CHAPTER 8

Managing Effects of Drought in the Midwest and Northeast United States

CO-LEAD AUTHORS

Heidi Asbjornsen and John L. Campbell

Anthony W. D'Amato, Jeff Garnas, John S. Gunn, Louis R. Iverson, Todd A. Ontl, Neil Pederson, Matthew P. Peters, and P. Danielle Shannon

H. Asbjornsen is an Associate Professor of Ecosystem Ecology and Ecohydrology, University of New Hampshire, Department of Natural Resources and the Environment and Earth Systems Research Center, Durham, NH 03824.

J.L. Campbell is a Research Ecologist, U.S. Department of Agriculture, Forest Service, Northern Research Station, Durham, NH 03824.

A.W. D'Amato is a Professor of Silviculture and Applied Forest Ecology and Forestry Program Director, University of Vermont, Rubenstein School of Environment and Natural Resources, Burlington, VT 05405.

J. Garnas is an Assistant Professor of Forest Ecosystem Health, University of New Hampshire, Department of Natural Resources and the Environment, Durham, NH 03824.

J.S. Gunn is a Research Assistant Professor of Forest Management, University of New Hampshire, New Hampshire Agricultural Experiment Station, Durham, NH 03824.

L.R. Iverson is a Landscape Ecologist, U.S. Department of Agriculture, Forest Service, Northern Research Station, Northern Institute of Applied Climate Science, Delaware, OH 43015.

T.A. Ontl is a Climate Hub Fellow, U.S. Department of Agriculture, Forest Service, Northern Research Station, Northern Institute of Applied Climate Science, Houghton, MI 49931.

N. Pederson is a Senior Ecologist, Harvard University, Harvard Forest, Petersham, MA 01366.

M.P. Peters is an Ecologist, U.S. Department of Agriculture, Forest Service, Northern Research Station, Northern Institute of Applied Climate Science, Delaware, OH 43015.

P.D. Shannon is the Climate Hub Northern Forests Coordinator, U.S. Department of Agriculture, Forest Service, Northern Research Station, Northern Institute of Applied Climate Science, and Michigan Technological University, Houghton, MI 49931.

BACKGROUND

Severe droughts are relatively rare in the Midwest and Northeast compared to other parts of the United States. This 20-State region, hereafter referred to as the Northern Region, is defined as the States bounded by Maine, Minnesota, Missouri, and Maryland. Although the Northern Region has a cool, wet climate and is generally considered to have an abundance of water, model projections suggest that droughts may become more frequent and severe in the future. The Northern Region is densely populated (39 percent of the U.S. population) (U.S. Census Bureau 2018), so changes in precipitation may be especially disruptive. Impacts will affect forest ecosystems and the services they provide, including timber and nontimber products, water regulation and supply, erosion and pollution control, biodiversity protection, and recreation.

Nearly 43 percent of the Northern Region is forested (Oswalt et al. 2014), so management of this key resource is central to maintaining the economy and quality of life. Unlike much of the Western United States, the majority (74 percent) of forest land in the Northern Region is privately owned, mostly as smaller family forest holdings (Oswalt et al. 2014). This model of ownership is challenging from a management perspective because of difficulties facilitating change at the landscape scale when so many individuals are involved. Unlike government agencies that can alter land management practices more directly, making changes in management of privately owned land is largely accomplished through education and using incentives to achieve desired outcomes (e.g., cost-share payments for implementing specific management practices).

Most forests in the Northern Region are not currently managed with drought in mind. Because drought has historically had less of an impact on forest health compared to other regions, drought management tools and techniques are not well established. The increased probability of future drought in the Northern Region has created a need for information about both the impacts of drought on forests and the options for land managers to cope with acute and chronic reductions in water availability.

Forests in the Northern Region are typically energy-limited rather than water-limited, and widespread drought-induced diebacks are rare. However, drought has caused widespread tree mortality in some ecosystems in this region, especially in the lower Midwest (e.g., oak forests in the Ozark Mountains of Missouri; Jenkins and Pallardy 1995). When drought triggers mortality, the affected trees have usually been predisposed to drought by other stressors. Some of these stressors are associated with the dense human population, such as air pollution and the prevalence of pests, pathogens, and invasive species (Haavik et al. 2015, Jenkins and Pallardy 1995, Pedersen 1998).

Despite these issues, the Northern Region has a diversity of tree species, which may help enhance resistance and resilience to drought (Peters et al. 2015). This high biodiversity also increases management options by providing a broader selection of drought-tolerant tree species. However, how the region's trees may fare in the future is difficult to predict for several reasons: the unprecedented projected changes in climate, interactions with multiple simultaneously changing drivers (e.g., atmospheric CO_2, ozone, nitrogen deposition), and the relative dearth of research on drought impacts on forests in the Northern Region. Given these complexities and uncertainty in future climate, drought poses a challenge to land managers in the Northern Region and warrants consideration in management decisions.

DROUGHT DEFINITIONS AND TRENDS

Drought can be defined from many different perspectives, and each approach will lead to a different understanding of how drought is expressed on the landscape across both temporal and spatial scales. In this report, we consider three types of drought that are especially relevant to forest managers—meteorological, hydrological, and ecological drought—and describe past and projected future drought trends.

Meteorological Drought

Meteorological drought is often defined solely by precipitation, based on the degree of dryness and duration of the dry period (Wilhite and Glantz 1985). The thresholds for the duration and severity of meteorological drought are site-specific and are identified by evaluating deviations from normal (i.e., average historical) climatic conditions. An extreme drought or wet spell can be quantified statistically as the tails of the historical rainfall distribution (Smith 2011). Although this approach is limited by the availability of reliable data, analysis of tree rings (which can serve as historical proxies spanning centuries) and modeling (for projecting future climate trends) can greatly expand the capacity to assess longer term drought trends. Several indices (e.g., Palmer Drought Severity Index [PDSI] and Standardized Precipitation Evapotranspiration Index [SPEI]) have been developed to identify periods of meteorological drought and are valuable for monitoring long-term trends (e.g., Donat et al. 2013, Palmer 1965). However, these indices often require variables, such as soil moisture, that are rarely available for long periods across broad regions.

Within the Northern Region, tree-ring records indicate that severe meteorological droughts occurred before the 20th century (Cook and Jacoby 1977, Pederson et al. 2013, Stahle et al. 2007). There is evidence of a megadrought in the 1500s (Stahle et al. 2000) and then a series of repeated severe droughts during the middle of the 1600s (McEwan et al. 2011, Pederson et al. 2014). Over the 20th century, the frequency and magnitude of droughts have declined. Conditions in the early 21st century have been wetter (Pederson et al. 2015), and although droughts still occur (e.g., Sweet et al. 2017), they have not been as severe as the megadroughts of the past.

Average annual precipitation across States in the region ranges from 178 to 330 inches (NCDC 2017). Although some areas of the United States, such as parts of the Southeast and Northwest, receive more annual precipitation, the Northern Region is becoming wetter at a faster rate than any other region. Between the periods of 1901–1960 and 1986–2015, precipitation increased by more than 15 percent in the Northern Region (Easterling et al. 2017). Additionally, the Ohio River and Hudson River valleys have had more days with rain during the summer in the most recent 20 years than during the previous 40 years (Bishop and Pederson 2015).

These past trends in precipitation are consistent with future projections from general circulation models (GCMs) that show increases in precipitation through the end of the 21st century (Fan et al. 2015; Hayhoe et al. 2007, 2010). In contrast with the historical record, however, much of the future increase is expected to occur during winter, with either little change or slight declines in summer precipitation (depending on the model and greenhouse gas emissions scenario used). If recent trends continue, summer precipitation events are expected to come increasingly as short bursts of heavy, intense rainfall, with longer intervening dry periods (Easterling et al. 2017). Therefore, even though the Northern Region is getting more precipitation on average, there is heightened concern about future drought effects on forests because of both projected variability and extremes in precipitation and warming due to warming temperatures.

Future drought trends for the Northeast and Midwest areas of the Northern Region were also evaluated by modeling PDSI through the end of the century. A common issue with characterizing trends using drought and aridity indices (such as PDSI) is that they produce location-based, time series datasets that cannot be easily compared at broader spatial scales or among time periods. To remedy this, PDSI time series datasets were aggregated into weighted values, such that the frequency of drought events is weighted by their intensity. Using this approach, a single cumulative value can represent the relative potential for drought of a location, (see chapter 2 for details of the Cumulative Drought Severity Index [CDSI] calculations). Cumulative Drought Severity Index values were compared for two models each under two future greenhouse gas emissions scenarios—representative concentration pathway (RCP) scenarios 4.5 and 8.5 (Moss et al. 2008)—and for three 30-year periods: 2010–2039, 2040–2069, and 2070–2099. The 30-year period of 1980–2009 was used as a baseline. These four models, developed for the 2020 Resources Planning Act Assessment (Joyce et al. 2018), represent scenarios of warm-wet, hot-wet, hot-slightly dry, and hot-dry.

Results from this analysis show a projected rise in drought conditions for the second half of the 21st century, during which percentages of the Northeast and Midwest under some form of drought more than double spatially and/or temporally compared to the baseline period of 1980–2009 (fig. 8.1). None of the scenarios show great changes in drought or moist conditions through 2040, but change markedly after that. The hot-slightly dry and especially the hot-dry scenarios show the largest increases in extreme and severe drought in both regions by end of century, though the wetter scenarios also show increasing drought (fig. 8.1).

Most of the scenarios also show a reduction of moist classes, especially after mid-century (fig. 8.1). This trend appears to be more prominent in the Midwest as compared to the Northeast. These patterns generally agree with observed recent regional increases in precipitation and flooding, with moisture stress further exacerbated in some places because of higher temperatures and longer periods between significant precipitation events.

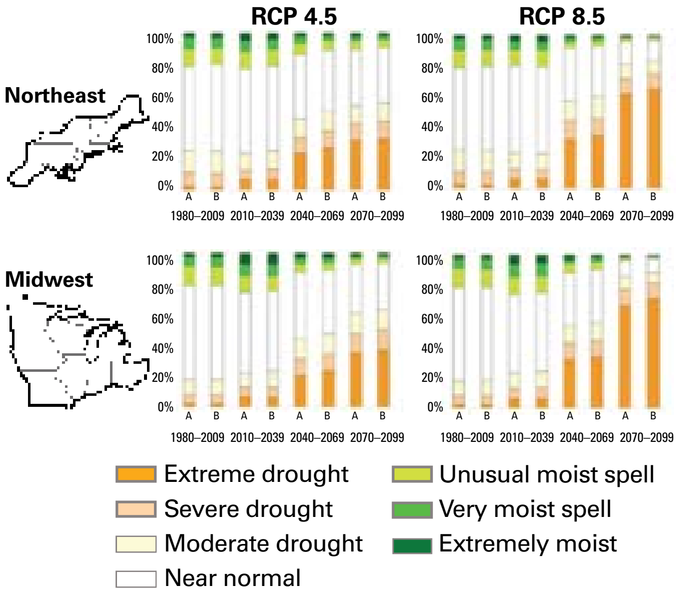


Figure 8.1—Palmer Drought Severity Index (PDSI) for the Northeast and Midwest regions under two greenhouse gas emissions scenarios, representative concentration pathways (RCP) 4.5 and 8.5. Palmer Drought Severity Index was calculated (A) with and (B) without a snowmelt function applied. Following Wells et al. (2004), drought classifications are as follows: extreme drought (PDSI ≤-4.00), severe drought (PDSI -3.9 to -3.0), moderate drought (PDSI -2.9 to -2.0), near normal (PDSI -1.9 to 1.9), unusual moist spell (PDSI 2.0 to 2.9), very moist spell (PDSI 3.0 to 3.9), and extremely moist (PDSI ≥4.0). Bars represent each PDSI class as a percentage of months out of each 30-year time period. See chapter 2 for more detail.

Hydrological Drought

Hydrological drought occurs when periods of low precipitation cause a reduction in surface and subsurface water supplies (i.e., streams, rivers, lakes, reservoirs, soil moisture, groundwater) (Van Loon 2015). This definition differs from meteorological drought in that hydrological droughts are influenced not only by a lack of rainfall, but also by other processes that affect water supply, such as evaporation, transpiration, storage, and runoff. Although hydrological records are not as old as some other indicators of drought, such as tree rings, they are the most useful for identifying water deficits.

Modeling is the only practical way to assess the effect of climate change on future trends in hydrological drought. However, two challenges are the uncertainty in the models and the climate scenarios used. Plant transpiration strongly regulates streamflow in the Northern Region, so a lack of understanding about how vegetation may change under future climate conditions complicates the ability to determine how changes in climate affect hydrology.

Analyses of hydrological data from the recent past have shown no obvious evidence of an increase in drought frequency within the Northern Region. In fact, because of increasing trends in precipitation in the Northern Region, stream and river flows have also generally increased (Burns et al. 2007, Campbell et al. 2011, Collins 2009). Other hydrological evidence of increasingly wetter conditions includes greater average annual soil moisture (Groffman et al. 2012) and higher groundwater levels (Dudley and Hodgkins 2013). Collectively, these records suggest that hydrological drought is becoming less common in the Northern Region.

Perhaps more important than changes in annual hydrological values are the seasonal shifts in the water balance that have occurred and their net effect on water supply. In the more northerly areas of the region, warming has caused a decline in the amount and duration of snowpack (Burakowski et al. 2008, Campbell et al. 2010, Hodgkins and Dudley 2006), resulting in a more muted spring snowmelt peak and higher winter flows (Campbell et al. 2011, Hodgkins et al. 2003, Novotny and Stefan 2007). A decline in snowmelt runoff could reduce groundwater recharge, which, when combined with a longer growing season and greater transpiration, could increase the risk of late-summer drought. However, historical evidence of this trend is lacking. Further, baseflows have generally increased during the growing season, at least in some portions of the Northern Region (Campbell et al. 2011, Novotny and Stefan 2007) because of higher precipitation in the spring, summer, and fall (Hayhoe et al. 2007). Whether this pattern will continue in the future is unclear. Results from models typically indicate increases in hydrological drought frequency (Hayhoe et al. 2007) and a greater tendency for drought stress in late summer through the end of the 21st century (Campbell et al. 2009).

The efficiency of tree water use depends on factors such as forest composition, amount of biomass, tree health, and the influence of changing atmospheric CO_2. Uncertainty about these factors makes future changes in hydrology difficult to predict. As a result, hydrological models have shown a broad range of responses to changing climate, with some showing increases in annual water yield and others showing decreases (Blake et al. 2000, Hayhoe et al. 2007, Ollinger et al. 2008, Pourmokhtarian et al. 2017). Future changes in climate, especially precipitation, will undoubtedly influence the hydrological drought regime, but the direction and extent of change remain highly uncertain.

Ecological Drought

Ecological drought is a relatively new term that more fully addresses the ecological impacts of drought, without the more constrained, human-centric emphasis of other definitions of drought (e.g., socioeconomic, agricultural, hydrological). Crausbay et al. (2017) defined ecological drought as an "episodic deficit in water availability that drives ecosystems beyond thresholds of vulnerability, impacts ecosystem services, and triggers feedbacks in natural and/or human systems." Thus, the definition of ecological drought integrates the ecological, climatic, hydrological, socioeconomic, and cultural dimensions of drought. Ecological drought emphasizes the underlying mechanisms that control individual or ecosystem responses to drought, and it is not directly tied to actual historical or projected future trends in precipitation. This definition also accounts for site-specific edaphic, topographic, and climatic characteristics that affect responses to drought, such as physical factors of a site that influence soil moisture available to vegetation (Gerten et al. 2008, Zeppel et al. 2014).

Although the Northern Region has experienced droughts in the past, they have usually not been severe enough to elicit a widespread threshold response with lasting broad-scale ecological impacts, such as a shift from forest to grassland. The impacts of past droughts have typically been subtler, but they nevertheless have had important ecological consequences. These include reductions in forest production, increased fire frequency and severity, outbreaks of pests and pathogens, spread of invasive species, and changes in the cycling of water and nutrients. Perhaps some of the most notable ecological droughts in the Northern Region are those that have caused tree dieback and mortality. The following sections highlight past observations and current understanding of potential future impacts of ecological drought on forests in the Northern Region.

FOREST DROUGHT IMPACTS

Vulnerability and Resilience

It is difficult to anticipate the full range of impacts of increasing future drought on forests within the Northern Region. Based on recent reviews and modeling experiments, possible responses include high vulnerability (e.g., Charney et al. 2016, Janowiak et al. 2018, Liénard et al. 2016, Martín-Benito and Pederson 2015, Rogers et al. 2017, Swanston et al. 2017), substantial resilience (e.g., Duveneck and Scheller 2016, Duveneck et al. 2017), or a mix of these two extremes (e.g., Brandt et al. 2014, Clark et al. 2016).

Evidence indicates that past droughts have caused tree mortality across the Eastern United States, including parts of the Northern Region (Millers et al. 1989). New research has shown that multiannual drought (defined as more than one standard deviation less than the long-term mean of summer precipitation) preceded observed tree mortality. This relationship holds regardless of where the observations occurred or when they occurred within the last 100 years (i.e., early 20th century compared to later 20th century) (Druckenbrod et al. 2019).

In the Southeastern United States, severe droughts in the 1980s and 2000s caused tree mortality (e.g., Berdanier and Clark 2016, Clinton et al. 1993, Jenkins and Pallardy 1995, Spetich 2004, Stringer et al. 1989), providing further insight about the effect of increased drought frequency or severity on forests in the Northern Region. However, latitudinal analyses of climatic sensitivity during the 20th century indicate that trees in more southern locations are more vulnerable to maximum temperatures than are trees farther north (Martín-Benito and Pederson 2015, Williams et al. 2010), suggesting that the impact of warming on the climatic balance in the Northern Region may be less than what has been observed elsewhere.

Several factors may help to mitigate consequences of drought to Northern Region forests. For example, northern temperate forests are characterized by high species diversity or structural complexity, which could help to offset impacts of extreme climate events, given the overall positive effects of diversity and heterogeneity on stability and resilience (e.g., Hautier et al. 2015, Isbell et al. 2015, Martín-Benito et al. 2008, Morin et al. 2014, Ratcliffe et al. 2017, Tilman 1999). Moreover, trees from different canopy layers have different sensitivities to climate (Canham and Murphy 2016, Orwig and Abrams 1997). For example, in a South American temperate broadleaf forest, severe drought-induced mortality in the canopy trees allowed understory trees, which were less vulnerable to moisture stress, to grow into the canopy (Rodríguez-Catón et al. 2015). Therefore, ecosystems may have the capacity to rapidly recover new vegetation following widespread mortality, albeit potentially with different species composition.

Long-lived trees growing in northern forests could also improve drought resilience. Old trees typically maintain large reserves of carbohydrates in their tissues that may be accessed during stressful periods to maintain critical growth and metabolic functions (Hoch et al. 2003, Richardson et al. 2013), providing an inherent safeguard against extreme events such as droughts. At the ecosystem scale, high biodiversity may provide a buffer to drought through shifts in species composition from drought-intolerant species to more drought-tolerant species, while maintaining critical ecosystem functions such as carbon cycling and hydrological regulation. Another consideration is that the projected trend of increasing drought in the future may overestimate the influence of warming in a mesic region if potential changes in future water-use efficiency result in wetter than anticipated soils (Mankin et al. 2017). Thus, such long-term increases in changes in species composition or efficiency of water use may help to offset future drought impacts. However, a recent analysis of yellow-poplar (*Liriodendron tulipifera*) and northern red oak (*Quercus rubra*) in southern New York indicated that soil moisture was still the dominant limiting growth factor, despite increased atmospheric CO_2 and a potential associated increase in water-use efficiency (Levesque et al. 2017). With longer and more severe drought events predicted for the future, once certain thresholds in warming or drying are reached, the inherent resistance of long-lived trees and the potential for physiological adjustments to changing environmental conditions may be exceeded.

Shifts in Species Composition and Diversity

Species compositional changes in the Northern Region over the past century have been dominated by "mesophication," defined as the gradual replacement of shade-intolerant, fire-adapted species (e.g., pines [*Pinus* spp.], oaks [*Quercus* spp.], hickories [*Carya* spp.]) with shade-tolerant, fire-sensitive species (e.g., maples [*Acer* spp.], birches [*Betula* spp.], beeches [*Fagus* spp.]) (Nowacki and Abrams 2008). This trend of mesophication is likely a response to two variables: wetter growing conditions (Pederson et al. 2015) and the closing up of overstory canopies following abandonment of widespread agriculture and grazing (Nowacki and Abrams 2015). Over long time scales, an increasing prevalence of drought could lead to shifts in species composition and diversity as more vulnerable species decline and more drought-resistant species increase in abundance.

Mesophytic species have been hypothesized to be especially vulnerable to future severe droughts (Abrams and Nowacki 2016). This hypothesis is consistent with the broad classification of drought tolerance based on each species' distributional range, optimal site conditions, physiological responses, and traits (Matthews et al. 2011, Niinemets and Valladares 2006, Peters et al. 2015). However, there are inconsistencies among studies in how different species are classified (Klein 2014, Loewenstein and Pallardy 1998, Martínez-Vilalta et al. 2014, Roman et al. 2015). Further, predictions based on drought-tolerance classification and actual field observations of drought impacts do not always agree (Gu et al. 2015, Hoffmann et al. 2011, Pedersen 1998, Roy et al. 2004, Voelker et al. 2008). More targeted research is needed to improve understanding of how species respond to drought and the long-term implications for forest community dynamics.

One approach to assess the effect of drought on forest composition is to use models to predict species responses to potential changes in suitable habitat (https://www.fs.fed.us/nrs/atlas; Iverson et al. 2008a, 2008b, 2011, 2019). Results from these modeling studies generally show that, under most scenarios of climate change, boreal species (e.g., black spruce [*Picea mariana*], red spruce [*P. rubens*], and balsam fir [*Abies balsamea*]) are projected to lose suitable habitat, but more southern species (e.g., American basswood [*Tilia americana*], black cherry [*Prunus serotina*], and northern red oak) are expected to gain suitable habitat. We used this modeling approach to assess the capability of tree species to cope with a changing climate, especially drought, at eight national forests across the Northern Region (table 8.1). Four variables were considered to develop a capability class: (1) projected change in suitable habitat by 2100, according to models using the RCP 8.5 scenario of emissions (Iverson et al. 2019); (2) adaptability of the species to a changing climate according to a literature review (Matthews et al. 2011); (3) reliability of the model as determined by a statistical analysis (Iverson et al. 2008b); and (4) current abundance of the species based on Forest Service, Forest Inventory and Analysis (FIA) data. We assumed that a species' capability to cope with a changing climate was decreased when the species showed a loss of suitable habitat following warming according to RCP 8.5, especially when it was an uncommon to rare species that was not particularly adapted to drought conditions. Each species was classified by its capacity to cope with changing conditions using the following scale: very good, good, fair, poor, very poor, lost, or new habitat. For example, if a species was modeled to gain substantial habitat according to the RCP 8.5 scenario of emissions, had some characteristics (e.g., resistance to drought or pests)

Table 8.1—Number of tree species (sorted west to east) by capability class (i.e., their ability to cope with a changing climate and drought) for nine national forests in the Northern Region, and current, potential (new habitat), and total number of species modeled in each national forest.

			CAPABILITY CLASS					NUMBER OF MODELED SPECIES		
National Forest	Latitude	Longitude	Very good	Good	Fair	Poor	Very poor	Lost	New habitat	Total
Hoosier	38	86	8	9	16	15	10	4	14	76
Wayne	39	82	4	7	18	20	10	5	8	72
Allegheny	41	79	5	8	7	16	5	7	17	65
Finger Lakes	42	76	4	9	8	23	5	9	13	71
Green Mountain	43	72	4	7	6	13	9	2	17	58
White Mountain	44	71	3	7	7	9	7	3	28	64
Chequamegon	46	91	2	6	15	7	4	1	19	54
Chippewa	47	94	2	8	9	10	1	1	15	46

that provided adaptability, had a reliable statistical model, and was currently abundant in the national forest in question, it was rated as 'very good' in capability to cope with the projected changes in climate.

Results indicated that current species abundance and drought tolerance predicted greater potential of a given species to remain. Species projected to experience a severe loss in habitat are those less able to cope with the changing climate. Northernmost forests (latitude >42 N) tended to have both less species diversity and fewer species with ratings of either very good or good drought-coping capability (table 8.1). The northeastern forests (Green Mountain and White Mountain National Forests) were, however, predicted to provide suitable habitat for more species that could increase diversity as species from the south move northward. This pattern of more potential migrations was not so true for the northwestern forests (Chippewa, Chequamegon, and Nicolet National Forests), however, as the tree diversity south of these forests is less due to its historic prairie state, lower rainfall, and high proportion of agriculture. Despite these predicted shifts in suitable habitat, however, other modeling studies (Iverson et al. 2004) and empirical evidence (Zhu et al. 2012) largely suggest that migration rates of tree species are far too slow to track such rapid changes in a suitable climate niche. The largest differences among forests for capacity to cope was from east to west. For the Hoosier, Wayne, Chequamegon, Nicolet, and Chippewa National Forests (longitude >80 W), an average of 58 percent of the current species rated fair or better, but for the Allegheny, Finger Lakes, Green Mountain, and White Mountain National Forests, only 40 percent of the current species had a rating of fair or better. These results support the hypothesis that species composition could change under the changing climate and that the Northeast may undergo the largest changes because of more potential migrations and fewer species with at least a fair capability to cope with climate change.

Insects, Pathogens, and Invasive Species

The most important drivers of forest disturbance in the Northern Region are wind, ice, insects, pathogens, invasive species, and to a lesser degree, fire (Dukes et al. 2009). Among the least well-understood aspects of forest responses to climate change, including drought, is how insect pests, pathogens, and invasive species will respond. Disturbances caused by these agents will continue and are likely to increase with global climate change, exacerbated by the gradual accrual of novel species introduced into forests (Aukema et al. 2010, Liebhold et al. 1995). Threats to forest resilience and sustainability include higher air temperatures, more variable and extreme weather events, biological invasion, shifting ranges, and local climatic mismatching (e.g., of remnant populations and/or species that are slower to migrate). Set against this backdrop, insects, pathogens, and invasive species rank among the top threats (Dukes et al. 2009, Lovett et al. 2006).

For both insects and pathogens, consequences of environmental change are likely to be complex. Changes in air temperature, as well in as the duration and severity of drought, can act either directly or indirectly on populations. Direct effects on insects and pathogen growth rates, fecundity, and survival are simple enough to document and examine experimentally, although extrapolating from lab or semi-field conditions can be challenging (Koricheva et al. 1998b). Indirect effects are much more difficult to predict and are likely to affect many systems (Kolb et al. 2016). For example, multiple changes are hypothesized in response to water stress (e.g., host plant nutritional quality, constitutive or induced defenses, and physiological responses to herbivory or pathogen attack), and these changes are likely to be nonlinear and context-dependent (Kolb et al. 2016, Mattson and Haack 1987). The rare empirical studies tend to show variable results that appear to be specific to the feeding guild or tissue preference of the insect or pathogen.

Other likely strong influences on phytophagous insect and pathogen populations are changes in the abundance, distribution, and seasonality within natural enemy and competitor populations (Weed et al. 2013) and, in some cases, alternative hosts (e.g., for rust fungi; Kinloch 2003). Insect and pathogen population responses to intermittent water stress differ from responses to long-term water stress and are influenced by the timing and duration of dry periods (Kolb et al. 2016). Despite these multifactorial challenges, research is improving understanding, and some general patterns are beginning to emerge (Jactel et al. 2012, Koricheva et al. 1998a).

One explanation for the lack of a clear relationship between drought and insect or pathogen abundance is that droughts are relatively rare, as is true for many other types of drought impacts in the Northern Region. However, some evidence suggests that interannual variation in precipitation is correlated with either the abundance of insects or pathogens or the severity of the damage they cause (e.g., Dukes et al. 2009). One example comes from a large-scale assessment of the role of climate in driving the dynamics of beech bark disease. For both causal agents of the disease (the scale insect, *Cryptococcus fagisuga*, and *Neonectria* fungi), a spatially replicated time series showed that both spring and fall precipitation were important predictors of three key population parameters: the strength of density dependence, predicted equilibrium abundance, and the contribution of exogenous (climatic) variation (Garnas et al. 2011).

As a second example, in areas that experience periodic water stress such as the forests adjacent to the Great Plains, wood-boring insects appear to increase under drought conditions (Haavik et al. 2015). Despite the relative lack of empirical evidence, these two examples suggest that drought has the capacity to influence pest and pathogen population dynamics in eastern forests. Other evidence for the effect of insects on forests comes from a recent meta-analysis of insect responses to plant stress, including drought (Chakraborty et al. 2014). The results suggest that, generally, cambium feeders may benefit the most from plant stress, followed by sucking, mining, and then chewing insects, with galling insects having the lowest relative survivorship. Drought may therefore favor insect pests such as the emerald ash borer (*Agrilus planipennis*, a phloem/cambial feeder) and hemlock woolly adelgid (*Adelges tsugae*, a sucking insect), while the black oak gall wasp, along with myriad species of defoliators, might be expected to decline. For the emerald ash borer, some empirical evidence suggests increased success during drought, at least under controlled conditions. Further, even where insects respond only minimally to drought (e.g., defoliators such as the gypsy moth [*Lymantria dispar*] or spruce budworm [*Choristoneura* spp.]), the effects of repeated defoliation on tree growth and survival are likely to be higher under water stress (Davidson et al. 1999). Trees with repeated and/or severe defoliation are less able to respond physiologically to drought and to recover during periods of high water availability (Jacquet et al. 2014), representing an alternative pathway by which drought and insects may impact forests.

The gypsy moth is one insect that has shown clear, though primarily indirect, positive responses to drought. Introduced near Boston, MA, in the late 1860s, the gypsy moth has become one of the most damaging tree defoliators in the United States. Although temporal patterns of gypsy moth outbreaks have not shown obvious correlations with periods of reduced precipitation or water stress, limited evidence suggests that increased drought frequency or severity could affect their population dynamics. For example, the introduced biocontrol fungus, *Entomophaga maimaiga*, suppresses caterpillar populations in most years. However, it strongly depends on high humidity, especially in spring (Hajek and Webb 1999). Therefore, drought could substantially limit suppression by this important top-down control on gypsy moth populations. In 2015–2016, drought was correlated with gypsy moth outbreaks in a number of Eastern States, and this relationship was largely attributed to drought-related reduction in *Entomophaga maimaiga* infection during this period (Reilly et al. 2014).

As with insects, few empirical examples suggest that drought is currently a driver of disease dynamics in forests in the Northern Region. In general, however, obligate biotrophs (microbes that feed primarily on living plant tissue) often need periods of high humidity and/or soil or leaf surface moisture for infection to occur. Thus, these pathogens might be expected to decline in response to drought, while those that respond to tree stress (i.e., early colonizing saprophytes, often living endophytically within tree tissues) are more likely to increase (e.g., Desprez-Loustau et al. 2006, Kolb et al. 2016). Similar to effects of defoliation, biotrophs that successfully invade trees may cause higher rates of mortality because they deplete the energetic and/or nutrient reserves of their hosts under stress conditions.

The response of invasive plants to drought is also likely to be idiosyncratic and complex. Invasive plants tend to be fast-growing and vigorous which often correlates with high water requirements (even with high-efficiency water use). This relationship suggests that invasive plants may suffer during drought. However, invasive plants are also often characterized by high phenotypic plasticity (specifically, the ability to tolerate a wide range of abiotic conditions) and more efficient water use, and they are often strongly associated with disturbance (Cordell et al. 2002, Davidson et al. 2011, Funk 2013, Heberling and Fridley 2013). Thus, if future drought causes widespread tree decline and mortality, invasive plants could respond quickly to elevated nutrient pulses and reduced shade. Invasive plants would also probably benefit if fire becomes increasingly relevant in these systems, at least in certain parts of the range (Flory et al. 2015).

Socioeconomic Impacts of Drought

Drought can influence the character, quality, and species composition of forests as well as the timing of many management practices. These changes could affect local and regional economies that depend on forest products. For example, changes in forest composition in the Central Hardwoods region (southern Missouri, Illinois, Indiana, and Ohio) are projected to destroy wildlife habitat and cause steep declines in the value of timber (Ma et al. 2016).

The consequences of drought can also affect a variety of forest-based cultural traditions, tourism, recreation, and seasonal activities. For example, drought-induced changes in species composition could affect Tribal communities that depend on certain tree species for their culture and livelihoods, such as paper birch (*Betula papyrifera*), northern white cedar (*Thuja occidentalis*), and quaking aspen (*Populus tremuloides*) (Fisichelli et al. 2014; Handler et al. 2014a, 2014b; Janowiak et al. 2014). Droughts can lower water levels in lakes and streams, affecting recreational activities such as boating, swimming, and fishing. During winter, even short-term droughts can have large consequences for winter recreational activities (e.g., snowmobiling, skiing), which are often critical to the economy of rural communities in the Northern Region. Droughts can alter the timing and duration of autumn leaf color (Xie et al. 2015) as well as wildlife tourism (e.g., hunting, fishing, birding) affected by shifting habitats and altered migratory patterns (Rodenhouse et al. 2008, Thomas et al. 2013).

Maple syrup production, another economically and culturally important forest-based activity in the Northern Region, will likely be affected by increasingly frequent drought in the future. Historical trends in sugar maple (*Acer saccharum*) decline may be related to drought, frequently in association with other, often interacting stressors such as insects, pathogens, and nutrient deficiencies (Bishop et al. 2015, Pitel and Yanai 2014). Changes in snowpack depth can alter the timing and length of the growing season and the occurrence of soil freeze-thaw dynamics in early spring. These variables in turn affect sugar maple health (Brown et al. 2015, Hufkens et al. 2012), the technical and operational activities related to sugar maple management, and the quantity and quality of syrup produced (Duchesne and Houle 2014, Matthews and Iverson 2017, Skinner et al. 2010).

Forest management decisions should also take into consideration the logistical and technical challenges of drought. From one perspective, drought may offer some advantages to logging operations. Winter drought may be helpful to loggers because a shallower snowpack may improve access to tree boles. If air temperatures are sufficiently cold, winter drought would also promote the development of soil frost, which reduces erosion and compaction from logging operations. A shallower snowpack may also shorten the duration of the mud season that follows spring snowmelt, when logging operations are typically curtailed.

Future projections, however, suggest more precipitation in winter, with more rain than snow, and an intermittent snowpack. These conditions could lead to longer periods of high soil water content that are unfavorable for logging, causing compaction and affecting the stability of forest roads. Further, although frozen soil may be beneficial for logging access, it has negative ecological effects, resulting in root mortality (e.g., Tierney et al. 2001), nutrient leaching (e.g., Fitzhugh et al. 2001), and decreased plant productivity (e.g., Kreyling et al. 2012). In the past 70 years in the upper Midwest, with warming winters, the duration of snowpack or frozen ground conditions suitable for winter harvest has been shortened by 2 to 3 weeks (Rittenhouse and Rissman 2015). This trend has had economic impacts on the forest industry, where forest operations are limited by lack of snow cover or frozen ground conditions necessary to access sites and operate harvesting equipment. With less winter snow cover and frozen ground conditions, seasonal restrictions on forest operations have increased (Evans et al. 2016), resulting in economic consequences to both forest industry and woodland landowners through reduced timber values (Conrad et al. 2017).

Many socioeconomic factors will dictate the degree and extent to which management is able to influence the vulnerability of forests across the Eastern United States to future drought events. Forest ownership patterns across this region are complex, with family forest owners owning the vast majority of forested areas. Large public and private ownerships are also important, particularly in the Lake States, northern Maine, and the Adirondack region of New York. Given the wide variety of landowner objectives, this ownership pattern complicates how forest management aimed at increasing drought adaptation will occur. Similarly, many silvicultural treatments for increasing drought adaptation either require investments in management (i.e., planting, tending treatments) or rely on markets for lower grade materials, creating potential economic barriers to widespread implementation. For these reasons, the likelihood is high that the most common management response on privately owned forests, especially small ones, will be to do nothing to prepare for increased future risk of severe drought. The management options outlined in the following section, and summarized in table 8.2, are in part to encourage forest managers to consider more active approaches to drought preparedness. Table 8.3 gives examples of drought-related management strategies that have been implemented as part of the Northern Institute of Applied Climate Science's Climate Change Response Framework (https://forestadaptation.org/demos).

Table 8.2—Summary of potential management practices to reduce drought impacts and enhance resilience in forest stands in the Northern Region

MANAGEMENT PRACTICE	GUIDELINES	DESIRED OUTCOMES
Thinning	• Regional stocking guides or density management diagrams provide optimal density targets. • Avoid excessively heavy thinning: trees with very large crowns and high leaf area-to-sapwood area ratios may be more vulnerable to drought.	• Optimal stand densities support healthy trees and sufficient water availability during dry periods. • Thinned forests are often more drought-resilient than unthinned forests.
Natural or artificial regeneration	• Facilitate natural regeneration of adapted local genotypes or species via seedbed treatment and/or microclimate amelioration. • Plant seedlings of genotypes or species better adapted to moisture stress. • Use assisted migration to introduce new species from habitats representing future conditions.	• Silvicultural practices with natural or artificial regeneration help to shift composition towards more drought-adapted species or genotypes, establishing more resilient forests.
Carbon sequestration	• Enroll forests in carbon offset programs to provide an economic benefit while contributing to climate change mitigation.	• Forests in the region provide long-term, significant increases in carbon sequestration.

CASE STUDY 8.1

Providence Water: Adapting forests to drought

Providence Water, in Rhode Island, is managing forests to be better adapted to future drought conditions. In keeping with goals to maintain and protect water yield and water quality, the public water utility is managing for a diversity of species, selecting those that may best tolerate extended drought conditions, and actively planting tree species from southerly seed zones on selected experimental sites within the project area.

The Scituate Reservoir and five smaller tributary reservoirs are the primary drinking water sources to approximately 600,000 people. The reservoirs are surrounded by 5 261 ha of mostly forested public land (formerly agricultural lands) that serves as "green infrastructure" filtering surface runoff, acting as the first step ("first barrier" in water resources engineering parlance) in the water treatment process.

The woodlands surrounding the reservoir are currently experiencing hardwood regeneration failure due to pests and pathogens (e.g., red pine scale, red pine adelgid, gypsy moth, orange-striped oakworm, chestnut blight) along with intense herbivory pressures. Anticipated future shifts in climate may interact to increase severe weather events and drought risks, further challenging regeneration of local species. Warming and altered precipitation patterns may result in less winter snow and persistence of drier conditions later into the growing season. Prolonged warm, dry, and drought conditions may harm forest species unable to tolerate hotter and drier conditions. A changing climate is likely to intensify forest stressors, including insect pests, forest diseases, invasive plant species, and deer herbivory.

Shown in autumn 2014, this site on Providence, Rhode Island, Scituate Reservoir watershed property shows the effects of multiple forest health stressors, including dry conditions, deer herbivory, and insect pests. (Photo courtesy of Christopher Riely)

Providence Water is experimenting with actions that promote ecosystem transition to a diverse forest that could be better adapted to future conditions. Using the U.S. Department of Agriculture, Forest Service publication, "*Forest Adaptation Resources: Climate Change Tools and Approaches for Land Managers*," (Swanston et al. 2016; https://www.nrs.fs.fed.us/pubs/52760), Providence Water designed the following specific management actions to prepare forests for a changing climate:

- In oak forests with regeneration failure, guide changes in species composition by planting tree species expected to be better adapted to future conditions (e.g., black oak, black locust, white oak, pin oak, persimmon, sweetgum, eastern red cedar, sassafras, and loblolly, pitch, and shortleaf pines), and tend/treat tree seedlings as needed.
- Plant tree seedlings better adapted to expected future conditions in areas where Providence Water could manage herbivory and protect these seedlings, including an oak forest within an existing deer exclosure fence (constructed prior to the adaptation project).
- In upland oak stands, harvest declining and poor-quality trees, and conduct enrichment planting with future-adapted tree seedlings (e.g., black locust, black oak, chestnut oak, persimmon, shortleaf pine, sweetgum, Virginia pine, white oak).

Providence Water will monitor success of these tactics, going beyond the forest inventory data they were already collecting to assess deer browse impacts and the growth and survival of the planted future-adapted seedlings. Two sites on Providence Water land have been planted with future-adapted species, and plans for more planting are under consideration (see https://www.forestadaptation.org/providence).

Table 8.3—Management strategies for drought in the Northern Region[a]

DROUGHT MANAGEMENT THEME	MANAGEMENT GOAL	MANAGEMENT TACTIC	CASE STUDIES
Soil moisture	• Reduce competition for moisture, nutrients, and light. • Promote diverse age classes.	• Cut shelterwood with reserves to increase structural and species diversity while maintaining aspects of the mature forest. • Focus on removing crowded, damaged, or stressed trees. • Manage aspen in multiple blocks, with the goal of creating several age classes in 5-year increments.	**Massachusetts Dept. of Conservation & Recreation: Bristol Lot Timber Sale** (https://www.forestadaptation.org/bristol) **Gogebic County: Mosinee Grouse Enhanced Management System** (https://www.forestadaptation.org/node/544)
Heat- and drought-tolerant tree species	• Favor native species adapted to future conditions. • Introduce species expected to be adapted to future conditions.	• Harvest declining and poor-quality trees to improve the growth of the residual stand. • Conduct enrichment planting and seeding of tree species expected to be better adapted to future conditions.	**Florence County: Climate-informed Forest Restoration** (https://www.forestadaptation.org/flo-co) (case study 8.2) **Providence Water: Planting Future-Adapted Forests** (https://www.forestadaptation.org/providence) (case study 8.1)
Pest and pathogen pressures	• Maintain or improve the ability of forests to resist pests and pathogens.	• Created a mix of species, age classes, and stand structures to reduce the availability of host species for pests and pathogens (e.g., blight-resistant American chestnut [*Castanea dentata*] that is more resistant to gypsy moth). • Implement forest management practices to reduce the long-term effects of hemlock woolly adelgid and maintain stream shading.	**Massachusetts Dept. of Conservation & Recreation: Bristol Lot Timber Sale** (https://www.forestadaptation.org/bristol) **Trout Unlimited: Adapting the Riparian Areas and Water of the North River** (https://www.forestadaptation.org/tu-ne)
Herbivory	• Manage herbivory to promote regeneration of desired species.	• Plant tree species expected to be better adapted to future conditions within an existing deer exclosure.	**Providence Water: Planting Future-Adapted Forests** (https://www.forestadaptation.org/providence) (case study 8.1)
Invasive species	• Prevent introduction and establishment of invasive plant species; remove existing invasive species.	• Control existing invasive species; map and monitor populations of new invasive species across the property. • Seed logging trails after harvest to reduce erosion and prevent invasive species.	**Leopold Foundation: Leopold-Pine Island Important Bird Area** (https://www.forestadaptation.org/leopold)
Fire	• Restore or maintain fire in fire-adapted ecosystems. • Guide changes in species composition at early stages of stand development.	• Use prescribed fire to sustain a mixed-oak ecosystem and control invasive exotic or undesirable species.	**Massachusetts Dept. of Conservation & Recreation: Bristol Lot Timber Sale** (https://www.forestadaptation.org/bristol) **Leopold Foundation: Leopold-Pine Island Important Bird Area** (https://www.forestadaptation.org/leopold) **Michigan Dept. of Natural Resources: Barry State Game Area** (https://www.forestadaptation.org/Barry)

[a] Management strategies are likely to be case-specific and dependent on site characteristics and the values of the landowner (Northern Institute of Applied Climate Science's Climate Change Response Framework, https://forestadaptation.org/demos).

(*continued*)

Table 8.3 (*continued*)—Management strategies for drought in the Northern Region[a]

DROUGHT MANAGEMENT THEME	MANAGEMENT GOAL	MANAGEMENT TACTIC	CASE STUDIES
Shorter winters, altered harvest timing	• Reduce damage to soils and nutrient cycling. • Realign significantly disrupted ecosystems to meet expected future conditions.	• Reduce site impacts by using tracked equipment. • Protect soils to maintain water storage capacity by minimizing disturbance to sensitive areas (seeps or enriched areas) during harvest. • Prioritize areas most likely to support a summer harvest given ground conditions and potential costs.	**Vermont Land Trust: Increasing Opportunities for Sustainable Timber Harvest on the Atlas Timberlands** (https://www.forestadaptation.org/atlas)
Diversity and density management	• Promote diverse age classes. • Maintain and restore diversity of native species.	• Use variable-density thinning to improve structural and species diversity. • Diversify planting to improve species diversity in gaps and openings.	**Superior National Forest: Mesabi Project** (https://www.forestadaptation.org/mesabi)
Biological legacies	• Retain biological legacies.	• Retain habitat elements of the mature forest (e.g., mast production, vertical structural diversity, large-diameter trees).	**Massachusetts Dept. of Conservation & Recreation: Bristol Lot Timber Sale** (https://www.forestadaptation.org/bristol)
New mixes of native tree species	• Establish or encourage new mixes of native species.	• Use red pine and jack pine (*Pinus banksiana*) as nurse trees for oak plantings; harvest the pines as the oak establishes.	**Michigan Dept. of Natural Resources: Barry State Game Area** (https://www.forestadaptation.org/Barry)
Infrastructure for stream crossings	• Restore hydrology. • Design infrastructure to meet expected conditions.	• Assess and upgrade road-stream crossings to handle lower and higher peak streamflows and enhance aquatic organism passage. • Decommission roads to increase groundwater recharge.	**Chequamegon-Nicolet National Forest: Marengo and Twentymile Creek Watersheds** (https://www.forestadaptation.org/cnnf-water) **Monongahela National Forest: Lambert Restoration Project** (https://forestadaptation.org/LambertDemo) **Trout Unlimited: Adapting the Riparian Areas and Water of the North River** (https://forestadaptation.org/tu-ne)
Wildlife habitat	• Prioritize and maintain sensitive or at-risk species or communities. • Reduce landscape fragmentation. • Manage habitats over a range of sites and conditions.	• Establish a savanna complex of 60 ha in collaboration with adjacent landowners. • Enhance available habitat for migratory waterfowl available in dry fall migrations.	**Michigan Dept. of Natural Resources: Barry State Game Area** (https://www.forestadaptation.org/Barry) **Ducks Unlimited, Inc.: Improving Bottomland Hardwood Forest and Wetland Resiliency** (https://forestadaptation.org/BottomlandHardwoods)

[a] Management strategies are likely to be case-specific and dependent on site characteristics and the values of the landowner (Northern Institute of Applied Climate Science's Climate Change Response Framework, https://forestadaptation.org/demos).

CASE STUDY 8.2

Florence County, WI: Restoring a forest after drought

Florence County foresters manage more than 14 570 ha of forest land in northeast Wisconsin for timber production and a range of public uses such as hunting, fishing, and camping. The county is restoring 160 ha of forest lands that were significantly affected by drought and forest pests, with the goal of becoming better adapted to future drought conditions. Florence County contains large forested areas on sandy, low-fertility sites. The declining precipitation in northern Wisconsin over the past several decades has stressed forests, causing mortality in some areas. The stands selected for this project had experienced close to 90-percent mortality because of a combination of persistent drought and forest pest infestations (e.g., two-lined chestnut borer [*Agrilus bilineatus*]). Into the future, this site may continue to be susceptible to drought and forest health stressors due to sandy soils and a changing climate trending towards warmer temperatures, earlier snowmelt, and longer, drier growing seasons.

Florence County foresters are motivated to keep this area forested, so they worked with partners to use the online Adaptation Workbook (https://www.adaptationworkbook.org) to devise adaptation tactics to improve forest resilience to drought. Florence County foresters chose to salvage the stand, reserving healthy pockets of scrub oak and northern red oak. They conducted a large-scale planting of native species expected to be better adapted to future drought conditions (jack pine, red pine, and white pine in the uplands, and white pine and swamp white oak in lower, wetter areas). They also added wood-based soil amendments (wood ash and biochar) to 40 ha of the project area to improve soil water-holding capacity, nutrient exchange, and microbial communities.

This is the first large-scale field trial of soil amendments in midwestern forests. Monitoring is underway to measure the survival and growth of planted seedlings, as well as soil factors such as water-holding capacity, bulk density, soil pH, and cation exchange in soil amendment areas (Richard et al. 2018).

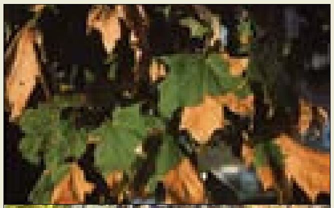
This project is a collaborative partnership with the Sustainable Resources Institute, Forest Service, Michigan Technological University, Wisconsin Department of Natural Resources, Verso Paper Corporation, and the Northern Institute of Applied Climate Science. Project funds were awarded through the Wildlife Conservation Society Climate Adaptation Fund in 2014. The support to establish the Climate Adaptation Fund was provided by the Doris Duke Charitable Foundation.

Florence County maintains dual certification under Sustainable Forestry Initiative (SFI) standard and the Forest Stewardship Council (FSC) standard.

Identifying symptoms of drought

Symptoms of tree drought stress can be difficult to identify because they may vary by species and location and look similar to symptoms of other stressors (e.g., insect pests, pathogens, nutrient deficiencies). Some key indicators are:

- Leaves turn from shiny to dull
- Loss of leaf turgor—wilted or drooping foliage
- Leaf scorch—leaves turn brown, often along the edges
- Chlorosis—paling or yellowing of green leaves
- Early fall color
- Premature leaf or needle drop
- Dieback of twigs or whole branches



Leaf scorch on sugar maple leaves. (Photo by Robert L. Anderson, USDA Forest Service)

Drought-stressed saplings begin to shed their leaves early in a Michigan forest. (Photo courtesy of USDA Forest Service)

DROUGHT MANAGEMENT OPTIONS AND CONSIDERATIONS

Thinning Treatments

The use of thinning has long been advocated as a strategy to maintain the growth and vigor of residual trees by reducing levels of resource competition in forest stands (Smith et al. 1997). Thinning is a proposed strategy to mitigate potential drought impacts in that it reduces moisture stress, thus minimizing growth declines and mortality (Aussenac and Granier 1988, Grant et al. 2013, Kohler et al. 2010, McDowell et al. 2006). Early experience with this strategy in U.S. forests was primarily in semi-arid regions (McDowell et al. 2006). However, recent studies from temperate forests in the Lake States and New England have demonstrated the benefit of density management to minimize growth declines during droughts and enhance postdrought recovery (Bottero et al. 2017, D'Amato et al. 2013, Gleason et al. 2017, Magruder et al. 2013). The ability to use thinning to minimize drought impacts in the Northern Region will hinge on the availability of markets for the low-grade materials that are often a large proportion of the volumes removed by these treatments. Thinning can also have unintended consequences, such as stimulating understory growth that may reduce soil water available for residual trees (Nilsen et al. 2001).

The effectiveness of thinning to mitigate drought impacts varies across regional aridity gradients of the Northern Region (i.e., from the Lake States [Michigan, Minnesota, Wisconsin]) to the Northeastern States. Overall, the greatest benefit of thinning has been observed in more arid climates. For example, research on effects of stand density on drought responses across pine-dominated forests suggests that thinning was more likely to reduce drought vulnerability on drier sites; however, thinned forest stands in temperate areas were also more resilient to drought than unthinned stands (Bottero et al. 2017). Similarly, drought had a greater effect on growth in thinned forests in the more arid midwestern forests than it did in New England (Gleason et al. 2017). Thinned, lower density stands had less depressed growth during drought in northeastern forests (northern hardwood and Acadian spruce-fir; Gleason et al. 2017). Thus, thinning may be an important management strategy to enhance resilience during drought, even in more humid parts of the Northern Region.

Beyond regional climate effects on thinning, forest developmental stage and structural conditions may also influence the effectiveness of thinning at reducing drought impact. For example, in a study of the long-term influence of density management on the drought resilience of red pine (*Pinus resinosa*) forests in Minnesota, stands thinned to very low densities (31–61 square feet per acre) were less affected by drought at young stand ages but were more vulnerable at older ages, relative to stands thinned to higher residual densities (92–153 square feet per acre; D'Amato et al. 2013) (fig. 8.2). This age-related shift in the benefits of thinning reflects the influence of early heavy thinning on long-term development of tree-level architecture: larger and older trees are often more vulnerable to drought (Skov et al. 2004). The greater drought vulnerability of larger, older trees in low-density stands has been attributed to their larger leaf areas and high leaf area-to-sapwood area ratios, which create water demands that are difficult to meet during drought periods (Kolb et al. 2007, McDowell et al. 2006). Increased allocation of biomass to crown development in response to greater resource availability has recently been linked to drought-related dieback around the globe (Jump et al. 2017).

These findings further underscore the potential vulnerability in the Northern Region, where sustained or severe drought has been largely absent over the last few decades: larger trees that have long experienced little drought stress are more vulnerable to future drought. Based on these and other findings, thinning to more moderate densities may be an effective strategy to reduce moisture stress and encourage the development of sustainable tree-level architecture. An encouraging finding, based on much of the research on thinning and drought, is that ideal densities for minimizing drought impacts correspond to the densities recommended by regional stocking guides and density management diagrams for generating optimal stand-level growth (Clark et al. 2016).

Artificial Regeneration of Adapted Genotypes or Species

One consequence of increased drought frequency and severity is that microclimate conditions may change in ways that limit natural regeneration by affecting processes of seed germination and seedling establishment. Local genotypes may also be maladapted to future climate conditions, limiting the potential for new seedlings to successfully regenerate following natural disturbance or harvesting. Further, the

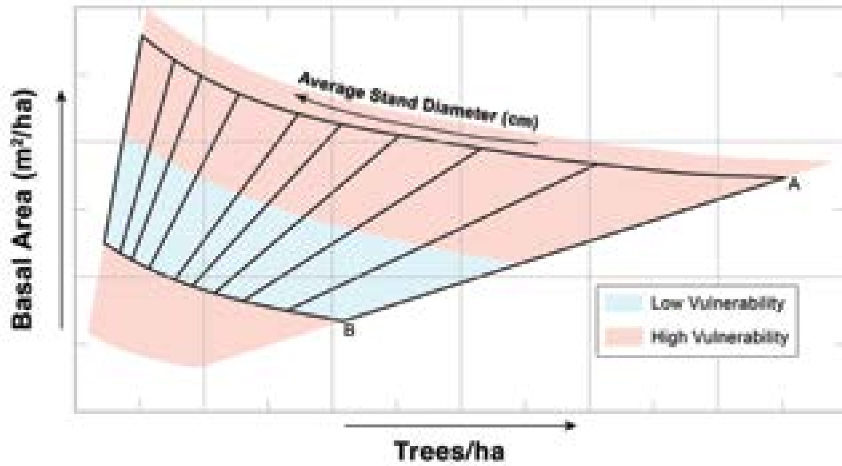


Figure 8.2—Generalized stocking guide showing zones of low and high drought vulnerability based on long-term research in red pine and northern hardwood forest ecosystems (D'Amato et al. 2013, Gleason et al. 2017). Zones of low vulnerability generally correspond to levels of residual stocking traditionally recommended for maintaining high levels of stand-level growth and vigor. Zones of high vulnerability correspond with highly stocked stand conditions in which inter-tree competition for resources causes drought-induced declines in growth, increased mortality, and low stocking conditions that favor tree-level architecture (high leaf area-to-sapwood area ratio) vulnerable to moisture stress.

projected rate of climate change will likely be greater than the migration rates of trees; thus, the potential for better adapted genotypes or tree species to move quickly enough to keep up with their bioclimatic envelope (i.e., future habitats that are suitable for their growth and survival) may also be severely limited (Dobrowski et al. 2013, Loarie et al. 2009). These potential impacts are especially relevant for forest management because sustainability of the production of timber and other forest products directly depends on the capacity of forest managers to successfully promote seedling regeneration and forest growth. Three forest management options for addressing these concerns are microclimate manipulation to facilitate natural regeneration, artificial regeneration of existing species, and assisted migration of non-local species, as well as a combination of these approaches (Grady et al. 2015).

Silvicultural treatments can improve microclimate conditions that favor seed germination, seedling establishment, and growth of desired species that have seed sources already present. For example, when conditions are safe, the seedbed can be improved using prescribed burning (Hutchinson et al. 2012, Iverson et al. 2017), manipulation of harvest residues and mulching (D'Amato et al. 2012), or mechanical scarification (Willis et al. 2015, Zaczek and Lhotka 2004). These treatments can facilitate access by roots to a stable moisture supply and reduce competition. To ameliorate moisture stress and buffer temperature extremes, additional shade can be provided by extending the period during which overstory trees are maintained on the stand (with shelterwood or variable retention harvest systems), especially during drought years and in the early phases of seedling establishment (Kellner and Swihart 2016). Moreover, given that forest stands with high species diversity may be more resilient to climate change, using silvicultural systems that promote high species diversity may enhance the sustainability of forest production under changing climate regimes. Examples include irregular shelterwood systems (Arseneault et al. 2011, Raymond and Bédard 2017) and adapting silvicultural treatments to existing environmental variation and species regeneration dynamics (e.g., Frey et al. 2007).

To promote the establishment of individuals that are more likely to survive and adapt to more frequent future drought, artificial regeneration can be used to seed or plant seedlings of genotypes or species considered better adapted to soil moisture deficit. Combining artificial regeneration with underplanting seedlings beneath existing canopies may provide additional benefits. Often, species expected to have greater success under future climate conditions are also intolerant to moderately intolerant of shade. As such, harvesting of overstory trees as part of a regeneration method should be included in silvicultural prescriptions that underplant these species. Another consideration when making decisions about artificial regeneration is that certain species, such as oaks, often have more limited success when planted compared to seedlings established by natural regeneration (Craig et al. 2014).

Assisted migration (i.e., assisted gene flow) involves the translocation of individual species or genotypes from outside a geographic region to facilitate adaptation of planted forests to climate change (Aitken and Bemmels 2016). Sources to identify promising genotypes to target for assisted migration plantings include information from provenance trials and knowledge about the environmental conditions within a species' distributional range (Aitken and Bemmels 2016, Aitken et al. 2008). For example, white pine (*Pinus strobus*) populations are predicted to decline in response to climate change. A proposed viable management response is to transfer white pine provenances from southern regions (e.g., Virginia) into more northern regions (e.g., Ontario) that are predicted to maintain habitats similar to the current distribution of those provenances (Joyce and Rehfeldt 2013).

Maintaining Timber Production

Models of forest productivity under a changing climate in the Northern Region generally show increases in net primary productivity (NPP) over the next century (Ollinger et al. 2008). However, those potential increases may be mitigated or even negated because of confounding and interacting factors, such as native and nonnative pests and pathogens, invasive plant species competition, disturbance from windthrow and ice storms, and increased drought stress (Rustad et al. 2012). The magnitude of potential impacts on forest productivity is uncertain, so landowners in the Northeast with timber objectives must adopt management strategies that facilitate the resilience of forest stands to a changing climate (e.g., Gunn et al. 2009). However, forest product markets and the ecological context in the Northern Region present some limitations on how forest managers can ameliorate consequences of drought.

The economic importance of the forest products sector in the Northern Region is well documented. The economic output of the forest products sector within the Midwest alone has been estimated at over $122 billion annually (Ballweg 2016, Deckard and Skurla 2011, DIS 2016, Henderson and Munn 2012, Leatherberry et al. 2006, Leefers 2017, McConnell 2012, Settle et al. 2016). Across the region, much of this economy was underpinned by the production of pulp and paper, which also supports a sawmill economy tied to the building sector. Between 2014 and 2017, closures of mills (and shutdown of paper machines within extant mills) and biomass power facilities have reduced the marketplace for low-grade wood by 40 percent in New England. Between 2006 and 2016, Minnesota has seen a similar decrease (-38 percent; MN DNR 2016). This state of the market for low-grade wood is compounded by a slow recovery of the housing market in the United States since bottoming out in 2007–2009 (U.S. Endowment for Forestry and Communities 2017).

Although the harvest of high-quality and high-value sawlog material for the building sector is fundamental to the bottom line of landowners and loggers, investment in growth and yield to improve silvicultural practices on investment ownerships has been minimal, implying that productivity is becoming a minor concern for landowners interested in timber value (D'Amato et al. 2018). For example, in Maine the acreage devoted to timber stand improvement and herbicide treatments remains low relative to the acreage harvested annually (Maine Forest Service 2017). This trend is further emphasized in a recent study that documented a lack of clear silvicultural goals for recent harvests throughout New England and New York (Belair and Ducey 2018). More than one-third of the recent harvesting in Maine can be categorized as using nonsilvicultural practices such as "commercial clearcut" or "high-grade" (Belair and Ducey 2018). Similar trends have been observed in studies examining family forest ownerships in the region (Maker et al. 2014). Still, global demand for forest products remains high, and the Northern Region forest sector infrastructure will likely be maintained for the foreseeable future, although perhaps at levels lower than in recent years (Levesque 2018). If the sector is to remain economically viable in the long term, forest managers will need to adjust management practices accordingly.

Silviculture in the Northern Region is rarely intensive, and investments in intermediate treatments are generally minimal. Therefore, management options for coping with drought, such as the density management option suggested earlier, are difficult to implement in the Northern Region because of the associated economic challenges. Industrial forest owners may be in the best position to execute more costly management strategies to mitigate the impacts of drought stress on productivity, such as shortening rotation durations and implementing thinning treatments. In contrast, small family forest owners may not have professional forestry assistance to support long-term management decisions in a changing climate with increased risk from drought (Butler et al. 2016). The management dynamics of forests in the Northern Region would need to change dramatically to implement practices that reduce impacts from drought and climate change in general.

Carbon Sequestration

Net sequestration of atmospheric carbon by forests in the United States offsets the equivalent of nearly 10 percent of carbon emissions from the transportation and energy sectors combined (Wear and Coulston 2015). Through conservation, restoration, and improved management, forests in the Northern Region have the potential to be even more influential in mitigating climate change (Griscom et al. 2017, Nave et al. 2018). Any potential increase in tree mortality and decrease in forest productivity places this crucial carbon sink at risk. Although this risk is important to understand for broader carbon accounting purposes, a more practical concern is the emerging carbon offset marketplace, which is a critical component of climate change mitigation efforts (Anderson et al. 2017).

Developing carbon markets and regional climate change policies allow emitters of greenhouse gases to offset their emissions through forest-based carbon sequestration projects. In 2015, the worldwide market in forest carbon offset trading was $761 million (Goldstein and Ruef 2016). Several significant transactions have occurred in the Northern Region on private forest lands that demonstrate the potential benefits of this market to landowners. For example, the Downeast Lakes Land Trust of Grand Lake Stream, ME, recently achieved the first formal forest carbon offset project verification in the Northern Forest, on 19,119 acres in eastern Maine. The project received an initial issuance of nearly 200,000 compliance-eligible carbon offsets, which are expected to have a value of over $2 million upon conversion to the California Air Resources Board (ARB) program. As this project illustrates, significant revenues could be available to landowners under specific circumstances. Identifying these opportunities requires a comprehensive understanding of how financial and legal risks are influenced by natural disturbances.

Each carbon offset program has its own rules and requirements for premature, intentional, and unintentional project termination (termed reversal). The Climate Action Reserve (Forest Project Protocol) requires a 100-year commitment to maintain stocks. To address reversal risk, a percentage of credits is set aside as a buffer in case of a reversal, based on a project-specific risk evaluation (this can be reduced further by the use of a qualified conservation easement). Offset projects using the American Carbon Registry require a 40-year commitment, and a project-specific risk assessment determines the amount of credits that must be placed in the buffer pool, secured from an approved alternate source of offsets, or the level of insurance coverage that must be purchased.

Participation in the carbon marketplace is limited by uncertainties surrounding the risk of reversal for a given project and compounded by the long commitment periods. The risk of reversal of carbon offset projects is influenced by at least three factors: (1) the severity, duration, and frequency of natural disturbances, including fire, insect damage, and severe weather; (2) the response of trees to increasing atmospheric CO_2 concentrations and changes in climatic conditions; and (3) landowner behavior (Galik and Jackson 2009). Landowner behavior can be addressed through legal mechanisms. However, to support both carbon offset project development and policies that seek to use forests as part of a regional climate mitigation strategy, more understanding is needed of reversal risk based on natural disturbance regimes in a changing climate (e.g., increased risk of ice storms, microbursts, and fire or stress related to severe summer droughts).

The nature of the standards and methodologies that govern how forest offset credits are generated may put these projects at risk. The most financially viable forest carbon offset projects involve forests that have higher than average biomass volume (i.e., carbon stocks) at the time of project initiation (Russell-Roy et al. 2014). These forests with larger and older trees are typically also at higher risk from drought stress (Skov et al. 2004). Uncertainty and risk are two major factors that may hinder more widespread use of this climate mitigation tool, even though it has many co-benefits for forest conservation and associated ecosystem services. Forest owners who want to engage in the carbon offset marketplace will need to develop and implement management strategies to minimize or mitigate that risk.

Fire Management and Risk

Historic fire data show a low probability of fire occurrence for the Northeast and upper Midwest, according to modeled outputs from limited fire scar data (Guyette et al. 2010, 2012). In these regions, high-severity fires occurred roughly every 30–75 years in much of the region, and the probability was much lower for certain ecosystems (e.g., up to 1,200 years for hemlock-white pine-northern hardwood forests [Whitney 1986] and more than every 800 years for northeastern spruce-fir forests [Lorimer 1977]).

Farther south within the Northern Region, the probability of fire occurrence was every 15–30 years. In recent times, fires throughout the region have become less frequent because of both human efforts to rapidly extinguish them and the highly fragmented nature of forest lands. Large fires can occur in the Eastern United States, however, as witnessed by the large complex of fires in the Great Smoky Mountains National Park and vicinity (2016) and significant fires in the White Mountain National Forest in New Hampshire (autumns of 2016 and 2017).

These kinds of exceptional fires seem to be on the rise worldwide. The 2017 wildfire season was especially unusual, with numerous severe fires occurring around the world, including Chile, the Mediterranean, Russia, Western North America, and even Greenland (Nature Climate Change Editors 2017). Climate projections, such as those presented in this document, indicate that the hot, dry conditions that facilitated these fires may become more common in the future. Thus, the forests of the Northeast and Midwest are likely to become more flammable. This flammability, with the very high human population density in the wildland interface, may increase the likelihood of fire, including catastrophic fire, challenging the institutional and infrastructural resources in place to manage them.

The fire season for the Northern Region tends to occur before leaf-out in spring, when solar radiation dries the forest floor. Wildfires in this region are generally small (<4 ha) and result from human activities, both intentional and unintentional, rather than by lightning (Cardille and Ventura 2001, Miranda et al. 2012, Peters and Iverson 2017). However, flash droughts, especially in the autumn, can lead to large fires, such as the Great Smoky Mountains National Park fires of 2016 (Wehner et al. 2017). Prescribed fire is used throughout the Northern Region—more so in the Midwest than in the more humid Northeast—as a tool to manage forests and savannas, often to promote oak and dissuade maples and other mesophytic species from dominating in the next forest (Brose et al. 2014). Using prescribed fire to restore communities is not easy, even in the drier Midwest. The burn windows are narrow, and multiple fires are often needed to obtain desired outcomes related to the diverse goods and services provided by forests (Hutchinson et al. 2012, Iverson et al. 2017).

Hydrological Functions and Services

Compared to other land uses, forests provide the cleanest and most stable supply of water for human uses (NRC 2008). In the Northern Region, the abundant, high-quality water filtered through forests serves multiple needs for residential, agricultural, industrial, and commercial uses, including drinking water, irrigation, recreation, wastewater assimilation, and power generation. Although severe droughts are relatively rare in the region, they affect water quality and quantity when they do occur because of the dense human population and the heavy reliance on water resources. Lakes, streams, and wetlands in forested watersheds are also critical habitat for many organisms and therefore enhance biodiversity. As drought severity progresses and surface waters dry out, water temperatures and nutrient concentrations increase, and refugia for aquatic species diminish (Vose et al. 2016). Because of the clear relationship between forests and water, management plans must be developed that sustain water resources. In some cases, such as watersheds that serve as a source of drinking water (see case study 8.1), the paramount forest management objective is to supply a sufficient amount of high-quality water for public consumption.

Linkages between forest management activities and streamflow and water quality have been evaluated comprehensively in the region (e.g., Brown et al. 2005, Hornbeck et al. 1986). Impacts of forest harvesting on streamflow are generally short-lived: increases in water yield seldom exceed 10 years after cutting (Hornbeck et al. 1993). Stream responses to cutting vary, depending on the harvesting intensity and site conditions (e.g., slope steepness, soil characteristics, forest cover type), and they can be extended with herbicide use and by making intermediate cuts. Transpiration rates of the regenerating forest generally recover rapidly, though, so harvesting practices are not typically considered a long-term, economically viable management strategy to minimize drought impacts on water yield. However, if the regenerating forest contains species with different transpiration rates or canopy interception than the forest it replaced, long-term effects on streamflow are possible (Hornbeck et al. 1993). Given the increased likelihood of both high and low flows in the future, it is improbable that forest managers will select for tree species based on transpiration and interception alone. Rather, a more tactical approach would involve establishing a diversity of species and age classes to ensure the continued functionality of forests under a broad range of conditions. For example, increasing biodiversity makes forests less vulnerable to insects and disease. Maintaining the structural diversity of forests, including trees of different age classes and levels of shade tolerance, can enhance recovery after disturbance.

Although managing water resources is typically not the primary forest management objective, many of the best forest management practices that have been established are designed to maintain water supply and quality. Practices that avoid compaction and promote infiltration act to reduce surface runoff and replenish groundwater supplies that sustain streamflow during dry periods. Leaving buffer strips along stream channels helps to maintain stream temperatures and reduce nutrient inputs. Further habitat protection can include the addition and retention of coarse woody debris in streams to establish pools that serve as refugia for aquatic organisms during droughts (Warren et al. 2010).

CONCLUSIONS

Although drought has not been a major concern for forest managers in the Northern Region in recent memory, climate change projections suggest that the frequency and severity of drought will likely increase in this region in the future, especially under "worst case" climate change scenarios. Our understanding of how different tree species and whole ecosystems will respond to greater moisture stress is limited, largely because of the historical lack of drought in the region. However, based on climate change projections, future forest responses to drought are likely. These could include mortality of more sensitive species, shifts in forest composition towards more drought-tolerant species, including exotic species, and potential migration of tree species into more suitable habitats outside of current geographic ranges. Such drought-related effects could in turn impact many forest provisions, including timber and nontimber products, water supply, carbon sequestration, wildlife habitat, and cultural benefits. Consequently, forest managers, landowners, and other stakeholders should consider a range of potential actions to mitigate and adapt to drought conditions. In this review, we highlighted a range of management options available to enhance the adaptive capacity and resilience of forest ecosystems to drought in the Northern Region, and we presented case studies to show where some of these activities have already been implemented. We also identified areas where knowledge is currently lacking, and where more targeted research is needed to better inform management decisions. Finally, a key theme throughout this chapter is that, even though the Northern Region is currently relatively wet and not moisture-limited, forest managers are likely to face new challenges related to water availability. Efforts should be directed at preparing forests for uncertain future conditions, while also taking measures to reduce the rate of climate change.

LITERATURE CITED

Abrams, M.D.; Nowacki, G.J. 2016. An interdisciplinary approach to better assess global change impacts and drought vulnerability on forest dynamics. Tree Physiology. 36: 421–427.

Aitken, S.N.; Bemmels, J.B. 2016. Time to get moving: assisted gene flow of forest trees. Evolutionary Applications. 9: 271–290.

Aitken, S.N.; Yeaman, S.; Holliday, J.A. [et al.]. 2008. Adaptation, migration or extirpation: climate change outcomes for tree populations. Evolutionary Applications. 1: 95–111.

Anderson, C.M.; Field, C.B.; Mach, K.J. 2017. Forest offsets partner climate-change mitigation with conservation. Frontiers in Ecology and the Environment. 15: 359–365.

Arseneault, J.E.; Saunders M.R.; Seymour R.S. [et al.]. 2011. First decadal response to treatment in a disturbance-based silviculture experiment in Maine. Forest Ecology and Management. 262: 404–412.

Aukema, J.E.; McCullough, D.G.; Von Holle, B. [et al.]. 2010. Historical accumulation of nonindigenous forest pests in the continental United States. BioScience. 60: 886–897.

Aussenac, G.; Granier, A. 1988. Effects of thinning on water stress and growth in Douglas-fir. Canadian Journal of Forest Research. 18: 100–105.

Ballweg, J. 2016. Forest economy Wisconsin. Madison, WI: Wisconsin Department of Natural Resources. 1 p. https://dnr.wi.gov/topic/ForestBusinesses/documents/factSheets/FactSheetWisconsin.pdf. [Date last accessed: January 2019].

Belair, E.; Ducey, M.J. 2018. Patterns in forest harvesting in New England and New York: using FIA data to evaluate silvicultural intent. Journal of Forestry. 116: 272–282.

Berdanier, A.B.; Clark, J.S. 2016. Multiyear drought-induced morbidity preceding tree death in southeastern U.S. forests. Ecological Applications. 26: 17–23.

Bishop, D.A.; Beier, C.M.; Pederson, N. [et al.]. 2015. Regional growth decline of sugar maple (*Acer saccharum*) and its potential causes. Ecosphere. 6: 1–14.

Bishop, D.A.; Pederson, N. 2015. Regional variation of transient precipitation and rainless-day frequency across a subcontinental hydroclimate gradient. Journal of Extreme Events. 2: 1550007.

Blake, R.; Khanbilvardi, R.; Rosenzweig, C. 2000. Climate change impacts on New York City's water supply system. Journal of the American Water Resources Association. 36: 279–292.

Bottero, A.; D'Amato, A.W.; Palik, B.J. [et al.]. 2017. Density-dependent vulnerability of forest ecosystems to drought. Journal of Applied Ecology. 54: 1605–1614.

Brandt, L.; He, H.; Iverson, L.; Thompson, F.R. [et al.]. 2014. Central Hardwoods ecosystem vulnerability assessment and synthesis: a report from the Central Hardwoods Climate Change Response Framework project. Gen. Tech. Rep. NRS-124. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 254 p.

Brose, P.H.; Dey, D.C.; Waldrop, T.A. 2014. The fire-oak literature of eastern North America: synthesis and guidelines. Gen. Tech. Rep. NRS-135. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 98 p.

Brown, A.E.; Zhang, L.; McMahon, T.A. [et al.]. 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. Journal of Hydrology. 310: 28–61.

Brown, L.J.; Lamhonwah, D.; Murphy, B.L. 2015. Projecting a spatial shift of Ontario's sugar maple habitat in response to climate change: a GIS approach. The Canadian Geographer. 59: 369–381.

Burakowski, E.A.; Wake, C.P.; Braswell, B. [et al.]. 2008. Trends in wintertime climate in the northeastern United States: 1965–2005. Journal of Geophysical Research. 113: D20114. doi:10.1029/2008JD009870.

Burns, D.A.; Klaus, J.; McHale, M.R. 2007. Recent climate trends and implications for water resources in the Catskill Mountain region, New York, USA. Journal of Hydrology. 336: 155–170.

Butler, B.J.; Hewes, J.H.; Dickinson, B.J. [et al.]. 2016. Family forest ownerships of the United States, 2013: findings from the USDA Forest Service's national woodland owner survey. Journal of Forestry. 114: 638–647.

Campbell, J.L.; Driscoll, C.T.; Pourmokhtarian, A. [et al.]. 2011. Streamflow responses to past and projected future changes in climate at the Hubbard Brook Experimental Forest, New Hampshire, USA. Water Resource Research. 47: W02514.

Campbell, J.L.; Ollinger, S.V.; Flerchinger, G.N. [et al.]. 2010. Past and projected future changes in snowpack and soil frost at the Hubbard Brook Experimental Forest, New Hampshire, USA. Hydrological Processes. 24: 2465–2480.

Campbell, J.L.; Rustad, L.E.; Boyer, E.W. [et al.]. 2009. Consequences of climate change for biogeochemical cycling in forests of northeastern North America. Canadian Journal of Forest Research. 39: 264–284.

Canham, C.D.; Murphy, L. 2016. The demography of tree species response to climate: seedling recruitment and survival. Ecosphere. 7: e01424.

Cardille, J.A.; Ventura, S.J. 2001. Occurrence of wildfire in the northern Great Lakes Region: effects of land cover and land ownership assessed at multiple scales. International Journal of Wildland Fire. 10: 145–154.

Chakraborty, S.; Whitehill, J.G.A.; Hill, A.L. [et al.]. 2014. Effects of water availability on emerald ash borer larval performance and phloem phenolics of Manchurian and black ash. Plant Cell and Environment. 37: 1009–1021.

Charney, N.D.; Babst, F.; Poulter, B. [et al.]. 2016. Observed forest sensitivity to climate implies large changes in 21st century North American forest growth. Ecology Letters. 19: 1119–1128.

Clark, J.S.; Iverson, L.; Woodall, C.W. [et al.]. 2016. The impacts of increasing drought on forest dynamics, structure, and biodiversity in the United States. Global Change Biology. 22: 2329–2352.

Clinton, B.D.; Boring, L.R.; Swank, W.T. 1993. Canopy gap characteristics and drought influences in oak forests of the Coweeta Basin. Ecology. 74: 1551–1558.

Collins, M.J. 2009. Evidence for changing flood risk in New England since the late 20th century. Journal of the American Water Resources Association. 45: 279–290.

Conrad, J.L.; Demchik, M.C.; Vokoun, M.M. [et al.]. 2017. Foresters' perceptions of the frequency, cost, and rationale for seasonal timber harvesting restrictions in Wisconsin. Forest Science. 63: 331–341.

Cook, E.R.; Jacoby, G.C. 1977. Tree-ring-drought relationships in the Hudson Valley, New York. Science. 198: 399–401.

Cordell, S.; Cabin, R.; Hadway, L.J. 2002. Physiological ecology of native and alien dry forest shrubs in Hawaii. Biological Invasions. 4: 387–397.

Craig, J.M.; Lhotka, J.M.; Stringer, J.W. 2014. Evaluating initial responses of natural and underplanted oak reproduction and a shade-tolerant competitor to midstory removal. Forest Science. 60: 1164–1171.

Crausbay, S.D.; Ramirez, A.R.; Carter, S.L. [et al.]. 2017. Defining ecological drought for the 21st century. Bulletin of the American Meteorological Society. 98: 2543–2550.

D'Amato, A.W.; Bradford, J.B.; Fraver, S. [et al.]. 2013. Effects of thinning on drought vulnerability and climate response in north temperate forest ecosystems. Ecological Applications. 23: 1735–1742.

D'Amato, A.W.; Jokela, E.J.; O'Hara, K.L.; Long, J.N. 2018. Silviculture in the United States: an amazing period of change over the past 30 years. Journal of Forestry. 116: 55–67.

D'Amato, A.W.; Segari, J.; Gilmore, D. 2012. Influence of site preparation on natural regeneration and understory plant communities within red pine shelterwood systems. Northern Journal of Applied Forestry. 29: 60–66.

Davidson, A.M.; Jennions, M.; Nicotra, A.B. 2011. Do invasive species show higher phenotypic plasticity than native species and, if so, is it adaptive? A meta-analysis. Ecology Letters. 14: 419–431.

Davidson, C.B.; Gottschalk, K.W.; Johnson, J.E. 1999. Tree mortality following defoliation by the European gypsy moth (*Lymantria dispar* L.) in the United States: a review. Forest Science. 45: 74–84.

Decision Innovation Solutions [DIS]. 2016. Economic contributions of Missouri agriculture and forestry. Decision Innovation Solutions. 30 p. https://agriculture.mo.gov/economicimpact/county-pdf/MissouriAgForestryEconomicContributionStudy.pdf. [Date last accessed: January 2019].

Deckard, D.L.; Skurla, J.A. 2011. Economic contributions of Minnesota's forest products industry - 2011 edition. Minnesota Department of Natural Resources - Division of Forestry. 18 p. files.dnr.state.mn.us/forestry/um/economiccontributionMNforestproductsindustry2011.pdf. [Date last accessed: January 2019].

Desprez-Loustau, M.; Marçais, B.; Nageleisen, L. [et al.]. 2006. Interactive effects of drought and pathogens in forest trees. Annals of Forest Science. 63: 597–612.

Dobrowski, S.Z.; Abatzoglou, J.; Swanson, A.K. [et al.]. 2013. The climate velocity of the contiguous United States during the 20th century. Global Change Biology. 19: 241–251.

Donat, M.G.; Alexander, L.V.; Yang, H. [et al.]. 2013. Updated analyses of temperature and precipitation extreme indices since the beginning of the twentieth century: the HadEX2 dataset. Journal of Geophysical Research: Atmospheres. 118: 2098–2118.

Druckenbrod, D.L.; Martin-Benito, D.; Orwig, D.A. [et al.]. 2019. Redefining temperate forest responses to climate and disturbance in the Eastern United States: new insights at the mesoscale. Global Ecology and Biogeography. 28: 557–575.

Duchesne, L.; Houle, D. 2014. Interannual and spatial variability of maple syrup yield as related to climatic factors. PeerJ. 2: e428.

Dudley, R.W.; Hodgkins, G.A. 2013. Historical groundwater trends in northern New England and relations with streamflow and climatic variables. Journal of the American Water Resources Association. 49: 1198–1212.

Dukes, J.S.; Pontius, J.; Orwig, D. [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, M.J.; Scheller, R.M. 2016. Measuring and managing resistance and resilience under climate change in northern Great Lake forests (USA). Landscape Ecology. 31: 669–686.

Duveneck, M.J.; Thompson, J.R.; Gustafson, E.J. [et al.]. 2017. Recovery dynamics and climate change effects to future New England forests. Landscape Ecology. 32: 1385–1397.

Easterling, D.R.; Kunkel, K.E.; Arnold, J.R. [et al.]. 2017. Precipitation change in the United States. In: Wuebbles, D.J.; Fahey, D.W.; Hibbard, K.A. [et al.], eds. Climate Science Special Report: Fourth National Climate Assessment, Volume I. Washington, DC: U.S. Global Change Research Program: 207–230.

Evans, A.M.; Lynch, M.; Clark, F. [et al.]. 2016. Economic and ecological effects of forest practices and harvesting constraints on Wisconsin's forest resources and economy. Madison, WI: Forest Stewards Guild. 120 p.

Fan, F.; Bradley, R.S.; Rawlins, M.A. 2015. Climate change in the Northeast United States: an analysis of the NARCCAP multimodel simulations. Journal of Geophysical Research: Atmospheres. 120: 10,569–10,592.

Fisichelli, N.A.; Abella, S.R.; Peters, M. [et al.]. 2014. Climate, trees, pests, and weeds: change, uncertainty, and biotic stressors in eastern U.S. National Park forests. Forest Ecology and Management. 327: 31–39.

Fitzhugh, R.D.; Driscoll, C.T.; Groffman, P.M. [et al.]. 2001. Effects of soil freezing disturbance on soil solution nitrogen, phosphorus, and carbon chemistry in a northern hardwood ecosystem. Biogeochemistry. 56: 215–238.

Flory, S.L.; Clay, K.; Emery, S.M. [et al.]. 2015. Fire and non-native grass invasion interact to suppress tree regeneration in temperate deciduous forests. Journal of Applied Ecology. 52: 992–1000.

Frey, B.R.; Ashton, M.S.; McKenna, J.J. [et al.]. 2007. Topographic and temporal patterns in tree seedling establishment, growth, and survival among masting species of southern New England mixed-deciduous forests. Forest Ecology and Management. 245: 54–63.

Funk, J.L. 2013. The physiology of invasive plants in low-resource environments. Conservation Physiology. 1: 1–17.

Galik, C.S.; Jackson, R.B. 2009. Risks to forest carbon offset projects in a changing climate. Forest Ecology and Management. 257: 2209–2216.

Garnas, J.R.; Houston, D.R.; Ayres, M.P. [et al.]. 2011. Disease ontogeny overshadows effects of climate and species interactions on population dynamics in a nonnative forest disease complex. Ecography. 35: 412–421.

Gerten, D.; Luo, Y.; Le Maire, G. [et al.]. 2008. Modelled effects of precipitation on ecosystem carbon and water dynamics in different climatic zones. Global Change Biology. 14: 2365–2379.

Gleason, K.E.; Bradford, J.B.; Bottero, A. [et al.]. 2017. Competition amplifies drought stress in forests across broad climatic and compositional gradients. Ecosphere. 8: e01849.

Goldstein, A.; Ruef, F. 2016. View from the understory: state of forest carbon finance 2016. Washington, DC: Forest Trends' Ecosystem Marketplace. 51 p. http://www.forest-trends.org/documents/files/doc_5388.pdf. [Date last accessed: January 2019].

Grady, K.C.; Kolb, T.E.; Ikeda, D.H. [et al.]. 2015. A bridge too far: cold and pathogen constraints to assisted migration of riparian forests. Restoration Ecology. 23: 811–820.

Grant, G.E.; Tague, C.L.; Allen, C.D. 2013. Watering the forest for the trees: an emerging priority for managing water in forest landscapes. Frontiers in Ecology and the Environment. 11: 314–321.

Griscom, B.W.; Adams, J.; Ellis, P.W. [et al.]. 2017. Natural climate solutions. Proceedings of the National Academy of Sciences. 114: 11,645–11,650.

Groffman, P.M.; Rustad, L.E.; Templer, P.H. [et al.]. 2012. Climate change effects are manifest in complex and surprising ways in the northern hardwood forest. BioScience. 62: 1056–1066.

Gu, L.; Pallardy, S.G.; Hosman, K.P. [et al.]. 2015. Drought-influenced mortality of tree species with different predawn leaf water dynamics in a decade-long study of a central U.S. forest. Biogeosciences. 12: 2831–2845.

Gunn, J.S.; Hagan, J.M.; Whitman, A.A. 2009. Forestry adaptation and mitigation in a changing climate: a forest resource manager's guide for the northeastern United States. Report ESRC-2009-1. Brunswick, ME: Manomet Center for Conservation Sciences. 16 p.

Guyette, R.; Spetich, M.A.; Dey, D.C. 2010. Developing and using fire scar histories in the Southern and Eastern United States. Joint Fire Science Program Research Project Report Project # 06-3-1-16. Final Report. 112 p. http://digitalcommons.unl.edu/jfspresearch/. [Date last accessed: January 2019].

Guyette, R.P.; Stambaugh, M.C.; Dey, D.C. [et al.]. 2012. Predicting fire frequency with chemistry and climate. Ecosystems. 15: 322–335.

Haavik, L.J.; Billings, S.A.; Guldin, J.M. [et al.]. 2015. Emergent insects, pathogens and drought shape changing patterns in oak decline in North America and Europe. Forest Ecology and Management. 354: 190–205.

Hajek, A.E.; Webb, R.E. 1999. Inoculative augmentation of the fungal entomopathogen *Entomophaga maimaiga* as a homeowner tactic to control gypsy moth (Lepidoptera: Lymantriidae). Biological Control. 14: 11–18.

Handler, S.; Duveneck, M.; Iverson, L. [et al.]. 2014a. Minnesota forest ecosystem vulnerability assessment and synthesis: a report from the Northwoods Climate Change Response Framework. Gen. Tech. Rep. NRS-133. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 228 p.

Handler, S.; Duveneck, M.J.; Iverson, L. [et al.]. 2014b. Michigan forest ecosystem vulnerability assessment and synthesis: a report from the Northwoods Climate Change Response Framework. Gen. Tech. Rep. NRS-129. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 229 p. http://www.treesearch.fs.fed.us/pubs/45688.

Hautier, Y.; Tilman, D.; Isbell, F. [et al.] 2015. Anthropogenic environmental changes affect ecosystem stability via biodiversity. Science. 348: 336–340.

Hayhoe, K.; VanDorn, J.; Croley, T. [et al.]. 2010. Regional climate change projections for Chicago and the U.S. Great Lakes. Journal of Great Lakes Research. 36: 7–21.

Hayhoe, K.; Wake, C.P.; Huntington, T.G. [et al.]. 2007. Past and future changes in climate and hydrological indicators in the U.S. Northeast. Climate Dynamics. 28: 381–407.

Heberling, J.M.; Fridley, J.D. 2013. Resource-use strategies of native and invasive plants in Eastern North American forests. New Phytologist. 200: 523–533.

Henderson, J.E.; Munn, I.A. 2012. Forestry in Illinois–the impact of the forest products industry on the Illinois economy: an input-output analysis. Illinois Forestry Development Council. 22 p. ifdc.nres.illinois.edu/wp-content/uploads/2013/10/illinois-forest-products-impact_2012.pdf. [Date last accessed: January 2019].

Hoch, G.; Richter, A.; Körner, C. 2003. Non-structural carbon compounds in temperate forest trees. Plant, Cell & Environment. 26: 1067–1081.

Hodgkins, G.A.; Dudley, R.W. 2006. Changes in late-winter snowpack depth, water equivalent, and density in Maine, 1926–2004. Hydrological Processes. 20: 741–751.

Hodgkins, G.A.; Dudley, R.W.; Huntington, T.G. 2003. Changes in the timing of high river flows in New England over the 20th century. Journal of Hydrology. 278: 244–252.

Hoffmann, W.A.; Marchin, R.M.; Abit, P. [et al.]. 2011. Hydraulic failure and tree dieback are associated with high wood density in a temperate forest under extreme drought. Global Change Biology. 17: 2731–2742.

Hornbeck, J.W.; Adams, M.B.; Corbett, E.S. [et al.]. 1993. Long-term impacts of forest treatments on water yield: a summary for northeastern USA. Journal of Hydrology. 150: 323–344.

Hornbeck, J.W.; Martin, C.W.; Pierce, R.S. [et al.]. 1986. Clearcutting northern hardwoods: effects on hydrologic and nutrient ion budgets. Forest Science. 32: 667–686.

Hufkens, K.; Friedl, M.A.; Keenan, T.F. [et al.]. 2012. Ecological impacts of a widespread frost event following early spring leaf-out. Global Change Biology. 18: 2365–2377.

Hutchinson, T.F.; Yaussy, D.A.; Long, R.P. [et al.]. 2012. Long-term (13-year) effects of repeated prescribed fires on stand structure and tree regeneration in mixed-oak forests. Forest Ecology and Management. 286: 87–100.

Isbell, F.; Craven, D.; Connolly, J. [et al.]. 2015. Biodiversity increases the resistance of ecosystem productivity to climate extremes. Nature. 526: 574.

Iverson, L.R.; Hutchinson, T.F.; Peters, M.P. [et al.]. 2017. Long-term response of oak-hickory regeneration to partial harvest and repeated fires: influence of light and moisture. Ecosphere. 8: e01642.

Iverson, L.R.; Peters, M.P.; Prasad, A.M.; Matthews, S.N. 2019. Analysis of climate change impacts on tree species of the Eastern US: results of DISTRIB-II modeling. Forests. 10(4): 302. doi:10.3390/f10040302.

Iverson, L.R.; Prasad, A.M.; Matthews, S.N. 2008a. Modeling potential climate change impacts on the trees of the Northeastern United States. Mitigation and Adaptation Strategies for Global Change. 13: 487–516.

Iverson, L.R.; Prasad, A.M.; Matthews, S.N. [et al.]. 2008b. Estimating potential habitat for 134 eastern U.S. tree species under six climate scenarios. Forest Ecology and Management. 254: 390–406.

Iverson, L.R.; Prasad, A.M.; Matthews, S.N. [et al.]. 2011. Lessons learned while integrating habitat, dispersal, disturbance, and life-history traits into species habitat models under climate change. Ecosystems. 14: 1005–1020.

Iverson, L.R.; Schwartz, M.W.; Prasad, A.M. 2004. How fast and far might tree species migrate under climate change in the Eastern United States? Global Ecology and Biogeography. 13: 209–219.

Jacquet, J.S.; Bosc, A.; O'Grady, A. [et al.]. 2014. Combined effects of defoliation and water stress on pine growth and non-structural carbohydrates. Tree Physiology. 34: 367–376.

Jactel, H.; Petit, J.; Desprez-Loustau, M.L. [et al.]. 2012. Drought effects on damage by forest insects and pathogens: a meta-analysis. Global Change Biology. 18: 267–276.

Janowiak, M.K.; D'Amato, A.W.; Swanston, C.W. [et al.]. 2018. New England and New York forest ecosystem vulnerability assessment and synthesis: a report from the New England Climate Change Response Framework. Gen. Tech. Rep. NRS-173. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 234 p. https://doi.org/10.2737/nrs-gtr-173.

Janowiak, M.K.; Iverson, L.R.; Mladenoff, D.J. [et al.]. 2014. Forest ecosystem vulnerability assessment and synthesis for northern Wisconsin and western Upper Michigan: a report from the Northwoods Climate Change Response Framework project. Gen. Tech. Rep. NRS-136. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 247 p.

Jenkins, M.A.; Pallardy, S.G. 1995. The influence of drought on red oak group species growth and mortality in the Missouri Ozarks. Canadian Journal of Forest Research. 25: 1119–1127.

Joyce, L.A.; Abatzoglou, J.T.; Coulson, D.P. 2018. Climate data for RPA 2020 assessment: MACAv2 (METDATA) historical modeled (1950–2005) and future (2006–2099) projections for the conterminous United States at the 1/24 degree grid scale. In: U.S. Department of Agriculture, Forest Service Research Data Archive. Fort Collins, CO: https://doi.org/10.2737/RDS-2018-0014. [Date accessed: June 21, 2018].

Joyce, D.G.; Rehfeldt, G.E. 2013. Climatic niche, ecological genetics, and impact of climate change on eastern white pine (*Pinus strobus* L.): guidelines for land managers. Forest Ecology and Management. 295: 173–192.

Jump, A.S.; Ruiz-Benito, P.; Greenwood, S. [et al.]. 2017. Structural overshoot of tree growth with climate variability and the global spectrum of drought-induced forest dieback. Global Change Biology. 23: 3742–3757.

Kellner, K.F.; Swihart, R.K. 2016. Timber harvest and drought interact to impact oak seedling growth and survival in the Central Hardwood Forest. Ecosphere. 7: e01473.

Kinloch, B.B. 2003. White pine blister rust in North America: Past and prognosis. Phytopathology. 93: 1044–1047.

Klein, T. 2014. The variability of stomatal sensitivity to leaf water potential across tree species indicates a continuum between isohydric and anisohydric behaviours. Functional Ecology. 28: 1313–1320.

Kohler, M.; Sohn, J.; Nägele, G.; Bauhus, J. 2010. Can drought tolerance of Norway spruce (*Picea abies* (L.) Karst.) be increased through thinning? European Journal of Forest Research. 129: 1109–1118.

Kolb, T.E.; Agee, J.K.; Fulé, P.Z. [et al.]. 2007. Perpetuating old ponderosa pine. Forest Ecology and Management. 249: 141–157.

Kolb, T.E.; Fettig, C.J.; Ayres, M.P. [et al.]. 2016. Observed and anticipated impacts of drought on forest insects and diseases in the United States. Forest Ecology and Management. 380: 321–334.

Koricheva, J.; Larsson, S.; Haukioja, E. 1998a. Insect performance on experimentally stressed woody plants: a meta-analysis. Annual Reviews in Entomology. 43: 195–216.

Koricheva, J.; Larsson, S.; Haukioja, E. [et al.]. 1998b. Regulation of woody plant secondary metabolism by resource availability: hypothesis testing by means of meta-analysis. Oikos. 83: 212–226.

Kreyling, J.; Haei, M.; Laudon, H. 2012. Absence of snow cover reduces understory plant cover and alters plant community composition in boreal forests. Oecologia. 168: 577–587.

Leatherberry, E.C.; Moser, K.W.; Perry, C. [et al.]. 2006. Iowa's forests 1999–2003 (part A). Resour. Bull. NC-266A. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Research Station. 84 p.

Leefers, L. 2017. Statewide report: forest products industries' economic contributions to Michigan's economy - 2017 Update. Lansing, MI: Michigan Department of Natural Resources, Forest Resources Division. 36 p. https://www.michigan.gov/documents/dnr/2016ForestProductsIndustriesContributions_535055_7.pdf. [Date accessed: April 30, 2019]

Levesque, C. 2018. Whither low-grade markets in the Northeast—silviculture be damned? New England Society of American Foresters News Quarterly. 79(1): 8–12. http://www.nesaf.org/userfiles/NQ_January_2018.pdf.

Levesque, M.; Andreu-Hayles, L.; Pederson, N. 2017. Water availability drives gas exchange and growth of trees in Northeastern U.S., not elevated CO^2 and reduced acid deposition. Scientific Reports. 7: 46158.

Liebhold, A.; Macdonald, W.; Bergdahl, D. [et al.]. 1995. Invasion by exotic forest pests: a threat to forest ecosystems. Forest Science. 41: 1–49.

Liénard, J.; Harrison, J.; Strigul, N. 2016. U.S. forest response to projected climate-related stress: a tolerance perspective. Global Change Biology. 22: 2875–2886.

Loarie, S.R.; Duffy, P.B.; Hamilton, H. [et al.] 2009. The velocity of climate change. Nature. 462: 1052.

Loewenstein, N.J.; Pallardy, S.G. 1998. Drought tolerance, xylem sap abscisic acid and stomatal conductance during soil drying: a comparison of canopy trees of three temperate deciduous angiosperms. Tree Physiology. 18: 431–439.

Lorimer, C.G. 1977. The presettlement forest and natural disturbance cycle of northeastern Maine. Ecology. 58: 139–148.

Lovett, G.; Canham, C.; Arthur, M. [et al.]. 2006. Forest ecosystem responses to exotic pests and pathogens in eastern North America. BioScience. 56: 395–405.

Ma, W.; Liang, J.; Cumming, J.R. [et al.]. 2016. Fundamental shifts of central hardwood forests under climate change. Ecological Modelling. 332: 28–41.

Magruder, M.; Chhin, S.; Palik, B. [et al.]. 2013. Thinning increases climatic resilience of red pine. Canadian Journal of Forest Research. 43: 878–889.

Maine Forest Service. 2017. 2016 silvicultural activities report. 8 p. http://www.maine.gov/dacf/mfs/publications/annual_reports.html. [Date accessed: January 9, 2018].

Maker, N.F.; Germain, R.H.; Anderson, N.M. 2014. Working woods: a case study of sustainable forest management on Vermont family forests. Journal of Forestry. 112: 371–380.

Mankin, J.S.; Smerdon, J.E.; Cook B.I. [et al.]. 2017. The curious case of projected twenty-first-century drying but greening in the American West. Journal of Climate. 30: 8689–8710.

Martín-Benito, D.; Cherubini, P.; del Río, M.; Cañellas, I. 2008. Growth response to climate and drought in *Pinus nigra* Arn. trees of different crown classes. Trees. 22: 363–373.

Martín-Benito, D.; Pederson, N. 2015. Convergence in drought stress, but a divergence of climatic drivers across a latitudinal gradient in a temperate broadleaf forest. Journal of Biogeography. 42: 925–937.

Martínez-Vilalta, J.; Poyatos, R.; Aguadé, D. [et al.]. 2014. A new look at water transport regulation in plants. New Phytologist. 204: 105–115.

Matthews, S.N.; Iverson, L.R. 2017. Managing for delicious ecosystem service under climate change: can United States sugar maple (*Acer saccharum*) syrup production be maintained in a warming climate? International Journal of Biodiversity Science, Ecosystem Services and Management. 13: 40–52.

Matthews, S.N.; Iverson, L.R.; Prasad, A.M. [et al.]. 2011. Modifying climate change habitat models using tree species-specific assessments of model uncertainty and life history-factors. Forest Ecology and Management. 262: 1460–1472.

Mattson, W.J.; Haack, R.A. 1987. The role of drought in outbreaks of plant-eating insects. BioScience. 37: 110–118.

McConnell, E. 2012. Ohio's forest economy. Columbus, OH: School of Environment and Natural Resources, The Ohio State University. 8 p. https://ohiowood.osu.edu/images/OH_Forest_Economy.pdf.

McDowell, N.G.; Adams, H.D.; Bailey, J.D. [et al.]. 2006. Homeostatic maintenance of ponderosa pine gas exchange in response to stand density changes. Ecological Applications. 16: 1164–1182.

McEwan, R.W.; Dyer, J.M.; Pederson, N. 2011. Multiple interacting ecosystem drivers: toward an encompassing hypothesis of oak forest dynamics across eastern North America. Ecography. 34: 244–256.

Millers, I.; Shriner, D.S.; Rizzo, D. 1989. History of hardwood decline in the Eastern United States. Gen. Tech. Rep. NE-126. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 75 p.

Minnesota Department of Natural Resources [MN DNR]. 2016. Minnesota's forest resources 2016. Minnesota Department of Natural Resources. 101 p. http://files.dnr.state.mn.us/forestry/um/forest-resources-report-2016.pdf. [Date last accessed: January 2019].

Miranda, B.R.; Sturtevant, B.R.; Stewart, S.I. [et al.]. 2012. Spatial and temporal drivers of wildfire occurrence in the context of rural development in northern Wisconsin, USA. International Journal of Wildland Fire. 21: 141–154.

Morin, X.; Fahse, L.; de Mazancourt, C. [et al.]. 2014. Temporal stability in forest productivity increases with tree diversity due to asynchrony in species dynamics. Ecology Letters. 17: 1526–1535.

Moss, R.; Babiker, W.; Brinkman, S. [et al.]. 2008. Towards new scenarios for the analysis of emissions, climate change, impacts, and response strategies. Technical Summary Rep. 978-92-9169-124-1. Geneva: Intergovernmental Panel on Climate Change. 25 p.

National Climatic Data Center [NCDC]. 2017. Climate data online. National Climatic Data Center. http://www.ncdc.noaa.gov/cdo-web/. [Date accessed: November 30, 2017].

National Research Council [NRC]. 2008. Hydrologic effects of a changing forest landscape. Washington, DC: National Research Council. The National Academies Press. https://doi.org/10.17226/12223.

Nature Climate Change Editors. 2017. Spreading like wildfire. Nature Climate Change. 7: 755.

Nave, L.E.; Domke, G.M.; Hofmeister, K.L. [et al.]. 2018. Reforestation can sequester two petagrams of carbon in U.S. topsoils in a century. Proceedings of the National Academy of Sciences. 115: 2776–2781.

Niinemets, Ü.; Valladares, F. 2006. Tolerance to shade, drought, and waterlogging of temperate Northern Hemisphere trees and shrubs. Ecological Monographs. 76: 521–547.

Nilsen, E.T.; Clinton, B.D.; Lei, T.T. [et al.]. 2001. Does *Rhododendron maximum* L. (Ericaceae) reduce the availability of resources above and belowground for canopy tree seedlings? American Midland Naturalist. 145: 325–343.

Novotny, E.V.; Stefan, H.G. 2007. Stream flow in Minnesota: indicator of climate change. Journal of Hydrology. 334: 319–333.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the Eastern United States. BioScience. 58: 123–138.

Nowacki, G.J.; Abrams, M.D. 2015. Is climate an important driver of post-European vegetation change in the Eastern United States? Global Change Biology. 21: 314–334.

Ollinger, S.V.; Goodale, C.L.; Hayhoe, K. [et al.]. 2008. Potential effects of climate change and rising CO_2 on ecosystem processes in northeastern U.S. forests. Mitigation and Adaptation Strategies for Global Change. 13: 467–485.

Orwig, D.A.; Abrams, M.D. 1997. Variation in radial growth responses to drought among species, site, and canopy strata. Trees. 11: 474–484.

Oswalt, S.N.; Smith, W.B.; Miles, P.D. [et al.]. 2014. Forest resources of the United States, 2012: a technical document supporting the Forest Service 2015 update of the RPA Assessment. Gen. Tech. Rep. WO-91. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 218 p. https://doi.org/10.2737/wo-gtr-91.

Palmer, W.C. 1965. Meteorological drought report. Weather Bureau Research Paper No. 45. Washington, DC: U.S. Department of Commerce. 58 p.

Pedersen, B.S. 1998. The role of stress in the mortality of midwestern oaks as indicated by growth prior to death. Ecology. 79: 79–93.

Pederson, N.; Bell, A.R.; Cook, E.R. [et al.]. 2013. Is an epic pluvial masking the water insecurity of the greater New York City region? Journal of Climate. 26: 1339–1354.

Pederson, N.; D'Amato, A.W.; Dyer, J.M. [et al.]. 2015. Climate remains an important driver of post-European vegetation change in the Eastern United States. Global Change Biology. 21: 2105–2110.

Pederson, N.; Dyer, J.M.; McEwan, R.W. [et al.]. 2014. The legacy of episodic climatic events in shaping temperate, broadleaf forests. Ecological Monographs. 84: 599–620.

Peters, M.P.; Iverson, L.R. 2017. Incorporating fine-scale drought information into an Eastern U.S. wildfire hazard model. International Journal of Wildland Fire. 26: 393–398.

Peters, M.P.; Iverson, L.R.; Matthews, S.N. 2015. Long-term droughtiness and drought tolerance of eastern U.S. forests over five decades. Forest Ecology and Management. 345: 56–64.

Pitel, N.E.; Yanai, R.D. 2014. Abiotic and biotic factors influencing sugar maple health: Soils, topography, climate, and defoliation. Soil Science Society of America Journal. 78: 2061–2070.

Pourmokhtarian, A.; Driscoll, C.T.; Campbell, J.L. [et al.]. 2017. Modeled ecohydrological responses to climate change at seven small watersheds in the Northeastern United States. Global Change Biology. 23: 840–856.

Ratcliffe, S.; Wirth, C.; Jucker, T. [et al.]. 2017. Biodiversity and ecosystem functioning relations in European forests depend on environmental context. Ecology Letters. 20: 1414–1426.

Raymond, P.; Bédard, S. 2017. The irregular shelterwood system as an alternative to clearcutting to achieve compositional and structural objectives in temperate mixedwood stands. Forest Ecology and Management. 398: 91–100.

Reilly, J.R.; Hajek, A.E.; Liebhold, A.M.; Plymale, R. 2014. Impact of *Entomophaga maimaiga* (Entomophthorales: Entomophthoraceae) on outbreak gypsy moth populations (Lepidoptera: Erebidae): the role of weather. Environmental Entomology. 43: 632–641.

Richard, R.P.; Potvin, L.R.; Kane, E.S. [et al.]. 2018. Biochar and wood ash amendments for forestry in the lake states: field report and initial results. Journal of Forestry. 116(3): 222–227.

Richardson, A.D.; Carbone, M.S.; Keenan, T.F. [et al.]. 2013. Seasonal dynamics and age of stemwood nonstructural carbohydrates in temperate forest trees. New Phytologist. 197: 850–861.

Rittenhouse, C.D.; Rissman, A.R. 2015. Changes in winter conditions impact forest management in north temperate forests. Journal of Environmental Management. 149: 157–167.

Rodenhouse, N.L.; Matthews, S.N.; McFarland, K.P. [et al.]. 2008. Potential effects of climate change on birds of the Northeast. Mitigation and Adaptation Strategies for Global Change. 13: 517–540.

Rodríguez-Catón, M.; Villalba, R.; Srur, A.M. [et al.]. 2015. Long-term trends in radial growth associated with *Nothofagus pumilio* forest decline in Patagonia: integrating local- into regional-scale patterns. Forest Ecology and Management. 339: 44–56.

Rogers, B.M.; Jantz, P.; Goetz, S.J. 2017. Vulnerability of eastern U.S. tree species to climate change. Global Change Biology. 23: 3302–3320.

Roman, D.T.; Novick, K.A.; Brzostek, E.R. [et al.]. 2015. The role of isohydric and anisohydric species in determining ecosystem-scale response to severe drought. Oecologia. 179: 641–654.

Roy, G.; Larocque, G.R.; Ansseau, C. 2004. Retrospective evaluation of the onset period of the visual symptoms of dieback in five Appalachian sugar maple stand types. The Forestry Chronicle. 80: 375–383.

Russell-Roy, E.T.; Keeton, W.S.; Pontius, J.A. [et al.]. 2014. Rehabilitation forestry and carbon market access on high-graded northern hardwood forests. Canadian Journal of Forest Research. 44: 614–627.

Rustad, L.; Campbell, J.; Dukes, J.S. [et al.]. 2012. Changing climate, changing forests: the impacts of climate change on forests of the Northeastern United States and eastern Canada. Gen. Tech. Rep. NRS-99. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 48 p.

Settle, J.; Gonso, C.; Seidl, M. 2016. Indiana's hardwood industry: its economic impact. Indianapolis, IN: Indiana Department of Natural Resources. 25 p.

Skinner, C.B.; DeGaetano, A.T.; Chabot, B.F. 2010. Implications of twenty-first century climate change on Northeastern United States maple syrup production: impacts and adaptations. Climatic Change. 100: 685–702.

Skov, K.R.; Kolb, T.E.; Wallin, K.F. 2004. Tree size and drought affect ponderosa pine physiological response to thinning and burning treatments. Forest Science. 50: 81–91.

Smith, D.M.; Larson, B.C.; Kelty, M.J. [et al.]. 1997. The practice of silviculture. 9th ed. New York: John Wiley. 960 p.

Smith, M.D. 2011. An ecological perspective on extreme climatic events: a synthetic definition and framework to guide future research. Journal of Ecology. 99: 656–663.

Spetich, M.A. 2004. Forest dynamics at the epicenter of an oak decline event in the Boston Mountains, Arkansas—year one. In: Yaussy, D.A.; Hix, D.M.; Long, R.P.; Goebel, P.C., eds. Proceedings, 14th central hardwood forest conference. Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station: 21–26.

Stahle, D.W.; Cook, E.R.; Cleaveland, M.K. [et al.]. 2000. Tree-ring data document 16th century megadrought over North America. EOS, Transactions American Geophysical Union. 81: 121–125.

Stahle, D.W.; Fye, F.K.; Cook, E.R. [et al.]. 2007. Tree-ring reconstructed megadroughts over North America since a.d. 1300. Climatic Change. 83: 133.

Stringer, J.W.; Kimmerer, T.W.; Overstreet, J.C. [et al.]. 1989. Oak mortality in eastern Kentucky. Southern Journal of Applied Forestry. 13: 86–91.

Swanston, C.; Brandt, L.A.; Janowiak, M.K. [et al.]. 2017. Vulnerability of forests of the Midwest and Northeast United States to climate change. Climatic Change. 146: 103–116.

Swanston, C.W.; Janowiak, M.K.; Brandt, L.A. [et al.]. 2016. Forest adaptation resources: climate change tools and approaches for land managers, 2nd ed. Gen. Tech. Rep. NRS-87-2. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 161 p. http://dx.doi.org/10.2737/NRS-GTR-87-2.

Sweet, S.K.; Wolfe, D.W.; DeGaetano, A. [et al.]. 2017. Anatomy of the 2016 drought in the Northeastern United States: implications for agriculture and water resources in humid climates. Agricultural and Forest Meteorology. 247: 571–581.

Thomas, D.S.K.; Wilhelmi, O.V.; Finnessey, T.N. [et al.]. 2013. A comprehensive framework for tourism and recreation drought vulnerability reduction. Environmental Research Letters. 8: 044004.

Tierney, G.L.; Fahey, T.J.; Groffman, P. [et al.]. 2001. Soil freezing alters fine root dynamics in a northern hardwood forest. Biogeochemistry. 56: 175–190.

Tilman, D. 1999. The ecological consequences of changes in biodiversity: a search for general principles. Ecology. 80: 1455–1474.

U.S. Census Bureau. 2018. Annual estimates of the resident United States population for the United States, regions, States, and Puerto Rico: April 1, 2010 to July 1, 2018. [Online database]. https://www.census.gov/newsroom/press-kits/2018/pop-estimates-national-state.html. [Date accessed: May 3, 2019].

U.S. Endowment for Forestry and Communities. 2017. Final report: The Blue Ribbon Commission on Forest and Forest Products Research & Development in the 21st Century. 21 p. http://www.usendowment.org/images/blue_ribbon_commission_report-web.pdf. [Date accessed: January 9, 2018].

Van Loon, A.F. 2015. Hydrological drought explained. Wiley Interdisciplinary Reviews: Water. 2: 359–392.

Voelker, S.L.; Muzika, R.M.; Guyette, R.P. 2008. Individual tree and stand level influences on the growth, vigor, and decline of red oaks in the Ozarks. Forest Science. 54: 8–20.

Vose, J.M.; Miniat, C.F.; Luce, C.H. [et al.]. 2016. Ecohydrological implications of drought for forests in the United States. Forest Ecology and Management. 380: 335–345.

Warren, D.R.; Mineau, M.M.; Ward, E.J. [et al.]. 2010. Relating fish biomass to habitat and chemistry in headwater streams of the northeastern United States. Environmental Biology of Fishes. 88: 51–62.

Wear, D.N.; Coulston, J.W. 2015. From sink to source: regional variation in U.S. forest carbon futures. Scientific Reports. 5: 16518.

Weed, A.S.; Ayres, M.P.; Hicke, J.A. 2013. Consequences of climate change for biotic disturbances in North American forests. Ecological Monographs. 83(4): 441-470.

Wehner, M.F.; Arnold, J.R.; Knutson, T. [et al.]. 2017. Climate Science Special Report: Fourth National Climate Assessment, Volume I. Washington, DC: U.S. Global Change Research Program: 231–256.

Whitney, G.G. 1986. Relation of Michigan's presettlement pine forests to substrate and disturbance history. Ecology. 67: 1548–1559.

Wilhite, D.A.; Glantz, M.H. 1985. Understanding the drought phenomenon: the role of definitions. Water International. 10: 111–120.

Williams, A.P.; Michaelsen, J.; Leavitt, S.W. [et al.]. 2010. Using tree rings to predict the response of tree growth to climate change in the continental United States during the twenty-first century. Earth Interactions. 14: 1–20.

Willis, J.L.; Walters, M.B.; Gottschalk, K.W. 2015. Scarification and gap size have interacting effects on northern temperate seedling establishment. Forest Ecology and Management. 347: 237–247.

Xie, Y.; Wang, X.; Silander, J.A. 2015. Deciduous forest responses to temperature, precipitation, and drought imply complex climate change impacts. Proceedings of the National Academy of Sciences. 112: 13,585–13,590.

Zaczek, J.J.; Lhotka, J.M. 2004. Seedling reproduction established with soil scarification within an oak overwood after overstory removal. Northern Journal of Applied Forestry. 21: 5–11.

Zeppel, M.J.B.; Wilks, J.V.; Lewis, J.D. 2014. Impacts of extreme precipitation and seasonal changes in precipitation on plants. Biogeosciences. 11: 3083–3093.

Zhu, K.; Woodall, C.W.; Clark, J.S. 2012. Failure to migrate: lack of tree range expansion in response to climate change. Global Change Biology. 18: 1042–1052.