies may include reductions in fuel loads and promotion of species structure (e.g., large fire resistant conifers) that will be better adapted to a fire regime that is expected to be different than those of the past due to potential effects of climate change, while at the same time managing landscapes in ways that allow ecologically and socially important forests and associated species to persist as long as possible (Spies et al. 2006; Matlack 2013). If climate warms to the point where hemlock and hardwoods are favored over spruce-fir, there may be less fire but only after a possible large conflagration burning through extensive dead and down spruce-fir. It is the transition phase between what exists now and what will be in the future that deserves attention. Considering the mounting environmental and social pressures on northeastern US forests, it is increasingly important to maintain and, where possible, foster their adaptive capacity in order to facilitate the recovery of ecosystems after disturbance, and support functional, structural, and compositional continuity (Paine et al. 1998; Buma & Wessman 2011; DeRose & Long 2014; Cavard et al. 2019). The maintenance of diverse forest compositions and age class distributions will be a key element for maintaining forest resilience (Heinselman 1973; Frelich & Lorimer 1991; Shugart et al. 1992; Gimingham & Johnson 1993; Attiwill 1994; He & Mladenoff 1999; Dale et al. 2001; Jayen et al. 2006; Dey et al. 2019). At ANP after 69 years of post-fire development, burned stands were dominated by birch-aspen, northern hardwoods, and in small isolated pockets the fire dependent pitch pine. Stands which did not burn in the 1947 fire were dominated primarily by spruce-fir and mixed hardwood-conifer community types, suggesting that cover type conversions occurred within the bounds of the 1947 fire creating a landscape mosaic of forest types and successional stages, which can be attributed to the fire.

One of the main goals of forest management is to use silvicultural techniques that are appropriate for maintaining ecosystems relatively close to their natural state (Figures 6a, 6b), or to recover that state when they have been modified by human activity (Bergeron 2000; Spies et al. 2006; Matlack 2013; DeRose & Long 2014). Because the composition and spatial structure of forest ecosystems depend strongly upon disturbance regimes, it is often recommended that silvicultural techniques that produce effects similar to those of natural disturbances be used (Bergeron 2000). Some undesirable climate change impacts (e.g., increased fire severity and frequency; impacts on seed production, germination, and seedling growth and survival) may also be avoided or reduced through distinct strategies based on a forest's historical disturbance regime (Stephens et al. 2013).

Fire management practices that maintain resilience by mimicking natural disturbance regimes include fire suppression, prescribed burning, salvage logging, and regeneration improvement (e.g., create germination seedbeds, reduce competing vegetation cover)

(Gauthier 2009). Prescribed fire is an example of a management action to reduce vulnerability or enhance recovery after disturbance (Gauthier 2009). Fire management aims to either increase or decrease (depending on specific goals) total area burned through the manipulation of forest structure and composition (Pyne et al. 1996). Prescribed burning, which consists of fire intentionally ignited, can be used to alter fuel structure and composition (e.g., reduce fuel loads, reduce fuel continuity, or enhance stand age distribution across a landscape) (Pyne et al. 1996). Another option resides in manipulative vegetation treatments. For example, mechanical removal of fuels is an approach that was initially developed to protect timber reserves against high fire risk (Hirsch et al. 2004; Krawchuk & Cummings 2011). This was done on MDI in the 1930s. The removal of dead or dying trees and downed woody debris can reduce the risk of fire as well as alter insect and disease dynamics (Pyne et al. 1996; Payette et al. 2001; Allen et al. 2010). Density management can also reduce drought stress (though it may make forests more susceptible to wind) (Gustafson & Sturtevant 2013; Varner et al. 2016; Lienard et al. 2016; Rogers et al. 2017; Abrahamson, 2018).

In the aftermath of a disturbance, recovery can be enhanced by adding structural elements that create shade or other safe sites that serve necessary for reestablishing vegetation (Grubb 1977). Late successional species can be planted to speed up succession (Egler 1954; Keeton et al. 2011). Further, deciduous stands are characterized by lower flammability than coniferous stands (Hély et al. 2001), therefore deciduous stands in ANP could be used as strategic barriers combined with pre-existing fire breaks (e.g., roads and water bodies) to decrease the ANP landscape susceptibility to fire (Herberger & Patterson 1998). Forest management strategies such as continuous cover silviculture and the enhancement of tree species diversity and of landscape heterogeneity may aid in the maintenance of forest cover, the conservation of C stocks, and biodiversity (Law et al., 2001, 2003; Humphreys et al., 2005; Gough et al. 2007; Dore et al. 2008; Hundiburg et al. 2009; Tang et al. 2014). However, like many important drivers of disturbance regimes (e.g., species composition), response to management can take centuries (Turner 1987; Mladenoff et al. 1993; Robertson et al. 1993; Fleming 1996). Thus, management considerations need to take long lead-times into account.

The question is: Can resource management agencies adapt and mitigate the impacts of this potential transitory increase in fire activity through proactive forest management? Forest management and fire suppression in northeastern US forests began in the 1920s at a time when fire activity was low (Fahey & Reiners 1981; Clark 1988; Flannigan & van Wagner 1991; Flannigan et al. 1998, 2001; Bergeron et al. 2001; Irland 2014; Hart et al. 2019). Knowledge of and experience with wildfire by the public, communities and resource management sectors may hence be significantly underestimated. Most of all areas burned are due to fire that escaped from human control (e.g., escaped prescribed fires and brush pile burning), and those that grow to high intensities are costly and difficult to suppress (Podur & Wotton 2010). Fire suppression agencies can be expected to be increasingly overwhelmed as climate change creates fire danger conditions similar to those experienced historically (e.g., 1947), leading to higher numbers of escaped and intense fires (Le Goff et al. 2005; Jasinski & Payette 2005; Podur & Wotton 2010; Miller 2019). Nevertheless, the history of severe fire activity is not expected to be exceeded in the next few years (Girardin & Mudelsee 2008). Thus, there is time for preparation and exploring new management strategies that could ensure the sustainability of forest management under a changing climate (Girardin et al. 2013). Even in National Parks like

ANP where our understanding of the forest's successional trajectories is strong, forest management, though limited by regulation, should develop plans that will mitigate the impact of large scale fire.

**Conclusion**

The results of this study may be used to inform and support future management decisions at ANP. The biomass and necromass data may be applied to fuel and fire management. The data informs the fuel load and fire hazard as presented in the following chapter.

It is important to recognize that many forms of disturbances are important to ecosystem health and functioning. Thus, management efforts to preserve biodiversity and ecosystem services should include explicit consideration of disturbance processes (Wallington et al. 2005). Increasing the resilience of ecosystems through deliberate and proactive management to climate-mediated changes in disturbance regimes is important (Spies et al. 2012; Stephens et al. 2013) for the safety of human structures and life that now occupy the landscape.

Periodic monitoring is essential to assess the condition of the forests and improve our understanding of the interactions and feedback among ecosystem processes. Post-disturbance regeneration phases deserve particular attention, as they may provide early warnings of forest health degradation and allow the rapid implementation of remedial actions to prevent, for instance, the loss of forest cover and shifts to undesirable states. Likewise, they may reflect desirable change which should not be interfered with by short-sighted attempts to 'fix what is not broken'.

## Chapter 5: Wildfire Risk Assessment

Note: This is a stand-alone chapter written for the National Park Service. It went through the peer review process and is presented here after it was revised and accepted.


**Abstract**

Federal and state fire managers have a critical need for a general baseline geospatial assessment of wildfire risk that identifies areas of paticularily high fuel loading. The projected increase in population, pressure for land use change, uncertainties associated with climate change, and declining State and Federal budgets will result in more complex fire suppression strategies. Fire management programs must continue to operate strategically and efficiently to meet this challenge. The main objective of this study was to identify areas in Acadia National Park and the surrounding landscape with high wildfire risk so that managers could identify where mitigation practices would be most effective in reducing wildfire risk. Based on the findings wildfire risk is generally very high to high with small pockets of moderate to low risk outside of the 1947 fire boundary. Within the 1947 fire boundary, the wildfire risk is primarily moderate to low with small areas of high to very high risk which is primarily due to topography. The wildfire risk maps presented are an accurate broad scale representation of the wildfire risk across Acadia National Park. Due to variances in the reliability of the input data, the scale at which this analysis was conducted, and the range of fuels present throughout the study area, conclusions based on the findings of this wildfire risk assessment should be carefully considered at the prescribed fire burn unit scale. Mount Desert Island has and will continue to experience a marked increase in human development and visitation thereby increasing the likelihood of

human-caused ignitions. This coupled with the uncertainties of climate change may significantly increase the vulnerability of wildfire occurrence across the Park.

**Introduction**

Wildfire risk is defined as areas with the potential for destruction or damage from wildfire (U.S. Forest Service 2010). To reduce wildfire risk to communities, land managers must identify and prioritize areas for hazardous fuels reduction to minimize catastrophic damage to natural resources and infrastructure in the event of a wildfire (Hardy 2005; Hessburg et al. 2007; Keane et al. 2010, Thompson et al. 2011; Hmielowski et al. 2016). Spatial data describing fuel loads are critical for strategic decision-making during wildfire events and planning to mitigate wildfire risk across the landscape (Keane et al. 2001; Morgan et al. 2001; Rollins et al. 2004). An approach to mapping wildfire risk that integrates extensive field data, remote-sensing technologies, and biophysical modeling is an effective method for identifying priority areas for fuels reduction (Keane et al. 2001, 2002; Morgan et al. 2001; Finney 2005; Rollins et al. 2006; Rollins 2009). Although ANP data suggest that catastrophic large fires historically occurred infrequently (Davis 1962, 1967; Patterson et al. 1983; Patterson & Backman 1988; Drake & Patterson 1994; Patterson 2006; Schauffler et al. 2007; Marlon 2015), the potential exists for high-intensity fires in the future.

While large devastating wildfires are rare in the northeastern US they do occur. For example, in the summer of 1908, more than 300,000 acres (121,405 ha) burned in the Adirondacks of New York, 142,000 acres (57,465 ha) burned in Maine, and 16,000 acres (6,475 ha) burned in Vermont (Long 2016). New Hampshire’s worst fire year was 1903 when 84,000 acres (33,994 ha) burned. In the 1940s, large landscape-scale wildfires fed on fuel left by the September 1938 hurricane occurred across the Northeast region (Long 2016). In 1941, the largest post-hurricane wildfire, the Marlow-Stoddard fire in New Hampshire burned 27,000 acres (10,927 ha) during the last three days of April before a May 1 precipitation event extinguished it (Long 2016).

The worst wildfire season ever recorded in the Northeast came six years later in October 1947 when a prolonged drought gave way to wildfires which burned an additional 20,000 acres (8,094 ha) across New Hampshire (Long 2016), and more than 212,000 acres (96,000 ha) in Maine, of which more than 17,000 acres (7,730 ha) burned on Mount Desert Island (MDI) (Drake & Patterson 1994; Herberger & Patterson 1998). The precipitation on MDI for the entire month of October was 0.02 inches, the lowest on record (Drake & Patterson 1994). This created extreme drought conditions which, along with strong winds and a cold dry front, resulted in a catastrophic fire that burned for over a week (October 17-27, 1947) (Drake & Patterson 1994). The fire claimed three lives, burned one-third of the town of Bar Harbor destroying 237 homes and the Jackson Laboratory, and caused an estimated $23,000,000 in damage (in 1947 dollars) (Herberger & Patterson 1998). The fire burned nearly 30% of the land area of the largest island off the coast of Maine, and nearly 20% of NPS land on the island, during a period when the Keetch-Byram Drought index (KBDI) was estimated to have exceeded 500 [typical of late spring, early growing conditions (Patterson et al. 1983; http://www.wfas.net).

Predictions of future fire activity are largely in agreement and suggest that annual burned area and fire occurrence will increase in the Northeast by the end of this century, and trends will be statistically detectable by the mid-21st century (McKenzie et al. 2004; Girardin &

Mudelsee 2008; Gauthier et al. 2015). In Maine, we may expect the mid-Holocene hemlock-northern hardwood forest fire regime (1,000 yr return intervals) with less fire in the future (Marlon 2015), but the transition to less fire might be preceded by catastrophic decline of mature spruce-fir and the associated increased fire risk. Many conifer stands on portions of MDI not burned in 1947 are now approaching 150 years of age, so a major fire could occur during the next century as forests become over-mature and fuel loads increase. On land that burned in 1947, fire might be delayed 100 years, but only where deciduous trees remain (aspen/birch or longer-lived maple/beech). Under severe drought conditions (as in 1947), even deciduous forests of ANP will burn with great severity (i.e., with consumption of most soil organic matter) (Patterson personal communication). In this era of rapid climate change, understanding past and predicting future fire activity are scientific challenges that are central to the development of forest management practices and policies. The first step to the development of such practices and policies is to map current fire risk across the landscape.

Fuel is the most important single factor influencing fire ignition, spread, and behavior (Pyne et al. 1996). Fuel loading is defined as the amount of fuel present expressed quantitatively in tons per acre. Fuel loading in any given area does not necessarily mean a high-intensity fire will burn. The quality of fuels (e.g., size class, live or dead, wet or dry) available for combustion is more influential in the likelihood of fire than the quantity (Pyne et al. 1996). Thus, fire behavior is dependent on certain fuel characteristics: type, loading, and availability. Fuel arrangement is also very important in determining wildfire risk and potential fire behavior. There must be horizontal and vertical continuity for a high-intensity fire to burn (Pyne et al. 1996). Since fuel stratum relationships are extremely complex, fire managers often describe fuels by grouping vegetation communities, based upon potential fire behavior, into fuel types or fuel models (Anderson 1982; Riaño et al. 2002; Scott & Burgan 2005). The fuel type, along with topography and weather, will determine the rate of spread and intensity of fire (Rothermel 1983).

Many wildfire risk assessments map spatial complexity of fuels indirectly by assigning fuel load characteristics to vegetation types (Miller et al. 2003). This is because remotely sensed data, such as Landsat, do not penetrate the forest canopy thus limiting their utility for mapping surface fuels where tree canopies are present (Keane et al. 2000; Miller et al. 2003). As a result, most wildfire risk assessments first classify an image into vegetation categories and then assign fuel types to each category, thereby producing a vegetation-type based map (Miller et al. 2003; Falkowski et al. 2005). The greatest challenge in mapping fuels accurately is the high variability of fuels across the landscape especially within vegetation types due to localized disturbance history and succession (Miller et al. 2003; Falkowski et al 2005). Extensive field data are therefore useful to adequately portray this variability.

In this report I present methodology for mapping current wildfire risk at Acadia National Park (ANP) and the surrounding MDI, Schoodic Peninsula, and Isle au Haut, Maine landscape using an ArcGIS model that extends field data of current fuel conditions and spatially portrays wildfire risk across MDI and beyond. Wildfire risk is evaluated as a function of three primary topics: fuel load (T/acre), topography, and the wildland-urban interface. The main objective of this study was to identify areas in and nearby ANP with high wildfire risk to focus management resources in the areas of greatest need of fuel reduction. A secondary objective was to evaluate and improve upon available fuel type data (i.e., Scott & Burgan 2005) at the local level. The results of this study can be used to prioritize areas for applying fuels mitigation practices based on anticipated potential ecological benefits and the estimated management effort and cost required to reduce hazardous fuel loading at ANP.

## Methods

### *Data collection*

To establish plots representing the primary forest cover types within ANP vegetation maps interpreted from aerial photos were used to identify 23 homogeneous, approximately 5-to-10 acre (2- to 4-ha) stands (Table 56) (Patterson et al. 1983). The characteristics of these sample stands, including fire histories, are described in Patterson et al. (1983). These stands were resampled in 2016 to characterize current forest composition, structure, and fuel load (Figure 7). At least 20 plots were sampled per stand though the number depended on the size and shape of the stand. Within each stand, sampling points for standing live and dead, and dead downed woody fuels were located on a grid with points at two-chain intervals along transects running two chains apart. Locations were marked on the 1:50,000 Park USGS map, and GPS waypoint locations were archived with the National Park Service (NPS).

Table 56. Stand ID, number of plots, stand type, burn date(s) and stand initiation for sample stands in ANP (from Patterson et al. 1983).

| Stand ID | n plots | Stand Type | Burn Date(s) | Initiation Date |
|---|---|---|---|---|
| AC01 | 20 | spruce-fir | | 1850 |
| AC02 | 20 | Northern hardwoods | 1901, 1948 | 1870-1910 |
| AC03 | 20 | Northern hardwoods | 1948 | 1870-1900 |
| AC04 | 21 | spruce-fir | 1864 | 1840-1890 |
| AC05 | 20 | Northern white cedar | 1864, 1889 | 1889 |
| AC06 | 20 | Northern white cedar | | 1840-1900 |
| AC07 | 20 | red oak | 1948 | 1901 |
| AC08 | 20 | pitch pine | 1948 | 1948 |
| AC09 | 20 | birch-aspen | 1948 | 1948 |
| AC11 | 20 | mixed conifer | 1780, 1820 | 1820-1825 |
| AC12 | 20 | spruce-fir | | 1860-1870 |
| AC13 | 20 | spruce-fir | | 1860-1915 |
| AC15 | 20 | mixed conifer | | 1840-1865/1890-1910 |
| AC16 | 20 | mixed hardwood - conifer | fire scars, no date | 1840-1910 |
| AC17 | 20 | spruce-fir | fire scars, no date | 1780-1845 |
| AC18 | 20 | spruce-fir | | 1830-1840/1890-1900 |
| AC19 | 30 | mixed hardwood - conifer | 1880 | 1830-1880 |
| AC20 | 20 | spruce-fir | 1855, 1910 | 1855-1910 |
| AC22 | 20 | spruce-fir | | 1820-1860 |
| AC23 | 20 | mixed hardwood - conifer | 1880 | 1830-1920 |
| AC24 | 20 | pitch pine | 1860, 1885 | 1885 |
| AC25 | 20 | birch-aspen | 1948 | 1948 |
| AC26 | 20 | spruce-fir | unknown | unknown |

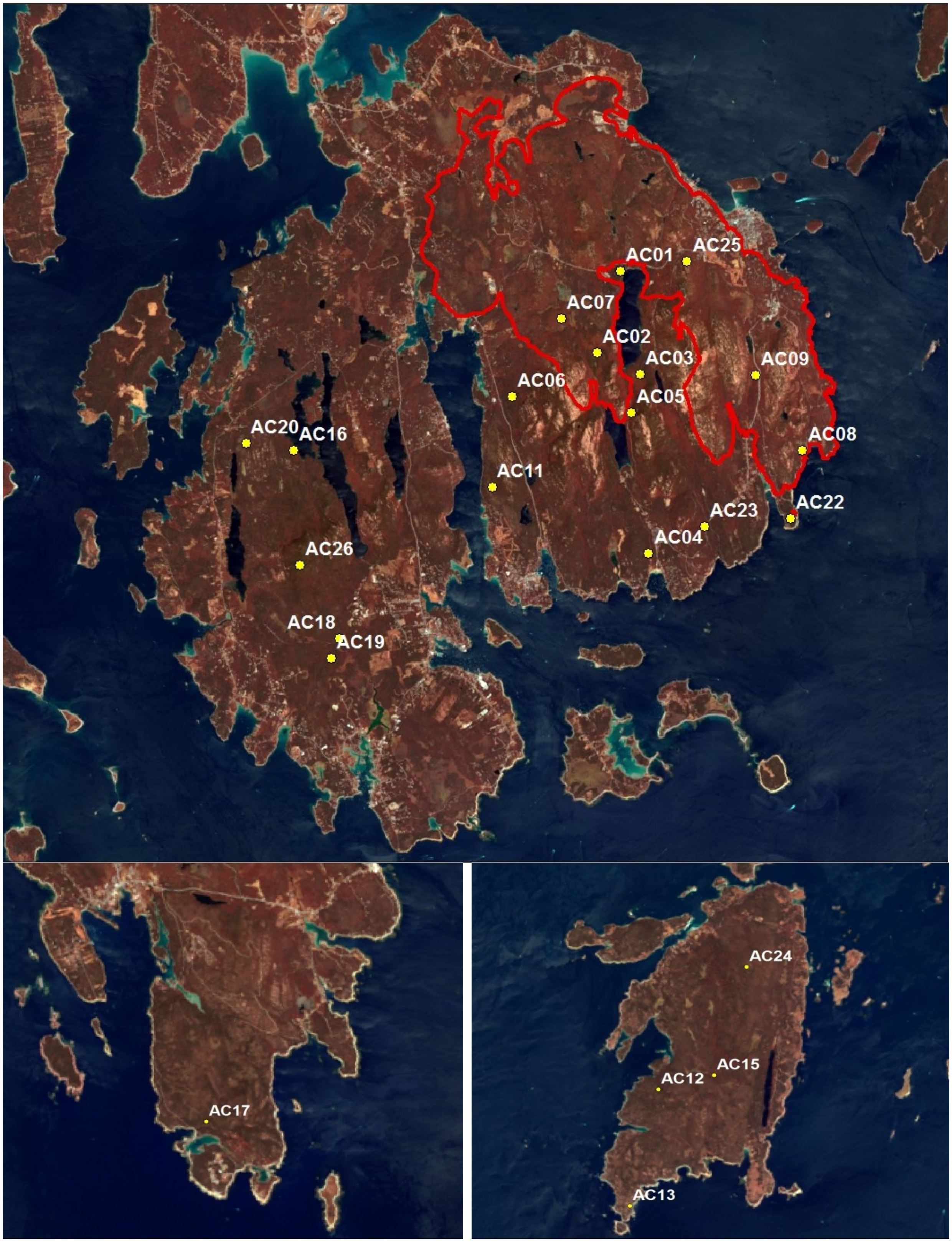



Figure 7. Map showing the sample stand locations for Mount Desert Island (top), Schoodic Peninsula (bottom left), and Isle au Haut (bottom right). The extent of the 1947 fire is outlined in red. Background: Landsat 8 satellite imagery acquired August 23, 2016.

Tree species composition was characterized at each point using a ten-factor angle gauge (Cruz-All). Diameters at 4.5 ft (1.4 m) above the ground (dbh) to the nearest 0.1 in. (0.25 cm) were recorded by species for stems ≥ 1 in. (2.54 cm) in diameter. Saplings >4.5 ft (1.4 m) tall and ≤ 1 in. (2.54 cm) dbh were tallied by species in 0.1 in. (0.254 cm) size classes in the 0.01-acre (0.004 ha) radius plots (Figure 8). The number of shrubs and tree seedlings ≤ 4.5 ft (1.4 m) tall were recorded by 1 ft (0.3 m) height classes in 0.001-acre (0.0004 ha) radius plots (Figure 8).

To sample dead, downed woody fuels, a transect was established in a randomly determined direction originating at each point (Figure 8). Along each transect fuel parameters were surveyed using the planar-intercept method (Brown 1974). Sampling transects were 50 ft (15.2 m) long for 1000-hr fuels [>3 in. (7.6 cm) in diameter], 12-ft (3.7 m) long for 100-hr fuels [1-to-3 in. (2.5-to-7.6 cm) diameter], and 6 ft (1.8 m) long for 10- and 1-hr fuels [<1 in. (2.5 cm) and 0.25 in. (0.64 cm) in diameter] (Figure 8). Fuel up to 4.5 ft (1.4 m) in height was counted if it intersected the plane and was measured at its maximum height. Duff depth (in.) and fuel height (in.) was measured at 15 ft (4.57 m) and 30 ft (9.15 m) along the fuel transect.

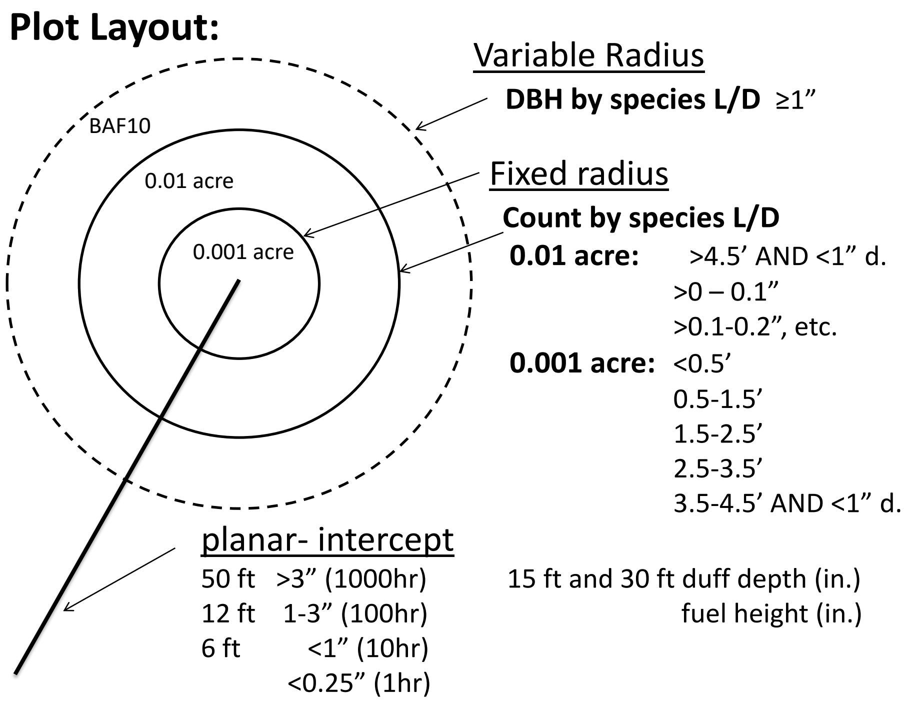



Figure 8. Plot layout for the vegetation and fuels inventory in ANP. L/D = live or dead.

*Data analysis*

Biomass estimates in tons (T per acre of individual stems for trees, saplings, and shrubs) were calculated using regression equations and specific gravity coefficients from Young et al. (1980):

[(Ln weight = A+B (Ln DBH or Ln Height)]

Where: Ln= natural logarithm to the base e, A= dry weight aboveground specific gravity coefficient from Young et al. (1980), B= dry weight aboveground specific gravity coefficient from Young et al. (1980), DBH= diameter measured in inches (in.) at 4.5’ above ground, Height= total tree height measured in feet (ft). For shrub biomass midpoints of the height classes were used (i.e., 0.25, 1, 2, 3, 4). Methods for shrub calculations has the potential to inflate biomass of individual plots with high stem counts.

Calculated biomass was converted to stand mass densities by averaging plot sums. Specific gravities for most species were obtained from the U.S. Forest Products Laboratory (Brown 1974). Specific gravity estimates for jack pine and pitch pine were found in Whittaker and Woodwell (1968), Alban (1978), and Ledig et al. (1975). Where specific gravity estimates did not exist for a species, respective general hardwood or softwood weights from Tritton and Hornbeck (1982) were used. Calculations of downed fuel loads by approximate timelag class at the plot level followed Brown (1974). Average secants were taken from Brown (1974). The constants $d^2$ (squared average-quadratic-mean diameters for slash and non-slash ground fuels) and s (average slope correction factor) are presented in Patterson et al. (1983). Outliers were not removed because they represented areas of particularly high or low biomass accumulation.

*Wildfire risk model development*

A combination of field data, remote sensing, and ESRI ArcGIS software were used to predict fuel loading spatially across ANP. The variables in the model included critical factors that affect wildland fire: fuel (i.e., biomass in T/acre from plot data as measured in the field); topography (i.e., slope and aspect) and the wildland-urban interface (Figure 9). The integration of these variables was applied in a hierarchical scheme.% influence on the model for fuel, topography, and the wildland-urban interface were determined by the Northeast Wildfire Risk Assessment Geospatial Workgroup (U.S. Forest Service 2010). The following% influence and weights were assigned for each input layer: fuels – 80% influence on the model; topography – 10% influence on the model; and wildland-urban interface – 10% influence on the model (U.S. Forest Service 2010).

The wildfire risk assessment model followed closely to that of the regional Northeast Wildfire Risk Assessment (U.S. Forest Service 2010). The objective of the model was to improve upon the Scott and Burgan fuel models (2005) which were the fuel base used by the U.S. Forest Service (2010). I did this by using on-the-ground field data to ground-truth remotely sensed fuels data. The purpose was to produce an accurate wildfire risk map of the ANP landscape. Given that the fuel base receives 80% influence on the model it is critical that the input data are accurate. The map is intended to be a general depiction of the wildfire risk for the study area.

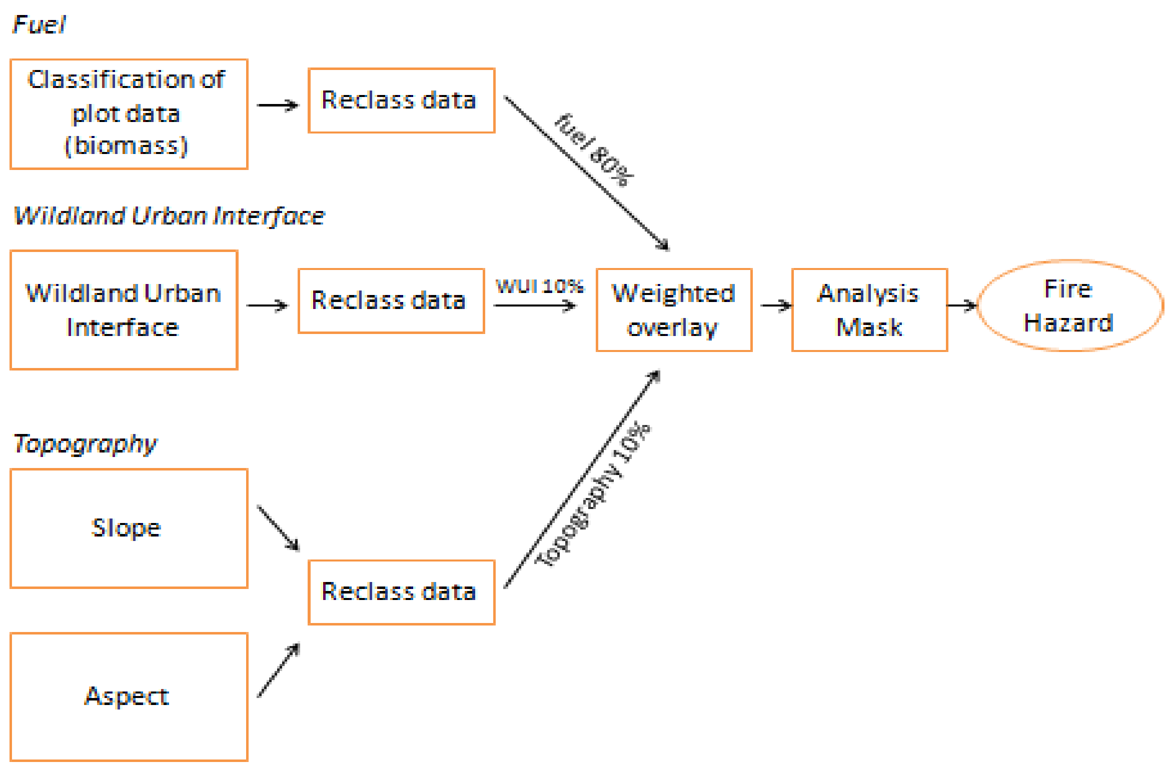



Figure 9. Acadia National Park wildfire risk assessment model (adapted from U.S. Forest Service 2010).

Fuel model selection

In order to evaluate what fuel base to use in the wildfire risk model, I first assessed whether remotely sensed numeric biomass data was an accurate predictor of biomass measured on-the-ground using visual plots and mixed effects models. I then used ANOVA to evaluate whether the best mixed-effects model was better at predicting biomass than a model without the numeric, remotely sensed biomass data. Finally, I chose the remotely sensed categorical vegetation type data that best reflected on-the-ground sampling.

To determine the best layers for accurately representing conditions on the ground, vegetative characteristics of each sampling location where field data were collected were assessed using GIS spatial data consisting of both categorical vegetative community type and numeric biomass data. Vegetative community type data were measured at each point using a) The Nature Conservancy's habitat types (Ferree & Anderson 2013), b) Scott and Burgan fuel models (2005), and c) vegetation community types at ANP (National Park Service 2003). Remotely sensed biomass data were estimated at each sampling point using a) raster values from a 2016 Landsat 8 image processed with Normalized Difference Vegetation Index (NDVI [NIR-R/NIR+R]) methodology, b) the Woods Hole NACP Aboveground National Biomass and Carbon Baseline Data V.2 estimate of biomass (NBCD) (Kellndorfer et al. 2013), and c) the regionally refined current biomass grid based on the Woods Hole NACP data produced by the University of Massachusetts' Landscape Ecology Lab (UMASS) (McGarigal et al. 2017). The Landsat image selected for NDVI processing was acquired on August 23, 2016. This image was captured at the same season as the data were collected and presented good cloud-free coverage for discriminating different vegetation communities. Values from vegetative communities and raster images were extracted in ArcMap 10.6.

To account for the spatially autocorrelated nature of the field data (i.e., plots were grouped in stands), linear mixed effects models, with stand as a random factor, were used to evaluate how well the field-derived biomass measurements can be predicted from numeric remotely sensed biomass data. Because the relationship between biomass and remotely sensed data likely varies by vegetation type, fixed predictor variables included both the categorical community data as well as the numeric, remotely sensed biomass data, and the interaction between them. Models were built for each combination of a single catagorical and single numeric layer, as well as for each layer individually. Tree data were normally distributed. Saplings, shrubs, ground fuel, and total biomass (T/acre) were right skewed and thus $Log_{10}$ transformed. Outliers were not removed because they represented areas of particularly high or low biomass accumulation. Akaike Information Criterion (AIC) was calculated for each model using the CAIC function in the cAIC4 package (Saefken & Ruegamer 2018) and was used to determine the best model for predicting biomass across the landscape. Models were built using the lmer function in the lme4 package (Bates et al. 2015) with R statistical software (R Core Team 2018). $R^2$ values were calculated for each model using the r.squaredGLMM function in the MuMIn package (Barton 2019) with R statistical software (R Core Team 2018). ANOVAs were used to determine whether the inclusion of the numeric remotely sensed biomass data performed better than the categorical data alone. Assumptions of the best model were checked by visually assessing the residuals. Because biomass could not be adequately predicted using remotely sensed numeric biomass data, I instead chose to include only vegetation class in our flammability model.

Fuel base selection

The best fit categorical vegetation type data were selected by looking at the correct classification rate as identified on-the-ground. Using species flammability categories (Appendix 1), the vegetation data were given weights associated with wildfire risk whereby spruce/fir/cedar were assigned a weight of 5 (very high risk), mixed conifer-deciduous were assigned a weight of 4 (high risk), red/white/jack/pitch pine were assigned a weight of 3 (moderate risk), birch/aspen/northern hardwood were assigned a weight of 2 (low risk), and health/grass/shrub were assigned a weight of 1(very low risk) in the weighted overlay. The fire risk weights assigned to the fuels were determined by basic fire ecology knowledge (e.g., needle and branch structure, bark characteristics, pitch (Brown & Davis 1973, Pyne et al. 1996), and species specific fire ecology and fire effects literature published by the USDA Fire Effect Information System (FEIS) (www.feis-crs.org) and Patterson (personal communication). FEIS synthesizes fire ecology and fire regimes in the United States and includes all species of ANP, and in some cases such as red spruce, discuss ANP specifically. The wildfire risk scores were visually compared with the total average biomass by stand as measured in the field as a means of assessing the relationship between the flammability categories and on-the-ground biomass.

Topography

Topography has a large impact on fire behavior, of which slope is considered the critical factor. The steeper the slope the faster the fire will burn (Pyne et al. 1996). South and southwest slopes are the most critical in terms of start and spread of wildfires (Chuvieco & Congalton

1989; Pyne et al. 1996). For these reasons, two LANDFIRE data layers were modified and combined in the model: slope and aspect. The slope data layer identified only those pixels that represented a slope greater than 20% (Appendix 2). The aspect data layer identified only those pixels which represented aspect values between 135 and 315 degrees (Appendix 3). The data used in the model thus contained only those pixels that had both a slope greater than 20% and an aspect between 135 and 315 degrees (U.S. Forest Service 2010) (Appendix 4).

Wildland Urban Interface

The wildland urban interface (i.e., developed areas, urban, residential, mixed urban, built-up, roads), and open water as identified by the National Park Service (2003) received a weight of zero and were masked in the model to eliminate non-burnable covers from analysis to produce the final wildfire risk assessment. These data were combined using a weighted overlay and masking technique to develop the output wildfire risk assessment (Figure 9).

## Results

### *Fuel model*

Visual plot and mixed effects models evaluation of the best available regional remotely sensed numeric biomass data (i.e., NBCD, NDVI, UMASS) showed weak correlations between NDVI, UMASS, NBCD and the plot data used in the mixed models (Table 57). All models were weak based on $R^2$ ranging from 0 to 0.04 (Table 57).

Mixed effects model 16 and model 22, both of which used the NPS categorical data, model 16 with the NBCD data and model 22 without the NBCD data, had the lowest AIC values (Table 57). Assumptions of these models were visually assessed by plotting the residuals. With the exception of six outliers, which were intentionally left in the dataset because they represent areas of particularly high fuel loading, residuals were normally distributed. Using ANOVA to evaluate whether the inclusion of the numeric remotely sensed biomass data performed better than the categorical data alone, I found that there is no significant difference between model 16 (NBCD with NPS categorical vegetation data) and model 22 (NPS categorical vegetation data alone) ($X^2$= 4.05,df=2, p=0.13).

*Fuel base*

I intended to predict biomass from the plot data on a 30m cell grid from the best available remotely sensed data; however, due to the findings of the mixed effects models, vegetation categories alone were used to extrapolate the horizontal spatial arrangement of biomass in the study area. The best fit categorical vegetation type data were selected by looking at the correct classification rate as identified on-the-ground at the sample stands. The TNC (2013) data were 60% accurate, Scott and Burgan (2005) data were 52% accurate, and NPS (2003) data were 70% accurate. Thus, I chose to use the NPS vegetation layer as the fuel base of the fire risk assessment model.

*Wildfire risk model*

The input GIS base layers that were used in the weighted overlay and analysis mask are shown in Appendices 1 through 4. I used vegetation flammability categories, which was a reclassification of the vegetation community types at ANP (National Park Service 2003) to portray fuels in the final wildfire risk assessment because of the overall accuracy of the NPS vegetation data and its correlation with successional stage and subsequent relationship to total fuel loads.

The wildfire risk for the entire study area with plot data symbolized by fuel loads (average total T/acre) as a means of visually assessing the accuracy of the final fire risk map is shown in Figure 12. Across all stands sampled, total dead downed woody fuel ranged from 1.32 to 28.04 T/acre. Standing live trees ranged from 12.9 to 78.1 T/acre. Standing live saplings ranged from 0.04 to 0.62 T/acre. Standing live shrubs ranged from no measurable to 16.9 T/acre. Standing dead trees ranged from 0.10 to 17.6 T/acre. Standing dead saplings ranged from 0 to 0.16 T/acre. Standing dead shrubs ranged from 0 to 0.3 T/acre (Table 58). The values presented in Table 58 are the values associated with the total fuel loads by stand shown in Figure 12. The average duff depth for all sampled stands ranged from 1.24 to 7.70 inches. Average fuel height ranged from 0.84 to 9.97 inches. Average depth to bedrock ranged from 2.24 to 17.4 inches. Average% canopy closure ranged from 0.42 to 0.96% closure (Table 59).

Based on the model, outside the 1947 fire boundary the wildfire risk for MDI is generally very high to high with small pockets of moderate to low risk. Within the fire boundary, the wildfire risk is primarily moderate to low with small areas of high to very high risk due primarily due to topography (Figure 10). The wildfire risk for Schoodic Peninsula and Isle au Haut is very high to high with small pockets of moderate to low risk (Figure 11).

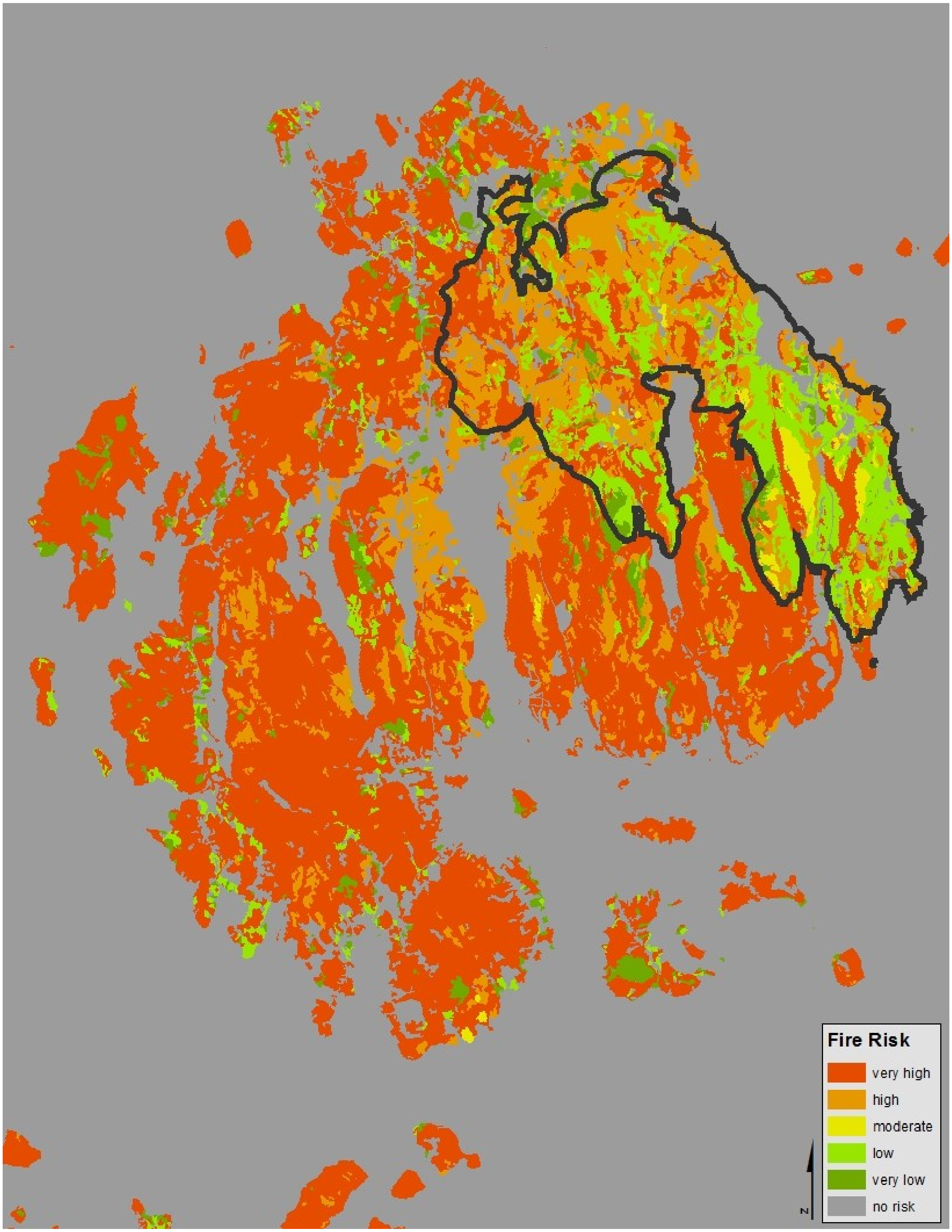


Figure 10. Wildfire risk for Mount Desert Island, Maine. Wildfire risk ranges from very high (red) to moderate (yellow) to very low (green). Stand data were collected in summer 2016.

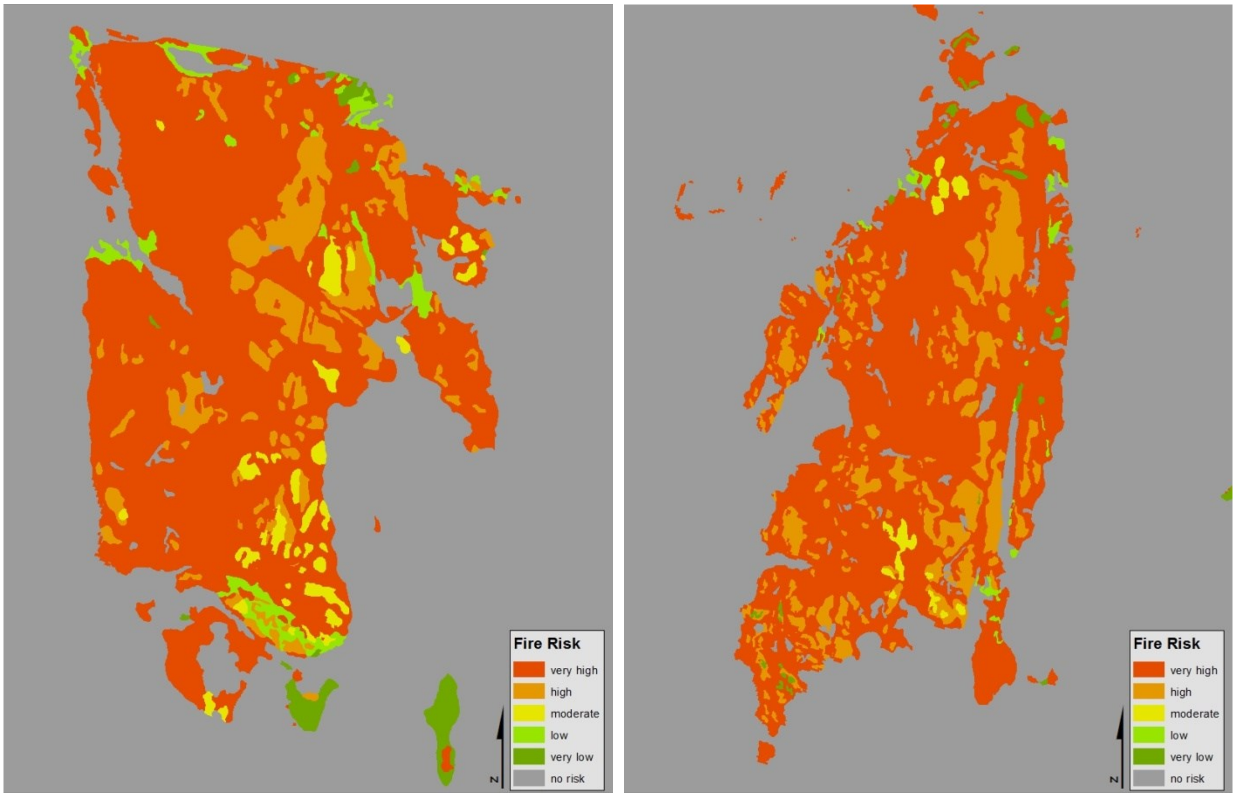


Figure 11. Wildfire risk for Schoodic Peninsula (left) and Isle au Haut (right), Maine. Wildfire risk ranges from very high (red) to moderate (yellow) to very low (green). Stand data were collected in summer 2016.

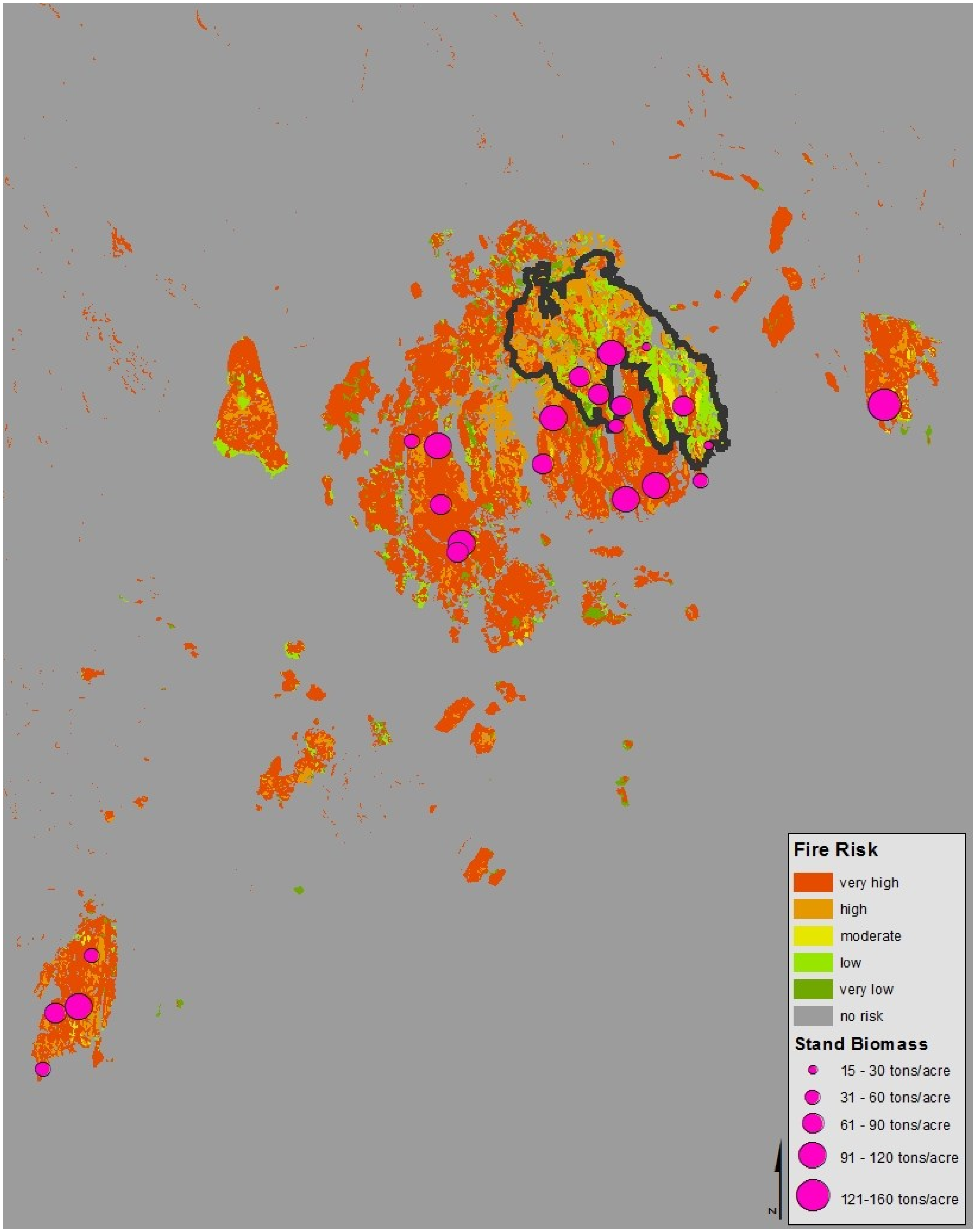


Figure 12. Wildfire risk for Mount Desert Island, Isle au Haut and Schoodic Peninsula with plot data symbolized by fuel loads (average total T/acre). Wildfire risk ranges from very high (red) to moderate (yellow) to very low (green). Stand data were collected in summer 2016.

Table 57. Mixed effects model outputs. Biomass is total biomass (T/acre) as measured in the field. Vegetation community types at ANP (NPS), The Nature Conservancy's habitat types (TNC), and Scott and Burgan fuel models (SB) were the input data layers for vegetation types. The biomass data by classified vegetation types were evaluated with the associated raster values from a Normalized Difference Vegetation Index (NDVI), the Woods Hole NACP Aboveground National Biomass and Carbon Baseline Data V.2 estimate of biomass (NBCD), and the regionally refined current biomass grid produced by the University of Massachusetts' Landscape Ecology Lab (UMASS). Df is degrees of freedom. Akaike Information Criterion (AIC) was calculated using the CAIC function in the cAIC4 package (Saefken & Ruegamer 2018). $R^2$ values were calculated for each model using the r.squaredGLMM function in the MuMIn package (Barton 2019) with R statistical software (R Core Team 2018).

| Model | Mixed Effects Models (input data) | df | classified vegetation types | AIC | ΔAIC | $R^2$ |
|---|---|---|---|---|---|---|
| 16 | biomass= NBCD*NPS reclassified 2 | 23.62 | conifer, mixed | 4529 | 0 | 0.04 |
| 22 | biomass= NPS reclassified 2 | 21.74 | conifer, mixed | 4529.8 | 0.72 | 0.03 |
| 4 | biomass= NDVI*NPS reclassified 2 | 23.30 | conifer, mixed | 4530 | 0.99 | 0.04 |
| 21 | biomass= NPS reclassified | 22.41 | conifer, mixed, deciduous | 4531.1 | 2.11 | 0.03 |
| 15 | biomass= NBCD*NPS reclassified | 25.26 | conifer, mixed, deciduous | 4532.3 | 3.31 | 0.04 |
| 3 | biomass= NDVI*NPS reclassified | 25.07 | conifer, mixed, deciduous | 4532.5 | 3.45 | 0.04 |
| 10 | biomass= UMASS*NPS reclassified 2 | 23.68 | conifer, mixed | 4533.6 | 4.58 | 0.03 |
| 27 | biomass= NBCD | 21.79 | NA | 4533.7 | 4.67 | 0.01 |
| 25 | biomass= NDVI | 21.66 | NA | 4535.4 | 6.35 | 0.00 |
| 26 | biomass= UMASS | 21.93 | NA | 4536.3 | 7.26 | 0.00 |
| 19 | biomass= TNC reclassified | 22.5 | conifer, mixed, deciduous | 4536.7 | 7.7 | 0.01 |
| 9 | biomass= UMASS*NPS reclassified | 25.35 | conifer, mixed, deciduous | 4536.9 | 7.84 | 0.03 |
| 23 | biomass= NPS flammability reclassified | 22.06 | high, med., low (relative to species burnability) | 4537.9 | 8.87 | 0.02 |
| 13 | biomass= NBCD*TNC reclassified | 25.21 | conifer, mixed, deciduous | 4539.4 | 10.4 | 0.02 |
| 24 | biomasss= flammability reclassified 2 | 22.85 | 1) spruce/fir/cedar, 2) mixed conifer-deciduous, 3)pines, 4)birch-aspen/northern hardwoods, 5)health, grass, shrub (Appendix 1) | 4539.5 | 10.5 | 0.02 |
| 17 | biomass= NBCD*flammability reclassified | 24.76 | high, med., low (relative to species burnability) | 4540 | 11 | 0.02 |
| 1 | biomass= NDVI*TNC reclassified | 25.02 | conifer, mixed, deciduous | 4541.6 | 12.6 | 0.02 |
| 5 | biomass= NDVI*NPS flammability reclassified | 24.63 | high, med., low (relative to species burnability) | 4542 | 13 | 0.02 |
| 7 | biomass= UMASS*TNC reclassified | 25.41 | conifer, mixed, deciduous | 4542.3 | 13.2 | 0.02 |
| 18 | biomass= NBCD*flammability reclassified 2 | 26.53 | 1) spruce/fir/cedar, 2) mixed conifer-deciduous, 3)pines, 4)birch-aspen/northern hardwoods, 5)health, grass, shrub (Appendix 1) | 4543.3 | 14.3 | 0.02 |
| 11 | biomass= UMASS*flammability reclassified | 24.98 | high, med., low (relative to species burnability) | 4543.3 | 14.3 | 0.02 |
| 6 | biomas= NDVI*NPS flammability reclassified 2 | 26.38 | 1) spruce/fir/cedar, 2) mixed conifer-deciduous, 3)pines, 4)birch-aspen/northern hardwoods, 5)health, grass, shrub (Appendix 1) | 4545.5 | 16.4 | 0.02 |
| 12 | biomass= UMASS*flammability reclassified 2 | 26.77 | 1) spruce/fir/cedar, 2) mixed conifer-deciduous, 3)pines, 4)birch-aspen/northern hardwoods, 5)health, grass, shrub (Appendix 1) | 4547 | 17.9 | 0.02 |
| 20 | biomass= SB | 29.63 | original fuel models | 4551.9 | 22.9 | 0.00 |
| 14 | biomass= NBCD*SB | 39.25 | original fuel models | 4567 | 37.9 | 0.02 |
| 2 | biomass= NDVI*SB | 39.02 | original fuel models | 4568.4 | 39.3 | 0.01 |
| 8 | biomass= UMASS*SB | 39.43 | original fuel models | 4571.4 | 42.4 | 0.00 |

Table 58. Fuel loading in 2016 for dead, down woody material and standing live and dead material (T/acre).
* indicates stands burned in 1947.

| | | | Downded Woody Fuel (tons/acre) | | | | | | Standing Woody Fuel (tons/acre) | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Fuel | Model | | | | | Sound | Rotten | | Live | | | | | | Dead | | | | | |
| NFDRS | FBPS | Stand | 1-hr | 10-hr | 100-hr | 1000-hr | 1000-hr | Total | Trees | STDEV | Saplings | STDEV | Shrubs | STDEV | Trees | STDEV | Saplings | STDEV | Shrubs | STDEV |
| H | 8 | AC01 | 0.42 | 0.13 | 1.97 | 8.59 | 2.00 | 13.11 | 77.3 | 25.71 | 0.12 | 0.15 | 0.6 | 1.8 | 9.6 | 8.11 | 0.02 | 0.04 | 0.0 | 0.0 |
| R/E | 8/9 | *AC02 | 0.35 | 0.15 | 1.76 | 0.78 | 1.26 | 4.31 | 66 | 20.61 | 0.06 | 0.08 | 0.5 | 1.2 | 9.5 | 7.55 | 0.03 | 0.07 | 0.0 | 0.0 |
| R/E | 8/9 | AC03 | 0.37 | 0.11 | 1.16 | 3.32 | 3.03 | 7.99 | 73.3 | 22.92 | 0.10 | 0.07 | 0.0 | 0.1 | 6.1 | 6.79 | 0.01 | 0.03 | 0.0 | 0.0 |
| H | 8 | AC04 | 0.61 | 0.14 | 0.94 | 2.80 | 4.01 | 8.51 | 78.1 | 17.02 | 0.18 | 0.27 | 0.6 | 1.4 | 7.9 | 8.67 | 0.15 | 0.31 | 0.0 | 0.0 |
| H | 8 | AC05 | 0.41 | 0.10 | 1.41 | 2.88 | 1.79 | 6.59 | 43 | 15.48 | 0.10 | 0.12 | 0.6 | 2.3 | 6.7 | 4.71 | 0.01 | 0.03 | 0.0 | 0.0 |
| H | 8 | AC06 | 0.36 | 0.11 | 1.40 | 12.26 | 1.65 | 15.78 | 71.9 | 28.90 | 0.20 | 0.18 | 1.1 | 1.8 | 12.8 | 7.80 | 0.01 | 0.03 | 0.0 | 0.1 |
| R/E | 8/9 | *AC07 | 0.30 | 0.12 | 1.70 | 0.08 | 1.38 | 3.59 | 61.9 | 16.38 | 0.13 | 0.21 | 0.1 | 0.1 | 5.5 | 5.04 | 0.01 | 0.01 | 0.0 | 0.0 |
| Q | 6 | *AC08 | 0.52 | 0.08 | 1.17 | 0.05 | 2.22 | 4.05 | 21.6 | 13.23 | 0.04 | 0.06 | 0.9 | 1.1 | 0.3 | 0.78 | 0.03 | 0.05 | 0.0 | 0.0 |
| R/E | 8/9 | *AC09 | 0.28 | 0.10 | 1.34 | 0.98 | 1.05 | 3.76 | 72.6 | 23.80 | 0.10 | 0.17 | 0.0 | 0.1 | 2.8 | 3.92 | 0.02 | 0.05 | 0.0 | 0.0 |
| H | 8 | AC11 | 0.75 | 0.14 | 1.20 | 3.56 | 3.38 | 9.04 | 54.1 | 18.21 | 0.06 | 0.09 | 0.4 | 0.7 | 17.6 | 9.17 | 0.03 | 0.06 | 0.0 | 0.1 |
| H | 8 | AC12 | 1.21 | 0.17 | 1.40 | 4.24 | 5.79 | 12.81 | 45.7 | 19.05 | 0.14 | 0.21 | 2.1 | 5.0 | 6.1 | 5.89 | 0.01 | 0.04 | 0.1 | 0.3 |
| H | 8 | AC13 | 1.90 | 0.43 | 7.67 | 12.21 | 5.82 | 28.04 | 18.4 | 17.97 | 0.35 | 0.42 | 0.6 | 0.9 | 8.3 | 8.55 | 0.01 | 0.02 | 0.0 | 0.1 |
| H | 8 | AC15 | 0.74 | 0.16 | 2.27 | 4.30 | 4.82 | 12.29 | 65 | 24.28 | 0.12 | 0.20 | 8.7 | 22.1 | 13 | 10.72 | 0.00 | 0.01 | 0.0 | 0.0 |
| R/E | 8/9 | AC16 | 0.65 | 0.16 | 1.58 | 4.20 | 4.95 | 11.55 | 73 | 22.44 | 0.13 | 0.12 | 0.6 | 0.7 | 16.1 | 9.78 | 0.01 | 0.03 | 0.0 | 0.1 |
| H | 8 | AC17 | 0.85 | 0.12 | 1.61 | 9.34 | 8.01 | 19.92 | 76 | 20.77 | 0.48 | 0.60 | 13.0 | 34.3 | 15.4 | 11.46 | 0.07 | 0.13 | 0.0 | 0.1 |
| H | 8 | AC18 | 0.83 | 0.16 | 1.70 | 8.19 | 2.03 | 12.91 | 59.86 | 19.51 | 0.32 | 0.41 | 16.9 | 26.3 | 7.66 | 5.85 | 0.03 | 0.07 | 0.0 | 0.0 |
| R/E | 8/9 | AC19 | 0.69 | 0.10 | 1.30 | 3.80 | 1.40 | 7.30 | 65.71 | 17.80 | 0.36 | 0.64 | 2.9 | 5.0 | 9.50 | 9.21 | 0.08 | 0.15 | 0.0 | 0.0 |
| Q | 6 | AC20 | 0.67 | 0.09 | 0.18 | 1.41 | 0.06 | 2.41 | 38.6 | 12.15 | 0.10 | 0.16 | 0.5 | 0.5 | 4.8 | 6.59 | 0.00 | 0.00 | 0.0 | 0.0 |
| G | 10 | AC22 | 1.13 | 0.17 | 1.77 | 1.66 | 3.96 | 8.68 | 35.6 | 24.24 | 0.25 | 0.48 | 0.8 | 2.2 | 3.6 | 4.92 | 0.16 | 0.27 | 0.2 | 0.9 |
| R/E | 8/9 | AC23 | 0.55 | 0.12 | 2.26 | 3.70 | 5.82 | 12.45 | 73.8 | 16.74 | 0.10 | 0.11 | 0.1 | 0.2 | 12.5 | 10.21 | 0.01 | 0.03 | 0.0 | 0.0 |
| Q | 6 | AC24 | 0.53 | 0.13 | 0.61 | 0.05 | 0.00 | 1.32 | 35.7 | 13.02 | 0.05 | 0.05 | 3.7 | 2.4 | 1.8 | 2.02 | 0.00 | 0.00 | 0.0 | 0.0 |
| R/E | 8/9 | *AC25 | 0.23 | 0.06 | 0.94 | 0.05 | 0.25 | 1.53 | 12.9 | 9.97 | 0.06 | 0.14 | 0.4 | 0.5 | 0.1 | 0.66 | 0.01 | 0.02 | 0.0 | 0.0 |
| G | 10 | AC26 | 1.01 | 0.17 | 2.68 | 5.63 | 4.43 | 13.92 | 48.6 | 24.21 | 0.62 | 0.68 | 1.0 | 1.2 | 16.5 | 9.82 | 0.01 | 0.02 | 0.3 | 1.1 |

Table 59. Average duff depth, fuel height, depth to bedrock, and% canopy closure for stands sampled in 2016. * indicates stands burned in 1947.

| | Average | | Average | | Average | | Average | |
|---|---|---|---|---|---|---|---|---|
| Stand | Duff Depth (in.) | STDEV | Fuel Height (in.) | STDEV | Depth to Bedrock (in.) | STDEV | Canopy Density (%) | STDEV |
| AC01 | 2.51 | 1.99 | 3.68 | 7.11 | 9.20 | 3.25 | 0.92 | 0.14 |
| *AC02 | 3.39 | 1.70 | 1.49 | 1.07 | 17.40 | 5.30 | 0.93 | 0.02 |
| AC03 | 2.19 | 1.41 | 1.48 | 1.47 | 12.15 | 5.34 | 0.96 | 0.01 |
| AC04 | 4.52 | 1.55 | 1.10 | 0.94 | 10.07 | 3.09 | 0.88 | 0.04 |
| AC05 | 1.24 | 1.31 | 2.21 | 3.22 | 2.24 | 2.41 | 0.87 | 0.10 |
| AC06 | 3.66 | 1.63 | 5.94 | 10.52 | 11.01 | 3.82 | 0.84 | 0.13 |
| *AC07 | 3.05 | 1.03 | 1.11 | 0.76 | 7.91 | 3.31 | 0.86 | 0.05 |
| *AC08 | 3.61 | 1.48 | 0.84 | 0.81 | 5.44 | 2.81 | 0.42 | 0.33 |
| *AC09 | 1.97 | 0.99 | 1.66 | 3.61 | 11.06 | 3.70 | 0.92 | 0.04 |
| AC11 | 4.95 | 2.15 | 2.54 | 2.26 | 8.69 | 3.64 | 0.83 | 0.09 |
| AC12 | 7.70 | 2.44 | 4.31 | 7.46 | 8.64 | 2.67 | 0.74 | 0.16 |
| AC13 | 4.37 | 2.87 | 9.97 | 10.40 | 8.07 | 3.39 | 0.53 | 0.30 |
| AC15 | 5.92 | 2.49 | 5.18 | 6.81 | 7.36 | 3.43 | 0.81 | 0.09 |
| AC16 | 4.88 | 1.04 | 1.04 | 1.04 | 8.76 | 2.93 | 0.92 | 0.04 |
| AC17 | 7.32 | 4.07 | 1.70 | 3.29 | 10.01 | 5.98 | 0.82 | 0.10 |
| AC18 | 6.59 | 1.72 | 3.57 | 3.50 | 10.23 | 3.60 | 0.85 | 0.08 |
| AC19 | 5.48 | 1.91 | 2.51 | 3.89 | 12.57 | 4.56 | 0.89 | 0.04 |
| AC20 | 4.42 | 2.04 | 4.18 | 7.15 | 8.66 | 3.88 | 0.74 | 0.19 |
| AC22 | 4.59 | 2.45 | 2.84 | 4.57 | 6.66 | 3.30 | 0.67 | 0.30 |
| AC23 | 2.46 | 1.51 | 4.62 | 5.13 | 12.73 | 6.63 | 0.88 | 0.06 |
| AC24 | 5.06 | 2.34 | 2.84 | 2.46 | 5.39 | 2.23 | 0.60 | 0.19 |
| *AC25 | 3.49 | 2.34 | 1.15 | 1.29 | 5.23 | 3.47 | 0.42 | 0.35 |
| AC26 | 5.32 | 2.04 | 2.89 | 5.49 | 10.84 | 3.26 | 0.81 | 0.16 |

## Discussion

The purpose of this research was to identify areas in and near ANP with high wildfire risk so managers can focus resources in the areas of greatest need of fuels management. The methodology was moderately successful in mapping the delineation of fuel loads across the study area. This model met the goal to improve upon the Scott and Burgan (2005) fuel model layer (Table 57) which was used in the Northeast Wildfire Risk Assessment (U.S. Forest Service 2010). The greatest limitation to the model is that the best available remotely sensed data do not penetrate the canopy of the forest and so only measure above-ground live biomass. Assessment of surface fuels under tree canopies with remotely sensed data has proven to be difficult (Miller et al. 2003). My intent was to map the total biomass accounting for vertical structure of both live and dead fuels. Given the limitations of available remotely sensed data, I used existing vegetation type data as a surrogate for fuel loads. However, vegetation type alone is not an optimized proxy for fuel loading. The distribution and accumulation of fuels is highly variable (Brown & Bevins 1986) and depends upon vegetation type, disturbance regime or stand history, structural stage classes, soils, and/or moisture availability (Keane et al. 2000, 2001; Brandis & Jacobson 2003; Miller et al. 2003). Variability of fuels within a vegetation type can be greater than variation between types (Miller et al. 2003). Further, if vegetation types are misclassified in the underlying data, then errors will carry through to the final fire risk map.

Active remote sensing platforms such as light detection and ranging (lidar) have been used to predict fuel loads and generally achieve stronger relationships to vegetation than with other forms of high-resolution satellite imagery (Falkowski et al. 2005). Higher accuracies using lidar data are to be expected given their higher information content. Much of lidar energy penetrates the forest canopy. Thus, lidar systems can record information starting from the top of the canopy, through the canopy, all the way to the ground; thus, their value for understanding vertical structure of the forest fuels. Future fuels mapping efforts at ANP can be improved by integrating multispectral satellite data with hyperspectral data and/or lidar data. However, availability of such data sets is limited, and the costs of acquiring and analyzing such data is impractical for most land managers (Falkowski et al. 2005).

I recognize that the incidence of wildfire due to human-caused ignition while recreating is a missing component of this effort. Often the occurrence and location of wildfire ignitions at ANP reflects the activities of humans who cause fires and thus increase wildfire risk. Inclusion of historical fire occurrence data would enhance the human-ignition potential in the model. Although human behavior cannot accurately be modeled, fires resulting from malicious intentions or accidents can occur throughout the region when conditions are suitable. Education and preventative measures can mitigate threats to personal property, and regardless of risk levels.

Due to variation in the reliability of the input data, the scale at which this analysis was conducted, and the variability of fuels within the study area, conclusions based on the findings of this wildfire risk assessment should be applied only after ground-truthing a prescribed fire burn unit in the field. However, there is a consensus by ANP land managers familiar with local variation that this map generally depicts the wildfire risk at ANP. Although risk and risk mapping can provide insight into the potential long-term distribution of wildfires, managers and property owners need to continually assess local site conditions, which were beyond the scope of this study.

**Conclusion**

Wildfires pose a threat to property and resources within and surround ANP. Managers are faced with important management decisions about how to best allocate the limited resources available. Wildfire risk mapping provides supplemental information on which to base the limited resource allocations. This study aims to provide a tool to spatially project the risk of wildfires across ANP and surrounding landscape. The projected increase in population, pressure for land use change, and declining State and Federal budgets will result in more complex fire suppression strategies. This coupled with the uncertainties of climate change may significantly increase wildfire occurrence at ANP. Until more resources and technology are available the fire risk assessment map provided here is a good starting point for prioritizing fuel reduction options.

**Acknowledgements**

This project was a cooperative effort between the USDI National Park Service at Acadia National Park and Antioch University New England. I thank the National Park Service - Wildland Fire Research Reserve Fund - for funding this research. I also thank all who helped with the successful completion of this project including Kate Miller of the NPS Northeast Temperate Inventory and Monitoring Network, Shawn Fraver of University of Maine, and Nick Fisichelli of Schoodic Institute. I would especially like to acknowledge the superior efforts of Lisabeth Willey of Antioch University New England for her help with data analysis.

## Literature Cited

Abella SR, Fornwalt PJ. 2015. Ten years of vegetation assembly after a North American mega fire. Global Change Biology **21**:789–802.

Abrahamson, I. 2018. Fire Effects Information System (FEIS). Available from https://www.feis-crs.org/feis/.

Adams, MA. 2013. Mega-fires, tipping points and ecosystem services: Managing forests and woodlands in an uncertain future. Forest Ecology and Management **294**:250–261.

Ahlgren, IF. 1974. The effects of fire on soil organisms. Pages 47-72 *in* Kozlowski TT, Ahlgren CE, editors. Fire and Ecosystems. Academic Press, New York.

Alban, DH. 1978. Growth of adjacent red and jack pine plantations in the Lake States. Journal of Forestry **76**:418–421.

Allen CD, et al. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management **259**:660–684.

Amraoui MM, Liberato LR, Calado TJ, DaCamara CC, Coelho LP, Trigo RM, Gouveia CM. 2013. Fire activity over Mediterranean Europe based on information from Meteosat-8. Forest Ecology and Management **294**:62–75.

Anderson, HE. 1982. Aids to determining fuel models for estimating fire behavior. USDA Forest Service General Technical Report INT-122. 22 p.

Appenzeller, T. 2015. The new North: Stoked by climate change, fire and insects are remaking the planet's vast boreal forest. Science **6250**:806–806.

Attiwill, PM. 1994. The disturbance of forest ecosystems: The ecological basis for conservative management. Forest Ecology and Management **63**:247-300.

Ayres, MP, Lombardero MJ. 2000. Assessing the consequences of global change for forest disturbance from herbivores and pathogens. Science of the Total Environment **262**:263–286.

Baker, W. 1995. Long-term response of disturbance landscapes to human intervention and global change. Landscape Ecology **10**: 143-159.

Bale JS, et al. 2002. Herbivory in global climate change research: Direct effects of rising temperature on insect herbivores. Global Change Biology **8**:1–16.

Balshi MS, McGuire AD, Duffy P, Flannigan M, Walsh J, Melillo J. 2009. Assessing the response of area burned to changing climate in western boreal North America using a

Multivariate Adaptive Regression Splines (MARS) approach. Global Change Biology **15**:578–600.

Barnes BV, Zak DR, Denton SR, Spurr SH. 1998. Forest ecology. John Wiley, New York.

Barnicle, KS. 1984. Impacts of the 1947 Bar Harbor Fire and post-fire salvage operations on the vegetation of Acadia National Park, Mount Desert Island, Maine. Department of Forestry and Wildlife. University of Massachusetts.

Barton, K. 2019. MuMIn: Multi-Model Inference. R package version 1.43.6. https://CRAN.R-project.org/package=MuMIn

Barton AM, Keeton WS. 2018. Ecology and recovery of Eastern old-growth forests. Island Press, Washington, DC.

Barton AM, White AS, Cogbill CV. 2012. The changing nature of the Maine woods. Rhodora **116**:359–362.

Bates D, Maechler M, Bolker B, Walker, S. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software, **67**, 1-48.

Beckage B, Ellingwood C. 2008. Fire feedbacks with vegetation and alternative stable states. COMPLEX SYSTEMS -CHAMPAIGN- **18**:159–173.

Bergeron, Y. 2000. Species and stand dynamics in the mixed woods of Quebec's southern boreal forest. Ecology **81**:1500–1516.

Bergeron Y, Dansereau PR. 1993. Predicting the composition of Canadian southern boreal forest in different fire cycles. Journal of Vegetation Science **4**:827–832.

Bergeron Y, Gauthier S, Kafka V, Lefort P, Lesieur D. 2001. Natural fire frequency for the eastern Canadian boreal forest: consequences for sustainable forestry. Canadian Journal of Forest Research **31**:384–391.

Bergeron YP, Richard JH, Carcaillet C, Gauthier S, Flannigan M, Prairie YT. 1998. Variability in fire frequency and forest composition in Canada's southeastern boreal forest: A challenge for sustainable forest management. Available from http://www.ecologyandsociety.org/vol2/iss2/art6/.

Bessie WC, Johnson EA. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology **76**:747–762.

Birdsey RA, Heath LS. 1995. Carbon changes in U.S. forests. United States Department of Agriculture Forest Service General Technical Report RM: 56–70.

Boiffin J, Munson AD. 2013. Three large fire years threaten resilience of closed crown black spruce forests in eastern Canada. Ecosphere **4**:1–20.

Bonan, GB. 2008. Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. Science **320**:1444–9.

Brandis K, Jacobson C. 2003. Estimation of vegetative fuel loads using Landsat TM imagery in New South Wales, Australia. International Journal of Wildland Fire **12** DOI: 10.1071/WF03032.

Brose PH, Dey DC, Guyette RP, Marschall JM, Stambaugh MC. 2013. The influences of drought and humans on the fire regimes of northern Pennsylvania, USA. Canadian Journal of Forest Research **43**:757–767.

Brotak EA, Reifsnyder WE. 1997. Predicting major wildland fire occurrence. Fire Management Notes **38**:5-8.

Brown JK, Bevins CD. Intermountain Research Station (Ogden U, United States. Forest Service. 1986. Surface fuel loadings and predicted fire behavior for vegetation types in the northern Rocky Mountains. U.S. Dept. of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT.

Brown AA, Davis KP. 1973. Forest fire control and use, 2nd Edition. McGraw-Hill Book Co., New York.

Brown, JK. 1974. Handbook for inventorying downed woody material. USDA Forest Service General Technical Report. INT-16. 26 p.

Brown CD, Johnstone JF. 2012. Once burned, twice shy: Repeat fires reduce seed availability and alter substrate constraints on *Picea mariana* saplings. Forest Ecology and Management **266**:34–41.

Buma B, Wessman C. 2011. Disturbance interactions can impact resilience mechanisms of forests. Ecosphere **2**:64.

Butler, J. 2014. Wildfire loose: the week Maine burned. Down East Books, Camden, Maine.

Campbell JL, et al. 2009. Consequences of climate change for biogeochemical cycling in forests of northeastern North America. Canadian Journal of Forest Research **39**:264–284.

Carcaillet C, Ali AA, Richard PJH, Frchette B, Bergeron Y. 2010. Resilience of the boreal forest in response to Holocene fire-frequency changes assessed by pollen diversity and population dynamics. International Journal of Wildland Fire **19**:1026–1039.

Catanzaro P, D'Amato A. 2019. Forest carbon: An essential natural solution for climate change. University of Massachusetts.

Cavard X, Bergeron Y, Paré D, Nilsson MC, Wardle DA. 2019. Disentangling effects of time since fire, overstory composition and organic layer thickness on nutrient availability in Canadian boreal forest. Ecosystems **22**:33–48.

Chen J, John R, Sun G, McNulty S, Noormets A, Xiao J, Turner MG, Franklin JF. 2014. Carbon fluxes and storage in forests and landscapes. Pages 139–166 Forest landscapes and global change: Challenges for research and management. Springer New York, New York.

Chuvieco E, Congalton RG. 1989. Application of remote sensing and geographic information systems to forest fire hazard mapping. Remote Sensing of Environment **29**:147–159.

Clark, JS. 1988. Effect of climate change on fire regimes in northwestern Minnesota. Nature **334**:233–235.

Clark JS, Patterson WA. 1997. Background and local charcoal in sediments: Scales of fire evidence in the paleorecord. pp. 23-48 in J.S. Clark, H. Cachier, J.G. Goldammer, and B. Stocks (eds.) Sediment Records of Biomass Burning and Global Change. NATO ASI series. Series I, Global Environmental Change; no. 51. Springer-Verlag, Berlin.

Dale VH, et al. 2001. Climate change and forest disturbances. BioScience **51**:723–734.

Dale VH, Joyce LA, McNulty S, Neilson RP. 2000. The interplay between climate change, forests, and disturbances. Science of the Total Environment **262**:201–204.

D'Amato A, Fraver S, Palik B, Bradford J, Patty L. 2011. Singular and interactive effects of blowdown, salvage logging, and wildfire in sub-boreal pine systems. Forest Ecology and Management **262**:2070–2078.

Davis, RB. 1962. The spruce-fir forests of the coast of Maine. PhD Thesis. Cornell University. 307 p.

Davis, RB. 1967. Pollen studies of near-surface sediments in Maine lakes. Pages 143-173 in Cushing EJ, Wright EH, editors. Quaternary Paleoecology. Yale University Press, New Haven.

De Grandpré L, Kneeshaw DD, Perigon S, Boucher D, Marchand M, Pureswaran D, Girardin MP. 2019. Adverse climatic periods precede and amplify defoliator-induced tree mortality in eastern boreal North America. Journal of Ecology **107**:452–467.

DeRose RJ, Long JN. 2014. Resistance and resilience: A conceptual framework for silviculture. Forest Science **60**: 1205-1212.

Dey DC, Knapp BO, Battaglia MA, Deal RL, Hart JL, O'Hara KL, Schweitzer CJ, Schuler TM. 2019. Barriers to natural saplings in temperate forests across the USA. New Forests **50**:11–40.

Dilling L, et al. 2003.The role of carbon cycle observations and knowledge in carbon management. Annual Review of Environment and Resources, **28**, 521–558.

Dore S, Kolb TE, Montes-Helu M, Sullivan BW, Winslow WD, Hart SC, Kaye JP, Koch GW, Hungate BA. 2008. Long-term impact of a stand-replacing fire on ecosystem $CO_2$ exchange of a ponderosa pine forest. Global Change Biology **14**:1801–1820.

Drake NE, Patterson WA. 1994. Holocene fire and vegetation: History of Mount Desert Island, Maine. University of Massachusetts, Amherst, Massachusetts.

Dukes JS, et al. 2009. Responses of insect pests, pathogens, and invasive plant species to climate change in the forests of northeastern North America: What can we predict? Canadian Journal of Forest Research **39**:231–248.

Duveneck MJ, Scheller RM, White MA, Handler SD, Ravenscroft C. 2014. Climate change effects on northern Great Lake (USA) forests: A case for preserving diversity. Ecosphere Ecosphere **5**: art23.

Egler, FE. 1954. Vegetation science concepts I. Initial floristic composition, a factor in old-field vegetation development with 2 figs. Vegetation **4**: 412–417.

Euskirchen E, Chen J, Li H, Gustafson EJ, Crow TR. 2002. Modeling net carbon flows under alternative management regimes at a landscape level. Ecological Modeling, **154**: 75–91.

Fahey, TJ, Reiners WA. 1981. Fire in the forests of Maine and New Hampshire. Bulletin of the Torrey Botanical Club **108**:362–373.

Falkowski MJ, Gessler PE, Morgan P, Hudak AT, Smith AMS. 2005. Characterizing and mapping forest fire fuels using ASTER imagery and gradient modeling. Forest Ecology and Management **217**:129–146.

Ferree C, Anderson MG. 2013. A map of terrestrial habitats of the northeastern United States: Methods and approach. The Nature Conservancy, Eastern Conservation Science, Eastern Regional Office. Boston, MA. https://www.conservationgateway.org/ConservationByGeography/NorthAmerica/UnitedStates/edc/reportsdata/terrestrial/habitatmap/Pages/default.aspx

Finney, MA. 2005. The challenge of quantitative risk analysis for wildland fire. Forest Ecology and Management **211**:97–108.

Fischlin A, Midgley GF, Price JT, Leemans R, Gopal B, Turley C, Rounsevell MDA, Dube OP, Tarazona J, Velichko AA. 2007. Ecosystems, their properties, goods and services. *In* Climate Change 2007: impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate

Change. *Edited by* M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden, and C.E. Hanson. Cambridge University Press. UK. Pp. 211-272.

Fisichelli NA, Peters M, Iverson L, Matthews S, Hawkins Hoffman C. 2013. Climate change and forests of the Acadia National Park region: projected changes in habitat suitability for 83 tree species. National Park Service, Fort Collins, Colorado.

Flannigan MD, Bergeron Y, Engelmark O, Wotton BM. 1998. Future wildfire in circumboreal forests in relation to global warming. Journal of Vegetation Science **9**:469–476.

Flannigan M, Campbell I, Wotton M, Carcaillet C, Richard P, Bergeron Y. 2001. Future fire in Canada's boreal forest: paleoecology results and general circulation model - regional climate model simulations. Canadian Journal of Forest Research **31**:854–864.

Flannigan MD, Krawchuk MA, de Groot WJ, Wotton BM, Gowman LM. 2009. Implications of changing climate for global wildland fire. International journal of wildland fire.

Flannigan MD, Wagner CV. 1991. Climate change and wildfire in Canada. Canadian Journal of Forest Research **21**:66–72.

Foster, DR. 1983a. The phytosociology, fire history, and vegetation dynamics of the boreal forest of southeastern Labrador, Canada. University Microfilms International, Ann Arbor, Mich.

Foster, DR. 1983b. The history and pattern of fire in the boreal forest of southeastern Labrador.

Foster, DR. 1985. Vegetation development following fire in Picea Mariana (Black Spruce)-Pleurozium forests of south-eastern Labrador, Canada. Journal of Ecology **73**:517–534.

Foster DR, Aber JD. 2004. Forests in time: The environmental consequences of 1,000 years of change in New England. Yale University Press, New Haven.

Frelich LE, Lorimer CG. 1991. Natural disturbance regimes in hemlock-hardwood forests of the Upper Great Lakes region. Ecological Monographs **61**:145–164.

Frelich LE, Reich PB. 2010. Will environmental changes reinforce the impact of global warming on the prairie—forest border of central North America? Frontiers in Ecology and the Environment **8**:371–378.

Gao, Y. 2012. Projected changes of extreme weather events in the eastern United States based on a high resolution climate modeling system. Environmental Research Letters **7**.

Gauthier, S. 2009. Ecosystem management in the boreal forest. Presses de l'Université du Québec, Québec. Available from http://www.deslibris.ca/ID/434022.

Gauthier S, Bernier P, Kuuluvainen T, Shvidenko AZ, Schepaschenko DG. 2015. Boreal forest health and global change. Science **349**:819–822.

Gavin DG, Beckage B, Osborne B. 2008. Forest dynamics and the growth decline of red spruce and sugar maple on Bolton Mountain, Vermont: A comparison of modeling methods. Canadian Journal of Forest Research **38**:2635–2649.

Gimingham CH, Johnson EA. 1993. Fire and fegetation dynamics: Studies from the North American boreal forest. The Journal of Ecology **81**:384.

Girardin MP, Ali AA, Carcaillet C, Gauthier S, Hély C, Le Goff H, Terrier A, Bergeron Y. 2013. Fire in managed forests of eastern Canada: Risks and options. Forest Ecology and Management **294**:238–249.

Girardin MP, Mudelsee M. 2008. Past and future changes in Canadian boreal wildfire activity. Ecological Applications: A Publication of the Ecological Society of America **18**:391–406.

Goodale CL, et al. 2002. Forest carbon sinks in the northern hemisphere. Ecological Applications **12**:891–899.

Gough CM, Vogel CS, Harrold KH, George K, Curtis PS. 2007. The legacy of harvest and fire on ecosystem carbon storage in a north temperate forest. Global Change Biology **13**:1935–1949.

Gower ST, McMurtrie RE, Murty D. 1996. Aboveground net primary production decline with stand age: potential causes. Trends in Ecology & Evolution **11**:378.

Green, DG. 1982. Fire and stability in the postglacial forests of southwest Nova Scotia. Journal of biogeography **9**:29-40.

Grubb, PJ. 1977. The maintenance of species-richness in plant communities: The importance of the regeneration niche. Biological Reviews **52**:107–145.

Gunderson LH, Allen CR, Holling, CS. 2009. Foundations of Ecological Resilience. Island Press, Washington.

Gunn JS, Ducey MJ, Whitman AA. 2014. Late-successional and old-growth forest carbon temporal dynamics in the Northern Forest (Northeastern USA). Forest Ecology and Management **312**:40–46.

Gustafson E, Sturtevant B. 2013. Modeling forest mortality caused by drought stress: Implications for climate change. Ecosystems **16**.

Hamburg SP, Cogbill CV. 1988. Historical decline of red spruce populations and climatic warming. Nature **331**:428–431.

Harbor MAPB 177 B, Us M 04609 P-3338 C. 2019. Acadia National Park (U.S. National Park Service). Available from https://www.nps.gov/acad/index.htm.

Hardy CC. 2005. Wildland fire hazard and risk: Problems, definitions, and context. Forest Ecology and Management **211**:73–82.

Hart SJ, Henkelman J, McLoughlin PD, Nielsen SE, Truchon-Savard A, Johnstone JF. 2019. Examining forest resilience to changing fire frequency in a fire-prone region of boreal forest. Global Change Biology **25**:869–884.

Hayhoe K, et al. 2007a. Past and future changes in climate and hydrological indicators in the US northeast. Climate Dynamics **28**:381–407.

Hayhoe K, Wake C, Anderson B, Liang XZ, Maurer E, Zhu J, Bradbury J, DeGaetano A, Stoner AM, Wuebbles D. 2007b. Regional climate change projections for the northeast USA. Mitigation and Adaptation Strategies for Global Change **13**:425–436.

He HS, Mladenoff DJ. 1999. Spatially explicit and stochastic simulation of forest-landscape fire disturbance and succession. Ecology **80**:81–99.

Heinselman ML. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quaternary Research **3**:320-382.

Heinselman, ML. 1981a. Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. *In*: Fire Regimes and Ecosystem Properties. USDA Forest Service General Technical Report WO-26.

Heinselman, ML. 1981b. Fire and succession in the conifer forests of northern North America. In Forest Succession: Concepts and Application, Pages 374-405 in West DC, Shugart HH, Botkin DB, editors. Springer-Verlag, New York, NY.

Hély C, Flannigan M, Bergeron Y, McRae D. 2001. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. Canadian Journal of Forest Research **31**: 430–441.

Herberger M, Patterson WA. 1998. A vegetative fuelbreak protecting the town of Bar Harbor, Maine – Acadia National Park, ME. University of Massachusetts, Amherst, Massachusetts.

Hessburg PF, Reynolds KM, Keane RE, James KM, Salter RB. 2007. Evaluating wildland fire danger and prioritizing vegetation and fuels treatments. Forest Ecology and Management **247**:1–17.

Hirsch K, Kafka V, Todd B. 2004. Using forest management techniques to alter forest fuels and reduce wildfire activity: an exploratory analysis. In Fire in temperate, boreal, and montane ecosystems. Tall Timber Research Station, Tallahassee, FL.

Hisano M, Chen HYH, Searle EB, Reich PB. 2019. Species-rich boreal forests grew more and suffered less mortality than species-poor forests under the environmental change of the past half-century. Ecology Letters **22**: 999–1008.

Hmielowski TL, Carter SK, Helmers D, Radeloff VC, Spaul H, Zedler P. 2016. Prioritizing land management efforts at a landscape scale: A case study using prescribed fire in Wisconsin. Ecological Applications **26**: 1018–1029.

Holling, CS. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics **4**: 1-23.

Hooper DU, et al. 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecological Monographs **75**: 3-35.

Humphreys ER, Black AT, Morgenstern K, Li Z, Nesic Z. 2005. Net ecosystem production of a Douglas-fir stand for 3 years following clearcut harvesting. Global Change Biology **11**: 450–464.

Hudiburg T, Law B, Turner DP, Campbell J, Donato D, Duane M. 2009. Carbon dynamics of Oregon and northern California forests and potential land-based carbon storage. Ecological Applications **19**: 163–180.

Huntington TG, Richardson AD, McGuire KJ, Hayhoe K. 2009. Climate and hydrological changes in the northeastern United States: Recent trends and implications for forested and aquatic ecosystems. Canadian Journal of Forest Research **39:** 199-212.

Huston, M. 1979. A general hypothesis of species diversity. The American Naturalist **113**: 81–101.

Huston, MA. 1994. Biological diversity: The coexistence of species on changing landscapes. Cambridge University Press, Cambridge.

IPCC Sixth Assessment Report: Climate Change 2022. 2019. Available from https://www.ipcc.ch/report/sixth-assessment-report-cycle/.

Irland, LC. 2014. Analyzing size distribution of large wildfires. Fire Management Today **74**(1): 15-20.

Iverson LR, Schwartz MW, Prasad AM. 2004. How fast and far might tree species migrate in the eastern United States due to climate change? GEB Global Ecology and Biogeography **13**:209–219.

Jantz P, Monahan WB, Hansen AJ, Rogers BM, Zolkos S, Cormier T, Goetz SJ. 2016. Potential impacts of climate change on vegetation for national parks in the eastern United States. Pages 151–173 Climate change in wildlands: Pioneering approaches to science and management. Washington, DC: Island Press/Center for Resource Economics: Island Press.

Jasinski JP, and Payette S. 2005. The creation of alternative stable states in the southern boreal forest, Québec, Canada. Ecological Monographs **75**: 561–583.

Jayen K, Leduc A, Bergeron Y. 2006. Effect of fire severity on saplings success in the boreal forest of northwest Quebec, Canada. Ecoscience **13**: 143–151.

Johnstone JF, Chapin FS. 2003. Non-equilibrium succession dynamics indicate continued northern migration of lodgepole pine. Global Change Biology **9**: 1401–1409.

Johnstone JF, Chapin FS. 2006. Fire interval effects on successional trajectory in boreal forests of northwest Canada. Ecosystems **9**: 268–277.

Keane, RE. Fire Modeling Institute (Missoula M., Fire Sciences Laboratory (Missoula M). FER, Rocky Mountain Research Station. Fort Collins C. 2000. Mapping vegetation and fuels for fire management on the Gila National Forest complex, New Mexico. U.S. Dept. of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Keane RE, Burgan R, van Wagtendonk J. 2001. Mapping wildland fuels for fire management across multiple scales: Integrating remote sensing, GIS, and biophysical modeling. International Journal of Wildland Fire **10**: 301.

Keane RE, Rocky Mountain Research Station. Fort Collins C. 2002. Integrating ecosystem sampling, gradient modeling, remote sensing, and ecosystem simulation to create spatially explicit landscape inventories. U.S. Dept. of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Keane RE, Drury S, Karau EC, Hessburg PF, Reynolds KM. 2010. A method for mapping fire hazard and risk across multiple scales and its application in fire management. National Emergency Training Center, Emmitsburg, MD.

Keenan RJ. 2015. Climate change impacts and adaptation in forest management: a review. Annals of Forest Science **72**:145–167.

Keetch-Byram Drought Index. 1968. Available from http://www.wfas.net/index.php/keetch-byram-index-moisture--drought-49 (accessed May 2016).

Keeton WS, Whitman AA, McGee GC, Goodale CL. 2011. Late-successional biomass development in northern hardwood-conifer forests of the Northeastern United States.

Forest Science **57**: 489–505.

Keith H, Mackey BG, Lindenmayer DB, Likens GE. 2009. Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. Proceedings of the National Academy of Sciences of the United States of America **106**: 11635–11640.

Kellndorfer J, Walker W, Kirsch K, Fiske G, Bishop J, Lapoint L, Hoppus M, Westfall J. 2013. NACP Aboveground Biomass and Carbon Baseline Data, V.2 (NBCD 2000), U.S.A., 2000. Data set. Available online http://daac.ornl.gov from ORNL DAAC, Oak Ridge, Tennessee, U.S.A. http://dx.doi.org/10.3334/ORNLDAAC/1161.

Kent LY. 2014. An evaluation of fire regime reconstruction methods. ERI Working Paper No. 32. Ecological Restoration Institute and Southwest Fire Science Consortium, Northern Arizona University: Flagstaff, AZ. 15 p.

Krawchuk MA, Cumming SG. 2011. Effects of biotic feedback and harvest management on boreal forest fire activity under climate change. Ecological applications: a publication of the Ecological Society of America **21**: 122–36.

Laing, CP. 1993. The importance of fire in the maintenance of jack pine at its southeastern range limit in Acadia National Park, Maine. MS Thesis, University of Massachusetts, Amherst.

Law BE, Sun OJ, Campbell J, Van Tuyl S, Thornton PE. 2003. Changes in carbon storage and fluxes in a chronosequence of ponderosa pine. Global Change Biology, **9**: 510–524.

Law BE, Thornton PE, Irvine J, Anthoni PM, Van Tuyl S. 2001. Carbon storage and fluxes in ponderosa pine forests at different developmental stages. Global Change Biology, **7**: 755–777.

Ledig FT, Zobel BJ, Matthias MF. 1975. Geoclimatic patterns in specific gravity and tracheid length in wood of pitch pine. Canadian Journal of Forest Research. **5**: 318-329.

Le Goff H, Leduc A, Bergeron Y, Flannigan M. 2005. The adaptive capacity of forest management to changing fire regimes in the boreal forest of Quebec. Forestry Chronicle **81**: 582–592.

Lesmeister DB, Sovern SG, Davis RJ, Bell DM, Gregory MJ, Vogeler JC. 2019. Mixed-severity wildfire and habitat of an old-forest obligate. Ecosphere **10**.

Li W, Li L, Fu R, Deng Y, Wang H. 2011. Changes to the North Atlantic subtropical high and its role in the intensification of summer rainfall variability in the southeastern United States. Journal of Climate **24**: 1499–1506.

Lienard J, Strigul N, Harrison J. 2016. US forest response to projected climate-related stress: a tolerance perspective. Global Change Biology **22**: 2875–2886.

Long, S. 2016. Thirty-Eight: The hurricane that transformed New England. Yale University Press, New Haven, CT.

Lorimer, CG. 1977. The presettlement forest and natural disturbance cycle of northeastern Maine USA. Ecology **58**: 139-148.

Luken JO, Hinton AC, Baker DG. 1992. Response of woody plant communities in power-line corridors to frequent anthropogenic disturbance. Ecological Applications **2**: 356–362.

Luyssaert S, Schulze ED, Börner A, Knohl A, Hessenmöller D, Law BE, Ciais P, Grace J. 2008. Old-growth forests as global carbon sinks. Nature **455**: 213–215.

Maine State Planning Office. 1977. Maine coastal inventory handbook. Coastal Planning Program, Land Use Information Series IV.

Malevsky-Malevich SP, Molkentin EK, Nadyozhina ED, Shklyarevich OB. 2008. An assessment of potential change in wildfire activity in the Russian boreal forest zone induced by climate warming during the twenty-first century. Climatic Change: An Interdisciplinary, International Journal Devoted to the Description, Causes and Implications of Climatic Change **86**: 463–474.

Manabe S, Wetherald RT, Stouffer RJ. 1981. Summer dryness due to an increase of atmospheric CO2 concentration. Climatic Change **3**: 347–386.

Marlon, JR. 2015. North Atlantic Fire Science Exchange. Webinar: Response of forests and fire to climate change and human activities - North Atlantic Fire Science Exchange. Available from http://www.firesciencenorthatlantic.org/events-webinars-source/2015/7/22/save-the-date-nafse-webinar-response-of-forests-and-fire-to-climate-change-and-human-activities (accessed May 2016).

Matlack GR. 2013. Reassessment of the use of fire as a management tool in deciduous forests of eastern North America. Conservation Biology **27**: 916–26.

McGariga lK, Plunkett EB, Compton BW, Deluca WV, Grand J. 2017. Designing sustainable landscapes: modeling forest succession and disturbance. Report to the North Atlantic Conservation Cooperative, US Fish and Wildlife Service, Northeast Region. https://www.umass.edu/landeco/research/dsl/products/dsl_products.html

McKenzie D, Gedalof Z, Peterson DL, Mote P. 2004. Climatic change, wildfire, and conservation. Conservation Biology **18**: 890–902.

Messaoud Y, Goudiaby V, Bergeron Y. 2019. Persistence of balsam fir and black spruce populations in the mixedwood and coniferous bioclimatic domain of eastern North America. Ecology and Evolution **9**: 5118–5132.

Millar CI, Stephenson NL. 2015. Temperate forest health in an era of emerging megadisturbance. Science **349**.

Miller JD, Danzer SR, Watts JM, Stone S, Yool SR. 2003. Cluster analysis of structural stage classes to map wildland fuels in a Madrean ecosystem. Journal of Environmental Management **68**: 239–252.

Miller KN, Mitchel BR, Curtin PJ, Wheeler JS. 2014. Forest Health Monitoring in Acadia National Park. Northeast Temperate Network 2006-2013 Summary Report. Natural Resource Report NPS/NETN/NRR—2014/777. US Department of the Interior, National Park Service, Natural Resource Stewardship and Science, Fort Collins, CO.

Miller DR. 2019. Wildfires in the Northeastern United States: evaluating fire occurrence and risk in the past, present, and future. University of Massachusetts Amherst.

Mladenoff DJ, White MA, Pastor J, Crow TR. 1993. Comparing spatial pattern in unaltered old-growth and disturbed forest landscapes. Ecological Applications **3**: 294–306.

Mohan JE, Cox RM, Iverson LR. 2009. Composition and carbon dynamics of forests in northeastern North America in a future, warmer world. Canadian Journal of Forest Research **39**: 213–230.

Moore, TR. 1981. Controls on the decomposition of organic matter in subarctic spruce-lichen woodland soils. Soil Scientist, **131**: 107-113.

Moore JC, De Ruiter PC, Hunt HW. 1993. Influence of productivity on the stability of real and model ecosystems. Science **261**: 906.

Moore B, Taylor N. 1927. Vegetation of Mt. Desert Island, Maine and its environment. Brooklyn Bot. Garden Mem. No. 3.

Morgan P, Hardy CC, Swetnam TW, Rollins MG, Long DG. 2001. Mapping fire regimes across time and space: Understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire **10**: 329.

National Park Service. 2003. Spatial vegetation data for Acadia National Park vegetation Mapping Project, U.S. Geological Survey, Center for Biological Informatics. Vegetation Information for the Acadia National Park Vegetation Inventory Project.

Netherer S, Schopf A. 2010. Potential effects of climate change on insect herbivores in European forests—General aspects and the pine processionary moth as specific example. Forest Ecology and Management **259**: 831–838.

Nowacki GJ, Abrams MD. 2008. The demise of fire and “Mesophication” of forests in the

Eastern United States. BioScience **58**: 123–138.

Oliver CD, Larson BC, Miller HG. 1997. Forest stand dynamics (Update edition). Society of Foresters of Great Britain. **70**: 101.

Paine RT, Tegner MJ, Johnson EA. 1998. Compounded perturbations yield ecological surprises. Ecosystems **1**: 535–545.

Patterson WA, Backman AE. 1988. Fire and disease history of forests. pp. 603-662 *In*: B. Huntley, B. and T. Webb III, eds. Handbook of Vegetation Science, vol. 7. Vegetation history. Dordrecht, Netherlands: Kluwer Academic Publishers: 603-632.

Patterson WA, Backman AE, Davis RB. 1984. Sedimentary charcoal as an indicator of forest fires in Acadia National Park, Maine. 6th International Palynology Congress Abstract p. 126.

Patterson WA, Edwards KJ, Maguire DJ. 1987. Microscopic charcoal as a fossil indicator of fire. Quaternary Science Reviews **6**: 3–23.

Patterson WA, Saunders KE, Horton LJ. 1983. Fire regimes of the coastal Maine forests of Acadia National Park. USDI/NPS North Atlantic Region Office of Scientific Studies Report OSS 83-3. 259 p.

Patterson, WA. 1996. Ten-year re-evaluation of stands; Chapter 4: Vegetation, Chapter 5: Fuels. Unpublished.

Patterson, WA. 2006. The paleoecology of fire and oaks in eastern forests. *In*: Dickinson, M.B., ed. 2006. Fire in eastern oak forests: delivering science to land managers, proceedings of a conference; 2005 November 15-17; Columbus, OH. General Technical Report NRS-P-1. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: 2-19.

Patterson, WA. 2018. Drought and the 1947 Maine fires: Context based on 112 years of weather data for Bar Harbor. University of Massachusetts/Amherst. Available from https://www.firesciencenorthatlantic.org/events-webinars-source.

Payette S, Bhiry N, Delwaide A, Simard M. 2000. Origin of the lichen woodland at its southern range limit in eastern Canada: the catastrophic impact of insect defoliators and fire on the spruce-moss forest. Canadian Journal of Forest Research **30**: 288–305.

Payette S, Delwaide A. 2003. Shift of conifer boreal forest to lichen: Heath parkland caused by successive stand disturbances. Ecosystems **6**: 540–550.

Payette S, Fortin MJ, Gamache I. 2001. The subarctic forest-tundra: The structure of a biome in a changing climate. BioScience **51**: 709–718.

Plummer DA, Caya D, Frigon A, Côté H, Giguère M, Paquin D, Biner S, Harvey R, de Elia R. 2006. Climate and climate change over North America as simulated by the Canadian RCM. Journal of Climate **19**: 3112–3132.

Podur JJ, Martell DL. 2009. The influence of weather and fuel type on the fuel composition of the area burned by forest fires in Ontario, 1996—2006. Ecological Applications **19**: 1246–1252.

Podur J, Wotton M. 2010. Will climate change overwhelm fire management capacity? Ecological Modelling **221**: 1301–1309.

Pontius J, Schaberg PG, Halman JM. 2016. Seventy years of forest growth and community dynamics in an undisturbed northern hardwood forest. Canadian Journal of Forest Research **46**: 959–967.

Pyne SJ, Andrews PL, Laven RD. 1996. Introduction to wildland fire. 2nd edition. Wiley, New York.

R Core Team (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Raymond CL, McKenzie D. 2013. Temporal carbon dynamics of forests in Washington, US: Implications for ecological theory and carbon management. Forest Ecology and Management **310**: 796–811.

Riaño D, Chuvieco E, Salas J, Palacios-Orueta A, Bastarrika A. 2002. Generation of fuel type maps from Landsat TM images and ancillary data in Mediterranean ecosystems. Canadian Journal of Forest Research **32**: 1301–1315.

Robertson GP, Boyd GE, Crum JR. 1993. The spatial variability of soil resources following long-term disturbance. Oecologia **96**: 451–456.

Rogers BM, Neilson RP, Drapek R, Lenihan JM, Wells JR, Bachelet D, Law BE. 2011. Impacts of climate change on fire regimes and carbon stocks of the U.S. Pacific Northwest. Journal of Geophysical Research: Biogeosciences **116**.

Rogers BM, Jantz P, Goetz SJ. 2017. Vulnerability of eastern US tree species to climate change. Global Change Biology **23**: 3302–3320.

Rollins MG, Keane RE, Parsons RA. 2004. Mapping fuels and fire regimes using remote sensing, ecosystem simulation, and gradient modeling. Ecological Applications **14**: 75–95.

Rollins, MG. 2009. LANDFIRE: A nationally consistent vegetation, wildland fire, and fuel assessment. International Journal of Wildland Fire **18**(3): 235-249.

Rollins MG, Frame CK. Rocky Mountain Research Station (Fort Collins). 2006. The LANDFIRE Prototype Project: Nationally consistent and locally relevant geospatial data for wildland fire management. U.S. Dept. of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Rothermel, RC. 1983. How to predict the spread and intensity of forest range fires. General Technical Report INT- 143, United States Department of Agriculture Forest Service, Ogdon, UT.

Rustad, LE. 2012. Changing climate, changing forests: The impacts of climate change on forests of the northeastern United States and eastern Canada. Newtown Square, PA: USDA, Forest Service, Northern Research Station.

Ryan MG, Senock RS, Binkley D, Fownes JH, Giardina CP. 2004. An experimental test of the causes of forest growth decline with stand age. Ecological Monographs **74**: 393–414.

Saefken B, Ruegamer D. 2018. cAIC4: Conditional Akaike information criterion for lme4, R package version 0.3.

Schauffler M, Nelson SJ, Kahl JS, Jacobson GL, Haines TA, Patterson WA, Johnson KB. 2007. Paleoecological assessment of watershed history in PRIMENet watersheds at Acadia National Park, USA. Environmental Monitoring and Assessment **126**: 39-53.

Schimel DS, et al. 2001. Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. Nature, **414**: 169–176.

Schulte, LA, Mladenoff DJ. 2005. Severe wind and fire regimes in northern forests: historical variability at the regional scale. Ecology **86**: 431–445.

Scott JH, Burgan RE. Rocky Mountain Research Station. 2005. Standard fire behavior fuel models a comprehensive set for use with Rothermel's surface fire spread model. U.S. Dept. of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Seidl R, Schelhaas MJ, Lindner M, Lexer MJ. 2009. Modelling bark beetle disturbances in a large-scale forest scenario model to assess climate change impacts and evaluate adaptive management strategies. Regional Environmental Change **9**: 101–119.

Seidl R, et al. 2011a. Modelling natural disturbances in forest ecosystems: A review. Ecological Modelling **222**: 903–924.

Seidl R, Schelhaas MJ, Lexer MJ. 2011b. Unraveling the drivers of intensifying forest disturbance regimes in Europe. Global Change Biology **17**.

Shugart HH, Leemans R, Bonan GB. 1992. A systems analysis of the global boreal forest. Cambridge University Press, Cambridge.

Siefert A, Lesser MR, Fridley JD. 2015. How do climate and dispersal traits limit ranges of tree species along latitudinal and elevational gradients? Global Ecology & Biogeography **24**.

Simard M, Payette S. 2005. Reduction of black spruce seed bank by spruce budworm infestation compromises postfire stand saplings. Canadian Journal of Forest Research **35**: 1686–1696

Sirois L, Payette S. 1989. Postfire black spruce establishment in subarctic and boreal Quebec. Canadian Journal of Forest Research **19**: 1571–1580.

Soil Survey Staff. 2019. Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available from: https://websoilsurvey.sc.egov.usda.gov/. Accessed May 2019.

Spies TA, Hemstrom MA, Youngblood A, Hummel S. 2006. Conserving old-growth forest diversity in disturbance-prone landscapes. Conservation Biology **20**: 351–362.

Spies TA, Lindenmayer DB, Gill AM, Stephens SL, Agee JK. 2012. Challenges and a checklist for biodiversity conservation in fire-prone forests: Perspectives from the Pacific Northwest of USA and Southeastern Australia. Biological Conservation **145**: 5–14.

Star J, Fisichelli N, Bryan AM, Babson A, Cole-Will R, Miller-Rushing A. Acadia National Park climate change scenario planning workshop summary. 2015. Available from https://irma.nps.gov/DataStore/Reference/Profile/2230257. Accessed September 2017.

Stephens SL, Agee JK, Fule PZ, North MP, Romme WH, Swetnam TW, Turner MG. 2013. Managing forests and fire in changing climates. Science **342**: 41–42.

Stocks BJ, Fosberg MA, Lynham TJ, Mearns, L, Wotton BM, Yang, Q, Jin JZ, Lawrence K, Hartley GR, Mason JA, McKenny DW. 1998. Climate change and forest fire potential in Russian and Canadian boreal forests. Climatic Change **38**: 1–13.

Tang G, Beckage B, Smith B. 2014. Potential future dynamics of carbon fluxes and pools in New England forests and their climatic sensitivities: A model-based study. Global Biogeochemical Cycles **28**: 286–299.

The Wildland Fire Assesment System (WFAS). (n.d.). Available from http://www.wfas.net/ Accessed March 2020.

Thompson MP, Calkin DE, Finney MA, Ager AA, Gilbertson-Day JW. 2011. Integrated national-scale assessment of wildfire risk to human and ecological values. Stochastic Environmental Research and Risk Assessment **25**: 761–7.

Tilman, D. 1996. Biodiversity: population versus ecosystem stability. Ecology **77**: 1082-1087.

Tritton LM, Hornbeck JW. 1982. Biomass equations for major tree species of the northeast. Gen. Tech. Rep. NE-69. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station.

Trumbore S, Brando P, Hartmann H. 2015. Forest health and global change. Science (New York, N.Y.) **349**: 814–8.

Turner, MG. 1987. Landscape heterogeneity and disturbance: Using stem char to predict mortality and insect infestation of fire-damaged slash pines. New York: Springer-Verlag.

Turner MG, Baker WL, Peterson CJ, Peet RK. 1998. Factors influencing succession: Lessons from large, infrequent natural disturbances. Ecosystems **1**: 511–523.

Turner, MG. 2010. Disturbance and landscape dynamics in a changing world. Ecology **91**: 2833–49.

U.S. Forest Service. 2010. Northeast Wildfire Risk Assessment. NWRA Steering Committee. Area Assessment- Phase I. 20 p.

USGCRP. 2018. Fourth National Climate Assessment. Available from https://nca2018.globalchange.gov.

Van Nes EH, Scheffer M. 2004. Large species shifts triggered by small forces. American Naturalist **164**: 255-266.

Varner JM, Arthur MA, Clark SL, Dey DC, Hart JL, Schweitzer CJ. 2016. Fire in Eastern north American Oak Ecosystems: Filling the Gaps. Fire Ecology **12**: 1–6.

Viro, PJ. 1974. Effects of forest fire on soil. Pages 7-35 in Kozlowski TT Ahlgren CE, editors. Fire and ecosystems. Academic Press, New York.

Vogt K, et al. 1997. Ecosystems: Balancing science with management. Springer-Verlag, New York.

Wein RW, Burzynski MP, Sreenivasa BA, Tolonen K. 1987. Bog profile evidence of fire and vegetation dynamics since 3000 years BP in the Acadian Forest. Canadian Journal of Botany **65**: 1180–1186.

Wein RW, Moore JM. 1977. Fire history and rotations in the New Brunswick Acadian Forest. Canadian Journal of Forest Research **7**: 285–294.

Wells, CG. 1978. Effects of fire on soil: a state-of-knowledge review; prepared for the Forest Service, National Fire Effects Workshop, Denver, Colo., April 10-14, 1978. U.S. Department of Agriculture, Forest Service, Washington, D.C.

Whittaker RH, Woodwell GM. 1968. Dimension and production relations of trees and shrubs in the Brookhaven Forest, New York. Journal of Ecology **56**: 1–25.

Wilson, JB. 1994. The `Intermediate Disturbance Hypothesis' of species coexistence is based on patch dynamics. New Zealand journal of ecology **18**: 176.

Worrall JJ, Marchetti SB, Rehfeldt GE, Hamann A, Gray LK, Hogg EH, Michaelian M. 2013. Recent declines of Populus tremuloides in North America linked to climate. Forest Ecology and Management **299**: 35–51.

Wotton BM, Flannigan MD. 1993. Length of the fire season in a changing climate. Forestry Chronicle **69**: 187.

Young HE, Ribe JH, Wainwright K. 1980. MR230: Weight Tables for Tree and Shrub Species in Maine. Available from http://digitalcommons.library.umaine.edu/aes_miscreports/19/.

# Appendices

Appendix 1 – Acadia National Park 1947 Fire Photographic Repeat Collection. Photos from 1983 taken by K. Barnicle. Photos from 2016 taken by J. Charpentier. Photos from 1947/1948 available in National Park Service archives at Acadia National Park. Description of photos below.

1a – 1947; 1b- 1948: NPS archives.

2a-1- 1947; 2a-2 – 1947: NPS archives.

2b-1 – 1948; 2b-2- 1948: NPS archives.

3a-1947; 3b- 1948: NPS archives.

4a-1947; 4b-1948: NPS archives.

4c – 1983

4d-2016

5a-1947; 5b-1948: NPS archives.

5c -1983

5d-2016

6a – 1947; 6b – 1948: NPS archives.

6c- 1983

6d-2016

7a – 1947; 7b – 1948: NPS archives.

7c- 1983 7d-2016

8a – 1947; 8b – 1948: NPS archives.

8c- 1983 8d-2016

9a- 1947; 9b-1948: NPS archives.

9c- 1983 9d-2016

10a – 1947; 10b- 1948: NPS archives.

10c- 1983 10d-2016

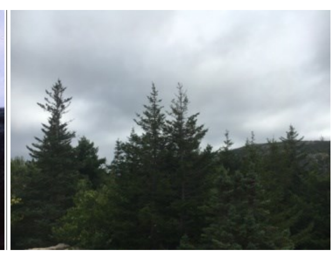
11a- 1947; 11b- 1948: NPS archives.


11c- 1983 11d-2016

12a- 1947; 12b-1948: NPS archives.

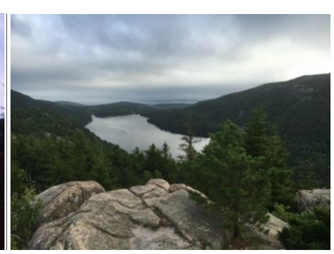

12c-1983 12d-2016

13a- 1947; 13b-1948: NPS archives.

13c- 1983 13d-2016

14a- 1947; 14b-1948: NPS archives.

14c-1983 14d-2016

15a-1947; 15b-1948; 15c- 1948: NPS archives.


15d-1983


15e-2016

16a-1947; 16b-1948: NPS archives.


16c-1983


16d-2016

17a- 1948: NPS archives.


17b-1983


17c-2016

18a- 1948: NPS archives.


18b-1983


18c-2016

19a- 1948: NPS archives.

19b-1983 19c-2016

20a- 1948: NPS archives.

20b-1983 20c-2016

21a- 1948: NPS archives.

21b-1983 21c-2016

22a- 1948: NPS archives.

22b-1983 22c-2016

23a-1948: NPS archives.


23b-1983 23c-2016

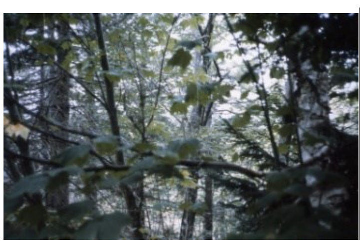
24a- 1948: NPS archives.


24b- 1983 24c-2016

25a- 1948: NPS archives.


25b-1983 25c-2016

26a- 1948: NPS archives.

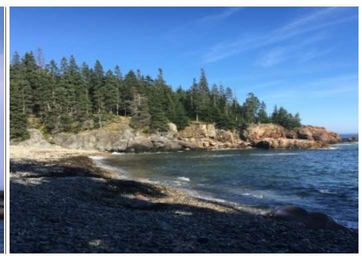



26b-1983 26c-2016

28a- 1948: NPS archives.

28b-1983 28c-2016

29a- 1948: NPS archives.



29b-1983 29c-2016

Descriptions of 1947 Mount Desert Island Fire Slides.

1a. 1947: This pitch pine, previously thought to be dead, sprouted adventitious buds one growing season following the fire. It is said that this tree still exists today. (Photo by G. Gordon Bruce)

lb. 1947: This oak tree showed prolific sprouts triggered by the fire. Decay usually follows when this species has been injured by fire. (Photo by G. Gordon Bruce)

2a. After fire: Two photographs showing the severity of wind damage following the fire and before logging and cleanup operations had been done. Photo on the right was taken on the Bubbles. (Photo by G. Gordon Bruce)

2b. After fire: Following one growing season many areas within the burned area showed tremendous regeneration as shown by the birch seedlings in the right photograph. The photo on the left shows the rapid growth of red alder (?) sprouts. (Photo by G. Gordon Bruce)

3a. After fire: Dolliver's Dump where the fire began. Burned area extends from the boxes to the vegetation in the background. (Photographer unknown)

3b. 1983: Dump is now abandoned and thickly grown over with shrubs, wildflowers, and an occasional aspen. Vegetation in the top photo is mostly hardwoods with a few survivors of the fire in the background. (Photo by K. Barnicle)

4a. Before treatment: Main entrance to Satterlee Estate. (Photo by Hironimus)

4b. After treatment: Same as above. Burned area treatment by NPS crews. (Photo by Hironimus)

4c. 1983: Hardwood stand present. Great Head trail marker to the right of photo. (Photo by K. Barnicle)

4d. 2016: Hardwood stand still present. (Photo by J. Charpentier)

5a. Before fire: Looking south to Otter Point from the site of the Satterlee Estate. (Photo by Hironimus)

5b. After fire: Dead trees and debris removed. Note Sand Beach at lower right and crews working along Ocean Drive. Burned area treatment by NPS crews. (Photo by Hironimus)

5c. 1983: Hardwood revegetation and possible spruce survivor to the right of photo. (Photo by K. Barnicle)

5d. 2016: Hardwood vegetation continuing to grow. (Photo by J. Charpentier)

6a. Before fire: Typical view from Ocean Drive showing fire damage to pitch pine. (Photo by Schroeder)

6b. After fire: With dead and burned trees removed, a beautiful vista is opened to the south. (Photo by Schroeder)

6c. 1983: Pitch pine to the left is still surviving. Strong pitch pine regeneration along roadside. (Photo by K. Barnicle)

6d. 2016: Pitch pine continuing to grow. (Photo by J. Charpentier)

7a. Before fire: Eastern portion of Sand Beach looking east to Satterlee Estate. (Photo by Hironimus)

7b. After fire: Same as above. Burned area treatment by NPS crews. (Photo by Hironimus)

7c. 1983: Conifers still dominant in the area, possible survivors of the fire. (Photo by K. Barnicle)

7d. 2016: Same as above. (Photo by J. Charpentier)

8a Before fire: Ocean Drive after Sand Beach parking lot. Severely burned pitch pine. (Photo by Hironimus)

8b. After fire: Same as above. Standing trees, although burned, may develop sufficient seeds to restock the area. Burned area treatment by NPS crews. (Photo by Hironimus)

8c. 1983: Larger trees are now gone but there is thick pitch pine regeneration in the area. (Photo by K. Barnicle)

8d. 2016: Pitch pine continues to grow. (Photo by J. Charpentier)

9a. Before fire: Junction Kebo Mountain Road and Ledgelawn Ext. West to USNPS weather station. (Photo by Doudna)

9b. After fire: Same area as above. Exposing weather station. Burned area treatment by NPS crews. (Photo by. Hironimus)

9c. 1983: Photo at slightly different angle. Strong hardwood regeneration. (Photo by Barnicle)

9d. 2016: Strong hardwood regeneration continues to grow. (Photo by J. Charpentier)

10a. Before fire: Cadillac Mt. Road looking west. Eagle Lake and McFarland Mountain in background. Fire-killed pitch pine type. (Photo by Hironimus)

10b. After fire: Here again a beautiful vista has been opened. This view of Eagle Lake, the mountains and panorama of the Blue Hill area on the mainland were previously concealed. Burned area treatment by NPS crews. (Photo by Hironimus)

10c. 1983: Predominately birch with scattered pitch pines, spruces and cedars. (Photo by K. Barnicle)

10d. 2016: Same as above but the birch seems to have dropped out of the stand. Predominately pitch pine, spruce, and cedar. (Photo by J. Charpentier)

11a. Before fire: Northwest from Cadillac Mountain Road to McFarland and Youngs Mountain. Severely burned mixed coniferous type in foreground. (Photo by Hironimus)

11b. After fire: Same as above: Removal of timber exposes heretofore unseen panorama to the northwest and northern portion of Eagle Lake seen at the left. Burned area treatment by NPS crews. (Photo by Hironimus)

11c. 1983: Exposed rocks are slowly being recovered by mosses, shrubs and eventually pitch pine. (Photo by K. Barnicle)

11d. 2016: Still considerable exposed rock. Pitch pine continues to grow. (Photo by J. Charpentier)

12a. Before fire: From foot of Jordan Pond north to the Bubble Mountains, showing burned timber on south slopes and summits of both Bubbles. (Photo by G. Gordon Bruce)

12b. After fire: Same as above after removal of burned timber from slopes and summits of both mountains. Burned area treatment by NPS crews. (Photo by Hironimus)

12c. 1983: Larger amounts of exposed rock on both Bubbles as compared to the previous photos. (Photo by K. Barnicle)

12d. 2016: Same as above, but Bubbles is revegetating with less exposed rock. (Photo by J. Charpentier)

13a. Before fire: Southwest to Sargent Mountain, south ridge, from summit of the south Bubble. Camera angle slightly lower than picture shown below. (Photo by G. Gordon Bruce)

l3b. After fire: Same as above. Extreme fire hazard removed. Burned area treatment by Rockefeller crews. (Photo by G. Gordon Bruce)

13c. 1983: A birch and poplar stand with an occasional spruce. (Photo by K. Barnicle)

13d. 2016: Predominately spruce which continues to grow. (Photo by J. Charpentier)

14a. Before fire: North to Eagle Lake from summit of North Bubble Mountain. Extremely unsightly from Cadillac Mountain Road. (Photo by G. Gordon Bruce)

14b. After fire: Same as above. Clears heavily used foot trails. Burned area treatment by Rockefeller crews. (Photo by G. Gordon Bruce) l4c. 1983: Birch and popular stands are present on the summit today with scattered spruces. (Photo by K. Barnicle)

14d. 2016: Same as above. Slightly different camera lens and angle. (Photo by J. Charpentier)

15a. Before fire: Saddle between North and South Bubbles. (Photo by G. Gordon Bruce)

15b. After fire: Same as above after treatment. Burned area treatment by Rockefeller crews. (Photo by G. Gordon Bruce)

15c. After fire: Slightly higher angle. After treatment but logs must still be removed. (Photo by G. Gordon Bruce)

15d. 1983: Hardwood stand with scattered spruce. Note the more exposed rock surfaces on the North Bubble in the background as compared to the previous photo. (Photo by K. Barnicle)

15e. 2016: Hardwood stand has completely grown up. North Bubble is still in the background and is visible through the trees when standing on site. (Photo by J. Charpentier)

16a. Before fire: Northwest face of the south Bubble. Burned mixed coniferous growth. Jordan Pond in left foreground. (Photo by Doudna)

16b. After fire: Same as above, burned timber removed. Bare face contrasts starkly with Sargent Mountain in the background. Burned area treatment by Rockefeller crews. (Photo by Doudna)

16c. 1983: A hardwood stand with some conifers has revegetated the area. More areas with exposed rock, possibly due to erosion. (Photo by K. Barnicle)

16d. 2016: Same as above. Slightly different camera lens and angle. (Photo by J. Charpentier)

17a. After fire: View from Ocean Drive overlooking Sand Beach. Satterlee estate in the background. (Photographer unknown)

17b. 1983: Greater amounts of pitch pine exist than in previous photo. Notice the greater amounts of exposed rock in the background. Area is difficult to revegetate due to the direct exposure of the ocean. (Photo by K. Barnicle)

17c. 2016: Pitch pine continues to grow. The ocean view from this location is almost gone. (Photo by J. Charpentier)

l8a. After fire: Cadillac Mountain Road nearing the summit. Mostly spruce, hemlock, and hardwoods. (Photographer unknown)

l8b. 1983: Thick birch regeneration with smaller components of hemlock and spruce. Growth is not as fast in this site due to the higher elevations. (Photo by K. Barnicle)

18c. 2016: Birch is almost gone and conifers continue to grow. (Photo by J. Charpentier)

19a. After fire: Cadillac Mountain summit walk. Small patch of protected vegetation burned by the fire was able to support both shrubs and trees. (Photographer unknown)

19b. 1983: Today only blueberry bushes and other shrubs exist in this protected patch. Tree species have not been able to reestablish back into the area because of the amount of exposure. (Photo by K. Barnicle)

19c. 2016: Same as above, not a lot of change. (Photo by J. Charpentier)

20a. After fire: View from Ocean Drive overlooking Great Head through severely burned pitch pines. (Photographer unknown)

20b. 1983: Thick pitch pine regeneration. This species was able to revegetate because of dormant seeds in the soil or from seeds from cones opened up in the heat of the fire. (Photo by K. Barnicle)

20c. 2016: Pitch pine continues to grow. (Photo by J. Charpentier)

21a. After fire: Photo taken following logging operations on McFarland Hill behind CCC camp (Site of current Park Headquarters). Carriage trail and Cadillac Mountain in the background. (Photographer unknown)

2lb. 1983: Mostly white pine, spruce, and birch have regenerated the previously logged area. Stumps and logs can still be found in the area. (Photo by K. Barnicle)

21c. 2016: Same as above. Near-ground has been cleared and mowed. (Photo by J. Charpentier)

22a. After fire: With root anchorage burned away by ground fire, spruce stands on Otter Point, the Bubbles, and elsewhere collapsed like jackstraws in subsequent winds. Note the unburned trunks and branches. More intense fires can now occur in such areas than before unless timber salvage and cleanup operations are accomplished. Stand located near Cadillac Cliffs. (Photo by W.H. Ballard)

22b. 1983: This stand was not burned by the fire but shows how some spruce stands are subject to severe blowdown, thus increasing the amount of fuel on the forest floor and heightening the fire danger in these areas. (Photo by K. Barnicle)

22c. 2016: Dense spruce stand continues to grow. (Photo by J. Charpentier)

23a. After fire: All that remains of some once attractive white pine stands is utter desolation. The crown fire burned high and the ground fire burned deep. Stand located at the north end of Great Meadow. (Photo by W.H. Ballard)

23b. 1983: Revegetation of hardwoods has built up the organic soil layers to once again support vegetation. Tree to right of photo is not the same as the previous photo. (Photo by K. Barnicle)

23c. 2016: Same as above. Hardwoods continue to grow. Stem density appears to have decreased since 1983. (Photo by J. Charpentier)

24a. After fire: Erstwhile picturesque groups of pitch pine trees along Ocean Drive were left standing, stark black and dead. Sand Beach is shown in the background. (Photo by W.H. Ballard)

24b. 1983: The pitch pine trees showed here did not die as was previously thought but produced sprouts triggered by the fire as a mechanism to survive. Note the increased branchiness in the older trees. Thick pitch pine regeneration in the understory. (Photo by K. Barnicle)

24c. 2016: Pitch pine continues to grow and is dense in this location. (Photo by J. Charpentier)

25a. After fire: View of Bar Harbor from Cadillac Mountain summit parking lot showing burned forest in the background. (Photo by W.H. Ballard)

25b. 1983: Similar species regeneration, such as pitch pine, has occurred but to a lesser extent due to exposed conditions at the top of the mountain. Trees and shrubs show some stunted growth. (Photo by K. Barnicle)

25c. 2016: Same as above. (Photo by J. Charpentier)

26a. After fire: Where the burn was relatively light, as in some youthful hardwood stands, few, if any, of the dominant trees are likely to succumb. Stand located near lower mountain road near junction with Cadillac Mountain Road. (Photo by W.H. Ballard)

26b. 1983: Many of the larger trees did survive and similar species were able to regenerate back that had previously dominated stand. (Photo by K. Barnicle)

26c. 2016: This hardwood stand continues to grow. Strong red spruce component in the understory. (Photo by J. Charpentier)

27. Photo series missing.

28a. After fire: At least one spring must pass before the full extent or fire damage in hardwood stands will be evident, particularly where the burn was intense as in this stand located along Eagle Lake Road. (Photo by W.H. Ballard)

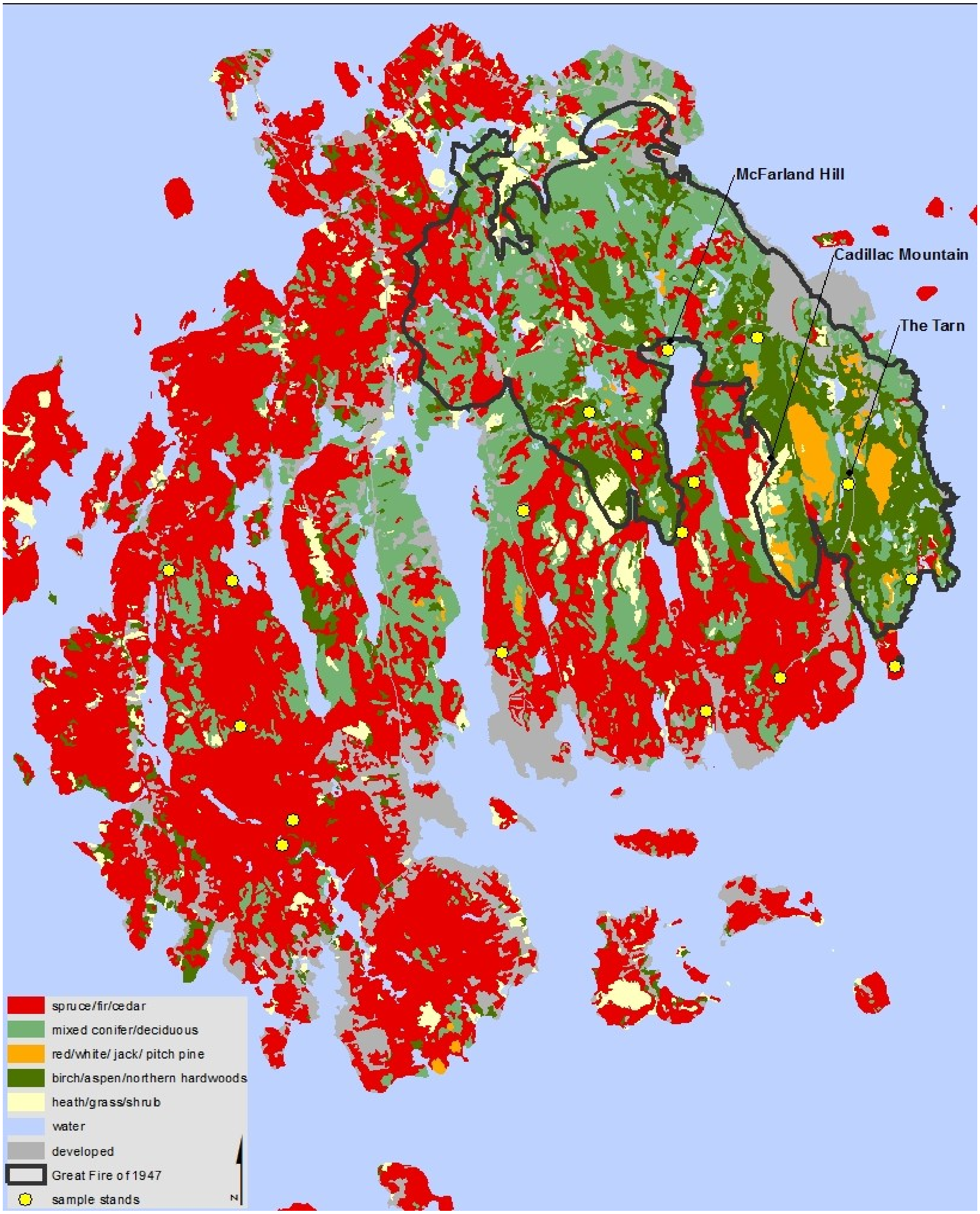
28b. 1983: Birch and aspen regeneration has resulted in these intensely burned hardwood areas. Stumps and cut logs can still be seen left form the cleanup operations. (Photo by K. Barnicle)

28c. 2016: Same as above. (Photo by J. Charpentier)

29a. After fire: Where fire and then wind struck stands in locations similar to this scenic attractiveness of spruce forest meeting rock- bound coast will not return for many years. Scene taken at Little Hunter's Beach. (Photo by W.H. Ballard)

29b. 1983: The unburned stand of mature spruce is now showing decline and dieback and some windthrow. This stand will become more susceptible to fire with its increasing fuel build up. (Photo by K. Barnicle)

29c. 2016: Same as above. (Photo by J. Charpentier)




Appendix 2. Vegetation flammability of Mount Desert Island. Source: Spatial Vegetation Data for Acadia National Park Vegetation Mapping Project, U.S. Geological Survey, Center for Biological Informatics. 2003. Vegetation Information for the Acadia National Park Vegetation Inventory Project.

To express vegetation flammability, cover types were grouped as follows:

(NOT by order of significance)

A. Spruce/fir/cedar (60-100% canopy closure)
Spruce-fir forest (conifer phase)
Spruce-fir forest (mixed phase)
White cedar woodland
Conifer swamp woodland (white cedar stage)
Conifer swamp woodland (spruce-mix phase)
Evergreen plantation

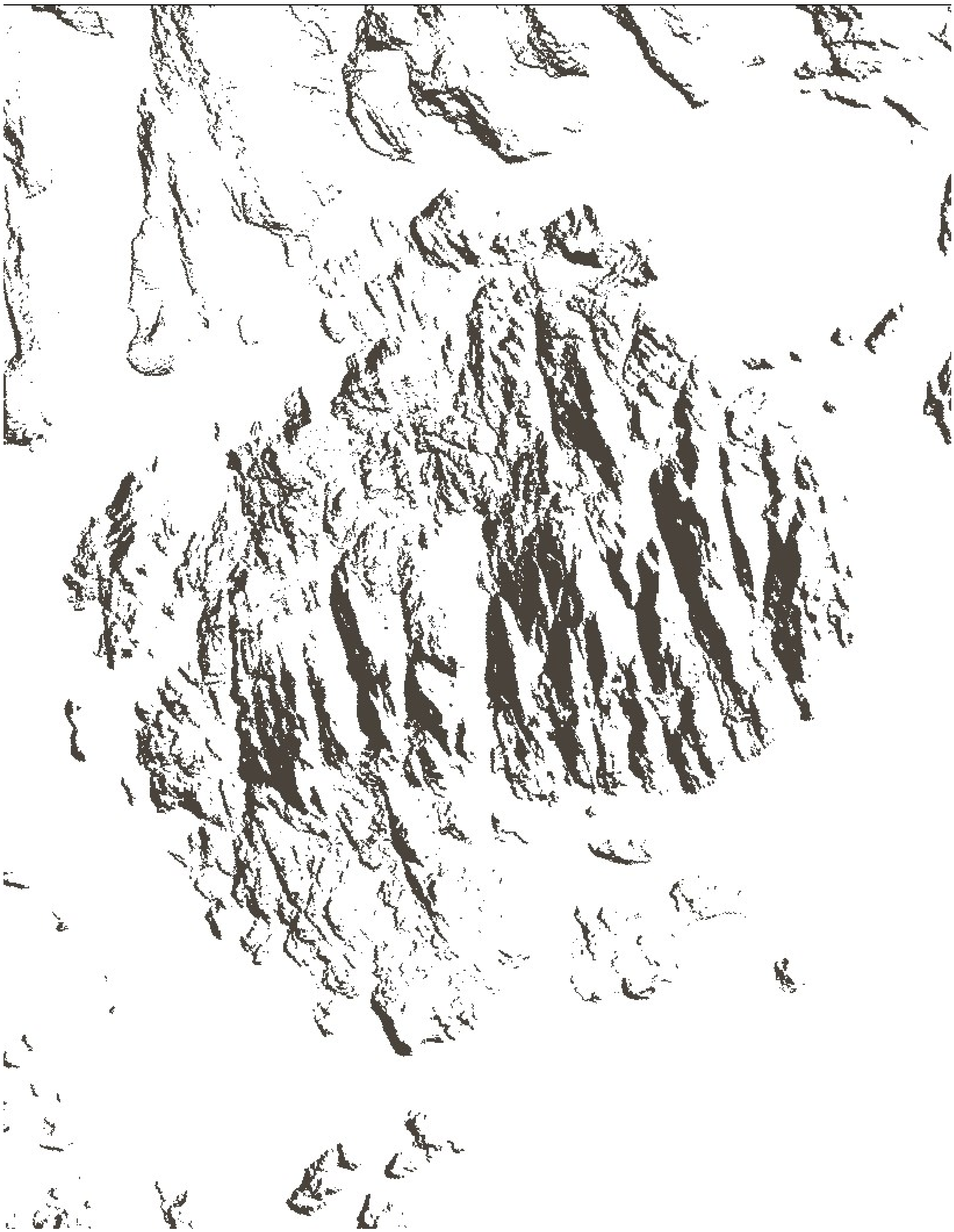
B. Mixed conifer-deciduous (25-60% canopy closure)
Mixed conifer woodland
Mixed conifer-deciduous woodland
White pine –hardwood forest
White pine-mixed conifer forest
Oak-pine forest

C. Red, white, jack, pitch pine:
Pitch pine heath barren
Red pine-white pine forest
Pitch pine-corema woodland
Pitch pine woodland
Jack pine woodland

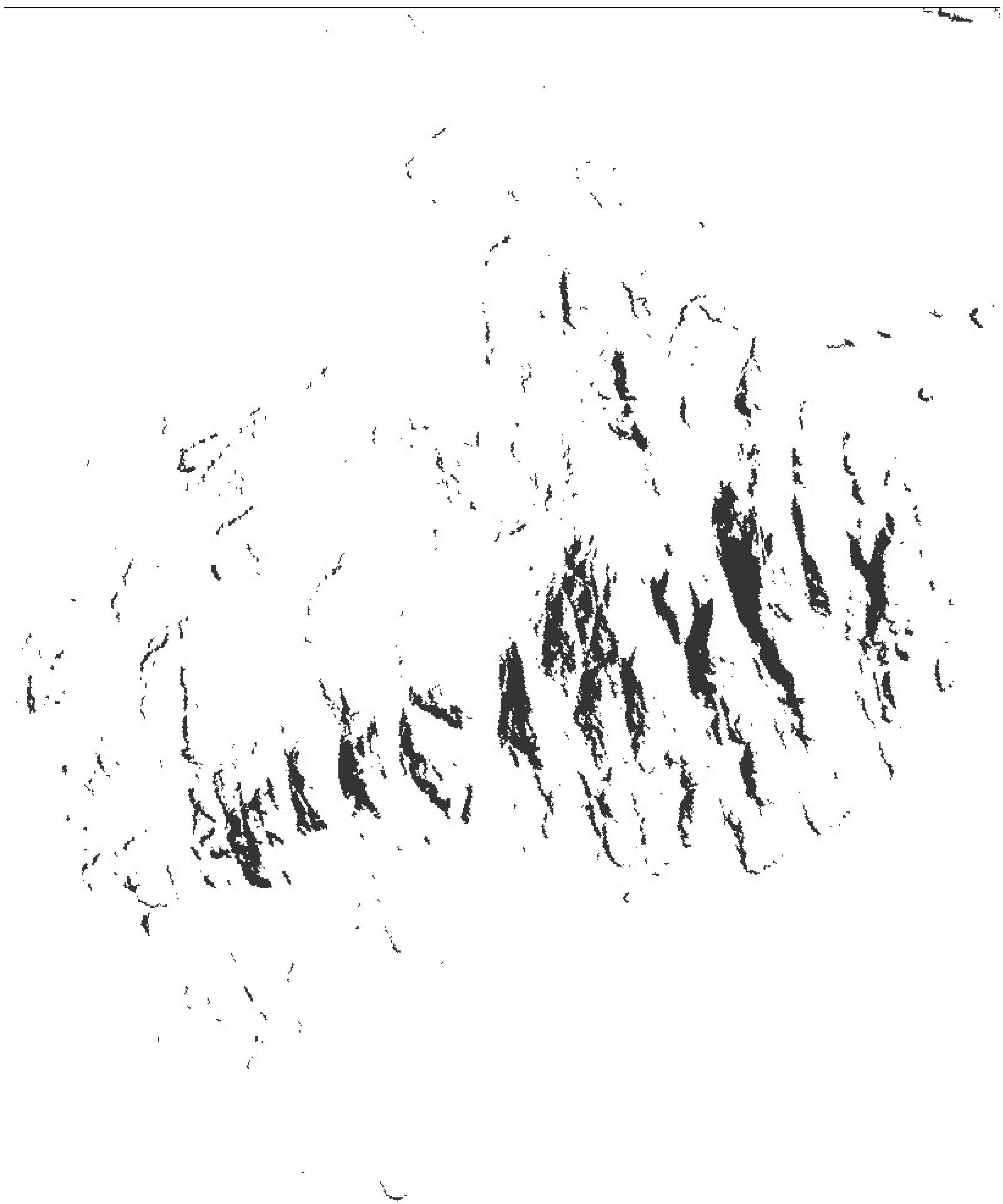
D. Birch-aspen/northern hardwoods:
Aspen-birch woodland/forest complex (woodland phase)
Aspen-birch woodland/forest complex (forest phase)
Aspen-birch woodland/forest complex (shrubland phase)
Beech-birch-maple forest
Red oak woodland
Mixed deciduous shrubland
Red maple-hardwood swamp

E. Health, grass, shrub
Alder shrubland
Perennial grass crops
Perennial grass crops with sparse shrubs
Mixed-grass forb
Fen Complex
Crowberry-bayberry headland
Dune grassland
Sweetgale mixed shrub fen
Sparsely vegetated talus
Dwarf shrub bog
Blueberry bald-summit shrubland complex
Other agricultural land

Appendix 3. LANDFIRE slope data reclassified to only identify those pixels that represent a slope greater than 20%.



Appendix 4. LANDFIRE aspect data reclassified to only include between 135 and 315 degrees. Within this range, solar heating of fuels is expected to contribute to an increase in wildfire risk.



Appendix 5. Topography map which combines LANDFIRE slope and aspect data. The topography map contains only those pixels which are greater than 20% slope and an aspect between 135 and 315 degree**s.**

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Bill Patterson

************************************************************
William A. Patterson III
Emeritus Professor of Forestry

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Thu, Dec 19, 2019 at 9:02 AM

Dear Jessica:

You have permission to use the repeat photo series I took circa 1982 at Acadia National Park comparing the 1947 post-fire photos.

This fire played a significant role in our lives as my mother's house burned in that fire and my uncle (on leave from the Navy) assisted in fighting the fire.

Good luck with your dissertation.

Thanks!

Katie

**Kathryn S. Barnicle**

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