PRESCRIBED FIRE, CLIMATE CHANGE, AND THE TRANSFORMATION OF OAK SOCIAL-ECOLOGICAL COMPLEX ADAPTIVE SYSTEMS

By

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A DISSERTATION

Submitted to
Michigan State University
in partial fulfillment of the requirements
for the degree of

Fisheries and Wildlife – Doctor of Philosophy

2021

ABSTRACT

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In the Anthropocene, human influence over wildlife and wild places is growing in intensity and speed. Wildlife and wild places are changing, and those changes feedback to affect humans in complex and sometimes counterintuitive ways. This feedback is most acute to those land managers who manage specific landscapes for specific wildlife, and it is critical that wildlife professionals develop a better understanding of changing wild systems. What makes systems wild, and how can wildness be protected or even cultivated in a conservation context? Even more important than keeping wildlife wild is keeping the feedback wild and understanding what makes a social-ecological system more wild or less wild. In this dissertation I build a model of an adaptive social-ecological system, and I apply it to the challenge of managing transforming oak ecosystems on state game areas in southern Michigan.

First, I interviewed land managers within the Michigan Department of Natural Resources and the Michigan Natural Features Inventory to develop a detailed and nuanced qualitative assessment of oak management and barriers to prescribed fire use on state lands in southern Michigan. From these interviews I developed a series of hypotheses in the form of simple causal loop diagrams, a common approach in systems dynamics modeling. These models provide four explanations for the persistent pattern of under-use of prescribed fire to manage oaks and savannas in southern Michigan.

Next, I developed an agent-based model of land managers using prescribed fire to manage oak ecosystems in a changing climate. The model illustrates how social interactions among agents stimulates some agents to adapt and blocks others from adapting to changes in the seasonal pattern of safe prescribed fire weather. Over time the number of burn days decreased because climate change increased danger of wildfire, adding another barrier to restoration of oak ecosystems.

I synthesized observations from the first two chapters with properties of complex adaptive systems to develop a description of wildness in social-ecological systems. In humid climates, like Michigan and much of the eastern United States, oak systems depend on human-mediated disturbance, and as such these systems are a useful example of an interdependent and interacting social-ecological system.

Important findings from this research include 1) the social system that causes fire exclusion is more complex than a lack of value or understanding of oak ecosystems by land managers, 2) both wet and dry regional climate models predict future fire weather than is poorer for prescribed fire because it will be too hot and dry to use fire safely, and 3) describing social-ecological systems as self-organizing provides a new paradigm to describe wildness of systems that include humans. Managing complex systems in the Anthropocene de-emphasizes prediction and control and emphasizes respect and interactive adaptation.

For all who wander,
literally or intellectually,
and are not lost

ACKNOWLEDGEMENTS

I would like to thank all of my friends, family, and colleagues that made sacrifices big and small to make this dissertation journey possible. I would especially like to thank my family. You gave me support when I needed to focus, and you provided welcome distractions when I needed to take a break.

I would like to acknowledge two people who played key roles in this research, but who sadly passed on before the dissertation was complete. Mark MacKay was a colleague and friend at the Michigan DNR, Wildlife Division, who was a kind and skilled conservationist with a passion for oak conservation. This dissertation grew from several long talks that we had on this subject between 2012 and 2016. The seeds of this dissertation were planted by Mark. The other person I would like to thank is Dr. William F. Porter, who was my advisor through most of this dissertation journey. Bill gave me space to develop this research project in a direction that is not typical of wildlife research projects. I appreciate his trust, his keen insights, and his wisdom. He provided necessary nudges at just the right moments. I wish I could have seen Mark's and Bill's reactions to the final product.

I would like to thank my colleagues at the DNR who have not only made space for me to pursue this project, but who put up with the many times that I turned water cooler banter to deep discussions of oaks, fire, and complex systems. My supervisors at the Michigan Department of Natural Resources (DNR), Pat Lederle and Amy Derosier, deserve special thanks for making this journey possible. Mike Donovan, Jesse Bramer, and Maria Albright were always willing to discuss oak conservation and helped keep the research grounded in the reality of land management. My extended network of adaptation professionals outside the DNR has also helped me understand the larger context of climate adaptation and adaptive capacity, especially Gregor Schuurman, Eric Beever, Lindsey Thurman, Ben Zuckerberg, Stephen Handler, and Olivia LeDee.

I would also like to thank all my colleagues at Michigan State University, especially my committee members Jianguo (Jack) Liu, Arika Ligmann-Zielinska, David Williams, and especially Gary Roloff who agreed to step up to advise me when I needed an advisor to take the project across the finish line. I would like to thank the Boone and Crockett Club Michigan University Programs Committee for their support and encouragement for me, the Quantitative Wildlife Center (QWC), and all the graduate students therein. I could not have done this without the support of my current and former QWC lab mates who helped with everything from programming to providing excuses to visit the MSU Dairy Store (conveniently located next door to the Natural Resources Building).

I would like to thank all my teachers, formal and informal, especially those that instilled in me a love for the outdoors and a life-long passion for learning. I want to thank one teacher especially, whom I will call "Aslan," for cultivating a deeper understand of the uncertainty inherent in complex systems, and the idea that complex systems are not tame systems.

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Dissertation Introduction

Oak forests, woodlands, and savannas are valuable wildlife habitat for a wide array of species, including popular game species such as the wild turkey (*Meleagris gallopavo*) and rare species such as the federally endangered Karner blue butterfly (*Lycaeides melissa samuelis*). Oak ecosystems are high in overall biodiversity, and they provide a wide array of economic and cultural values to society. In southern Michigan, as in much of the Midwest, most land use is not forest; the remaining forests exist as fragments in a matrix of agriculture and urban development. The largest remaining blocks of forest in this geography are on public lands. In southern Michigan, these are state game areas and state parks, where oak forest is the most common forest type.

Oak forests in southern Michigan, as elsewhere throughout their range, are changing. Oak ecosystems were created and maintained by centuries of land management using prescribed fire. These fire-dependent ecosystems are threatened by nearly a century of aggressive fire exclusion and fire suppression[1], which interacts with a variety of other stressors (e.g., invasive species, browse pressure, etc.) to cause a conversion of the forest subcanopy from white oak (*Quercus alba*) and black oak (*Quercus velutina*) to red maple (*Acer rubrum*), black cherry (*Prunus serotina*), and other vegetation typically found on more mesic and less drought-prone soils. This ecological process of transforming oak forest to maple forest is often termed "mesophication." Given ongoing climate change, there is concern that replacement of

[1] Fire ecologists distinguish between fire suppression, which is the act of extinguishing wildfires as soon as possible after they are ignited, and fire exclusion, which is the attempt to reduce the number of human-caused fire ignitions. Both processes reduce the area and frequency of wildland fire. Sometimes, especially outside the fire science discipline, the term suppression is used to encompass both processes, but this dissertation is concerned with the ways the two processes interact. Thus, the terms will not be used interchangeably.

oaks with red maple may reduce the resilience of forests to future climates that will be drier and more prone to growing-season drought. Mesophication is usually described as an ecological process, but it has its origin in fire exclusion and suppression, which are social processes.

In the first chapter, I extended the ecological mesophication hypothesis to make explicit a series of social mesophication hypotheses. I used a series of ten semi-structured interviews with land managers in the Michigan Department of Natural Resources (MDNR) to develop a series of hypotheses of barriers to greater use of prescribed fire and oak restoration on state lands directly managed by the agency. Conceptually these hypotheses can act as locally relevant, social science extensions of the ecological mesophication hypothesis. Together the social and ecological hypotheses comprise a social-ecological systems model of fire exclusion and its ecological consequences. The goal of this chapter was to develop a more nuanced and detailed understanding of the complex reasons that the agency consistently uses fire less often than would be necessary to reverse mesophication, despite goals and language valuing oak regeneration and recruitment.

The second chapter builds on the insights from the first chapter, focusing on the social dynamic of many land managers that compete for scarce staff and equipment to conduct prescribed fires. The social process of competition and the ecological process of mesophication were simulated in an agent-based model (ABM) driven by daily fire weather indices derived from downscaled regional climate models. The second chapter models social-ecological mesophication in the context of empirical ecological data from state land in southern Michigan and models of climate change. Whereas the first chapter was a qualitative analysis, the use of an agent-based modeling framework allows a quantitative description of the land management system as a complex adaptive system.

The third chapter synthesizes the insights from the qualitative and quantitative analyses. It extends one of the social hypotheses from Chapter 1 that focuses on regeneration failures as climate-driven ecological transformations. It also builds on an observation from Chapter 2 that the third most common subcanopy species across the entire study area was autumn olive (*Elaeagnus umbellata*), an invasive non-native species. The ubiquity of a nonnative species in this ecosystem raises questions. How should managers and wild species interact within the context of social-ecological systems that are transforming into novel communities? What is wildness in the context of a social-ecological complex adaptive system, especially a system that is (like most places in the world) thoroughly influenced by local human influences and distant telecoupling? What level of prediction and control should land managers try to exert in a transforming and complex adaptive system? In the third chapter I attempt to synthesize the concepts of wildness and self-organizing complex systems.

Through these three chapters, I build a case for interacting with lands managed for wild species from a paradigm based on complex adaptive systems: first by building human interaction explicitly into the ecological mesophication hypothesis; second by building a quantitative simulation model of the system as a complex adaptive system; and then finally by considering the properties of complex adaptive systems that allow them to adapt, self-organize, and thus act as self-willed or wild social-ecological systems.

Chapter 1: Conceptualizing Oak Mesophication as a Social-Ecological Process

ABSTRACT

Ecosystems structured by frequent, low-intensity fire are imperiled worldwide. Fire exclusion and suppression are degrading oak ecosystems in Michigan and throughout the northeastern United States through a process called mesophication. Mesophication is the replacement of a drier, fire-dependent ecosystem with a mesic, or moderately moist ecosystem. Oak mesophication is usually framed as an ecological dysfunction, but the fundamental driver of mesophication is fire exclusion, which is a social phenomenon. I used semi-structured interviews with public land managers in southern Michigan to describe their perspectives on the social, process, and policy drivers of mesophication in oak systems that they manage. From those descriptions, I developed hypotheses on drivers of system dynamics that result in social-ecological mesophication. Land managers described issues regarding procedural fairness, risk perceptions, fire suppression outside the study area, and implicit assumptions that ecological transformation equates to management failure. Consideration of mesophication as both a social and ecological phenomenon elucidated new opportunities to intervene to address oak mesophication by focusing on policy, procedure, and agency culture.

INTRODUCTION

Contemporary oak ecosystems in the Great Lakes and northeastern United States developed from an interaction of human land management decisions regarding fire exclusion and suppression (Nowacki and Abrams 2008, Abrams and Nowacki 2015) mediated by meteorology from monthly weather scales (Goldblum 2010) to century climate scales (Iverson et al. 2008, Handler et al. 2014). Although there is disagreement regarding the relative importance of climate or culture in a given place or time (Cronon 1983, Stewart 2002), in the Great Lakes region the two interact to drive ecosystems toward either closed-canopy maple-dominated hardwood forests or open-canopy oak-dominated woodlands and savannas. Although soil moisture holding capacity plays an important role in mediating the rate at which systems shift toward closed-canopy or open-canopy systems, disturbance regimes play an important role. For example, the humid microclimate near the Great Lakes allows mesic hardwood forest to grow on dunes of deep sand with little water holding capacity. Conversely, mesic burr oak savanna is an extirpated ecological community of xeric prairie species on mesic loam soils, which depended on annual cultural burning to maintain its open structure (Cohen et al. 2015). Forest and savannas with oak dominating both the canopy and subcanopy are fire dependent ecosystems. The current predominance of closed canopy oak canopy with mesic maple-dominated understories represents a novel ecosystem (Fitzpatrick et al. 2018), which is transitioning toward a red maple (*Acer rubrum*) dominated ecosystem. From a fire-exclusion and suppression perspective, this process is called mesophication (Nowacki and Abrams 2008). In this chapter I focus on the change in mesic or xeric vegetation along disturbance regimes over time, specifically anthropogenic origin fire and climate origin drought, rather than the influence of soils on vegetation in space.

Fire Exclusion, Suppression, and Ecological Change

Fire is and has been a social-ecological process that structured ecosystems throughout the world for millennia. Socially, contemporary fire is viewed as both a threat and land management tool wherever climate and fuels support ignition and fire spread. Fires were and are used to manage fuels near settlements, create forage for wildlife or domestic animals, or to clear undergrowth to facilitate travel, hunting, or agriculture (Cronon 1983, Stewart 2002, Abrams and Nowacki 2015). Ecologically, many species are adapted to specific fire regimes (i.e., patterns of fire frequency and intensity); in some cases, plant species create conditions that favor ignition and spread to reduce competition from less fire-adapted species. For example, oak (*Quercus* spp) leaves on the forest flood are highly flammable and roll in air currents in ways that spread fire far more readily than other deciduous tree species (Brose et al. 2014).

Globally, biomes associated with frequent fire, such as temperate grasslands and savannas, are among the most imperiled (Hoekstra et al. 2005). Decades of fire exclusion and suppression initially reduced fire but ultimately created fuel conditions that have increased fire intensity and behavior in ways that are imperiling ecosystems adapted to frequent low-intensity fires. In xeric climates, such as the southwestern United States and parts of Australia, fire exclusion and suppression have allowed woody fuels to accumulate to dangerous levels. In that geography, infrequent high-intensity fires threaten societal values (e.g., homes, air quality, and human lives) and ecological components (e.g., soil productivity and biodiversity.) In humid climates, like eastern North America, fine fuels often decompose before the next season, and fire suppressed ecosystems become less fire prone over time (Brose et al. 2014).

In the southern Great Lakes region, threats to remnant fire-dependent ecosystems are primarily from mesophication, or the process of xeric sites becoming more mesic (Nowacki and Abrams 2008). Historically, frequent fires maintained a mosaic of oak woodland (forests with 50%–80% canopy closure), oak savannas (5%–60% canopy closure), and treeless prairies (Cohen et al. 2015, Williams et al. 2019). As fire frequency decreased, these systems converted to greater canopy closure. Prairies became savannas; savannas and woodlands transformed into closed canopy oak forests (Packard and Mutel 1997, Nowacki and Abrams 2008, Abrams and Nowacki 2015). Mesic, fire-sensitive species, such as red maple, previously relegated by frequent low intensity fires to wetlands, invaded fire suppressed uplands to initiate a novel ecosystem. This ongoing process of mesophication is problematic, especially on public lands, where agency mandates emphasize maintaining or restoring oak forests for their ecological integrity and wildlife habitat values.

The oak mesophication hypothesis proposes that failure of oak regeneration and the phenomenon of mesic hardwood regeneration under predominately oak overstories are 1) widespread in the eastern United States and 2) driven by decades of widespread fire exclusion (Nowacki and Abrams 2008). Supporting evidence for the hypothesis is provided by many studies, recently reviewed by Hanberry et al. (2020).

Within the oak mesophication hypothesis, the role of humans is explicit. Current oak forests originated from indigenous use of fire, were perpetuated by unregulated use of fire by European settlers and wildfire, with these forests subsequently becoming threatened by policies starting in the mid-1900s to aggressively reduce wildland fire ignitions and increase fire exclusion and suppression (Abrams and Nowacki 2019). Despite this explicit role of people in realization of the mesophication hypothesis, it is rarely presented as a social-ecological hypothesis, in which the social and ecological systems interact to create a system that is more than simply the sum of the social and ecological subsystems (Norberg and Cumming 2014). People and culture are mentioned, but social science and social science tools are rarely integrated into studies of mesophication.

One notable exception to the lack of social science applied to oak mesophication is Knoot et al. (2010) who used semi-structured interviews to explore opportunities and barriers that affect human attempts to manage for oak ecosystems on private lands in Wisconsin, USA. They found that private landowners generally preferred mature oak forest over young forest and were unwilling to sacrifice the aesthetic values of mature forest for the ecological values of oak regeneration, especially if the silvicultural tools involved expensive short-term investments (i.e., invasive species control or prescribed fire). The study concluded that social phenomena (e.g., values and economic cost) were major barriers to reducing mesophication.

Understanding social processes is critical to developing relevant strategies for effective intervention in the social-ecological system of oak mesophication. Indeed, in the context of changing ecosystems (whether driven by climate, fire exclusion, or some combination), it is critical to integrate ecology, social science, and conservation practice in developing strategies for system change that are both theoretically grounded and relevant in practice (Norberg and Cumming 2014, Bonebrake et al. 2018).

Barriers to Oak Regeneration on State Lands

Landscape condition tends to reflect the values of those who own or are responsible for those lands (Cronon 1983, 1996, Howell 2001), and a mismatch between values and goals of land stewards and current composition of the landscape is an intriguing anomaly. I proposed two hypotheses to explain this mismatch between management goals and landscape reality. My first hypothesis was that land managers do not use prescribed fire to address mesophication because they do not recognize that mesophication is occurring, or that mesophication threatens wildlife habitat goals on state lands. The other hypothesis was that use of prescribed fire was limited primarily by annual budgets and work plans. In scoping this research project, both hypotheses were repeatedly rejected by interviewees, who noted that use of prescribed fire did not relate to level of training regarding oak mesophication, or size of the annual operating budget. Interviewees felt that barriers to oak management, and particularly use of prescribed fire to restore oak dominated ecosystems, were more complex and nuanced than annual budgets or staff training.

My objectives in this study were: 1) to use semi-structured interviews and qualitative analysis to develop a nuanced understanding of the social and policy system on public lands management of oak ecosystems at the state level; 2) to document the diversity of perspectives and mental models related to oak management and prescribed fire among individuals and among sub-agencies with the Michigan Department of Natural Resources (MDNR); and 3) develop hypotheses of the social systems that perpetuate oak mesophication.

METHODS

Study Area

The study area comprises the southern half of Michigan's lower peninsula, from approximately latitude 43.8N south to the state border with Ohio and Indiana. This area is approximately 69,000 km^2. Within this region, I focused on state public lands, which comprise 2000 km^2, or 3% of the region. Private land uses are predominately agricultural and urban. Public lands are predominately forest, with small remnants of savanna and prairie.

Like fire-maintained oak forests elsewhere in eastern North America, decades of fire exclusion and suppression has resulted in mesophication of oak woodlands, savannas, and prairies in Michigan (Lee and Kost 2008, Allen et al. 2018, Cohen et al. 2021). Formerly open canopies have become closed canopy forests dominated by black oak (*Quercus velutina*), white oak (*Quercus alba*), and big toothed aspen (*Populus grandidentata;* Figure 1). The most common subcanopy trees are less drought-tolerant (i.e., mesic), less fire tolerant species, like red maple and cherry (*Prunus serotina*). On public lands in southern Michigan, the most common cover type, based on canopy, is oak (Figure 1), but the most common subcanopy tree is red maple, which is a relatively minor component of the canopy (Figure 2). Forest inventory data on public lands in southern Michigan (Michigan Department of Natural Resources 2019) provide clear evidence of mesophication of oak woodlands.

This landscape condition, which developed from decades of fire exclusion, contrasts with explicit goals of the two primary public land management types in southern Michigan: state parks and state game areas. State parks comprise 480 km^2 in southern Michigan. Managers of these lands have a dual mandate to provide recreational opportunities and to preserve natural landscapes (Michigan Department of Natural Resources 2017). On state-owned lands that comprise state parks, the circa 1800 cover type (Albert et al. 2008) was often oak woodland, savanna, or prairie; thus, management goals for those sites are often open oak ecosystems. State game areas comprise 1,600 km^2 in southern Michigan, and managers also have a dual mandate to provide recreational opportunities for hunters and to create or restore habitats for wildlife (Michigan Department of Natural Resources 2015). Oak ecosystems provide high quality habitat for several valued game species, including white-tailed deer (*Odolcoileus virginianus*) and turkey (*Meleagris gallopavo*). Oak savannas are habitat for the federally endangered Karner blue butterfly (*Lycaeides melissa samuelis*) and many other state listed wildflowers and pollinators.

All state game areas (N=44) and several of the state parks (N=13) in the region were recently inventoried using the Michigan Forest Inventory system (Michigan Department of Natural Resources 2019). The Michigan Forest Inventory system is stand-based; stands are intended to represent relatively homogenous canopy vegetation conditions. For purposes of this study, I focused only on inventoried stands that were upland (7,183 stands covering 644 km^2). The most common forest cover types for these stands were oak association (238 km^2, or 37%), mixed upland deciduous (197 km^2, or 31%), aspen (106 km^2, or 16%), and northern hardwoods (41 km^2 or 6%). The most common canopy species included black oak (*Quercus velutina*), white oak (*Quercus alba*), and big tooth aspen (*Populus grandidentata,* Figure 1), and the most common subcanopy species were red maple (*Acer rubrum*), black cherry (*Prunus serotina*), and autumn olive (*Elaeagnus umbellata;* Figure 2). All forests in the region resulted from secondary succession, although contemporary stands that were mature forest in the earliest aerial imagery available (~1938) have a notably lower proportion of non-native species and often contain large mounds of Formicidae ants (Banschbach and Ogilvy 2014, Menke et al. 2015), which suggest that these areas were less intensively cultivated in the 1800s.

Michigan's Prescribed Burn Program

The MDNR is comprised of multiple Divisions, including Fisheries, Law Enforcement, Wildlife (WLD), Parks and Recreation (PRD), and Forest Resources (FRD). Within the study area, most state public lands are either state parks managed by PRD or state game areas managed by WLD. In southern Michigan, FRD field staff are fire officers, except for two foresters shared between FRD and WLD. Fire officers are tasked with wildfire suppression, and they also coordinate and implement the prescribed fire program. Because each Division operates primarily on restricted funds derived from their respective activities (e.g., park entrance fees, timber receipts, and hunting licenses), their budgets are distinct; by state law, restricted funds transferred from one Division to another must be accounted for. Thus, both PRD and WLD "purchase" prescribed fire services from FRD.

In southern Michigan, each land managing agency uses its own process to prioritize proposed prescribed burns, and FRD burns areas in priority rank order until the budget of a given agency is exhausted. PRD develops one list and ranks priorities based on their judgement as to the likelihood that prescribed fire will meet their restoration objectives. The WLD list is compiled from fire treatment proposals (FTP) that originate among the 13 WLD field biologists. Each biologist scores his or her own proposals according to a set of criteria. For each proposed burn, FRD converts each proposal into a formal burn plan, which lists the target season, fire intensity, and restrictions to temperature, humidity, wind speed, and wind direction to ensure the burn can be done safely while meeting land management objectives.

Interview Process

I interviewed land managers responsible for state parks and state game areas in southern Michigan. Scoping for interviews began at a facilitated workshop on the topic of barriers to oak regeneration and prescribed fire. Many of the participants in the workshop were state park and state wildlife area managers who provided assessments of the strengths and weaknesses of the MDNR program. However, I was concerned that some perspectives were potentially unshared in a public forum. I wanted to offer managers a space to provide more candid and in-depth answers without fear of peer judgement. Therefore, I initiated a series of semi-structured interviews (Rubin and Rubin 2005). Interviews were approximately 1 hour in length. I used a set of consistent questions to begin the conversation. Once the interview started, I gave interviewees wide latitude to pursue topics of interest. I continued to conduct interviews until information or insights consistently converged. I ceased scheduling after 10 interviews, which represented 42% of the DNR staff that attended the initial workshop.

I conducted interviews in March and April 2018; four interviews were in-person and six conducted by phone. I transcribed each interview and reread each transcript repeatedly until I was familiar with the material. I used textual analysis tools in QSR International's NVIVO 12 software to identify themes that were repeated across interviews. Themes were coded with the intention of documenting the diversity of answers among participants, rather than to create a representative picture of agency culture, policy, and processes. For that reason, results of the themes should be interpreted cautiously: a given point of view may represent the dominant view within a part of the agency, or it may reflect the views of a small segment of staff within the agency.

RESULTS

Several themes touched on social aspects of oak regeneration and prescribed fire that were broadly relevant. These themes included procedural fairness, risk, telecoupling, and control. I describe each theme, and develop an alternative hypothesis describing how each theme causes or mitigates oak mesophication. This interview process was reviewed by Michigan State University's Institutional Review Board (IRB) and on March 21, 2018, this study (STUDY00000544) was found to be exempt.

Procedural Fairness

Interview conversations often focused on critiques of the system by which prescribed burn resources were distributed among land managers. This was especially true of the process used internally within the WLD, in which each prescribed burn originates at the level of an individual wildlife biologist in the form of a fire treatment proposal (FTP) that is self-scored. Interviewees generally felt that their peers did not self-score in a fair and equitable manner. Two distinct perspectives were evident in the interviews. One perspective held that other biologists were not developing good proposals, leaving out key information that could have caused their proposals to score higher (a higher score results in greater likelihood of FTP implementation). To quote one interviewee:

> *"Yeah, and that is part of that frustration. You know, when I write my FTPs and stuff, I know what they are looking for. And I have tried to coach people in this. When you write FTPs, these are the things you have to highlight if you want your burn to rank out well. And I can lead the horse to water, but I can't make them drink… Because those things rank out high enough, and they are cheaper to do [per acre], and you get the habitat on the ground over a broader area. In a typical year I would put up 1,500 acres of burns over say 10 different game areas. The FRD folks see that and say, we can put a burn team in [the area] and pump out acres. We can do 2,000 acres. Boom, bang bang boom. And so… the bulk of the prescribed fire activity is… here. Because we do such big burns, you go [to another area] and they keep doing these smaller ones, and I have beat my head against the wall telling these people, 'Go big. Go big. Go big.' or, 'Develop a rotation. Write a fire plan. Give these guys some meat here.'"*

The opposite perspective was that those who were successful in getting highly ranked FTPs were intentionally interpreting the scoring criteria too loosely and "gaming the system" to unfairly influence the rank of their burns, and thus steal scarce fire resources from regions that interpreted the scoring criteria more strictly.

> *"In essence we are just trying to make sure that none of the [managers] gets left out in the cold. This is a game. It's nothing but a game. And [they] play it for whatever they think they can get away with. And [a particular land manager] for some very strange*

reason gets about two-thirds of all of the burns that are available to us, and the other [land managers] … are left with one-third. And it is mostly because of how aggressively they interpret the rules of the game."

Thus, both sides recognized that the current system led to a preferential attachment (i.e., a "rich-get-richer") dynamic. Those who were consistently allocated prescribed fire resources faulted those without access to fire for lack of initiative and for not using the process to their advantage. Conversely, those that tended to have few prescribed fire resources allocated blamed the other group for interpreting the rules unfairly to benefit that group.

Risk

The concept of risk and risk tolerance was mentioned by several land managers. Perspectives on risk and risk tolerance varied considerably among the three MDNR Divisions. Some managers felt that FRD adherence to National Wildfire Coordinating Group (NWCG) standards on all prescribed burns was too restrictive for fuel types and patch sizes typical of southern Michigan. NWCG standards require a high number of trained staff on each prescribed burn, and the NWCG sets high requirements for staff training. Some managers noted that the standards were originally created for federal agencies in the western United States, and that most state agencies set their own standards, which provides more flexibility for the low complexity and low intensity fires that are typical of oak forests in the eastern United States.

"They have a prescribed fire plan for whatever it is that they are going to burn. You know, for unit x. It's 300 acres of switchgrass and we are going to burn it. And so, they will have a basically a staffing plan that is spelled out in that burn plan. And they will have like a dozen people that are required on this burn. And you have to have a burn boss and a firing boss and then x number of engine bosses. Now those boss things, engine boss, firing boss, burn boss, those are national level qualifications… All those positions, those formal national positions require a ton of training and experience. And in order to pull off a burn, if you do not have those positions filled, you can't do the burn… If you do not have 4 or 5 engine bosses for the burn you are going to do, then the burn doesn't get done… If you go to other states, like [land manager], he came from

Indiana where their wildlife division did all their fires, did all their burns, unless it was super complicated and it had fuels that could cause problems. But these grassland burns, they did all of them themselves in house. There were no FRDs to come in and do them for them. So, it's... that is the culture here in Michigan."

The opposite perspective was also noted: the size and contiguity of forestland in Michigan, as well as flammability of common fuel types, are viewed by some as similar to western forestlands, and thus NWCG standards are appropriate from a statewide perspective. Although the study area of southern Michigan has less hazardous fuel types and less public lands relative to northern Michigan, the NWCG standards are set statewide, and those holding this perspective felt that an excess of caution was preferable to programmatic inconsistency.

"I do see the value in having certification and having people with a lot of qualification and training. Having gone through a lot of that myself and having been on prescribed burns that have gone bad... not really bad, but bad enough that it will spot across the line and to see people who don't have the training stand there and look and don't know what to do... that kind of freeze up mentality and not having that experience, that does hurt us. And I think it can be a safety issue sometimes. It is important to have people that have that experience on the line... I think it is important to have that experience on a wildfire. You need to have that experience and perspective after having run burns and wildfire experience too. I do see the value in that. I probably would not have said that 10 years ago… But I have been on the other side of the coin before and watching things go wrong, watching people not know what to do, scared me a little bit too."

Highlighting the two extremes of the position could give the impression that there were two distinct groups that viewed themselves in opposition. However, most managers who talked of risk also were careful to communicate both perspectives, and they were often unsure how to balance risk and agency mandates to manage fire dependent ecosystems.

"Yeah, I think we are being a little too risk-averse in some places like state-level qualifications to get more people qualified faster. But I also see the value in having a qualification system, not every manager should be able to run their own burns; I don't see that as safe necessarily. I'm kind of in the middle on that one."

Managers recognized that risk perceptions and risk tolerance varied among FRD, WLD, and PRD, and that those risk perceptions and tolerances were the underlying reason the state used NWCG standards on some relatively low complexity oak ecosystem prescribed burns in southern Michigan. Because the standards require relatively large numbers of relatively highly trained staff, the capacity of the agency to do prescribed fires was perceived to be limited by risk and adherence to high staffing, training, and equipment standards.

Teleconnections and Telecouplings

Teleconnections are strong causal linkages between geographically distant parts of a system. The term is most commonly in climate studies (Higgins and Vellinga 2004, Boers et al. 2019), but is sometimes used to describe climate-related connections in social systems (Butsic et al. 2015, Moser and Hart 2015). Teleconnections focus on the connection itself, usually within the domain a single discipline, domain, or dataset. The telecoupling concept is an extension of the teleconnection concept that is more integrative, drawing on multiple disciplines, domains, or datasets to describe distant feedbacks among coupled human and natural systems (Liu et al. 2013, 2016, Friis et al. 2016).

Managers noted that fire weather in distant parts of the state or in other states often limited availability of trained staff to do prescribed burns locally, even when local fire weather was suitable for prescribed fire. Fire suppression and prescribed fire draw from the same limited pool of equipment and NWCG qualified staff, and those staff are moved within or among states as needed to suppress wildfires.

> *"The risk of wildfire in the… Northern Lower [Peninsula] probably peaks in May. I would guess May, before complete green up. Where you have the chance to have periods of dry weather… after thaw. We tend to operate from March to April, then through April [in the Southern Lower Peninsula]. Then in May we are starting to pare down resources (pare*

down varies year to year), but get into May, into high fire risk now in the UP and the NLP. That fire risk in the UP can extend into June. Depending on the year, you have fire risk all the time…Let's take southern Michigan, where are we in summer? Well, we have moved our people out of… thinking about fire mostly. To be thinking about… crop management or other kinds of field work going on… You get different outcomes from early spring fires to late spring fires to summer fires to fall fires."

The linkages and feedbacks between the human system of flows of fire-trained staff and equipment and the natural system of ecological fire effects and mesophication can be conceptualized as a telecoupled system. This telecoupling operates both directly and indirectly. In a given year, staff and equipment for a prescribed burn are more available when they are not in-demand for fire suppression in other geographies, either on private lands away from the location of the proposed burn, in other parts of the state, or even in other states. This results in a direct effect on the flow of staff and equipment away from oak forests and savannas in southern Michigan. Telecoupling also has an indirect effect. Over several years, southern Michigan land managers learn that prescribed fire resources are most likely available only in early spring; they eventually plan their other land management activities around that season on the calendar, regardless of the availability to burn in any particular year. This is an example of path dependence in a complex system. Even as emerging science suggests burning during the growing season (Robertson and Hmielowski 2014, Miller et al. 2019, Fill and Crandall 2020) would create valuable wildlife habitats, telecoupling limits the fire season, operationally and based on previous experience, to a relatively short temporal window. Because staff, equipment, and weather are limiting, this telecoupling effectively limits the area that can be burned each year.

Control and Land Management

Another theme that managers returned to repeatedly during interviews was the ideas of "natural," "cultural," and the right relationship between human intervention and wildness of social-ecological systems, especially those managed explicitly for "wild" organisms (e.g., game areas or wildlife refuges). The interviews elicited impassioned responses from many of the interviewees; this theme was the only one that elicited strong language. One ecological perspective valued systems for their ecological integrity, which meant that the biotic and abiotic components of the ecosystem, including its processes, were similar to those in the early 1800s.

> *"…a system that has high ecological integrity is going to have intact natural processes that structure its composition in terms of I would say floristic composition and floristic structure. So, it is going to have an intact disturbance regime… its abiotic factors are going to be unperturbed, so soils and hydrology are going to be intact. It is going to be able to change over time and be resilient in that change."*

Other land managers exhibited a cultural perspective. They noted that the early 1800s were anomalous in terms of natural and human processes, and that oak woodlands, savannas, and prairies in the 1800s were the result of human land use choices by a significant and culturally complex Indigenous population that inhabited and burned the landscape over many millennia. They noted that defining ecological integrity by its "naturalness" is especially problematic for a system that would not exist in the relatively humid climate of Michigan without centuries of human intervention via cultural use of fire.

> *"And what I think they are failing to see is that there was a lot of anthropogenic burning that was going on for thousands of years before European settlement. …the national average of wildfires [is that] around 90% of the fires are caused by humans today. And you talk to our fire officers, and they say that 93% of the fires in Michigan are caused by humans. Then you think about having literally hundreds of thousands or even millions of people in the Midwest, Aboriginal, I mean, Native Americans living here. And having fire*

> *as a daily part of their life. It is almost impossible to imagine that either intentionally or unintentionally there would be at least 90% or 93% for the fires would be caused by humans. …it is very possible that we see oak savannas as a natural system was really an anthropogenic system… they take out of the picture what probably created that was human fire input.”*

Another manager built on this idea to note that the contemporary landscape is too fragmented and too sensitive to large wildfires to allow restoration at large extents. Although fire in the wildland-urban interface was an issue when these interviews occurred, the interviews in March and April 2018 sound prophetic given the extensive landscape fires and destruction of property in Australia and the western United States later in August 2018 and continuing through 2020.

> *“…but prescribed fire has really been the only way that you can get these stands to regenerate, but to do it on a scale... that existed before we had fire suppression... is probably not going to happen. The fires burned... uncontrollably in areas, sometimes quite dangerously. I don't think that would be acceptable in today's society to have those type of raging fires that would take out homes or decimate entire areas. But they regenerated oak very well… I am not really sure what the solutions are, but if they could basically ramp up... double or triple our prescribed burning program, I think that would certainly be a huge benefit.”*

The question of oaks as cultural artifacts or part of natural systems was more than philosophical. Land managers are tasked with deciding when to intervene in a social-ecological oak system. Thus, they struggle to decide when it is appropriate to intervene in a system to meet human goals and when to choose not to intervene, to let the natural processes determine the trajectory of the system. Managers would often use the phrase that a forest stand “wants to be oak” or no longer “wants to be oak,” suggesting that a desire toward a goal (i.e., teleology), or at least a recognition that the social-ecological oak system exhibited emergent self-organization (Alexander 2011).

> *"The way our state statutes are written we have more active management abilities in our natural and wilderness areas than at the federal level they do. Of course, we will do prescribed burning. We will do invasive plant control with chemicals. And yeah, it's still a struggle with that big question: what's the… the wilderness areas in Porcupine Mountains, for example, are dedicated to be that old growth forest. So, what happens… when it becomes young forest? What's natural when that happens? I think you have to go with what the system wants to be. You can't force the system to be something… where I would draw the line that the system is not sustainable easily or it would take a lot of inputs… then you might take the more hands-off approach. Let it sort itself out."*

Managers noted the challenges at the opposite end of the spectrum between control and naturalness, usually pointing to a management regime in northern Michigan, outside the study area. In that landscape, managers sometimes go to great lengths to ensure that forests regenerate to a similar species composition that existed prior to harvest or wildfire. When pine plantation forests regenerate oak or oak forests regenerate red maple, managers intervene to regenerate the same forests in the same place.

> *"It's red pine so we want to keep it red pine. And then it is oak, and we want to keep it oak. And so, what I had heard or seen happen a few times was a thinning on an oak stand, but part of the goal was regeneration. We didn't get any. We had red maple come up. So, then the treatment the next time it comes up in ten years is to clear-cut it and still not getting regeneration, so you just lost an oak stand. And here you had a red pine stand that had exactly what you wanted underneath if you wanted oak to be restarted, and it was roller chopped and sprayed and trenched and planted to red pine… there is definitely a mental barrier to cover type conversion."*

Some frustrated managers felt that both cultural and ecological perspectives were valid, and that a wise manager would focus intervention with prescribed fire in oak forests with high ecological integrity. These sites could be identified through the presence of plants sensitive to plowing or grazing, or by using other clues like presence of ant super-colonies or evidence from early twentieth century aerial imagery.

> *"…to key in on the places that were untouched. They were untilled, ungrazed probably, or lightly grazed, and they still have a composition reflecting what they looked like*

historically, probably, and so they also have a microbial community that is intact. And the proportion of non-native biomass is much lower… It's just going to keep getting shittier and shittier and we are going to keep losing components. As time goes on, we are going to lose them. It is like the wheels are falling off, but we are also losing like all the nuts and bolts on shit we don't even know is important… they're I believe especially resilient. Outside of there you essentially have what are basically novel ecosystems, tilled ecosystems that have reverted back to forested. And so, you have got novel ecosystems from the microbial community on up."

Perspectives varied greatly, and there was evidence that land managers were aware of broader conversations in society and academia regarding landscapes as both natural and cultural artifacts. They were thinking creatively, and often arguing with each other about how best to manage social-ecological systems for resilience, complexity, and ecosystem services.

DISCUSSION

The results of the interviews were the perceptions of land managers regarding the challenges of implementing prescribed fire at a scale necessary to regenerate and maintain oak-dominated ecosystems on state land in southern Michigan. Prescribed fire was important to restoring oak ecosystems, so these manager perceptions provided valuable insights into the complex challenge of management and restoration. I used the interview results to develop a set of hypotheses regarding how social systems influence fire, especially prescribed fire use. I present these hypotheses here as a set of causal loop models. As hypotheses, these models are explicitly proposing chains and feedbacks of causation.

A causal loop diagram (CLD) portrays generic system dynamics that often arise when feedback loops are combined in certain ways (Ford 2009, Rissman and Gillon 2017). Each arrow in the CLD denotes causation, specifically that a change in one entity causes a change in another. The direction of the arrow indicates the direction of causation. The positive sign indicates that change happens in the same direction (i.e., a decrease in A causes a decrease in B or an increase in A causes an increase in B). The negative sign indicates that causation happens in the opposite direction (i.e., a decrease in A causes an increase in B, or an increase in A causes a decrease in B). CLDs have been common in system dynamics modeling for several decades, and their strengths and weaknesses are well documented (Sterman 2001, Lane 2008). They can provide a useful heuristic to visualize patterns of influence in a system, but their simplicity masks important aspects of system dynamics, such as magnitudes of flows or diversity among agents. Here CLDs are used to model several alternative ways that the ecological mesophication hypothesis (Nowacki and Abrams 2008, Abrams and Nowacki 2015) can be extended to include social system dynamics based on qualitative interview data (Rissman and Gillon 2017, Tenza et al. 2017).

Ecological Mesophication Hypothesis

The ecological oak mesophication hypothesis (Nowacki and Abrams 2008, Abrams and Nowacki 2015) proposes that hardwood forests in eastern North America have multiple stable equilibria, and fire exclusion and suppression can shift forests from oak dominated ecosystems to ecosystems dominated by tree species more commonly found on mesic soils. I illustrate a simple form of the hypothesis using a CLD of four interacting and reinforcing feedbacks resulting in wildland fire (Figure 3). Oaks have other factors that adapt them to frequent fire (Brose et al. 2014), and more realistic, less parsimonious models of the hypothesis might incorporate deer browse, forest harvest, and invasive species (Arthur et al. 2012, Hanberry et al. 2020).

Because CLDs can be information dense, I will not describe each arrow in every diagram. I will however describe the first CLD (Figure 3) in detail for those readers for whom a CLD may be unfamiliar. At the far left of the diagram, an increase in fire exclusion and suppression causes a decrease in the amount of wildland fire on a given landscape. Note that this does not mean that the effect is "negative" as in undesirable; nor does negative mean that wildland fire always decreases. The negative sign means that a change in fire exclusion causes a change in the opposite direction in the amount of wildland fire on the landscape. Thus, the negative sign in the diagram could be interpreted to mean that a *decrease* in fire exclusion would cause an *increase* in the amount of wildland fire on the landscape, or it could mean that an *increase* in fire exclusion would cause a *decrease* in wildland fire. This meaning of positive and negative signs in the models is critical to understand CLDs.

The upper left feedback loop can be interpreted two ways. One can start presupposing an increase or a decrease in wildland fire. If one starts with a hypothetical decrease in wildland fire, the top arrow with a (-) sign can be read to mean that less fire causes an increase in subcanopy maple and mesic species coverage. More maple then causes a decrease in fine fuel loading because maple leaves rot during the winter and are not available as fine fuels the following season. The decrease in fine fuel loading then causes a decrease in wildland fire. (Note that the (+) sign indicates that change happens in the same direction; in this case, *decrease* causes *decrease*. It does not mean that wildland fire *increases*.) This completes a feedback loop. The loop could be interpreted the other way; an increase in wildland fire would work through the loop to cause an increase in wildland fire. Because the overall effect of the feedback loop is that less fire causes less fire and more fire causes more fire, the feedback loop is a reinforcing loop. Loops can be reinforcing or balancing, but in the mesophication example, the social and ecological systems are dominated by reinforcing loops that amplify the initial effect.

There are three other reinforcing loops in this CLD. A hypothetical increase in wildland fire results in increased subcanopy oak and increased pyric species cover. Because oak leaves persist through winter, they increase fine fuel loading, which completes the loop by increasing wildland fire. Similarly, increased subcanopy oak also results in leaves that dry faster, which decreases fine fuel moisture, which increases wildland fire. Conversely, there is a loop through maple and mesic vegetation that affects fuel moisture in a fourth reinforcing feedback loop. The net effect is four reinforcing loops that amplify the effect of fire exclusion and suppression (Figure 3).

Procedural Fairness Hypothesis

Several observers reported that the procedures for distributing scarce capacity to conduct prescribed burns was not fair. This was especially true regarding the internal process that WLD used to determine which proposed prescribed burns were the highest annual priority. Although the scoring system included many criteria, for simplicity I illustrate the effect of two criteria: restoration potential and habitat for a priority species (Figure 4). With this CDL, two levels of organization determine system operation. At the level of a proposed burn, if the proposed burn would restore a fire-suppressed system (as opposed to an agricultural field or an engineered dike around a wetland), or if it would create habitat for a priority species (e.g., pheasant *Phasianus colchicus* or Karner blue butterfly), that proposal would get a higher priority rank. At the level of a state game area, a proposal to burn a given site will be more likely to rank high and the site is more likely to get burned if it contains many fire suppressed systems and priority species. The WLD biologist in such an area perceives the priority system working well, which motivates the biologist to propose at least as many, if not more, burns the following year. The number of burn proposals funded in the next year thus becomes a combination of the number of proposals the prior year and how similar proposals from that biologist ranked during the previous year.

The Michigan Natural Features Inventory recently assessed the spatial distribution of fire needs among state game areas within our study area (Cohen et al. 2021). The proportions of fire-suppressed and priority species habitats were highly concentrated in two state game areas perceived to be gaming the procedural system. Thus, the highly skewed distribution of fire needs reflected the highly skewed distribution of access to prescribed fire resources. However, biologists were not familiar with the extent to which fire needs were concentrated in a few areas. They directly experience that relatively few of their proposals are approved, which results in increased perceptions of inequity. Biologists perceive their lack of success as resulting from cheating elsewhere in the system. Cohen et al. (2021) did not complete their study showing spatial clustering of fire needs until after my interviews were completed. The perceived inequity and lack of success in proposals thus becomes a demotivating and polarizing force for most biologists in the region.

I hypothesize that reinforcing feedbacks resulted in a preferential attachment (i.e., “rich-get-richer”) system dynamic, which rewarded a few biologists whose areas happened to have many priority species and high restoration potential, but penalized other managers who, at the time of the interviews, lacked data on the distribution of prescribed fire needs across the landscape. Additionally, the other managers lacked competing narratives about the “correct” way to write fire treatment proposals. Demoralization and perceptions of unfairness did not contribute to a broad base of support for expanding the existing prescribed fire program within WLD.

Agency Mandates and Risk Perception Hypothesis

The need to develop a proposal and ranking system for prescribed burns arises because the agency proposes more burns annually than they have capacity to execute. One of the limits on capacity is the agency decision to use NWCG standards statewide. Although the causal loop diagram (Figure 5) suggests a divide between perspectives of those within FRD (which is responsible to execute the fire suppression mandate of the agency) and WLD and PRD (whose mandates often require use of prescribed fire), interviewees from all three divisions described the same dynamic (albeit from different perspectives); agency mandates appeared to influence professional experience, which may have influenced risk perceptions, and ultimately risk perceptions resulted in a situation in which NWCG standards limit prescribed fire while making use of prescribed fire safer. Interestingly, because policies and agency mandates are largely static, there were few loops, except for a reinforcing loop in which higher tolerance leads to more prescribed fire, but lower risk tolerance leads to fewer fires and lower risk.

Telecoupling Hypothesis

Land managers noted a dynamic in the prescribed fire program that I describe as a spillover effect of a metacoupled fire suppression system in a changing climate (Liu 2017). Metacoupled systems involve both long distance telecoupling as well as coupling between adjacent systems (pericoupling) or within systems (intracoupling). Sometimes the coupling between two systems can have unintended consequences for a third system; these are referred to as spillover effects. Some interviewees described spillover effects, although they did not use the language of spillover and metacoupling. In the United States, NWCG qualified staff are moved seasonally to different geographies when fire suppression needs exceed local capacity. In the metacoupling conceptual framework, these shifts in staff and equipment can be described both as pericoupling (i.e., movement between southern to northern Michigan in spring) and telecoupling (i.e., southern Michigan to western states in summer and fall). The effect on the prescribed fire program is a spillover effect of flow of fire suppression staff and equipment to suppress wildfires adjacent and distant geographic areas. Thus, the mesophication in the natural system resulting from lack staff to do prescribed fire in the human system is a spillover in a metacoupled human and natural system (Liu et al. 2013, 2015, Liu 2017).

The metacoupling in this system is mediated by near-term seasonal weather and long-term climate-driven shifts in fire weather in geographies distant from southern Michigan. In early spring snow melts first in southern Michigan. Prescribed burns can occur in southern Michigan, and fire danger is low to non-existent further north where snow still covers the ground. In late spring after snow melts and before ground vegetation becomes green, fire danger rises in flammable fuel types (e.g., jack pine *Pinus banksiana* and red pine *Pinus resinosa*) in the northern part of the state. Fireline qualified staff and equipment are diverted north to be ready to fight wildfires, and this often limits capacity to conduct prescribed burns in the southern part of the state.

A similar pattern occurs on a national and international scale in autumn, when local conditions in southern Michigan could support a second prescribed fire season. Traditionally, wildfires in the western United States occur in the summer. Increasingly, the climate is changing such that the fire season in the western United States extends into fall and early winter, increasingly overlapping fire seasons in the southwestern United States and other countries, such as Australia (Westerling et al. 2006, Goss et al. 2020). Because countries and states share NWCG qualified fire suppression resources, the climate driven shifts in wildfire seasons draw fire trained staff out of Michigan during the potential fall prescribed burn season. Thus, fire weather in other parts of the state, or in other states, limits using prescribed fire in southern Michigan except during a brief period in late March through April.

Because these influences are over long distances, the feedbacks are often outside the control of local actors. In the case of prescribed fire, the social mesophication hypothesis is that seasonal patterns of fire risk triggered by changes at the scales of annual weather and of long-term climate change act together to limit prescribed fire to a brief spring season in southern Michigan (Figure 6). When combined with the already limited staff capacity related to risk perceptions and competing agency mandates, there is a limit to the number of burns that can occur, despite the best efforts of trained staff to do as much as they can in the seasonal period available to them. In the specific context of southern Michigan, distant effects of weather and climate limit FRD's capacity to meet the demands of WLD and PRD. In the context of mesophication as a social-ecological process, distant effects of fire suppression affect local capacity of the social system to reverse fire exclusion by increased use of prescribed fire.

Regeneration Goal Hypothesis

Issues of procedural fairness, risk perceptions, and telecouplings were relatively straightforward influences on the number of prescribed fires and thus level of mesophication occurring on public lands in southern Michigan. From a systems perspective (Meadows 2008), the factors operated at the level of materials and processes (Meadows 2008, Fischer and Riechers 2019). System change at the material and process level is relatively easy to enact, but often has little leverage in changing the system (Figure 7). My interviews often started at the material and process level, but often delved deeper into discussions of design or intent. Interventions at these deeper system levels are more difficult to enact, but often have more leverage in bringing about lasting system change (Meadows 2008, Fischer and Riechers 2019).

Land managers repeatedly pulled the prescribed fire conversation toward larger issues of forest regeneration, and it was not always clear why the conversation about oak regeneration in southern Michigan repeatedly wandered into discussions of pine or mesic hardwood regeneration in northern Michigan. One of the strengths of semi-structured interviews is that the interviewee has some control over the direction of the conversation, and repeated tangents in the same direction suggested relevance in parallel issues in other forest types and geographies. When the conversations were later coded, the themes of these tangents became apparent. The issue of oak regeneration was a special case of forest regeneration failures, which was part of a larger issue of goal setting and control in managing or preserving changing natural ecosystems.

The systems to which forest communities are tending under current fire exclusions, fire suppression, and silvicultural regimes could be described as novel communities (Hobbs et al. 2013, Fitzpatrick et al. 2018). Upland oak ecosystems in the past lacked a significant red maple component (Whitney 1994, Nowacki and Abrams 2008) . Fire exclusion and suppression coupled with intensive clearing and grazing in the nineteenth century allowed red maple, which persists as a minor component of black ash (*Fraxinus nigra*) dominated forested wetlands, to expand and dominate a wide range of wetland and upland forests (Whitney 1994, Abrams and Nowacki 2019). However, this change to a dominant species different than the current canopy is not limited to oaks and oak mesophication. In the interviews, pine systems were described as regenerating oaks well in their understories, and mesic hardwood systems were often described as regenerating to other tree species than the previous maple and beech canopy. Only when combining the different narratives did a more cohesive hypothesis emerge: regeneration failures in pine, oak, and maple systems are stepwise transformations (Figure 8); under a regime of fire-exclusion and climate change, pine systems tend to become oak, oak systems tend to become maple, and maple systems tend to become other mesic hardwoods, such as ironwood (*Ostrya virginiana*).

Forest management is often based on an assumption that the subcanopy should reflect the species composition of the canopy, and subcanopies are sometimes managed to reflect the species composition of the canopy more accurately. Loss of pine, expansion of oak, and failure of sugar maple (*Acer saccharum*) regeneration may be consistent with predicted responses of these systems to climate change (Iverson et al. 2008, Handler et al. 2014), but they are still interpreted as regeneration failures. The idea that system drivers are stationary (i.e., varying around a mean that does not systematically change through time), and that forest attributes like species composition should tend toward replacing themselves, has deep roots in conservation (Botkin 1992, 2012). Although many ecologists would reject the idea of climax communities or a balance of nature, the concept is still deeply embedded in natural resources law and policy, including goal setting within land management agencies. The assumption is that oak should be replaced by oak, pine by pine, and maple by maple; any deviation from this pattern is judged to be a regeneration failure. The implicit, and often explicit, goal is regeneration of the recent canopy or restoration to a historic baseline canopy.

Climate has changed, and it may be that community effects predicted thirty years ago (Botkin et al. 1991, He et al. 2002) are already manifesting. Conversely, current trends in forest community composition may be the result of non-climate drivers, such as invasive species, fire exclusion, or certain silvicultural practices. Regardless, changes are likely to accelerate as worldwide climates shift, often to combinations of temperature, precipitation, and phenology with no modern analogue. We know that in the past, no-analog climates resulted in no-analog ecological communities (Williams and Jackson 2007). In other words, we should not be surprised that dominant species in ecosystems are changing. It has happened in the past when climates shifted, and it will likely become more common in the future (Starzomski 2013). The Resist-Accept-Direct (RAD) framework was recently developed for land managers struggling to define success or failure in the context of climate driven ecological change (Schuurman et al. 2020, Thompson et al. 2021), and these concepts hold value for ecological transformations driven by non-climate challenges in an era of pervasive human influence on natural resources (i.e., the Anthropocene). They categorize three broad management approaches, from resisting to accepting to directing change. Managers often find themselves resisting change, but then questioning whether resistance is desirable or sustainable. Some land managers in southern Michigan appear to be questioning the wisdom and sustainability of goals that assume resist strategies when capacity is more appropriate for accept or direct.

Land management in the Anthropocene is increasingly unpredictable, in part because human systems often contain reinforcing feedback loops that create nonlinear system responses (Hull et al. 2015, Fischer and Riechers 2019). Despite significant unpredictability in system outcomes, complex adaptive systems have leverage points, and managing natural systems embedded in human-dominated social-ecological systems can illustrate useful leverage points in the system to attempt intervention. Thus, extending the oak mesophication hypothesis to incorporate social-ecological hypotheses can suggest point of intervention, either at the level of planning processes, policy, or mental models.

The leverage point in the burn proposal ranking system is relatively straightforward. Budgets can be divided among managers according to the amount of oak resource they manage. Because this intervention occurs at the level of materials and rewards (Figure 7), it has relatively little leverage. Managers may be less frustrated and may express more support for the prescribed fire program, but the quality of sites burned (as measured by the original ranking system) is likely to be lower. Thus, there is likely to be a trade-off in terms of quality.

Interventions in risk perceptions are likely to have higher leverage but are also likely to be harder to implement. The three sub-agencies within the DNR each have different mandates; their intent differs. One is primarily fire suppression; the other two are restoration of natural systems and provision of wildlife habitats. Their intents, and thus perceptions of what is at risk, differ. The prescribed fire program is based on compromise, in which one agency meets the needs to the best of its ability but is constrained within its mandate of minimizing risk from wildfire. One potential solution would be to create an exemption or exemptions to the NWCG standards. These might be geographical (only in southern Michigan), jurisdictional (only on PRD and WLD lands), or situational (only low complexity burns). However, this would require significant trust building and collaboration among the sub-agencies, and it would be fragile to mistakes, especially early in the process.

The telecoupling system dysfunction could be addressed by weakening the effect of links between the fire suppression and prescribed fire programs. This could be done in a couple ways. First, FRD could set policy such that prescribed fire takes precedent over stand-by during times when the weather is suitable for prescribed fire but risky for wildfire elsewhere in the state or country. This would require change all along the levels of leverage (Figure 7) from materials, to processes, to organization and intent. Fire suppression resources would likely need to be increased to allow more self-reliance within regions of the state. Second, a more targeted approach could be to create an additional and dedicated prescribed fire team in the southern part of the state that would be unavailable for stand-by when conditions change in other parts of the state. This would also require additional capacity development, but in the southern rather than the northern part of the state.

The way that land managers define regeneration “failure” is another leverage point at the scale of intent, mindset, and paradigm. Restoring a system to its past condition is a fundamental assumption in conservation, and the idea is shared across all three sub-agencies. In a forest context the idea is that, in a place, the future ecosystem should resemble the past ecosystem. Any deviation, especially in a managed ecosystem, is framed as failure. However, it could also be framed as an ecological transformation.

Accepting change in forest composition, however, is socially difficult. It requires a paradigm shift that climate change, fire exclusion, and other Anthropocene effects are changing forests here and now, and that the effects are not necessarily negative, even if the drivers of change are undesirable. Furthermore, it would require changing goals in plans throughout the agency, and those goals were developed with stakeholders over many decades. Even small changes to red pine management, for example, would have far reaching implications both for the state's forest products industry, local economies, and the operating budget of the state agency. Nevertheless, the alternative of continuing to invest significant management effort to resist change is probably not sustainable at a landscape level. Agencies will need to engage stakeholders in difficult conversations about agency capacity, goals, and expectations for ecosystem change.

Framing a conservation challenge in the Anthropocene, such as fire exclusion from oak ecosystems, as a social-ecological problem provides new and fresh perspectives to intractable problems. It suggests where and why prediction is difficult in such systems, but it also provides ways to frame interventions points, and the relative leverage and difficulty of such interventions. As climate change makes historical baselines increasingly irrelevant, conservation programs will need to be more explicit about the intervention points in social-ecological systems.

APPENDIX

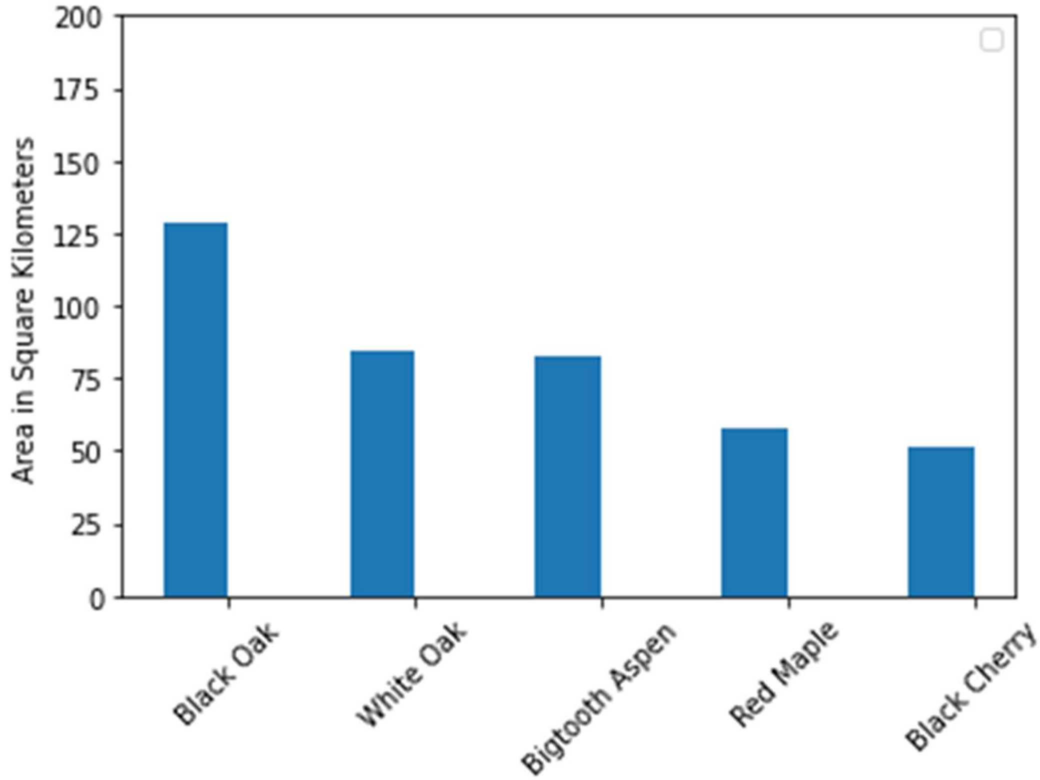

Figure 1. Sum of total area covered by the five most common canopy species on state-owned, inventoried uplands in southern Michigan. Area covered in each stand is derived from percent cover of each canopy species in a stand and the area of that stand. Data derived from the Michigan Forest Inventory as of November 2018.

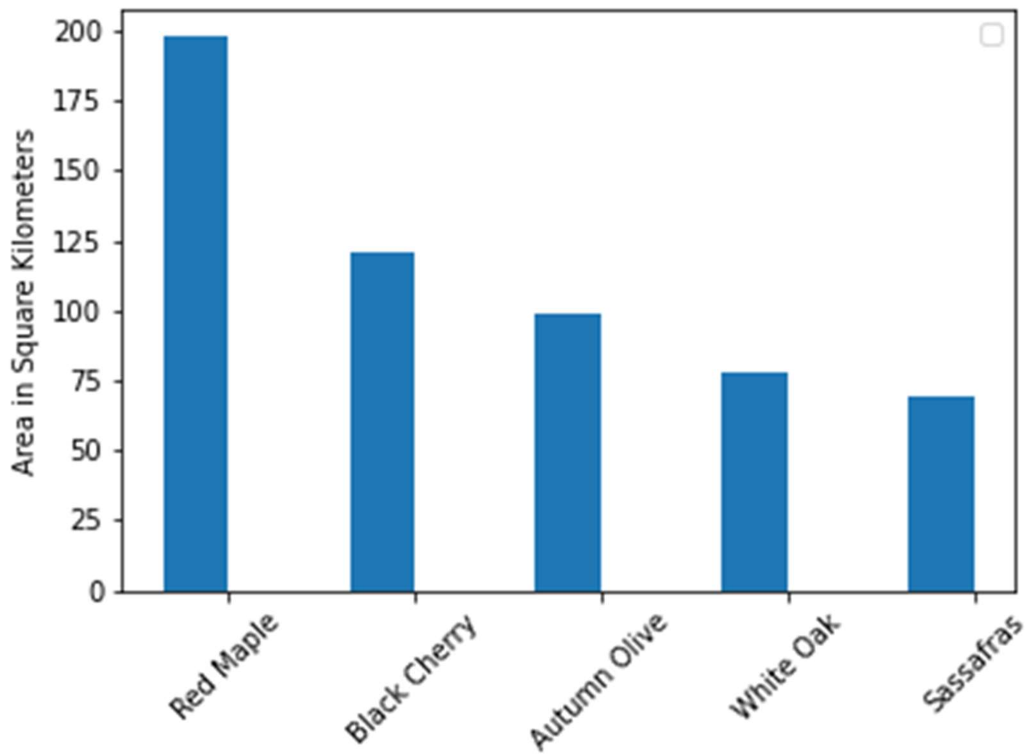

Figure 2. Sum of total area covered by the five most common subcanopy species on state-owned, inventoried uplands in southern Michigan. Area covered in each stand is derived from percent cover of each canopy species in a stand and the area of that stand. Data derived from the Michigan Forest Inventory as of November 2018.

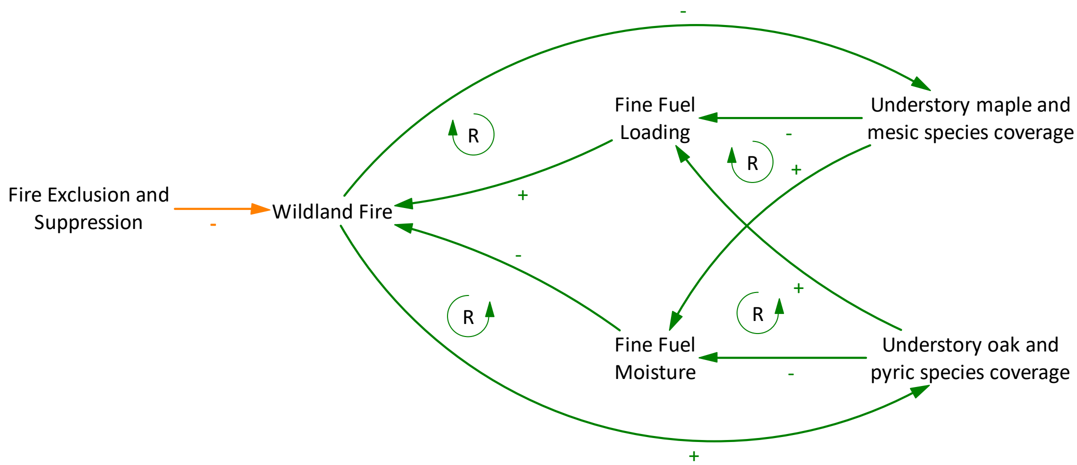

Figure 3. Causal loop diagram of the ecological oak mesophication hypothesis (Nowacki and Abrams 2008). “R” denotes a reinforcing (or positive) feedback loop. Orange arrows denote causation initiated from the social part of the system, and green arrows denote causation from the ecological part of the system. The “+” and “-“ denote that causation happens in the same (“+”) or opposite (“-“) direction.

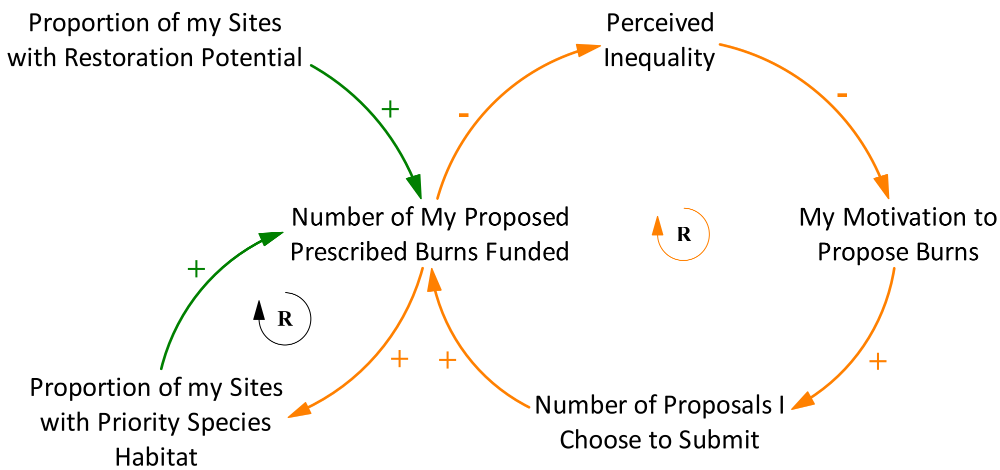

Figure 4. Causal loop diagram of perceived procedural fairness in the methods used by the Michigan DNR, Wildlife Division (WLD) in distributing scarce prescribed fire resources among land managers. R denotes a reinforcing (or positive) feedback loop. Orange arrows denote causation initiated from the social part of the system, and green arrows denote positive or negative causation between any two parts of the diagram linked by an arrow.

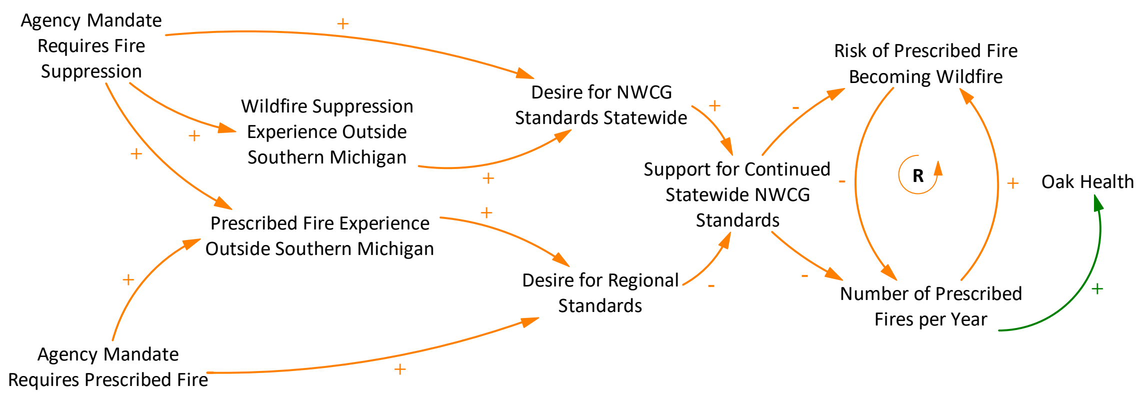

Figure 5. Causal loop diagram of agency mandates and experience with wildfire or prescribed fire, which influences both risk tolerance and number of prescribed fires. R denotes a reinforcing (or positive) feedback loop. Orange arrows denote causation initiated from the social part of the system, and green arrows denote causation from the ecological part of the system. “+” and “-“ denote positive or negative causation between any two parts of the diagram linked by an arrow.

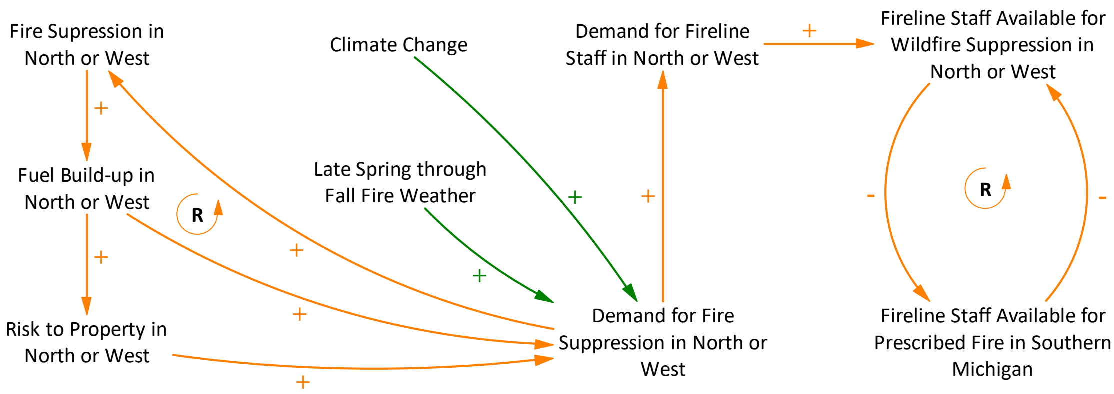

Figure 6. Causal loop diagram of telecouplings among prescribed fire and wildfire suppression programs in southern Michigan, northern Michigan, and the western United States. R denotes a reinforcing (or positive) feedback loop. Orange arrows denote causation initiated from the social part of the system, and green arrows denote causation from the ecological part of the system. "+" and "-" denote positive or negative causation between any two parts of the diagram linked by an arrow.

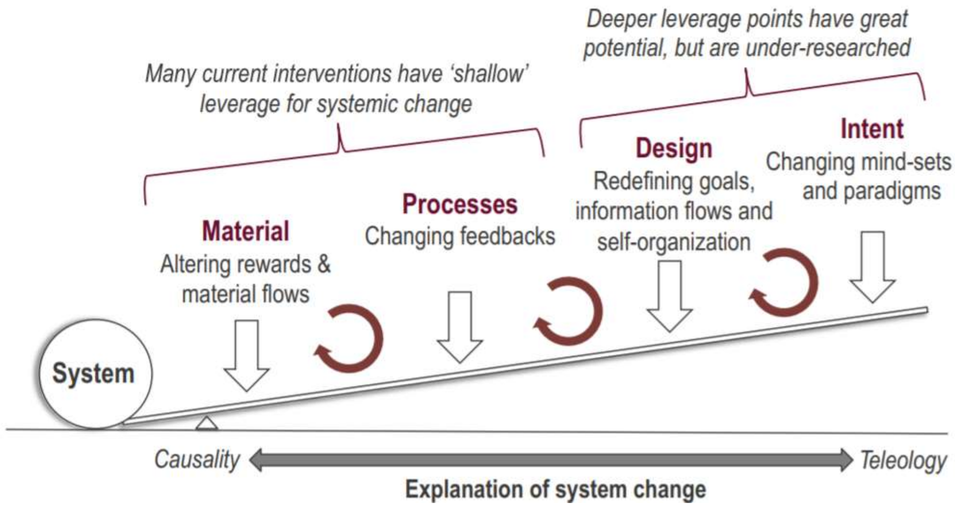

Figure 7. Leverage in driving change in systems with reinforcing and balancing feedback loops. From Fischer and Riechers (2019), based on a figure in Meadows (2008).

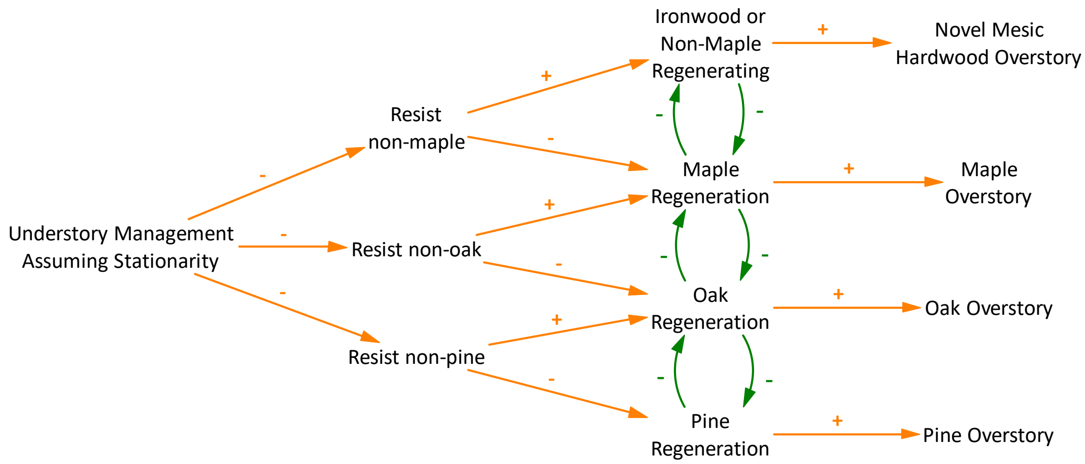

Figure 8. Causal loop diagram of transitions and trade-offs in common cover types in Michigan forests. R denotes a reinforcing (or positive) feedback loop. Orange arrows denote causation initiated from the social part of the system, and green arrows denote causation from the ecological part of the system. “+” and “-“ denote positive or negative causation between any two parts of the diagram linked by an arrow.

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Chapter 2: Fire Weather is Not Prescribed Fire Weather: An Agent-based Model of Land Manager Behavior and Oak Regeneration in a Changing Climate

ABSTRACT

The climate-niche for oak ecosystems appears to be expanding northward in the Great Lakes region, potentially benefiting current oak ecosystems along the northern boundary of the central hardwood region. However, oaks and the species associated with oak ecosystems depend on frequent low-intensity fires. Climate models project an increase in both mean annual precipitation and growing season drought, suggesting that weather suitable for low-intensity fires may increase in the future. I used two regional dynamically downscaled climate models to calculate daily burn weather, and then simulated land manager use of prescribed fire as a management tool in the late 20th century. I used an agent-based model to explore manager responses to climate change relative to the prescribed fire program and its effects on oak mesophication. The season during which the most prescribed burns could be conducted did not shift as expected under climate change. Instead, the number of potential burn days decreased. Using a business-as-usual scenario for annual budgets and daily capacity to conduct concurrent prescribed burns, oaks and other pyric vegetation continued to decline. Significant increases in annual and daily prescribed fire capacity are needed to reverse the effects of fire exclusion at the landscape scale under a rapidly changing climate.

INTRODUCTION

Fire-adapted ecosystems are threatened by contemporary anthropogenic and climate driven changes to fire regimes (Frelich 2017, Hanberry et al. 2020, McLauchlan et al. 2020), but it is unclear how climate and anthropogenic fire regimes interact to affect oak regeneration.

Fire exclusion and fire suppression have reduced the frequency and intensity of wildland fires (Abrams and Nowacki 2019). Unlike much of western North America, where fire exclusion and suppression can allow woody fuels to accumulate, the more humid forests of eastern North America experience faster decomposition rates. Exclusion and suppression of frequent, low intensity fires in this geography has resulted in the "mesophication" of formerly fire-adapted ecosystems; drought and fire-tolerant tree species are being replaced by fire-sensitive trees typical of more mesic (i.e., wetter) conditions (Nowacki and Abrams 2008, Allen et al. 2018), first in the forest subcanopy and eventually in the canopy. In the Great Lakes region, oak (*Quercus* spp.) dominated ecosystems are being replaced by maple (*Acer* spp.) dominated ecosystems (Leadbitter et al. 2002).

This change in tree species composition is cause for concern among land managers for many reasons. Oak dominated systems in the Great Lakes currently have high conservation value. Oak dominated habitats are important for a wide variety of wildlife species, including common species such as wild turkeys (*Meleagris gallopavo*) and squirrels, which are valued for various recreational pursuits, including hunting. Open oak habitats, such as oak savannas, are globally and locally imperiled (Nuzzo 1986, Hoekstra et al. 2005), and home to endangered species such as the Karner blue butterfly (*Lycaeides melissa samuelis*). Many of the common and rare species of oak-dominated ecosystems benefit from frequent low intensity fires (Pickens and Root 2009, Cohen et al. 2019, Bassett et al. 2020).

Oak-dominated ecosystems are more drought tolerant than the species replacing them. Oaks and their associated species are likely better adapted to drier growing seasons projected by most climate models for this region. Oaks and associated wildlife species will likely thrive along and just beyond the northern edge of range distributions (Duveneck et al. 2014, Handler et al. 2014, Toot et al. 2019). In Michigan this means that oaks are likely to be more resilient to climate change than northern hardwoods and mixed sub-boreal ecosystems within the state. Because oaks are perceived as providing valuable wildlife habitat elements and because they are likely more climate resilient, there is growing interest in oak habitat management and using management to reverse trends toward mesophication by red maple (*A. rubrum*), black cherry (*Prunus serotina*), and other fire-sensitive species more typical of mesic (wetter) soils.

The ecological context in which oak trees established themselves in the early 1900s is different from today. Fire regimes, climate, and land use have changed in significant ways. For example, passenger pigeons (*Ectopistes migratorius*), which were a major disturbance factor and significant seed predators, have been extinct for over a century (Ellsworth and McComb 2003, Buchanan and Hart 2012, Greenberg 2014). White-tailed deer (*Odocoileus virginianus*), which browse oak seedlings, were absent from much of the early 1900s landscape (Whitney 1994); they are now ubiquitous and abundant (Patton et al. 2018, Hanberry et al. 2020). Oak recruitment to the canopy in the contemporary landscape is relatively rare, except on the driest soils (Lee and Kost 2008, Knoot et al. 2010*a*, Bassett et al. 2020). Contemporary oak management requires intentional and carefully timed management. Forest management tools, including timber harvest, invasive species management, excluding herbivores, and prescribed fire are needed to reduce subcanopy competition, allow oak seedlings to establish themselves, and eventually allow oak recruitment into the forest canopy (Johnson et al. 2009, Knopp and Stout 2014).

One of these tools, prescribed fire, is particularly important (Brose et al. 2014, Abrams and Nowacki 2015), and is itself sensitive to changing climate . Prescribed fire is the intentional and planned use of fire as a management tool to meet specific objectives. Suitable weather is a critical and often limiting factor in both planning and using prescribed fire. Prescribed fire only occurs when weather conditions fall within the parameters of a fire prescription, which is a description of the conditions that must be met to ensure that a planned fire will be safe and meet management objectives (Packard and Mutel 1997, Sargent and Carter 1999). In addition to wind direction, weather parameters in a typical fire prescription include snow cover, wind speed, temperature, relative humidity, and time since precipitation (Lawson and Armitage 2008, De Jong et al. 2016). Although several studies project an increase in the risk of future wildfire as the climate warms (Barbero et al. 2015, Goss et al. 2020), few considered changes in weather suitable for prescribed fire (but see Clarke et al. 2019, Kupfer et al. 2020).

Climate limits the number of prescribed burn days per year, and managers tend to focus prescribed fire efforts in seasons that are most likely to have multiple consecutive burn days. In southern Michigan, consecutive burn days tend to occur in early spring after snowmelt and before trees leaf out, and in late fall after the first frost and before widespread snow cover. Historically, burning often occurred during burn days in the late fall (Cronon 1983, Whitney 1994). Contemporary managers focus on early spring. Changes in the timing or cluster of burn days could increase or decrease the number, timing, or seasonality of prescribed fires.

Climate vulnerability analyses for oak suggest that these species will thrive in future climates, which will better support fire in forested ecosystems (Duveneck et al. 2014, Handler et al. 2014, Rogers et al. 2016). However, there is an implicit assumption in these vulnerability analyses that more risk of wildfire will result in more frequent low-intensity fire and a disturbance regime that favors oak over maples and other mesophytic vegetation. In jurisdictions like southern Michigan with patchy forests and a high road density, wildfires quickly reach barriers and are quickly suppressed. Thus, prescribed fire dynamics and not wildfire dynamics are likely to drive fire effects on fire-dependent ecosystems, and climate change could indirectly affect oak or maple regeneration by changing the timing and amount of weather suitable for safe use of prescribed fire.

Prescribed fire is underused relative to the frequency and extent of fire necessary to maintain ecological communities that need recurring fire to persist, much less to reverse nearly a century of mesophication at a landscape level (Cohen et al. 2021). The barriers to use of prescribed fire are complex, involving perceptions among managers of procedural fairness, differences in risk perception, distant effects (i.e., telecouplings) of changing fire weather, and the ways the agency frames regeneration success and failure. Both risk perception and distant effects involved seasonal weather patterns that affect fire weather, and these may change as the climate changes (Chapter 1). The future of oaks in this region will be determined by interactions of land managers, changing weather patterns, and their combined effects on the use of prescribed fire as a land management tool. Assessing the vulnerability of oaks and oak-dependent wildlife thus requires models that explicitly incorporate the ways that weather and land managers interact day to day, as well as how managers adapt their behavior in the context of weather changes driven by changes in global climate patterns over years and decades. Natural processes alone are unlikely to capture the social-ecological processes that will lead toward or away from restoration and maintenance of valuable habitat for wildlife and plant species associated with oak-dominated ecosystems.

My objectives in this study were to develop a social-ecological agent-based model (ABM) of prescribed fire use and to use the model to predict the cover of upland oaks and other fire-dependent plant species under different scenarios of daily and annual burn capacity. The model was forced by climate to assess the adaptive capacity of the prescribed fire program to changes in climate. I used an agent-based model framework and downscaled climate models to simulate climate adaptation at two organizational levels: manager decision making based on daily weather for specific stands, and ecological effects across the public land base as climate changes at decade scales. This dual level simulation allowed incorporation of decision-making and weather that is relevant and transparent to land managers while still capturing adaptation or maladaptation at the multi-decade scale of climate and forest change.

METHODS

Study Area

The study area comprises state lands in Michigan in counties at the same latitude or south of Huron County and Oceana County. It is roughly the southern half of the lower peninsula of the state of Michigan in the USA. This landscape is a mix of row crop agriculture, forest, and urban development. The large blocks of contiguous forest land in this region are predominately public lands administered by the State of Michigan as wildlife areas or state parks.

Downscaled Fire Weather

Fire weather is a term used to describe a multidimensional weather space in which fire, once ignited, will continue to burn in natural vegetation on a given area. The Michigan Department of Natural Resources (MDNR) uses fire weather models developed by the Canadian Forest Service (Lawson and Armitage 2008; fire weather models developed by United States Forest Service are calibrated for the arid, mountainous regions of the western United States). Canadian fire weather is described by three statistics: a Fire Weather Index (FWI) describes the overall danger of wildfire, which is the sum of the Initial Spread Index (ISI), which describes the danger that a fire will spread rapidly once ignited, and the Build-Up Index (BUI), which describes the intensity (heat over time) of a fire. The MDNR uses ISI and BUI to plan allowable weather conditions for prescribed burns, as well as to plan when, where, and how to deploy fire suppression staff and equipment in the event of a wildfire.

The MDNR begins calculating and monitoring both ISI and BUI daily each year when the fire season for a given region begins, which the MDNR defines as the date when snow depth decreases to zero throughout the region (e.g., throughout southern Michigan). ISI is calculated from wind speed and fine fuel moisture codes, which are calculated from temperature, relative humidity, wind speed, and precipitation (Lawson and Armitage 2008). BUI is calculated from drought codes and duff moisture codes, which are derived from temperature, relative humidity, precipitation, month, and latitude (Lawson and Armitage 2008).

The weather that provides conditions suitable for prescribed fire is influenced in this region by distance from the Great Lakes. The lakes are large enough to significantly affect local weather but too small to incorporate into global climate models (Notaro et al. 2015). Thus, I used dynamically downscaled climate models (Notaro et al. 2014, 2015), which incorporated lake effects on weather, to generate realistic lake effect weather. These downscaled models were informed at the regional boundary by global climate models. The downscaled models provide debiased daily mean weather conditions for the variables needed to calculate ISI, BUI, and FWI for three two-decade time periods: 1980–2000, 2040–2060, and 2080–2100. I calculated fire weather (Lawson and Armitage 2008) from the driest (ACCESS-RegCM4) and wettest (CNRM-RegCM4) of the six models downscaled by Notaro et al. (2014). All models are based on the 8.5 representative concentration pathway (RCP), which is the high emissions scenario in the most recent set of IPCC reports (IPCC 2014).

To generate a time series of daily weather data on which to drive agent behavior in deciding when and how much to burn on a given area, I derived 14 separate series of daily weather for the climate model cell closest to the centroid of a simulated manager's multi-county management zone. These 14 zones were 1–4 county areas across the southern two-thirds of the lower peninsula of Michigan (Figure 9). Because the climate models were not temporally continuous, I repeated time series to create a continuous daily weather record from 1980– 2120. The 1980–2000 was repeated for 2000–2020 and 2020–2040; 2040–2060 was repeated for 2060–2080 and 2080–2100 was repeated for 2100–2120. Given that climate change is not continuous, I felt that this compromise was preferable to using climate data that did not include lake effect, or in which managers had an unrealistically short time (20 years) to adapt to changes that would occur over 40–60 years.

Wildlife Habitat Conditions

The ecological data for the model was derived from MDNR Michigan Forest Inventory (MiFI) data as of November 2018. MiFI is based on field surveys to 7,013 forest stands visited between 2005 and 2018, in the counties of Michigan including, south, and west of Huron County. Forest stands are areas with relatively homogenous vegetation and landform characteristics and range from less than 1 ha to 281 ha (mean size = 9 ha). For each stand, various forest measurements were recorded, including age of the dominant trees, tree diameter at breast height, and species composition. For this analysis I used only the precent cover of trees and herbaceous plant species in the forest canopy and on the ground, respectively. Every species with ≥2% cover within a forest stand was included. In most forest stands, the canopy was closed and had few gaps, and thus percent cover of all canopy species summed to 100%. However, subcanopies were often multilayered, and thus usually summed to over 100%.

Because this analysis was focused on prescribed fire and conversion of oak forests to mesic hardwoods, only data categorized as upland were used. This excluded floodplain forests and other wetland complexes. Savannas, barrens, and other forest openings, although rare, were included in MiFI and in the analysis. In total, 7,013 upland forest stands were included in the simulations, comprising all State Game Areas in the study area of southern Michigan, as well as some forested State Parks and State Recreation Areas (Figure 9). State Forests were not included.

Land Manager Agents and Behavior Rules

Land manager attributes and behavior were developed from interviews with land managers that occurred in March and April 2018. The methods and results from those interviews are discussed in Chapter 1 of this dissertation. The simulation model contains 14 agents, who manage forest stands on public lands in 14 different multi-county regions of southern Michigan (Figure 9). The number and shape of these regions corresponded with those of wildlife biologists in the Wildlife Division (WLD) of the MDNR, but I also simulated behavior of agents associated with land managers in the Parks and Recreation Division (PRD), and foresters and fire officers in the Forest Resources Division (FRD) (e.g., agents both choose which stands to burn and which days to burn). The agents in the agent-based model are a simplified amalgam of a complex professional network that collaborates to get prescribed fires accomplished. This simplification was necessary to make the model parsimonious, to reduce computational load, and because the interactions of FRD, PRD, and WLD staff were not the focus of this analysis (as compared to Chapter 1, in which differences among land managers in each agency and the ways they interacted was a research focus).

Model Description

The model is agent-based in which land manager agents obey a set of rules that determine which agents can use prescribed fire to manage individual forest stands within their counties. Within a stand, a fire increases cover of fire-dependent species and reduces cover of fire sensitive species. All stands each year experience fire exclusion effects, in which fire-dependent species decline in cover and fire-sensitive species increase in cover. Fire effects were calibrated to be much stronger than fire exclusion effects, such that repeated fires are necessary to maintain fire-dependent species on the landscape. Species specific fire sensitivities are a rough index developed from species lists of ecological communities and the typical fire return interval of those communities, and the relative strengths of mesophication and fire effects were calibrated such that 75 years of fire suppression resulted in mesophication similar to the status quo and fire frequencies for oak barrens and oak savannas resulted in species mixes typical of those communities.

The simulation occurs on two nested timesteps: daily and yearly. On a given day agents (i.e., simulated land managers) check to see if there is any remaining annual budget to do prescribed burns. The agency sets an upper limit on the number of burns that can occur each year. Each manager also sets two 14-day burn seasons when all staff will be on stand-by, ready to burn if the weather is suitable. If there are remaining burns in the budget for a given year and that day falls within one of the agent's planned burn seasons, then each manager checks the local weather in their multi-county area. When ISI is high enough to carry fire and low enough to be safe, and BUI is below a safety threshold, then a manager agent seeks to burn that day. All the managers make similar calculations. On a given day all managers, a subset, or no managers try to burn. However, there is a capacity limit to the number of simultaneous prescribed fires that can occur on a given day because fire-trained staff, fire engines, and other equipment are limited. Stands are selected randomly from the set of areas with agents who want to burn. Within the burned stands, fires reduce or increase the precent cover of each species according to their species-specific fire sensitivity (see Tables 1 and 2 for species specific fire sensitivities). The model then iterates over all days in the year, until day 365 (December 31), when the year step activities are re-triggered.

Once per year, the budget is reset, and managers review the prescribed burns that occurred (or did not occur) during the previous year. Managers may choose to adapt by trying to burn during a different season. Agents cannot increase budget or change daily capacity, but they can choose when they set their burn season. Setting a burn season earlier can be advantageous if an agent can tap more of the budget before their colleagues, but it can be disadvantageous if they set their season before fire weather is suitable. The annual window for fire weather changes at two scales: year to year, and with a changing climate for multiple decades.

I explored three scenarios that represented different policies regarding triggers of adaptive behavior. The control scenarios are that 1) all agents adapt every year and that 2) no agents adapt every year. Given that adaptation requires some effort, it is more reasonable to assume that agents adapt when dissatisfied with the status quo; 3) under the budget competition scenario, agents try to change their burn season start date for the next year after they get less than an equal share of the burn budget in the most recent year.

Because the model is stochastic regarding 1) which stands are selected to burn on a given day, and 2) the direction and magnitude of the fire season changes when agents adapt, I ran the model 10 times for each adaptation behavior. Additionally, I repeated the scenarios for how adaptation is triggered under two different downscaled Global Climate Models (ACCESS and CNRM), which represent the driest and wettest of six downscaled GCMs (Notaro et al. 2015). Finally, I explored the effects of annual capacity (e.g., budget) and daily capacity (e.g., staff and equipment) by running the model with status quo, or much increased (1,000%) sizes for both annual and daily capacity. These were also run 10 times each. Thus, I ran the model for three adaptation trigger scenarios, two climate models, two capacity scenarios for 3 * 2 * 2 * 10 = 120 model runs.

I used percent change in area covered by each species as a metric to show the effects of adapt rule triggers and the effects of different budget and capacity combinations. I focused on the response of subcanopy species because they are more sensitive to fire effects, and because canopy dynamics are determined by a complicated interaction of subcanopy competition, silvicultural practices, herbivory, and insect and disease dynamics that were beyond the scope of this model.

RESULTS

Downscaled Fire Weather

Snow on the ground precludes prescribed fire, especially during the winter season. Mean average snow depth decreased across the 14 management zones between the late 20th century (LTC; 1980–2000) and the late-twenty-first century (LTFC; 2080–2100; Figure 10). Decreases in snow depth were large across fall, winter, and spring. Other analyses of the same dataset show that extreme precipitation events in winter nearly offset snow depth losses due to melting (Notaro et al. 2014). Warming in the spring and fall reduces snowpack over potential fuels, which allows prescribed fire to potentially occur later and earlier in the season. However, the potential gains in spring were offset by rapid increases in ISI in the drier ACCESS model and the wetter CNRM model (Figure 11). BUI increased in the spring more in the drier ACCESS model as compared to the wetter CNRM model (Figure 12). Thus, the pattern of change in optimal burn dates for spring constricted from a plateau or multiple peaks to a narrower peak of optimal burn dates; the optimal burn season narrowed rather than shifted as the climate changed (Figure 13). The reason that fire weather was unsuitable changed, from too much spring snowpack to too high ISI and BUI in the ACCESS model.

The pattern of constriction was somewhat different in the ACCESS and CNRM models. In the ACCESS model the May peak was slightly higher than the April peak, and thus it could be interpreted as a shift in season as well as a constriction in season. The CNRM model showed a little change in the spring peak in April, and the decrease was similar in magnitude from May through the summer. The ACCESS model showed a larger decrease in potential days when prescribed fire would be safe because of increases in both ISI and BUI, especially during the summer season. The pattern was similar, but less pronounced in the CNRM model (Figure 13).

The overall pattern in both models was a decrease in burn days across all seasons, except on the best burn days in spring and fall. Thus, the ideal seasons for prescribed fire did not shift, as we expected, but instead narrowed considerably. This narrowing of the burn window was driven primarily by increased frequency of hot and dry conditions.

Emergent Adaptation

Two of the three rule sets for adaptation of fire plans resulted in managers attempting to adapt to a changing climate. The only rule set that did not show adaptation was the control, in which managers were never triggered to adapt. The rule set in which managers always adapt had the highest level of adaptation. The control and always adapt rule sets represented the lower and upper bounds of adaptation that was possible in the model.

The budget competition rule set resulted in an emergent pattern. Agents were triggered to adapt when they perceived that their colleagues, the other agents, were able to do more prescribed burns than they were. Forest stands were not distributed evenly among agents. Two agents had over 1,000 stands within their multi-county region, whereas two other agents had fewer than 100 stands. (These latter agents were from urban-dominated counties or areas with large wetland complexes and little forested upland). Because stands were selected randomly from the entire study area, agents with many upland forested stands usually received more than 1/14th of the burns and thus were rarely triggered to adapt; agents with few stands rarely had the opportunity to conduct any prescribed burns and were triggered to adapt almost every year. This pattern changed later in the model run because the adaptive agents used most of the budget by burning earlier in the spring. This triggered adaptation attempts by the agents who had not been triggered to adapt earlier. This pattern was similar in the ACCESS and CNRM models (Figure 14). However, because agents checked to see if an adaptation attempt was an improvement over the status quo, they adapted less often, and the emergent pattern was less pronounced (Figure 15).

In this system, agents were adaptive, but adaptation was not directional toward an earlier burn season because the prescribed fire season was not changing in the directional way I had expected. Furthermore, adaptation in burn season start was so small that it had no effect on the trajectory of oaks, maples, or invasive species on the landscape (Figure 16). Adaptation occurred, but the model was insensitive to this type of adaptation. The model was more sensitive to changes in annual budget and daily burn capacity.

Changing Oak Ecosystems Relative to Annual and Daily Burn Capacity

The decrease in suitable days for prescribed burning as the climate changes was only partially offset by adaptation in burn season. I also explored the effects of changing annual budget and changing daily capacity. Annual budgets and daily capacity in the model were based on number of stands, not dollar cost. To set a realistic budget based on stands, I set the business-as-usual (BAU) budget to the number and size of burns conducted by MDNR from 2016–2018. The average area burned over those three years was 1,850 ha. The average size of a forested stand in southern Michigan in MiFI was 9 ha. Thus, we set the budget at 1,850 ha / 9 ha or 206 stands burned per year. Within the constraints of implementing the prescribed fire program (see Chapter 1), the MDNR was able to conduct prescribed burns on an average of 18 days. Thus, the 2016–2018 baseline number of stands burned per year was 200, and capacity was 200 stands / 18 days or 11 stands per day. In the model, I rounded these to an annual budget of 200 and a daily capacity of 10 as the baseline scenario.

Within the MiFI dataset for this region, there were 7,013 stands containing percent cover data for 208 subcanopy species. By multiplying the size of each stand by the percent cover for each species, I derived a baseline metric of area covered by each canopy and subcanopy species throughout the study area (Tables 1 and 2). The canopy species that covered the most area were black oak (*Quercus velutina*), white oak (*Quercus alba*), and big-toothed aspen (*Populus grandidentata*; Figure 17); by contrast, subcanopy species that covered the most area were red maple, black cherry, and autumn olive (*Elaeagnus umbellata*; Figure 18). Comparing canopy and subcanopy for the ten most common species illustrates that black oak and big-toothed aspen in the canopy layer were overrepresented relative to the subcanopy. By contrast, red maple and black cherry were overrepresented in the subcanopy relative to the canopy, indicating that mesophication has reached an advanced state on state lands in southern Michigan (Figure 19). The area covered by white oak canopy and subcanopy were nearly the same.

Because the model does not include recruitment of subcanopy to canopy, I report only subcanopy values after 140 years. Model output was very similar between the ACCESS and CNRM models (see Figures 20, 21 and Tables 3, 4), and thus only ACCESS results are given hereafter. Business-as-usual levels of daily capacity and annual budget when simulated over 140 years resulted in increased area covered by red maple of 405.3 +/- 0.8 km^2, whereas white oak decreased to 1.4 +/- 0.2 km^2 and black oak decreased to 0.6 +/- 0.1 km^2 (Figure 20, Tables 3, 4). This represents extreme mesophication. By contrast, a ten-fold increase in both daily and annual fire capacity resulted in increased cover of white oak to 136.8 +/- 4.4 km^2 and black oak to 75.8 +/- 16.2 km^2 and a decrease of red maple to 724.4 +/- 51.5 km^2 (Figure 21, Tables 3, 4). This represents an increase in oak and reduction of maple in the subcanopy, but it is less oak and more maple than the 2018 areas in the canopy. Thus, even a ten-fold increase in prescribed fire still resulted in a persistent, low level of mesophication.

DISCUSSION

Wildfire Weather and Prescribed Fire Weather

Both the wettest and driest climate models indicated a future that will likely be more wildfire prone based on estimated ISI and BUI, and this future will offer fewer days suitable for prescribed fire. This pattern was apparent despite a net increase in mean annual precipitation in both models. Prior to this analysis, I was concerned that increased precipitation would limit prescribed fire activity more than dry conditions. However, mean annual precipitation appears to be a poor index for fire weather as modeled here. Indeed, the global pattern of increased mean precipitation and extreme precipitation events exists concurrent with a global increase in drought frequency and severity (Seneviratne et al. 2012, Alexander 2016). A rainy day is not a burn day, regardless of the amount of rain that falls. The number of suitable prescribed fire days is more complicated than estimated from mean annual precipitation alone.

The dry and wet models differed in the degree to which wildfire risk increased and the ways it shifted seasonally. In the wetter CNRM model, FWI increase was modest, except in the later summer and early fall (July – September). In the drier ACCESS model, FWI increased throughout the snow-free season (April-November). The pattern in number of days suitable for prescribed fire in each season shifted in ways that I did not expect. I expected and built agent behavior in the simulation model around the expectation that reduced snowfall would open new opportunities to do prescribed fire earlier in the spring and later in the fall; this was not what happened. Except for a small increase in opportunity in February in some parts of the study area associated with the ACCESS model, the prescribed fire seasons narrowed slightly on the winter end and significantly in the summer.

I induced land manager agents to adapt in the model, but when those agents compared a new potential start date for a prescribed fire season against their recent weather experience, they arrived at a local optimum and stopped adapting. A useful analogy is the theory of fitness landscapes in evolutionary biology, in which species often find themselves on local peaks where small evolutionary changes in either direction result in lower fitness and so they cease adapting (Kauffman 1996). Land manager agents quickly adapt to find the local peak of the best burn season. Then climate change erodes the landscape around the peak without shifting it earlier or later in the season. The agent therefore has less and less motivation to adapt.

Negative adaptation results in the agent-based model are insightful. Climate adaptation in the agency's prescribed fire program is not what I expected. It is not a question of individual land managers shifting their local burn season. Instead, adaptation will require strategies that result in prescribed fire with fewer annual number of suitable burn days and narrowing of the spring and fall seasons.

Reacting to Oak Mesophication in a Changing Climate

Under the business-as-usual levels of daily and annual capacity to do prescribed burns, land managers were unable to burn the number of stands each year necessary to offset mesophication according to the simulation model. Under both climate scenarios, the number of days per year with safe prescribed fire weather becomes even more limited in the future. Shifting the season to earlier burn dates or into the growing season has minimal effect given current levels of prescribed fire use. Thus, climate adaptation means increasing both annual and daily capacity, especially in the near term when there are more potential days per year with suitable weather. There are many ways to increase prescribed fire capacity. One way is to increase the annual budget for prescribed burns. However, the amount of budget increase needed, if capacity were met just by increased funds would be an order of magnitude larger than past budget increases in this program. Furthermore, in interviews with land managers and program leaders at the agency, budget was rarely highlighted as a significant limiting factor to increasing the amount of prescribed fire use on state agency lands in southern Michigan. Interviewees suggested other potential solutions (see Chapter 1 for more detail), which I review below in the context of the simulations.

Extending the current ABM to agents that adapt to proposed changes in annual budget or daily capacity would be a useful extension of this agent-based model, but such a model might need to incorporate adaptation at the organizational and individual manager levels. Some ways to increase capacity are management decisions that can be made primarily by field staff, either fire officers in FRD or wildlife biologists within WLD. However, other ways to increase capacity would require changes in planning, administration, or policy, which would need to occur at the level of the agency, not individual agents.

At the level of fire officers or wildlife biologists, burn prescriptions could be altered to magnify the fire effects on fire-sensitive vegetation. In the model, every fire has the same fire effect, and it can take 10 or more fires in the same stand to achieve the desired levels of mortality on fire-sensitive vegetation. In reality, there is a science and art to achieving desired fire effects with fewer burns (Ponisio et al. 2016, Frelich 2017, McLauchlan et al. 2020). One can focus burns on sandy soils, or where mesophication is less advanced. One can burn when weather allows more intense fires. Fires can be ignited such that they spread against the wind, which causes flames to linger longer and thus affect vegetation with more heat over time. Fire can be initiated in the growing season when plants are more sensitive to fire. Each of these involve trade-offs with other values (e.g., slower fires may require burning into the night, when smoke management becomes more challenging; or burning during the growing season can endanger turtles or other species that hibernate underground during the dormant season). Increasing capacity via fire effects is not without trade-offs, but it is a good example of a non-monetary cost that is mostly within the decision-space of local land managers.

Another intervention at the level of wildlife biologists and fire officers is to burn more stands in one day by burning larger areas. Individual stands could be clustered using adjacency rules and existing burn breaks like roads or rivers. Prescribed burns are usually ignited and controlled from the perimeter, and so the cost of a burn is determined largely by length of its perimeter. Because area increases exponentially relative to perimeter, more area (more stands) can be burned with relatively modest increases in cost. Larger burns require more staff, more planning, and thus have higher costs, but the per acre cost decreases exponentially relative to costs related to staffing the burn perimeter. Like fire effects, there are trade-offs. Larger burns are logistically challenging, and they may require trade-offs in management of specific species or special places. For example, permit restrictions for burning habitat of endangered butterflies often requires burning no more than 33% of the area occupied annually. A century of fire exclusion and suppression has reduced some of these populations to occupied areas measured in hectares and protected status as endangered. The nonlinear relationship between perimeter and area works in the opposite direction for smaller burns. The per acre cost increases exponentially as burn areas are decreased.

Although this model focuses on fire, other disturbances can be used in combination with prescribed fire to set back fire-sensitive trees and promote fire-dependent vegetation and wildlife species. Harvest of mature trees, herbicide treatments, mowing, or even targeted grazing (e.g., silvopasture) are management techniques that can magnify fire effects (Arthur et al. 2012, Hanberry et al. 2020). Many of these activities scale linearly with area (e.g., mowing requires staff to physically visit the entire area, not just the perimeter). This can be useful in targeted areas but prohibitive for large areas.

Other opportunities for increasing capacity occur at an administrative or policy level, especially those that allow more concurrent prescribed burns on days when the weather is suitable. As climate change reduces the number of suitable days and diverts staff to fire suppression, this will become ever more important. The same staff are used for prescribed fire and fire suppression, which means prescribed fire cannot occur when staff are fighting fires or on stand-by. One way to increase daily capacity in this regard would be to hire a burn crew (even if only seasonally) dedicated only to prescribed fire. This would allow burning when weather is suitable and safe in one part of the state, but dangerous in other parts of the state. An alternative to a dedicated DNR burn crew would be to develop an interagency burn crew dedicated to prescribed fire.

Another way to increase capacity would be to allow non-FRD staff with fire experience to conduct low complexity burns on marginally wet days. Some days, especially in fall, may have short daily windows in which fire carries through vegetation. Days are short and the window between dew drying from vegetation and rising humidity toward evening limits opportunities to engage a full burn crew. In addition, in the fall fire officers are increasingly called away to fight fires in the western United States via interagency cooperation. However, small, low risk fires may be possible. Small fires in the fall can be useful to attract game species to specific parts of the landscape (Sullivan et al. 2020, Mason and Lashley 2021), and this could be used to create recreational opportunity on state game areas. Allowing non-FRD staff to lead low complexity burns is a common practice in other states but would require some level of trust and tolerance for risk among administrators in land managing divisions within the MDNR. It would be budget neutral (and a way to direct staff to provide services to hunters in lieu of preparation for deer check.)

Finally, if existing budget, equipment, and staff capacity to conduct prescribed fire is fully used and capacity is still limited, then prescribed fire capacity could be increased with contractors from the private sector. The MDNR has a long history of working with many of these contractors through private lands programs and in the context of the Michigan Prescribed Fire Council. However, it is not clear that the agency has authority to contract for prescribed fire on state land with staff that are not employed by the agency. Changing this would require intervention at the policy and possibly the legislative level.

Model Assumptions and Future Directions for the Agent-based Model

My model exhibits realistic agent behavior and fire effects on landscapes in southern Michigan, including emergent agent behaviors arising from land managers competing for scarce budgets and daily burn capacity. However, data to validate specific components were lacking, and future studies could be used to test and better calibrate model components. For instance, fire effects on fire dependent and fire sensitive plant species were not scaled to geographic elements that can magnify or reduce fire effects. For example, modifying fire effects using soils, such that fire effects are intensified on dry sandy soils, and mesophication is more likely on richer or less drought prone soils, would lend more realism to geographic heterogeneity in species responses to fire. Similarly, fire effects on plant species were linear in the model, but fire effects are complicated and vary by species, fire intensity, plant phenology, soil water holding capacity, and other influences (US Forest Service n.d., Miller et al. 2019, McLauchlan et al. 2020). I focused on the relative fire sensitivity of species, but better modeling of fire effects would make the model more realistic.

Another improvement to the model would be to model tree recruitment from the subcanopy to the canopy. This is a complicated process that is well developed in other models, such as LANDIS (He 2009, Shifley et al. 2017). However, each species reacts to shade in different ways, and many subcanopy species in the model do not recruit to the canopy, either because of their life history (i.e., they are grass or shrubs) or because disease limits their recruitment (e.g., American elm *Ulmus americana*, beech *Fagus grandifolia*, and ash *Fraxinus* spp). Another challenge to modeling recruitment is extensive cover of the invasive exotic shrub, autumn olive, which covers an area in the subcanopy greater than all but one canopy tree species. This species introduces a novel ecosystem process because these systems formerly lacked a common nitrogen-fixing shrub. The effects of extensive autumn olive subcanopy on native tree recruitment are poorly understood, but one study found a negative effect of autumn olive on native tree diversity (Nickelson et al. 2015).

Stands for prescribed fire were selected randomly in the model. This creates a relatively homogenous effect of fire on vegetation across the landscape. However, managers use a complex system to assign priorities to different proposed burns, which focuses burning on a subset of the landscape. Extending the model such that agents focus effort would add more realistic heterogeneity to the landscape pattern of mesophication and oak conservation.

Finally, the model could be expanded with feedbacks between the effects of fire (or lack thereof) and agent decision making about where and how to use fire in the future. Managers could be made to monitor the success of fires in meeting their objectives, and that success could change how much fire they use, how they prioritize potential burn sites, and their focus on repeatedly burning one area until it meets restoration thresholds before trying to burn other areas. This change would move the model from being a social-ecological model (Ostrom 2009) to a subset of social-ecological models called coupled human and natural systems (CHANS) models (Liu et al. 2007*b*, An 2012). The current ABM includes interactions in which the climate systems affect the social system, which affects the natural system, but the feedback loop back to the social system is currently lacking.

CONCLUSIONS

The climate-agent-forest model of prescribed fire use on State Game Areas and State Parks in southern Michigan simulates important aspects of the management program, and it sheds light on management trajectories and trade-offs regarding the ways that daily fire weather emerges to restrict, rather than shift, prescribed burn season in a changing climate. It also highlights the counter-intuitive ways that budget competition incentivizes adaptation attempts among agents with the least upland oak forest resource. The model also shows that prescribed fire in the future will be limited by dry conditions, despite an increase in the mean annual precipitation in the region, and that budget and capacity to conduct multiple burns per day limit climate adaptation more than the ability of land manager agents to shift their burn season forward or backward in spring or fall. Ultimately, many of these insights would not have been possible with explicit consideration of the emergence of adaptation to changing climate from simulation of daily weather and land manager decisions.

APPENDICES

APPENDIX A: FIGURES

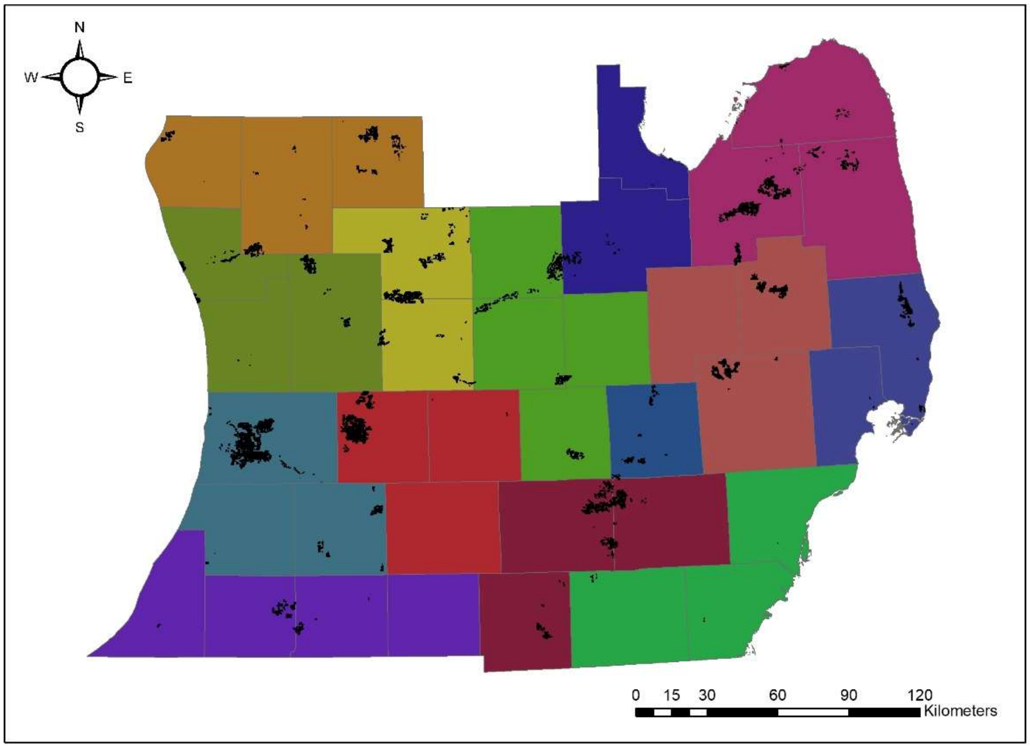


Figure 9. Southern Michigan showing the 14 multi-county regions managed by simulated land manager agents, and the 7,013 upland forest stands that they could manage using prescribed fire.

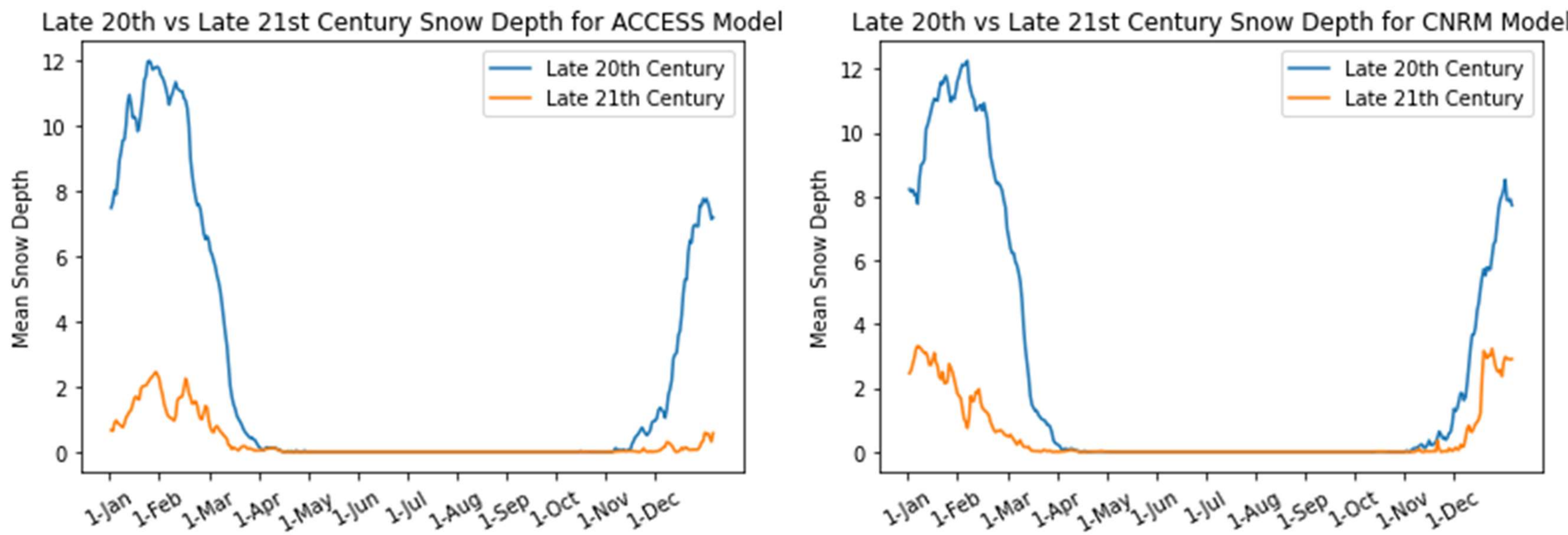


Figure 10. Mean modeled snow depth in centimeters on a given day for southern Michigan. The drier ACCESS-Reg4CM is on the left and the wetter CNRM-Reg4CM is on the right. Snow depths are shown for the late 20th century (1980-1999, in blue) and the late 21st century (2080-2099, in orange). Snow on the ground delimits the season when prescribed fire can occur.

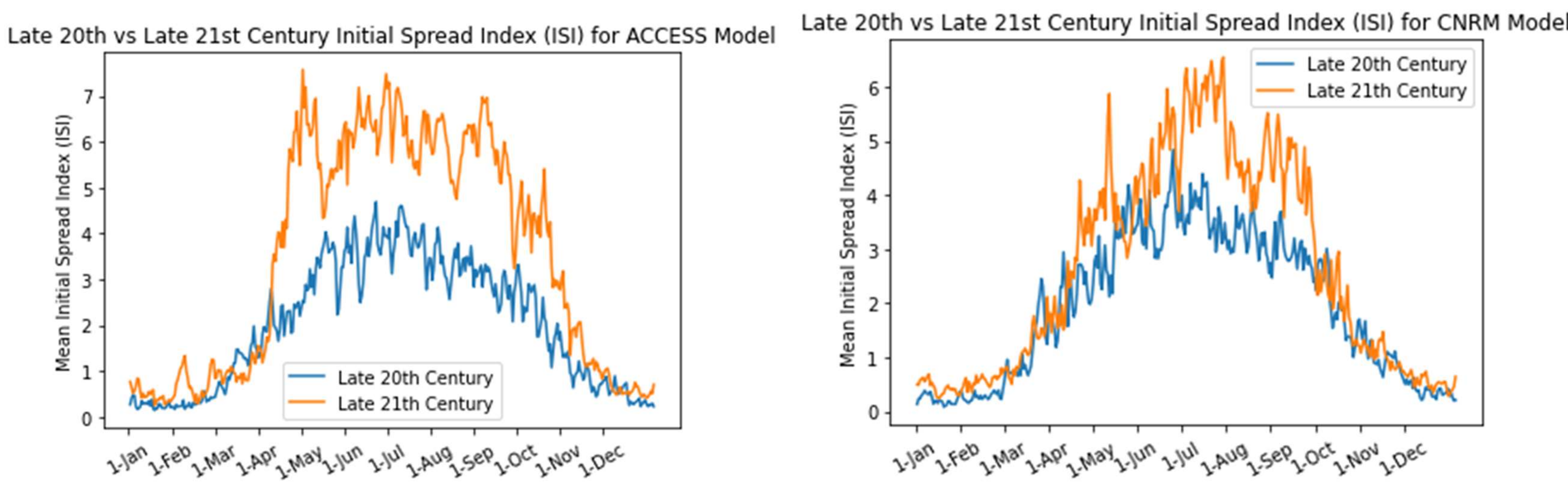
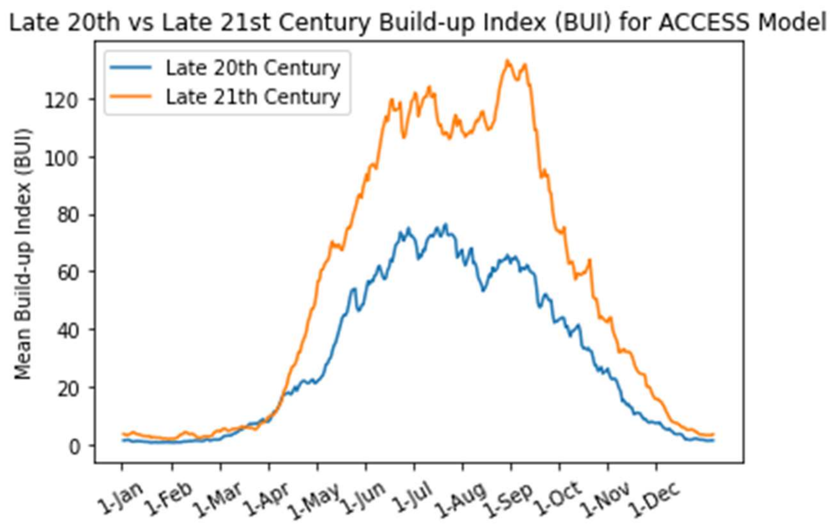


Figure 11. Initial Spread Index (ISI), a measure of the rate at which fires spread, which includes wind speed, temperature, and humidity. Prescribed fires are not sustainable at low ISI but are unsafe when ISI is too high. The drier ACCESS mode is on the left, and the wetter on the right. Blue are late 20th century values and orange are late 21st century values.



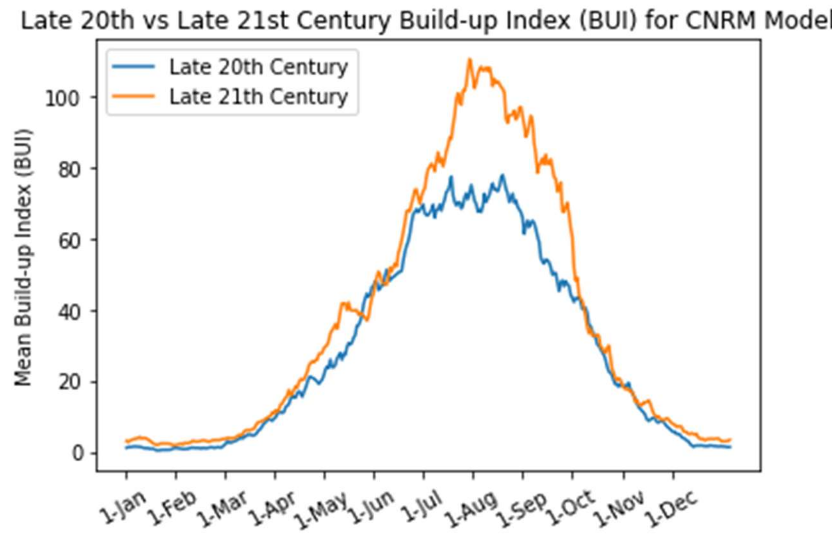


Figure 12. Build-Up Index (BUI), a measure of drought and the dryness of larger fuels, which accumulates over time and depends largely on the days since precipitation. Prescribed fires are less intense at low BUI, and more intense and less safe when BUI is high. The drier ACCESS mode is on the left, and the wetter on the right. Blue are late 20th century values and orange are late 21st century values.

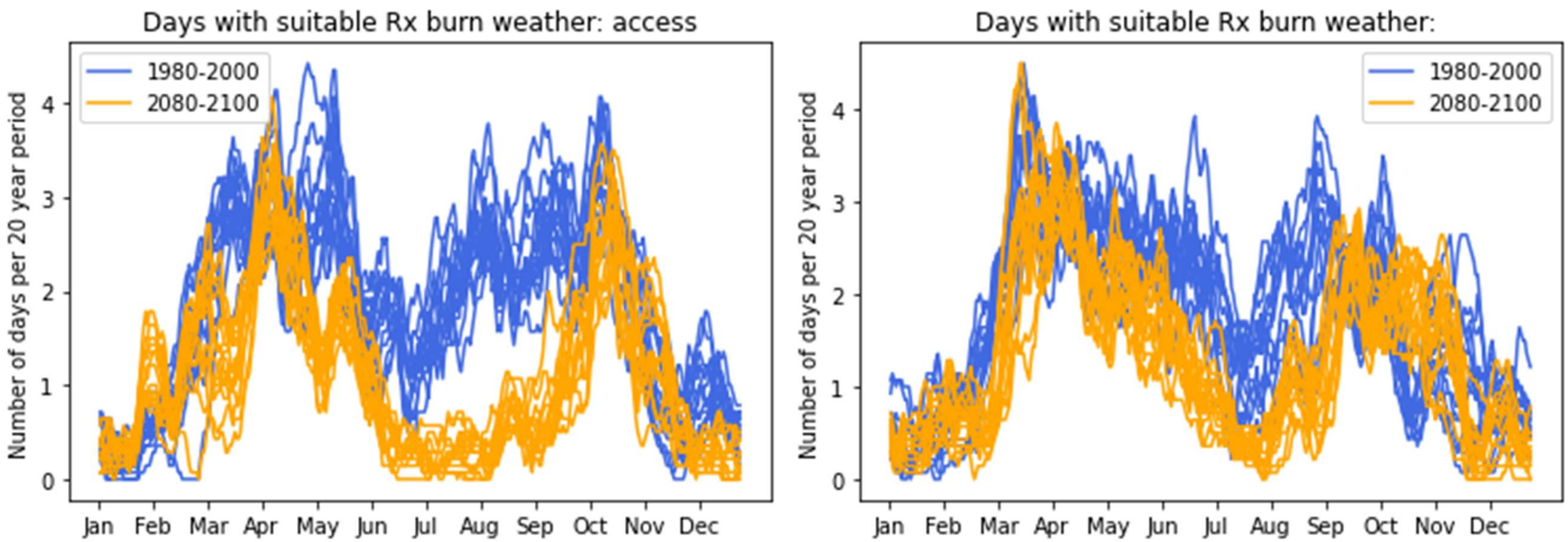


Figure 13. The number of days with suitable weather for prescribed burns (moderate ISI and BUI) for each calendar date out of the 20-year modeled weather. The drier ACCESS mode is on the left, and wetter CNRM on the right. Each blue line is the weather in an agent's multi-county area modeled for late 20th century, and orange for late 21st century. A peak on the graph represents a date of the year that is most likely to be suitable for being ready for prescribed fire. Summers become less suitable in both models, and winters do not become suitable, even in the drier ACCESS model by the late 21st century.

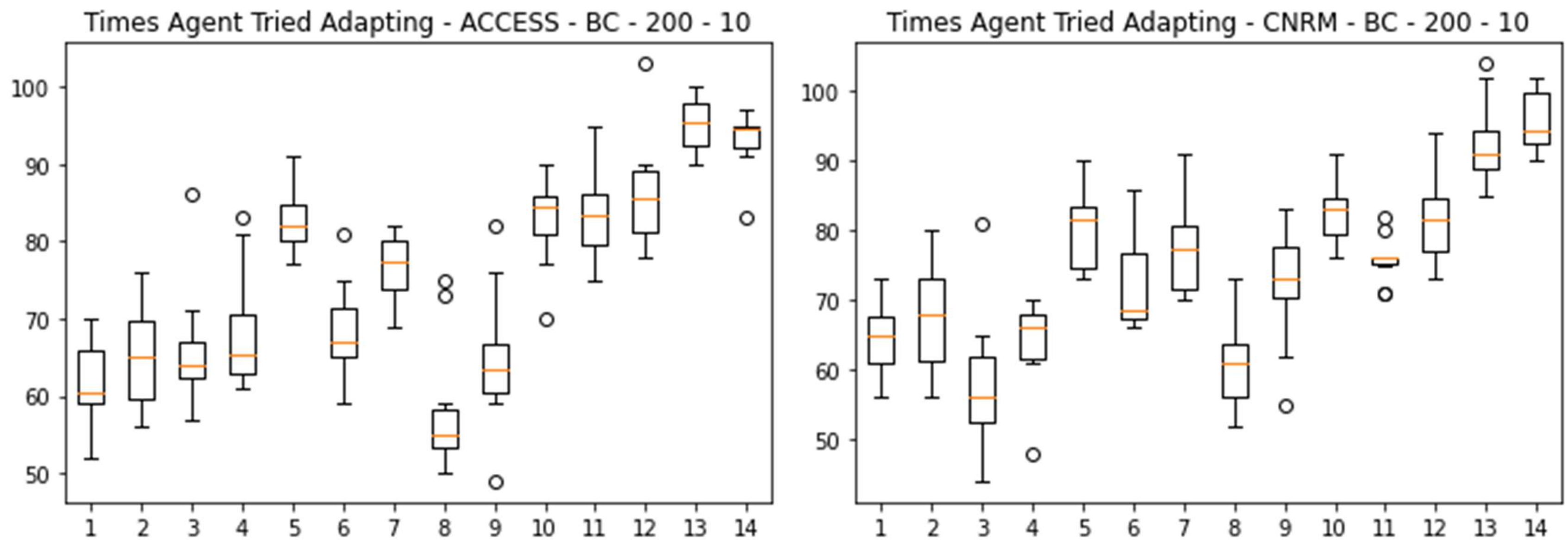


Figure 14. Number of times each agent tried to adapt in the drier ACCESS model and wetter CNRM model, under the scenario in which adaptation was triggered by competition for scarce budget resources. Agents are ordered from left to right by the number of forested stands that occur in their multi-county area of responsibility.

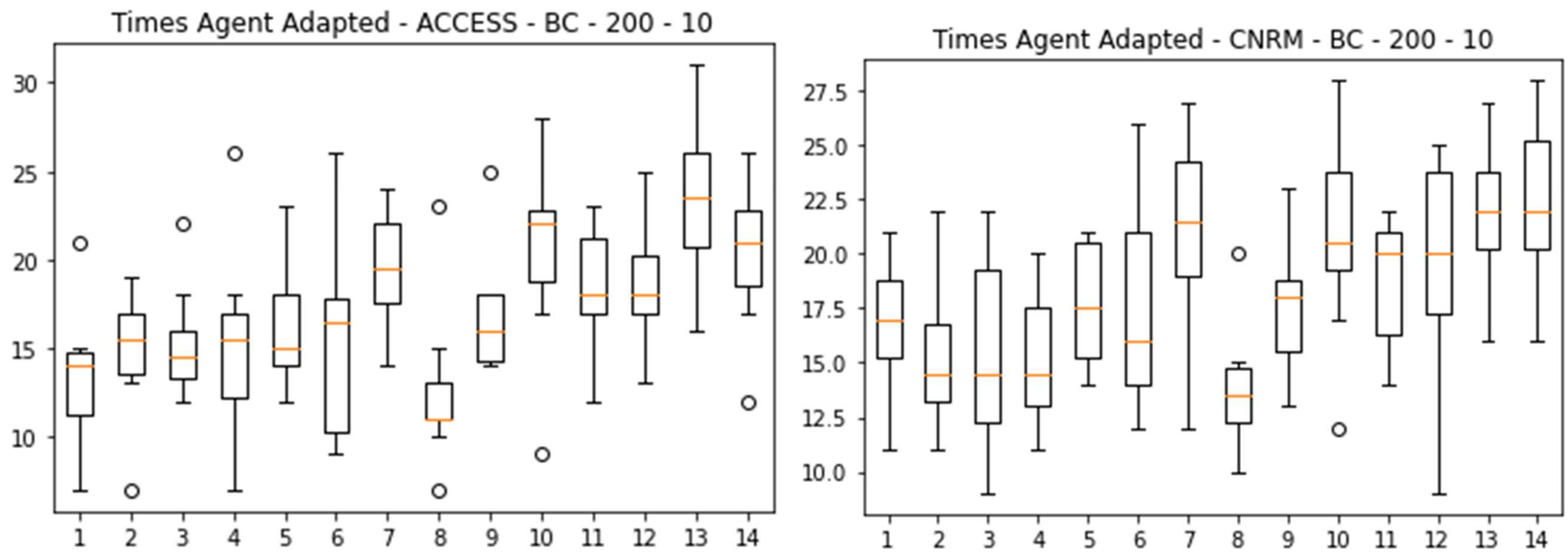


Figure 15. Number of times each agent adapted by changing the start date of the burn season in their area, in the drier ACCESS model and wetter CNRM model, under the scenario in which adaptation was triggered by competition for scarce budget resources. Agents are ordered from left to right by the number of forested stands that occur in their multi-county area of responsibility.

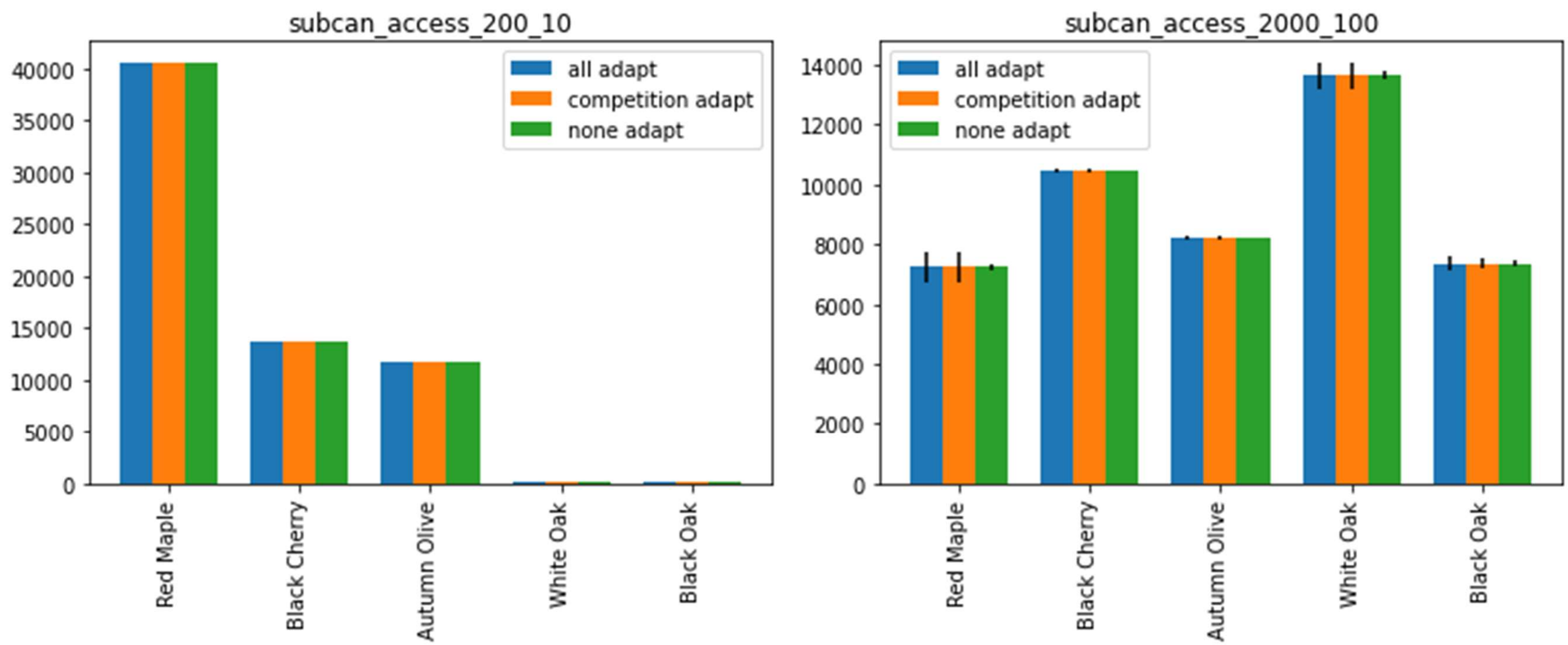


Figure 16. Effect of three adaptation trigger scenarios (all adapt, budget equity triggers adaptation, or no agents adapt) was negligible. The effect of budget and capacity scenario was much stronger. Left: annual capacity of 200 stands, and daily capacity of 10 stands. Right: annual capacity of 2000 stands and daily capacity of 100 stands.

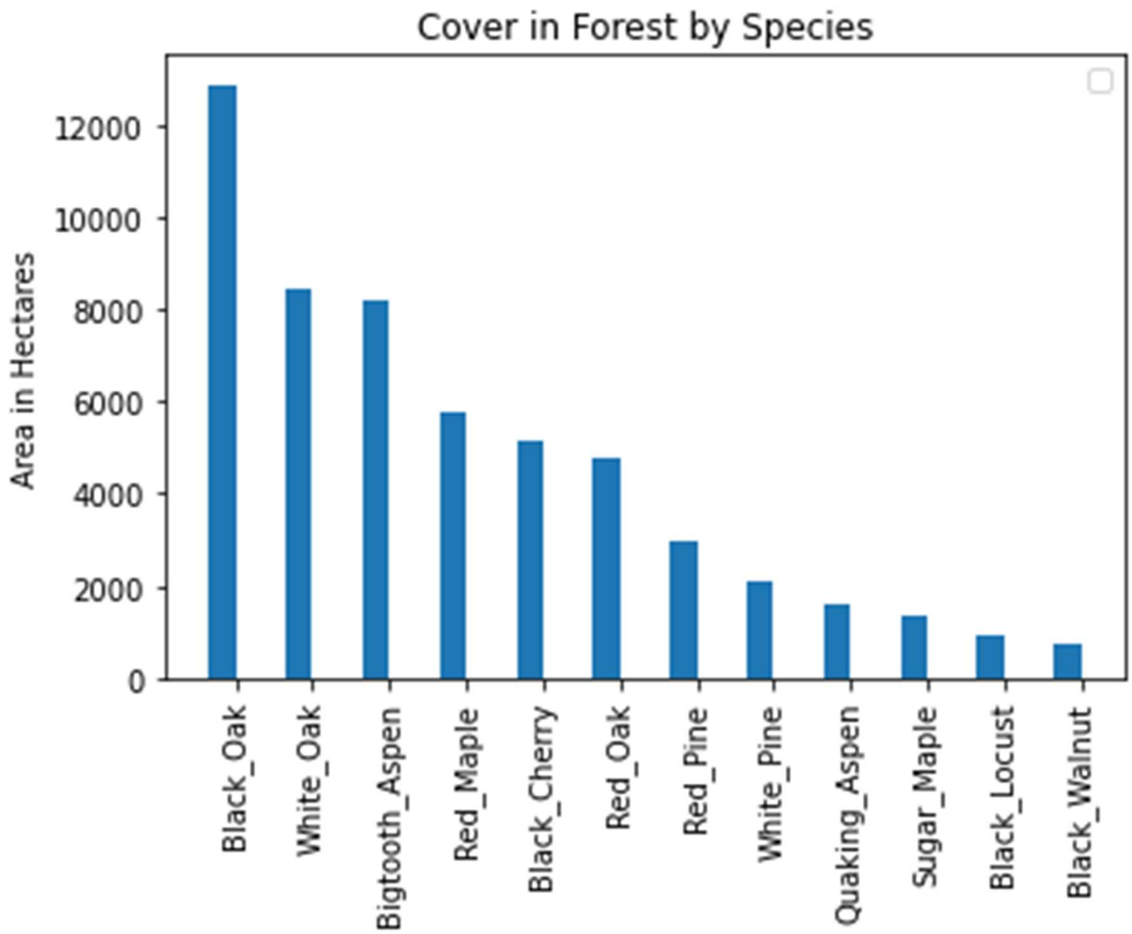


Figure 17. Most common 12 canopy species by area covered in the early 21st century in southern Michigan, USA, representing baseline conditions for an agent-based model. The three most common species are disturbance dependent species that need full exposure to sunlight to recruit from subcanopy to canopy.

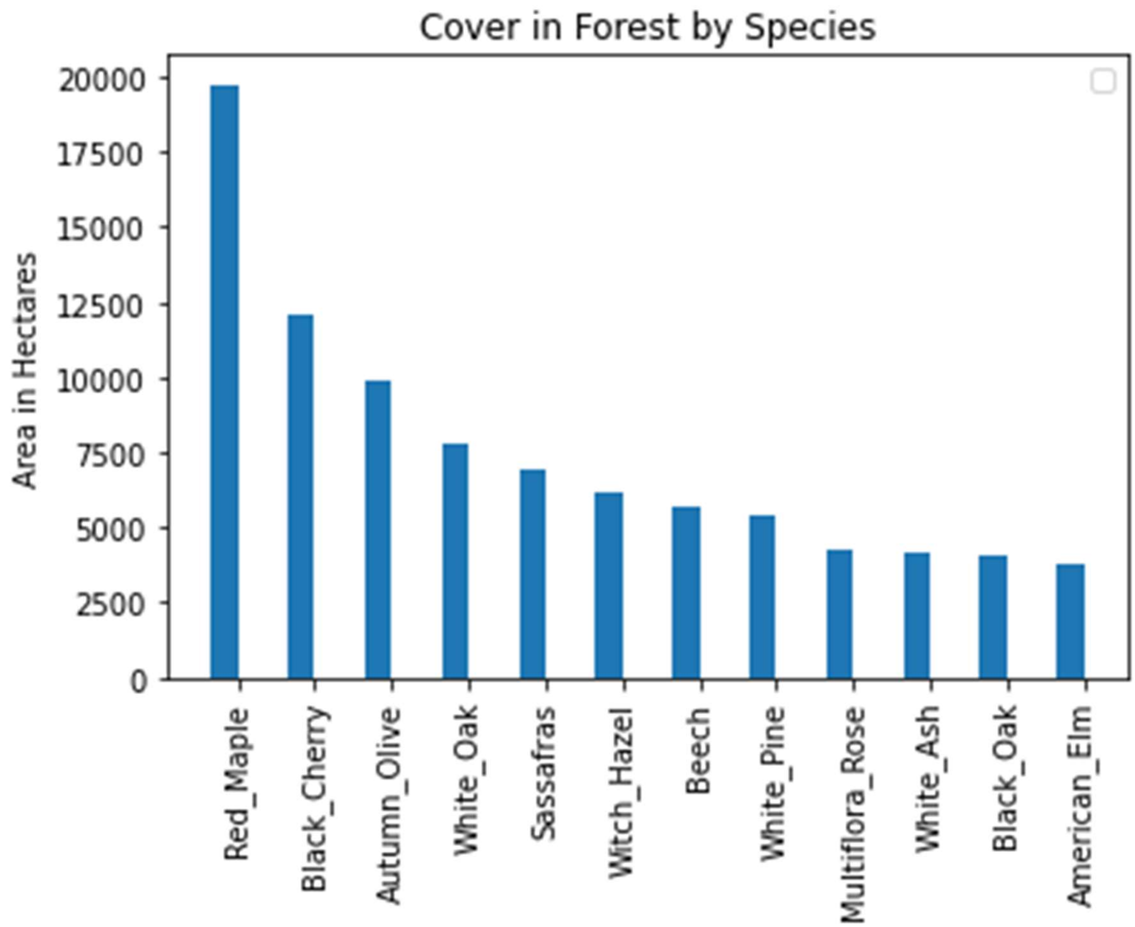


Figure 18. Most common 12 subcanopy species by area covered in the early 21st century in southern Michigan, USA, representing baseline conditions for an agent-based model. The two most common species are fire-sensitive species. The third most common species is an exotic shrub that covers more subcanopy area than all but one canopy species.

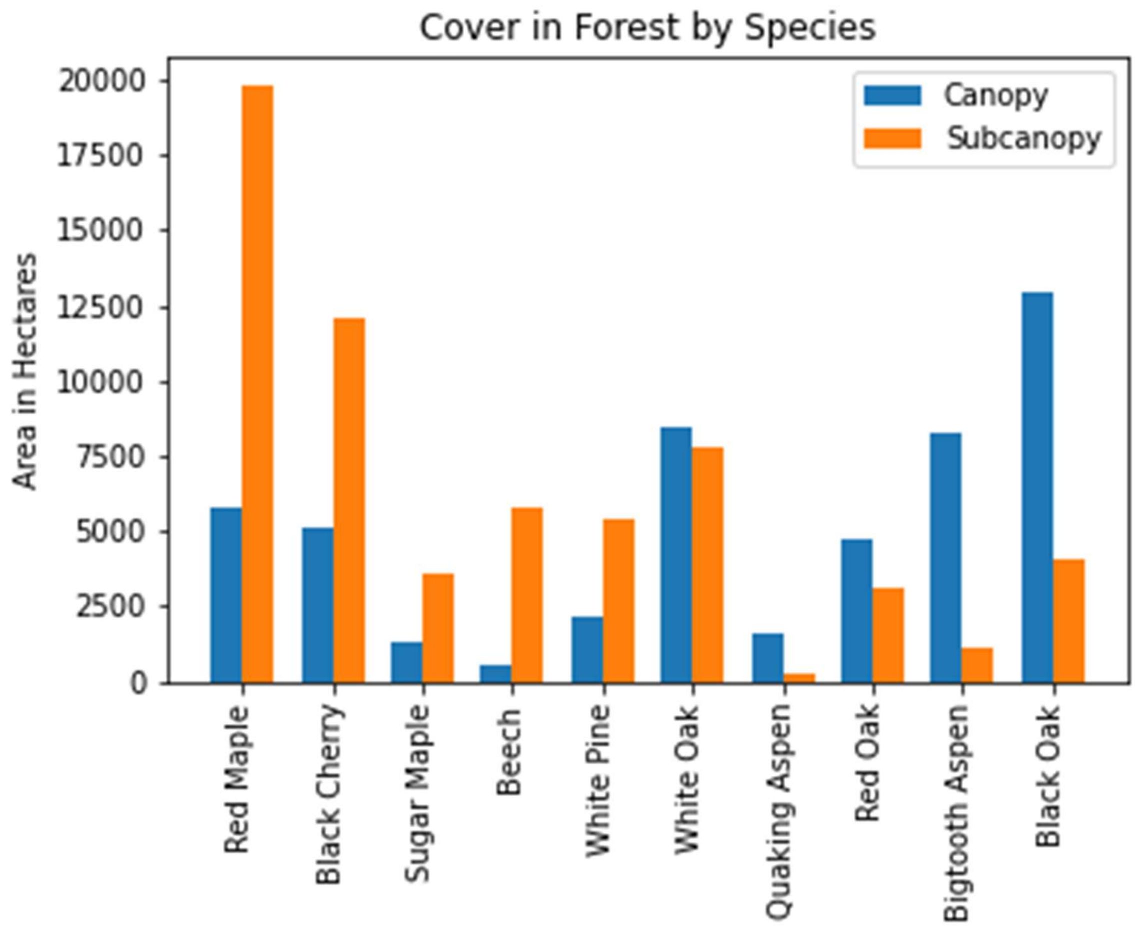


Figure 19. Ten common canopy and subcanopy tree species by area covered for the early 21st century, southern Michigan, USA. These Michigan Forest Inventory data represent baseline conditions for the agent-based model. The two most common species are fire-sensitive species. The greater area covered by red maple and black cherry in the subcanopy relative to the canopy, and the opposite pattern for red oak and black oak indicate an advanced state of mesophication on upland state lands in southern Michigan.

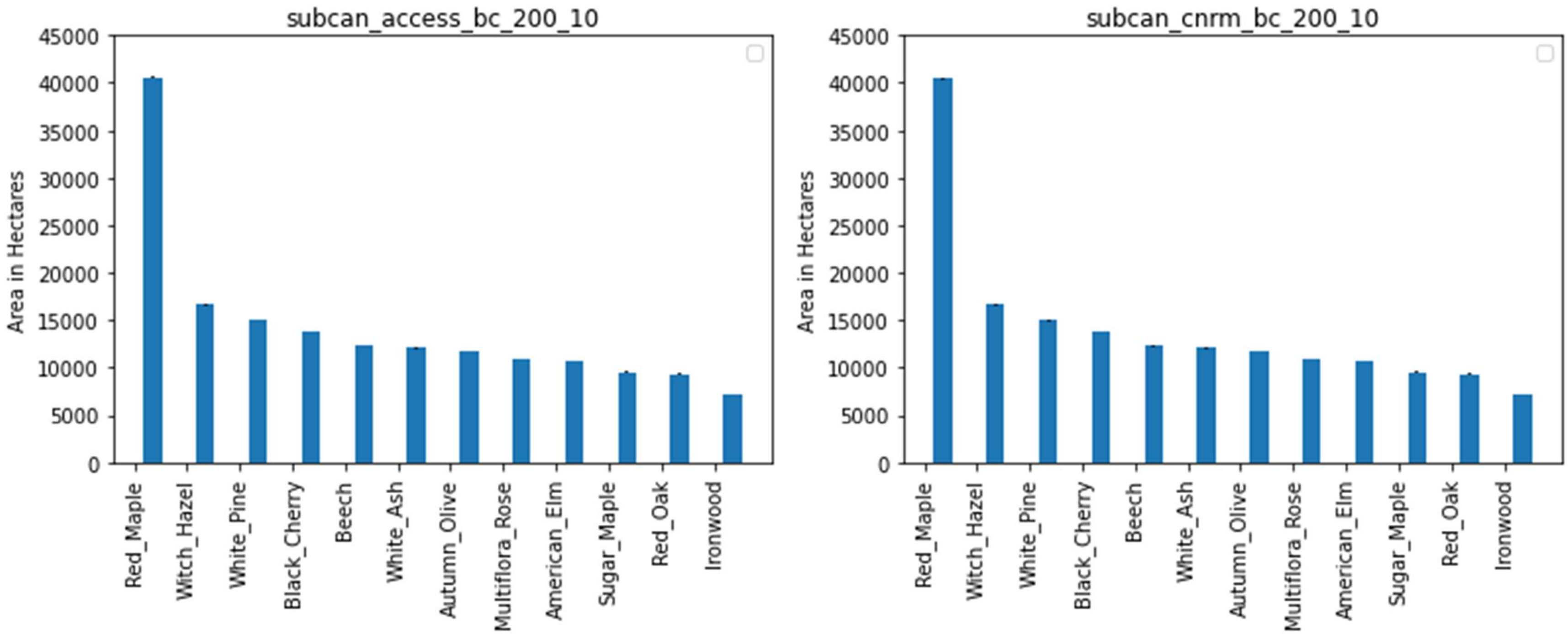


Figure 20. Most common 12 subcanopy species by area covered in the late 21st century. These are the model output conditions for the agent-based model with business-as-usual scenario levels of annual budget and daily capacity. All of the 12 most common species are at least slightly fire sensitive, and the two exotic species are more common than in the baseline early 21st century dataset. 95% confidence interval bars are too small to show for most species after 10 model runs, and differences between climate models are negligible.

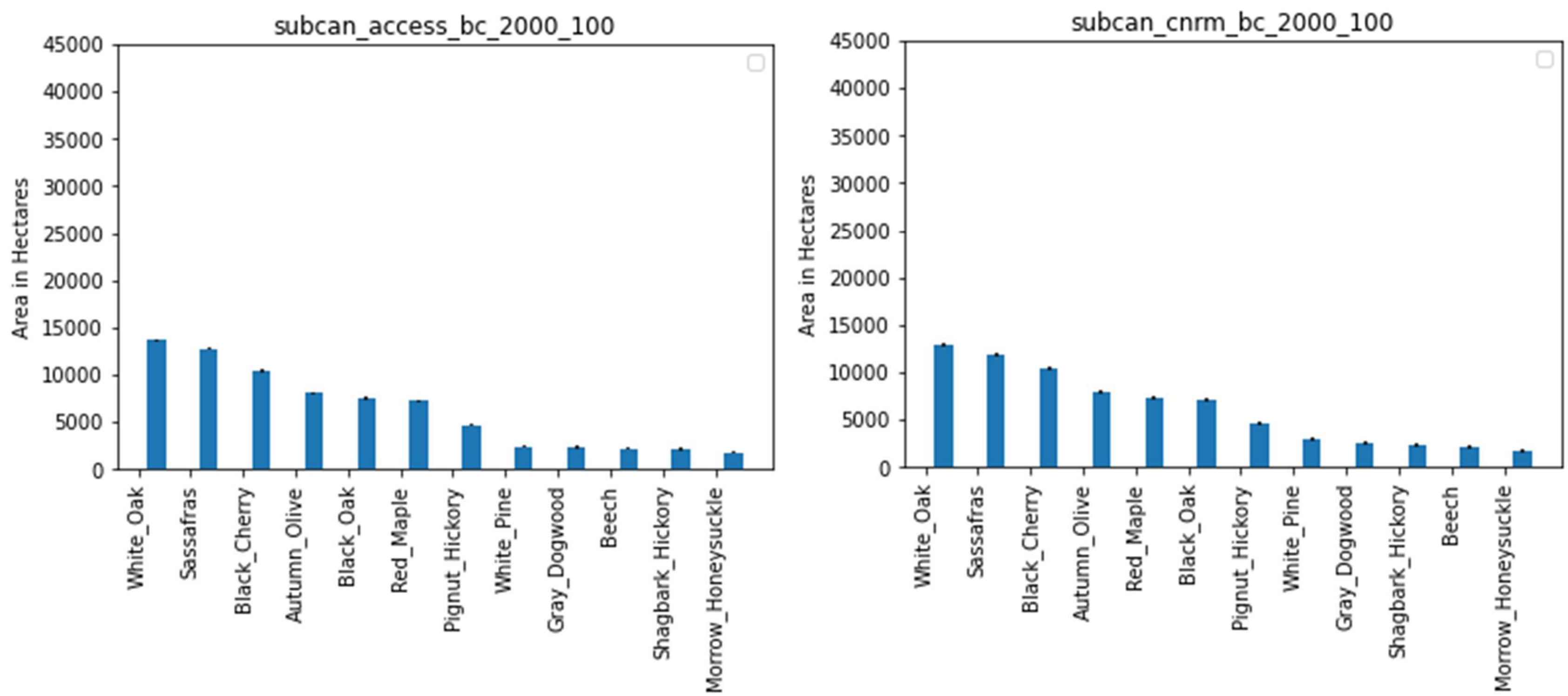


Figure 21. Most common 12 subcanopy species by area covered in the late 21st century. These are the model output conditions for the agent-based model with the scenario of a ten-fold increase in both annual budget and daily capacity. Red maple no longer dominates the subcanopy, and there is a mix of fire sensitive and fire dependent species. Exotic species are more similar to the baseline early 21st century dataset. 95% confidence interval bars are too small to show for most species after 10 model runs, and differences between climate models are negligible.

APPENDIX B: TABLES

Species Sensitivity Index Method

All 208 plant species in the Michigan Forest Inventory for the study area were initially ranked by fire sensitivity based on the literature (primarily US Forest Service n.d., Packard and Mutel 1997, Cohen et al. 2015). Species were first sorted into three bins: 1) species sensitive to fire or only present in mesic ecological communities with low fire return intervals; 2) species favored by a frequent fire return interval or commonly associated with pyric ecological communities like prairies or savannas; and 3) species that were described as fire neutral or were not described as having strong fire effects. Species were ordered within the three categories. Species that occur primarily in ecological communities that burn annually were given the lowest fire sensitivity (-10), whereas species that occur primarily in communities that have the longest fire return intervals were given the highest fire sensitivity (10). Relative fire sensitivities were refined by incorporating expert opinion of natural heritage program botanists and ecologists and land managers familiar with fire effects on vegetation in Michigan. Many tree species responded differently to fire at different life stages, and thus canopy species and subcanopy species were ranked differently. For example, mature jack pine reproduces vigorously following a fire, but the same species is very sensitive to fire before it reaches reproductive maturity.

Table 1. Fire sensitivity index for all canopy species in the Michigan Forest Inventory dataset.

Common Name	Scientific Name	Sensitivity Index
Sugar Maple	*Acer saccharum*	-9.99999
American Elm	*Ulmus americana*	-9.95213
Mountain Maple	*Acer spicatum*	-9.34202
Rock Elm	*Ulmus thomasii*	-9.21301
Slippery Elm	*Ulmus rubra*	-9.19997
Beech	*Fagus grandifolia*	-9.16218
Musclewood/Hornbeam	*Carpinus caroliniana*	-9.06921
Black Maple	*Acer nigrum*	-8.85312
Ironwood	*Ostrya virginiana*	-8.60939
Yellow Birch	*Betula alleghaniensis*	-8.48368
Balsam Fir	*Abies balsamea*	-8.41402
Red Mulberry	*Morus rubra*	-8.16792
Northern White Cedar	*Thuja occidentalis*	-8.1045
Douglas Fir	*Pseudotsuga menziesii*	-7.86227
Norway Spruce	*Picea abies*	-7.73431
White Mulberry	*Morus alba*	-7.66311
Sycamore	*Platanus occidentallis*	-7.36947
Eastern Red Cedar	*Juniperus virginiana*	-7.36435
Black Willow	*Salix nigra*	-7.35888
White Ash	*Fraxinus americana*	-7.21149
Bitternut Hickory	*Carya cordiformis*	-7.20561
Boxelder	*Acer negundo*	-6.75362
Hemlock	*Tsuga canadensis*	-6.73309
Silver Maple	*Acer saccharinum*	-6.70662
Swamp Cottonwood	*Populus heterophylla*	-6.53957
Basswood	*Tilia americana*	-6.45215
Black Ash	*Fraxinus nigra*	-6.41765
Honeylocust	*Gleditsia triacanthos*	-6.21816
Black/European Alder	*Alnus glutinosa*	-6.12682
Hackberry	*Celtis occidentalis*	-5.97307
Butternut	*Juglans cinerea*	-5.93948
Red Oak	*Quercus rubra*	-5.74396
Green Ash	*Fraxinus pennsylvanica*	-5.60318
Catalpa	*Catalpa speciosa*	-5.43203
Sweet Cherry	*Prunus avium*	-5.0331
Yellow Poplar (Tulip Tree)	*Liriodendron tulipifera*	-4.54216
Red Maple	*Acer rubrum*	-2.59037
Pin Oak (Southern)	*Quercus palustris*	-2.42438

Table 1 (cont'd)

Common Name	Scientific Name	Sensitivity Index
Scotch Pine	*Pinus sylvestris*	-1.83448
Tamarack	*Larix laricina*	-1.91661
Weeping Willow	*Salix sepulcralis*	-1.24964
Blue Spruce	*Picea pungens*	-0.5931
Tree of Heaven	*Ailanthus altissima*	-0.35176
Austrian Pine	*Pinus nigra*	-0.20108
Blackgum	*Nyssa sylvatica*	-0.12398
Black Cherry	*Prunus serotina*	-0.01764
Chinkapin Oak	*Quercus muehlenbergii*	0.1906
Black Locust	*Robinia pseudoacacia*	0.45785
American Chestnut	*Castanea dentata*	0.70136
Scarlet Oak	*Quercus coccinea*	0.70637
Osage Orange	*Maclura pomifera*	1.58086
Black Walnut	*Juglans nigra*	1.64321
Cottonwood	*Populus deltoides*	1.79621
Black Spruce	*Picea mariana*	2.60435
White Spruce	*Picea glauca*	2.78427
Balsam Poplar	*Populus balsamifera*	3.4365
Quaking Aspen	*Populus tremuloides*	3.6753
Bigtooth Aspen	*Populus grandidentata*	3.79299
White Pine	*Pinus strobus*	4.83329
Choke Cherry	*Prunus virginiana*	5.00364
Pin Cherry	*Prunus pensylvanica*	6.42832
Red Pine	*Pinus resinosa*	6.78334
Swamp White Oak	*Quercus bicolor*	7.33553
Northern Pin Oak	*Quercus ellipsoidalis*	7.67685
Paper Birch	*Betula papyrifera*	7.70507
Sassafras	*Sassafras albidum*	8.06081
White Oak	*Quercus alba*	8.33608
Pignut Hickory	*Carya glabra*	8.42681
Jack Pine	*Pinus banksiana*	8.5046
Bur Oak	*Quercus macrocarpa*	8.8
Shellbark Hickory	*Carya laciniosa*	9.29853
Black Oak	*Quercus velutina*	9.44286
Shagbark Hickory	*Carya ovata*	9.76172

Table 2. Fire sensitivity index for all subcanopy species in the Michigan Forest Inventory dataset.

Common Name	Scientific Name	Sensitivity Index
Black Maple	*Acer nigrum*	-9.99999
Sugar Maple	*Acer saccharum*	-9.99999
Norway Spruce	*Picea abies*	-9.99523
Alternate-leaved Dogwood	*Cornus alternifolia*	-9.82727
Maple Leaved Viburnum	*Viburnum acerifolium*	-9.82121
Ironwood	*Ostrya virginiana*	-9.74745
Spicebush	*Lindera benzoin*	-9.72357
Pawpaw	*Asimina triloba*	-9.60593
White Ash	*Fraxinus americana*	-9.48941
Eastern Red Cedar	*Juniperus virginiana*	-9.38723
Red Berried Elder	*Sambucus racemosa*	-9.38196
Mountain Maple	*Acer spicatum*	-9.28157
American Fly Honeysuckle	*Lonicera canadensis*	-9.27363
Red Pine	*Pinus resinosa*	-9.26555
White Mulberry	*Morus alba*	-9.20544
American Elm	*Ulmus americana*	-9.18208
Creeping Juniper	*Juniperus horizontalis*	-9.17881
Black Ash	*Fraxinus nigra*	-9.15608
Basswood	*Tilia americana*	-9.13046
Balsam Fir	*Abies balsamea*	-9.10418
Common Privet	*Ligustrum vulgare*	-8.76479
Red Mulberry	*Morus rubra*	-8.74095
Black Willow	*Salix nigra*	-8.73855
Black Haw	*Viburnum prunifolium*	-8.6021
Swamp Rose	*Rosa palustris*	-8.59094
Rock Elm	*Ulmus thomasii*	-8.58262
Yellow Birch	*Betula alleghaniensis*	-8.57739
Catalpa	*Catalpa speciosa*	-8.56716
Northern Gooseberry	*Ribes oxyacanthoides*	-8.55218
Musclewood/Hornbeam	*Carpinus caroliniana*	-8.5229
Prickly Or Wild Gooseberry	*Ribes cynosbati*	-8.49426
Jack Pine	*Pinus banksiana*	-8.47958
Norway Maple	*Acer platanoides*	-8.30125
Hemlock	*Tsuga canadensis*	-8.25426
Canadian Yew	*Taxus canadensis*	-8.24683
White Pine	*Pinus strobus*	-8.23574
Common Or Ground Juniper	*Juniperus communis*	-8.23283
Witch Hazel	*Hamamelis virginiana*	-8.14862
Yellow Poplar (Tulip Tree)	*Liriodendron tulipifera*	-8.00702

Table 2 (cont'd)

Common Name	Scientific Name	Sensitivity Index
Multiflora Rose	*Rosa multiflora*	-7.95687
Leatherwood	*Dirca palustris*	-7.68773
Chinkapin Oak	*Quercus muehlenbergii*	-7.62138
Northern White Cedar	*Thuja occidentalis*	-7.60694
Smooth Arrow Wood	*Viburnum dentatum*	-7.58871
Green Ash	*Fraxinus pennsylvanica*	-7.5641
Silver Maple	*Acer saccharinum*	-7.52662
Hazelnut (Beaked)	*Corylus cornuta*	-7.49379
Nannyberry	*Viburnum lentago*	-7.49046
Mountain Ash	*Sorbus americana*	-7.4465
Flowering Dogwood	*Cornus florida*	-7.39829
Bitternut Hickory	*Carya cordiformis*	-7.34521
Blue Ash	*Fraxinus quadrangulata*	-6.99785
Butternut	*Juglans cinerea*	-6.88708
Boxelder	*Acer negundo*	-6.82753
Wild Raisin	*Viburnum cassinoides*	-6.77315
Oval-leaved Privet	*Ligustrum ovalifolium*	-6.72662
Red Oak	*Quercus rubra*	-6.64918
Apple Rose	*Rosa villosa*	-6.39885
Dwarf Hackberry	*Celtis tenuifolia*	-6.34953
Paper Birch	*Betula papyrifera*	-6.24937
Buckeye	*Aesculus glabra*	-6.22885
American Highbush Cranberry	*Viburnum trilobum*	-6.01589
Pin Oak (Southern)	*Quercus palustris*	-5.85302
Japanese Barberry	*Berberis thunbergii*	-5.72872
Red Maple	*Acer rubrum*	-5.38259
Wormwood	*Artemisia pontica*	-5.3331
Wild Black Currant	*Ribes americanum*	-5.30382
Honeylocust	*Gleditsia triacanthos*	-5.25478
Black Currant	*Ribes nigrum*	-4.60955
Red Currant	*Ribes rubrum*	-4.38714
Weeping Willow	*Salix sepulcralis*	-4.26316
Southernwood	*Artemisia abrotanum*	-4.0467
European Highbush Cranberry	*Viburnum opulus*	-3.87548
Jetbead	*Rhodotypos scandens*	-3.79026
Slender Willow	*Salix petiolaris*	-3.70406
Sweet Cherry	*Prunus avium*	-3.56147
Beech	*Fagus grandifolia*	-3.52625
Michigan Holly	*Ilex verticillata*	-3.30587
Scotch Pine	*Pinus sylvestris*	-3.24471

Table 2 (cont'd)

Common Name	Scientific Name	Sensitivity Index
Trumpet Honeysuckle	*Lonicera sempervirens*	-2.93855
Chinese Buckthorn	*Rhamnus utilis*	-2.94105
Red Honeysuckle	*Lonicera dioica*	-2.91668
Glossy Buckthorn	*Rhamnus frangula*	-2.84344
Common Buckthorn	*Rhamnus cathartica*	-2.68596
Downy Arrow Wood	*Viburnum rafinesquianum*	-2.60255
European Honeysuckle	*Lonicera caprifolium*	-2.32914
Elderberry	*Sambucus canadensis*	-1.9891
Wayfaring Tree	*Viburnum lantana*	-1.71737
White Spruce	*Picea glauca*	-1.70879
Black Spruce	*Picea mariana*	-1.66687
Hackberry	*Celtis occidentalis*	-1.54237
Hairy Honeysuckle	*Lonicera hirsuta*	-1.45084
Amur Honeysuckle	*Lonicera maackii*	-1.29505
Osage Orange	*Maclura pomifera*	-1.26019
Garlic Mustard	*Alliaria petiolata*	-0.9951
Morrow Honeysuckle	*Lonicera morrowii*	-0.9315
Cottonwood	*Populus deltoides*	-0.74781
Autumn Olive	*Elaeagnus umbellata*	-0.70568
Oriental Bittersweet	*Celastrus orbiculata*	-0.68436
Smooth Tartarian Honeysuckle	*Lonicera tatarica*	-0.66194
Hybrid Honeysuckle	*Lonicera xbella*	-0.49024
Blackgum	*Nyssa sylvatica*	-0.47617
American Bittersweet	*Celastrus scandens*	-0.47321
Black Cherry	*Prunus serotina*	-0.42887
Spotted Knapweed	*Centaurea stoebe*	-0.33549
European Fly Honeysuckle	*Lonicera xylosteum*	-0.29468
Balsam Poplar	*Populus balsamifera*	-0.27201
Red Osier Dogwood	*Cornus stolonifera*	-0.19612
Bigtooth Aspen	*Populus grandidentata*	-0.03256
American Chestnut	*Castanea dentata*	0.12678
Quack Grass	*Elymus repens*	0.14167
Reed Canary Grass	*Phalaris arundinacea*	0.23908
Sand Dune Willow	*Salix cordata*	0.37917
Buttonbush	*Cephalanthus occidentalis*	0.44622
Poison Ivy	*Toxicodendron radicans*	0.50252
American Crab Tree	*Malus coronaria*	0.50339
Frost Grape	*Vitis vulpina*	0.51283
Swamp Dewberry	*Rubus hispidus*	0.52767
Summer Grape	*Vitis aestivalis*	0.53398

Table 2 (cont'd)

Common Name	Scientific Name	Sensitivity Index
Bristly Blackberry	*Rubus setosus*	0.7874
Sweet Gale	*Myrica gale*	0.56053
Thicket Creeper	*Parthenocissus inserta*	0.5926
Narrow-leaved Cattail	*Typha angustifolia*	0.73423
Quaking Aspen	*Populus tremuloides*	0.74534
Riverbank Grape	*Vitis riparia*	0.801
Black Locust	*Robinia pseudoacacia*	0.81479
Shrubby Cinquefoil	*Dasiphora fruticosa*	0.8611
Mountain Holly	*Nemopanthus mucronatus*	0.90204
Virginia Creeper	*Parthenocissus quinquefolia*	0.9102
Common Lilac	*Syringa vulgaris*	0.92888
Common Cattail	*Typha latifolia*	0.998
Yucca	*Yucca filamentosa*	1.05927
Tree of Heaven	*Ailanthus altissima*	1.1158
Phragmites (exotic)	*Phragmites australis*	1.13118
Chokeberry	*Aronia prunifolia*	1.17624
Beach Heath	*Hudsonia tomentosa*	1.19533
Ninebark	*Physocarpus opulifolius*	1.35556
Silky Dogwood	*Cornus amomum*	1.4976
Broom Sedge	*Carex scoparia*	1.50898
Timothy	*Phleum pratense*	1.65708
Wahoo/Burning Bush (Invasive)	*Euonymus alatus*	1.82406
Pussy Willow	*Salix discolor*	1.87714
Winged Wahoo	*Euonymus alata*	1.89996
Round Leaved Dogwood	*Cornus rugosa*	1.94563
Spindle Tree	*Euonymus europaea*	1.97252
Shrubby St. John's Wort	*Hypericum prolificum*	2.11121
Running Strawberry Bush	*Euonymus obovata*	2.28211
Bladdernut	*Staphylea trifolia*	2.58768
Fly Honeysuckle	*Lonicera villosa*	2.68793
Poison Sumac	*Toxicodendron vernix*	2.87752
Eastern Redbud	*Cercis canadensis*	2.89698
Labrador Tea	*Ledum groenlandicum*	3.05987
Black Walnut	*Juglans nigra*	3.37677
Prickly Ash	*Zanthoxylum americanum*	3.40459
Smooth Highbush Blueberry	*Vaccinium corymbosum*	3.72572
Pennsylvania Blackberry	*Rubus pensylvanicus*	4.44622
Swamp White Oak	*Quercus bicolor*	4.48556
Soapberry	*Shepherdia canadensis*	5.03024
Bristly Locust	*Robinia hispida*	5.59441

Table 2 (cont'd)

Common Name	Scientific Name	Sensitivity Index
Buffalo Berry	*Shepherdia argentea*	5.77152
Orchard Grass	*Dactylis glomerata*	5.59605
Hop Tree	*Ptelea trifoliata*	6.04954
Black Raspberry	*Rubus occidentalis*	6.21197
Common Blackberry	*Rubus allegheniensis*	6.26239
Wild Red Raspberry	*Rubus strigosus*	6.40583
Bush Honeysuckle	*Diervilla lonicera*	6.48849
Pin Cherry	*Prunus pensylvanica*	6.56616
Wintergreen	*Gaultheria procumbens*	6.60037
Japanese Bush Clover	*Lespedeza thunbergii*	6.68601
Staghorn Sumac	*Rhus typhina*	6.77435
Shellbark Hickory	*Carya laciniosa*	7.02574
Fragrant Sumac	*Rhus aromatica*	7.05249
Shingle Oak	*Quercus imbricaria*	7.11241
Common Green Brier	*Smilax rotundifolia*	7.11622
Shrubby Lespedeza	*Lespedeza bicolor*	7.30693
Smooth Blackberry	*Rubus canadensis*	7.32288
Choke Cherry	*Prunus virginiana*	7.34723
Pignut Hickory	*Carya glabra*	7.57488
Poverty Grass	*Danthonia spicata*	7.70942
Hillside Blueberry	*Vaccinium pallidum*	7.93533
Northern Pin Oak	*Quercus ellipsoidalis*	7.98174
Tag Alder	*Alnus rugosa*	8.06744
Common Lowbush Blueberry	*Vaccinium angustifolium*	8.11831
Huckleberry	*Gaylussacia baccata*	8.13585
Canada Blueberry	*Vaccinium myrtilloides*	8.15618
Gray Dogwood	*Cornus foemina*	8.3273
Smooth Sumac	*Rhus glabra*	8.36771
Sweet Fern	*Comptonia peregrina*	8.49504
White Oak	*Quercus alba*	8.65234
Bracken Fern	*Pteridium aquilinum*	8.75316
Shagbark Hickory	*Carya ovata*	8.9157
Alleghany Plum	*Prunus alleghaniensis*	8.9947
Meadowsweet	*Spiraea alba*	9.09612
Winged Sumac	*Rhus copallina*	9.09709
Sand Cherry	*Prunus pumila*	9.25255
Northern Dewberry	*Rubus flagellaris*	9.5864
Black Oak	*Quercus velutina*	9.70956
Hazelnut (American)	*Corylus americana*	9.77712
Bluejoint Reed Grass	*Calamagrostis canadensis*	9.83843

Table 2 (cont'd)

Common Name	Scientific Name	Sensitivity Index
Pasture Rose	*Rosa carolina*	9.87919
Indian Grass	*Sorghastrum nutans*	9.98363
Flowering Spurge	*Euphoria corollata*	9.92388
Bur Oak	*Quercus macrocarpa*	9.93743
Little Bluestem	*Schizachyrium scoparium*	9.9485
Sassafras	*Sassafras albidum*	9.97087
New Jersey Tea	*Ceanothus americanus*	9.97193
Big Bluestem	*Andropogon gerardii*	9.98972
Switchgrass	*Panicum virgatum*	9.99887

Table 3. Area in hectares and 95% confidence intervals after 10 runs of each set of 140-year simulations using the drier ACCESS climate model. Simulations included two capacity scenarios: business as usual (no more than 200 stands per year and no more than 10 stands per day) and much increased (2,000 stands per year and 100 stands per day).

		Business-as-usual		Much increased	
Common Name	Scientific Name	mean	95% CI	mean	95% CI
Black Maple	*Acer nigrum*	666.38	17.41	28.36	14.23
Sugar Maple	*Acer saccharum*	9,547.22	75.04	908.52	125.91
Norway Spruce	*Picea abies*	446.66	5.78	26.89	9.62
Alternate-leaved Dogwood	*Cornus alternifolia*	98.82	2.76	6.25	3.55
Maple Leaved Viburnum	*Viburnum acerifolium*	4,677.49	14.36	481.96	80.66
Ironwood	*Ostrya virginiana*	7,245.57	33.65	495.69	78.89
Spicebush	*Lindera benzoin*	2,343.60	19.61	313.88	54.21
Pawpaw	*Asimina triloba*	216.72	4.09	47.43	15.93
White Ash	*Fraxinus americana*	12,093.57	79.94	1,108.06	155.22
Eastern Red Cedar	*Juniperus virginiana*	2,075.84	21.41	237.53	56.39
Red Berried Elder	*Sambucus racemosa*	199.54	4.55	22.79	6.41
Mountain Maple	*Acer spicatum*	7.15	1.21	0.00	0.00
American Fly Honeysuckle	*Lonicera canadensis*	17.64	0.29	0.50	0.71
Red Pine	*Pinus resinosa*	1,055.95	7.78	158.29	22.17
White Mulberry	*Morus alba*	242.46	2.43	79.06	20.85
American Elm	*Ulmus americana*	10,730.55	27.14	1,054.88	187.89
Creeping Juniper	*Juniperus horizontalis*	65.09	10.09	0.00	0.00
Black Ash	*Fraxinus nigra*	378.28	2.66	70.78	14.48
Basswood	*Tilia americana*	2,605.79	56.08	238.64	42.14
Balsam Fir	*Abies balsamea*	44.32	0.89	0.34	0.45
Common Privet	*Ligustrum vulgare*	183.11	1.97	16.16	9.22
Red Mulberry	*Morus rubra*	20.72	0.32	0.00	0.00
Black Willow	*Salix nigra*	11.00	0.16	2.48	0.95
Black Haw	*Viburnum prunifolium*	62.36	0.58	13.34	9.88
Swamp Rose	*Rosa palustris*	24.35	0.74	5.92	3.85
Rock Elm	*Ulmus thomasii*	11.25	2.12	0.00	0.00
Yellow Birch	*Betula alleghaniensis*	344.67	15.29	18.50	6.71
Catalpa	*Catalpa speciosa*	238.32	0.84	97.15	26.21

Table 3 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much increased	
		mean	95% CI	mean	95% CI
Northern Gooseberry	*Ribes oxyacanthoides*	6.33	0.24	1.57	0.93
Musclewood/Hornbeam	*Carpinus caroliniana*	6,331.27	35.14	664.95	113.91
Prickly Or Wild Gooseberry	*Ribes cynosbati*	2,757.76	26.49	331.80	54.00
Big Bluestem	*Andropogon gerardii*	3.06	0.03	27.75	0.66
Jack Pine	*Pinus banksiana*	329.76	11.09	41.59	10.72
Norway Maple	*Acer platanoides*	43.35	0.94	8.36	7.38
Hemlock	*Tsuga canadensis*	1,795.30	39.94	121.86	20.37
Canadian Yew	*Taxus canadensis*	4.17	0.03	1.95	0.83
White Pine	*Pinus strobus*	15,012.23	34.89	2,456.11	353.68
Common Or Ground Juniper	*Juniperus communis*	600.12	6.16	65.50	12.94
Witch Hazel	*Hamamelis virginiana*	16,617.19	43.96	1,445.39	217.26
Yellow Poplar (Tulip Tree)	*Liriodendron tulipifera*	263.97	2.70	38.52	5.60
Multiflora Rose	*Rosa multiflora*	10,941.49	25.23	1,097.47	113.91
Leatherwood	*Dirca palustris*	20.38	1.36	0.53	0.52
Chinkapin Oak	*Quercus muehlenbergii*	57.68	0.82	10.73	4.69
Northern White Cedar	*Thuja occidentalis*	310.21	3.12	41.73	7.31
Smooth Arrow Wood	*Viburnum dentatum*	294.25	3.02	23.35	4.26
Green Ash	*Fraxinus pennsylvanica*	2,432.02	17.26	428.18	67.70
Silver Maple	*Acer saccharinum*	547.97	7.60	103.03	10.02
Hazelnut (Beaked)	*Corylus cornuta*	73.65	0.43	8.73	3.47
Nannyberry	*Viburnum lentago*	587.48	3.46	92.61	13.72
Mountain Ash	*Sorbus americana*	6.16	1.33	0.00	0.00
Flowering Dogwood	*Cornus florida*	1,447.17	7.38	296.10	48.87
Bitternut Hickory	*Carya cordiformis*	1,481.67	20.39	145.86	34.65
Blue Ash	*Fraxinus quadrangulata*	32.12	0.28	4.90	1.16
Butternut	*Juglans cinerea*	12.47	0.15	4.44	1.23
Boxelder	*Acer negundo*	1,512.72	11.58	208.10	26.07
Wild Raisin	*Viburnum cassinoides*	80.66	2.18	10.82	3.29

Table 3 (cont'd)

Common Name	Scientific Name	Business-as-usual mean	Business-as-usual 95% CI	Much increased mean	Much increased 95% CI
Oval-leaved Privet	*Ligustrum ovalifolium*	7.21	0.05	2.92	1.23
Red Oak	*Quercus rubra*	9,308.65	54.78	953.53	121.04
Apple Rose	*Rosa villosa*	8.70	0.21	1.19	0.97
Dwarf Hackberry	*Celtis tenuifolia*	328.43	3.48	33.73	11.52
Paper Birch	*Betula papyrifera*	2,154.61	18.70	238.54	50.43
Buckeye	*Aesculus glabra*	30.35	0.72	0.96	1.29
American Highbush Cranberry	*Viburnum trilobum*	54.70	1.78	5.27	2.42
Pin Oak (Southern)	*Quercus palustris*	277.68	8.25	12.31	4.23
Japanese Barberry	*Berberis thunbergii*	1,914.15	11.59	435.25	45.28
Red Maple	*Acer rubrum*	40,528.97	81.86	7,243.78	514.79
Wormwood	*Artemisia pontica*	1.86	0.04	0.17	0.17
Honeylocust	*Gleditsia triacanthos*	71.52	2.39	13.36	4.57
Wild Black Currant	*Ribes americanum*	83.78	2.17	11.65	6.48
Black Currant	*Ribes nigrum*	6.86	0.45	1.18	0.85
Red Currant	*Ribes rubrum*	64.47	5.89	0.00	0.00
Weeping Willow	*Salix sepulcralis*	3.91	0.21	0.00	0.00
Southernwood	*Artemisia abrotanum*	28.01	1.40	10.77	2.20
European Highbush Cranberry	*Viburnum opulus*	2.29	0.01	1.95	0.21
Jetbead	*Rhodotypos scandens*	9.53	0.66	1.40	0.95
Slender Willow	*Salix petiolaris*	10.91	0.86	0.00	0.00
Sweet Cherry	*Prunus avium*	216.61	2.95	51.42	3.37
Beech	*Fagus grandifolia*	12,315.63	56.75	2,219.45	159.16
Michigan Holly	*Ilex verticillata*	672.94	6.61	192.40	25.62
Scotch Pine	*Pinus sylvestris*	233.31	1.90	50.20	6.92
Chinese Buckthorn	*Rhamnus utilis*	48.70	1.76	1.43	1.66
Trumpet Honeysuckle	*Lonicera sempervirens*	2.06	0.10	0.26	0.16
Red Honeysuckle	*Lonicera dioica*	21.02	0.07	17.27	1.64
Glossy Buckthorn	*Rhamnus frangula*	914.47	7.26	135.35	15.30
Downy Arrow Wood	*Viburnum rafinesquianum*	301.06	2.95	124.59	5.85
Common Buckthorn	*Rhamnus cathartica*	1,128.35	6.94	331.13	14.95

Table 3 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much increased	
		mean	95% CI	mean	95% CI
European Honeysuckle	*Lonicera caprifolium*	1.59	0.08	0.53	0.11
Elderberry	*Sambucus canadensis*	131.55	1.80	55.58	4.35
Wayfaring Tree	*Viburnum lantana*	28.38	0.95	4.73	2.95
White Spruce	*Picea glauca*	350.03	5.53	115.55	5.81
Black Spruce	*Picea mariana*	7.17	0.19	0.82	0.40
Hackberry	*Celtis occidentalis*	612.63	3.99	275.81	11.59
Hairy Honeysuckle	*Lonicera hirsuta*	7.14	0.17	4.11	0.41
Amur Honeysuckle	*Lonicera maackii*	884.44	2.19	548.73	18.37
Osage Orange	*Maclura pomifera*	19.71	0.23	13.16	0.96
Garlic Mustard	*Alliaria petiolata*	1,031.07	2.99	606.01	10.05
Morrow Honeysuckle	*Lonicera morrowii*	3,055.51	2.93	1,816.93	21.43
Cottonwood	*Populus deltoides*	42.85	0.97	10.24	0.94
Autumn Olive	*Elaeagnus umbellata*	11,705.36	7.91	8,087.06	63.72
Oriental Bittersweet	*Celastrus orbiculata*	1,601.39	2.62	1,090.86	22.01
Smooth Tartarian Honeysuckle	*Lonicera tatarica*	291.73	1.13	185.73	4.79
Blackgum	*Nyssa sylvatica*	340.95	1.68	194.65	5.83
Hybrid Honeysuckle	*Lonicera xbella*	29.88	0.14	24.77	0.46
American Bittersweet	*Celastrus scandens*	22.20	0.13	18.26	0.24
Black Cherry	*Prunus serotina*	13,731.24	9.10	10,395.11	44.71
Spotted Knapweed	*Centaurea stoebe*	22.66	0.11	16.51	0.29
European Fly Honeysuckle	*Lonicera xylosteum*	10.54	0.10	7.24	0.24
Balsam Poplar	*Populus balsamifera*	15.42	0.08	12.77	0.26
Red Osier Dogwood	*Cornus stolonifera*	40.50	0.10	34.50	0.31
Bigtooth Aspen	*Populus grandidentata*	1,095.62	0.15	1,067.40	2.41
American Chestnut	*Castanea dentata*	73.19	0.06	75.38	0.17
Quack Grass	*Elymus repens*	2.17	0.01	2.31	0.02
Reed Canary Grass	*Phalaris arundinacea*	19.36	0.13	24.18	0.22
Sand Dune Willow	*Salix cordata*	7.85	0.05	8.13	0.01
Buttonbush	*Cephalanthus occidentalis*	15.28	0.17	21.88	0.20
Poison Ivy	*Toxicodendron radicans*	410.20	1.33	528.40	3.55

Table 3 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much increased	
		mean	95% CI	mean	95% CI
American Crab Tree	*Malus coronaria*	104.29	0.45	111.57	0.52
Frost Grape	*Vitis vulpina*	0.05	0.00	0.08	0.00
Summer Grape	*Vitis aestivalis*	17.83	0.25	24.18	0.81
Swamp Dewberry	*Rubus hispidus*	0.31	0.01	0.50	0.04
Sweet Gale	*Myrica gale*	3.06	0.15	5.31	0.49
Thicket Creeper	*Parthenocissus inserta*	6.94	0.05	7.63	0.06
Narrow-leaved Cattail	*Typha angustifolia*	0.34	0.00	0.36	0.00
Quaking Aspen	*Populus tremuloides*	216.77	1.00	310.63	4.95
Bristly Blackberry	*Rubus setosus*	7.48	0.15	15.44	0.53
Riverbank Grape	*Vitis riparia*	50.20	1.16	107.26	2.62
Black Locust	*Robinia pseudoacacia*	235.93	1.82	418.51	6.09
Shrubby Cinquefoil	*Dasiphora fruticosa*	3.67	0.08	6.62	0.23
Mountain Holly	*Nemopanthus mucronatus*	0.60	0.01	0.65	0.00
Virginia Creeper	*Parthenocissus quinquefolia*	291.13	2.62	515.68	7.52
Common Lilac	*Syringa vulgaris*	1.89	0.03	2.27	0.05
Common Cattail	*Typha latifolia*	1.06	0.16	3.45	0.45
Yucca	*Yucca filamentosa*	0.89	0.04	1.54	0.13
Tree of Heaven	*Ailanthus altissima*	91.94	0.82	119.76	1.29
Phragmites (exotic)	*Phragmites australis*	15.28	0.33	18.38	0.17
Beach Heath	*Hudsonia tomentosa*	16.20	1.41	44.94	3.35
Chokeberry	*Aronia prunifolia*	40.79	0.81	74.83	3.83
Ninebark	*Physocarpus opulifolius*	2.22	0.44	13.53	1.37
Silky Dogwood	*Cornus amomum*	61.14	1.30	166.74	8.83
Broom Sedge	*Carex scoparia*	1.82	0.18	4.52	0.17
Timothy	*Phleum pratense*	9.30	0.12	17.05	0.46
Wahoo/Burning Bush (Invasive)	*Euonymus alatus*	0.04	0.08	3.79	0.74
Pussy Willow	*Salix discolor*	0.32	0.28	13.51	1.83
Winged Wahoo	*Euonymus alata*	110.73	1.09	218.34	11.48
Round Leaved Dogwood	*Cornus rugosa*	30.44	0.45	66.89	3.44
Spindle Tree	*Euonymus europaea*	7.84	0.12	9.92	0.23

Table 3 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much increased	
		mean	95% CI	mean	95% CI
Shrubby St. John's Wort	*Hypericum prolificum*	2.40	0.13	12.36	1.09
Running Strawberry Bush	*Euonymus obovata*	0.45	0.22	8.87	0.31
Bladdernut	*Staphylea trifolia*	14.72	0.73	39.05	1.46
Fly Honeysuckle	*Lonicera villosa*	12.05	2.10	112.31	2.17
Poison Sumac	*Toxicodendron vernix*	6.20	0.28	25.23	1.17
Labrador Tea	*Ledum groenlandicum*	1.29	0.71	20.41	2.00
Eastern Redbud	*Cercis canadensis*	3.85	0.03	6.52	0.86
Black Walnut	*Juglans nigra*	89.22	2.13	544.10	28.94
Prickly Ash	*Zanthoxylum americanum*	119.87	4.64	1,278.08	29.07
Smooth Highbush Blueberry	*Vaccinium corymbosum*	38.04	1.38	278.57	7.99
Pennsylvania Blackberry	*Rubus pensylvanicus*	0.54	0.77	10.27	0.08
Swamp White Oak	*Quercus bicolor*	49.58	3.35	489.89	16.90
Soapberry	*Shepherdia canadensis*	0.00	0.00	1.61	1.08
Bristly Locust	*Robinia hispida*	0.00	0.00	5.74	1.05
Orchard Grass	*Dactylis glomerata*	0.00	0.00	0.59	0.17
Buffalo Berry	*Shepherdia argentea*	0.00	0.00	0.47	0.22
Hop Tree	*Ptelea trifoliata*	0.00	0.00	72.17	9.73
Black Raspberry	*Rubus occidentalis*	24.30	3.82	1,471.51	31.34
Common Blackberry	*Rubus allegheniensis*	34.90	5.64	1,347.92	20.85
Wild Red Raspberry	*Rubus strigosus*	0.96	0.63	122.23	2.73
Bush Honeysuckle	*Diervilla lonicera*	0.00	0.00	0.32	0.06
Wintergreen	*Gaultheria procumbens*	0.07	0.09	44.34	0.53
Pin Cherry	*Prunus pensylvanica*	0.77	0.84	141.36	6.20
Japanese Bush Clover	*Lespedeza thunbergii*	0.00	0.00	8.09	1.94
Staghorn Sumac	*Rhus typhina*	6.23	1.54	158.03	7.53
Shellbark Hickory	*Carya laciniosa*	1.77	1.45	172.82	9.05
Fragrant Sumac	*Rhus aromatica*	1.58	0.43	7.03	0.29

Table 3 (cont'd)

Common Name	Scientific Name	Business-as-usual mean	Business-as-usual 95% CI	Much increased mean	Much increased 95% CI
Shingle Oak	*Quercus imbricaria*	0.00	0.00	2.43	0.50
Common Green Brier	*Smilax rotundifolia*	1.01	0.66	662.02	93.94
Shrubby Lespedeza	*Lespedeza bicolor*	0.00	0.00	30.04	8.78
Smooth Blackberry	*Rubus canadensis*	2.38	2.22	117.38	12.13
Choke Cherry	*Prunus virginiana*	56.52	7.53	1,285.27	48.35
Pignut Hickory	*Carya glabra*	180.80	9.59	4,704.61	188.86
Poverty Grass	*Danthonia spicata*	0.00	0.00	18.20	0.29
Hillside Blueberry	*Vaccinium pallidum*	0.00	0.00	5.79	0.87
Northern Pin Oak	*Quercus ellipsoidalis*	7.62	1.69	783.99	15.88
Tag Alder	*Alnus rugosa*	0.17	0.21	306.05	37.58
Common Lowbush Blueberry	*Vaccinium angustifolium*	18.16	2.41	361.63	17.76
Canada Blueberry	*Vaccinium myrtilloides*	0.00	0.00	25.65	9.47
Huckleberry	*Gaylussacia baccata*	27.92	5.64	950.65	51.81
Gray Dogwood	*Cornus foemina*	45.60	6.82	2,353.10	70.11
Smooth Sumac	*Rhus glabra*	0.10	0.06	52.52	6.01
Sweet Fern	*Comptonia peregrina*	0.00	0.00	16.57	1.92
White Oak	*Quercus alba*	144.61	21.40	13,682.55	444.92
Bracken Fern	*Pteridium aquilinum*	10.13	5.26	684.18	16.20
Shagbark Hickory	*Carya ovata*	40.01	6.78	2,166.92	50.05
Alleghany Plum	*Prunus alleghaniensis*	0.00	0.00	2.80	0.74
Meadowsweet	*Spiraea alba*	0.00	0.00	14.34	3.56
Winged Sumac	*Rhus copallina*	3.08	0.03	18.90	3.16
Sand Cherry	*Prunus pumila*	0.00	0.00	28.31	3.01
Northern Dewberry	*Rubus flagellaris*	0.93	0.85	88.30	5.72
Black Oak	*Quercus velutina*	55.02	10.10	7,579.81	161.98
Hazelnut (American)	*Corylus americana*	31.73	1.36	661.19	43.45
Bluejoint Reed Grass	*Calamagrostis canadensis*	0.46	0.51	13.11	1.05
Pasture Rose	*Rosa carolina*	5.17	4.66	107.03	8.25
Bur Oak	*Quercus macrocarpa*	2.85	2.48	314.42	23.29
Flowering Spurge	*Euphoria corollata*	0.00	0.00	11.89	2.39

Table 3 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much increased	
		mean	95% CI	mean	95% CI
Little Bluestem	*Schizachyrium scoparium*	1.30	1.09	75.72	4.92
Sassafras	*Sassafras albidum*	109.96	13.19	12,791.63	391.86
New Jersey Tea	*Ceanothus americanus*	1.63	1.46	18.72	1.46
Indian Grass	*Sorghastrum nutans*	0.00	0.00	15.49	0.60
Switchgrass	*Panicum virgatum*	4.20	2.89	100.67	1.48

Table 4. Area in hectares and 95% confidence intervals after 10 runs of each set of 140-year simulations using the wetter CNRM climate model. Simulations included two capacity scenarios: business as usual (no more than 200 stands per year and no more than 10 stands per day) and much increased (2,000 stands per year and 100 stands per day).

		Business-as-usual		Much Increased	
Common Name	Scientific Name	mean	95% CI	mean	95% CI
Black Maple	*Acer nigrum*	653.88	32.72	31.76	5.03
Sugar Maple	*Acer saccharum*	9,478.80	79.58	700.78	66.38
Norway Spruce	*Picea abies*	439.37	18.47	23.84	9.44
Alternate-leaved Dogwood	*Cornus alternifolia*	99.95	0.71	5.13	1.85
Maple Leaved Viburnum	*Viburnum acerifolium*	4,663.50	23.09	546.12	68.63
Ironwood	*Ostrya virginiana*	7,224.68	43.56	439.58	53.30
Spicebush	*Lindera benzoin*	2,334.89	8.19	294.63	52.57
Pawpaw	*Asimina triloba*	212.67	5.19	54.45	7.43
White Ash	*Fraxinus americana*	12,044.40	69.07	769.81	125.48
Eastern Red Cedar	*Juniperus virginiana*	2,065.88	18.76	148.42	42.36
Red Berried Elder	*Sambucus racemosa*	198.88	2.61	23.57	6.16
Mountain Maple	*Acer spicatum*	7.28	0.83	0.00	0.00
American Fly Honeysuckle	*Lonicera canadensis*	17.55	0.36	1.72	1.49
Big Bluestem	*Andropogon gerardii*	1.74	1.56	27.77	0.90
White Mulberry	*Morus alba*	242.39	3.61	57.93	24.41
Red Pine	*Pinus resinosa*	1,057.54	7.05	130.11	27.35
American Elm	*Ulmus americana*	10,657.78	59.62	742.30	131.84
Creeping Juniper	*Juniperus horizontalis*	67.58	4.26	0.00	0.00
Black Ash	*Fraxinus nigra*	368.66	5.66	36.15	12.81
Basswood	*Tilia americana*	2,596.16	38.50	220.59	31.75
Balsam Fir	*Abies balsamea*	43.96	1.41	0.27	0.27
Common Privet	*Ligustrum vulgare*	178.45	5.10	13.40	12.27
Red Mulberry	*Morus rubra*	20.53	0.40	0.81	1.02
Black Willow	*Salix nigra*	11.03	0.12	2.84	1.61
Black Haw	*Viburnum prunifolium*	62.54	0.42	0.54	1.00
Swamp Rose	*Rosa palustris*	24.61	0.83	2.81	3.23

Table 4 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much Increased	
		mean	95% CI	mean	95% CI
Rock Elm	*Ulmus thomasii*	12.44	1.25	0.00	0.00
Yellow Birch	*Betula alleghaniensis*	343.11	9.22	17.65	8.94
Catalpa	*Catalpa speciosa*	236.89	1.18	54.58	17.36
Northern Gooseberry	*Ribes oxyacanthoides*	6.53	0.06	0.79	0.79
Prickly Or Wild Gooseberry	*Ribes cynosbati*	2,718.21	29.55	261.02	34.18
Musclewood/Hornbeam	*Carpinus caroliniana*	6,330.84	55.12	529.45	64.92
Jack Pine	*Pinus banksiana*	333.49	4.35	41.29	10.11
Norway Maple	*Acer platanoides*	43.88	0.61	6.80	5.75
Hemlock	*Tsuga canadensis*	1,767.43	51.10	83.72	23.91
Canadian Yew	*Taxus canadensis*	4.17	0.02	2.23	0.93
White Pine	*Pinus strobus*	14,979.47	48.69	3,046.23	266.12
Common Or Ground Juniper	*Juniperus communis*	589.71	10.31	49.90	16.94
Witch Hazel	*Hamamelis virginiana*	16,597.83	65.28	1,666.46	153.66
Yellow Poplar (Tulip Tree)	*Liriodendron tulipifera*	252.16	3.40	31.08	5.09
Multiflora Rose	*Rosa multiflora*	10,845.03	37.20	1,042.62	70.96
Leatherwood	*Dirca palustris*	19.54	1.87	0.38	0.48
Chinkapin Oak	*Quercus muehlenbergii*	58.54	0.91	13.36	3.20
Northern White Cedar	*Thuja occidentalis*	305.62	4.85	25.68	6.79
Smooth Arrow Wood	*Viburnum dentatum*	296.91	2.89	20.73	3.69
Silver Maple	*Acer saccharinum*	543.65	8.51	58.00	16.84
Green Ash	*Fraxinus pennsylvanica*	2,429.75	13.93	266.83	82.15
Hazelnut (Beaked)	*Corylus cornuta*	72.94	0.45	16.66	4.68
Nannyberry	*Viburnum lentago*	582.78	3.66	59.03	15.86
Mountain Ash	*Sorbus americana*	7.15	0.90	0.00	0.00

Table 4 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much Increased	
		mean	95% CI	mean	95% CI
Flowering Dogwood	*Cornus florida*	1,446.75	6.29	239.47	29.40
Bitternut Hickory	*Carya cordiformis*	1,466.05	25.78	165.36	30.15
Blue Ash	*Fraxinus quadrangulata*	32.18	0.31	4.04	1.63
Butternut	*Juglans cinerea*	12.51	0.13	3.31	0.76
Boxelder	*Acer negundo*	1,496.83	17.04	180.14	24.62
Wild Raisin	*Viburnum cassinoides*	83.43	1.28	4.09	3.84
Oval-leaved Privet	*Ligustrum ovalifolium*	7.21	0.07	1.33	1.01
Red Oak	*Quercus rubra*	9,324.93	84.93	879.95	94.13
Apple Rose	*Rosa villosa*	7.70	0.67	0.78	0.86
Dwarf Hackberry	*Celtis tenuifolia*	321.17	4.29	31.90	7.22
Buckeye	*Aesculus glabra*	29.54	0.85	1.50	1.50
Paper Birch	*Betula papyrifera*	2,166.32	17.23	129.16	45.56
American Highbush Cranberry	*Viburnum trilobum*	53.58	1.77	3.61	2.59
Pin Oak (Southern)	*Quercus palustris*	284.51	12.22	16.16	3.41
Japanese Barberry	*Berberis thunbergii*	1,872.19	17.13	359.98	34.44
Red Maple	*Acer rubrum*	40,433.05	70.04	7,314.79	466.75
Wormwood	*Artemisia pontica*	1.84	0.04	0.37	0.17
Wild Black Currant	*Ribes americanum*	82.15	2.13	8.53	3.47
Honeylocust	*Gleditsia triacanthos*	71.28	2.50	6.47	2.22
Black Currant	*Ribes nigrum*	6.86	0.31	2.24	0.85
Red Currant	*Ribes rubrum*	73.23	2.87	0.00	0.00
Weeping Willow	*Salix sepulcralis*	3.84	0.29	0.18	0.33
Southernwood	*Artemisia abrotanum*	28.01	0.99	14.32	2.36
European Highbush Cranberry	*Viburnum opulus*	2.27	0.03	1.65	0.26
Jetbead	*Rhodotypos scandens*	9.37	0.48	0.37	0.69
Sweet Cherry	*Prunus avium*	213.96	4.74	55.49	4.61

Table 4 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much Increased	
		mean	95% CI	mean	95% CI
Slender Willow	*Salix petiolaris*	11.20	0.66	0.00	0.00
Beech	*Fagus grandifolia*	12,285.11	45.64	2,214.10	140.69
Michigan Holly	*Ilex verticillata*	671.72	6.37	135.20	24.76
Scotch Pine	*Pinus sylvestris*	230.54	3.59	43.02	4.51
Chinese Buckthorn	*Rhamnus utilis*	48.76	2.38	1.70	2.03
Trumpet Honeysuckle	*Lonicera sempervirens*	2.01	0.11	0.23	0.18
Red Honeysuckle	*Lonicera dioica*	21.05	0.01	16.76	2.23
Glossy Buckthorn	*Rhamnus frangula*	919.32	8.60	161.22	25.97
Common Buckthorn	*Rhamnus cathartica*	1,116.28	10.54	327.22	19.65
Downy Arrow Wood	*Viburnum rafinesquianum*	297.51	3.90	108.84	8.49
European Honeysuckle	*Lonicera caprifolium*	1.61	0.06	0.60	0.25
Elderberry	*Sambucus canadensis*	131.95	1.49	56.95	4.48
Wayfaring Tree	*Viburnum lantana*	29.13	0.77	5.63	1.80
White Spruce	*Picea glauca*	347.93	6.61	106.68	8.20
Black Spruce	*Picea mariana*	7.25	0.23	1.47	0.55
Hairy Honeysuckle	*Lonicera hirsuta*	6.81	0.22	3.12	0.57
Hackberry	*Celtis occidentalis*	610.20	3.74	290.11	12.35
Amur Honeysuckle	*Lonicera maackii*	880.47	2.77	513.20	16.16
Osage Orange	*Maclura pomifera*	19.56	0.31	12.20	0.91
Garlic Mustard	*Alliaria petiolata*	1,028.89	3.08	613.94	15.33
Morrow Honeysuckle	*Lonicera morrowii*	3,050.15	4.35	1,811.68	19.89
Cottonwood	*Populus deltoides*	43.06	0.86	9.75	1.07
Autumn Olive	*Elaeagnus umbellata*	11,685.08	10.35	8,037.73	41.27
Oriental Bittersweet	*Celastrus orbiculata*	1,597.45	2.96	1,081.33	18.79
Smooth Tartarian Honeysuckle	*Lonicera tatarica*	289.60	1.29	178.88	3.95

Table 4 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much Increased	
		mean	95% CI	mean	95% CI
Hybrid Honeysuckle	*Lonicera xbella*	29.70	0.18	24.18	0.62
Blackgum	*Nyssa sylvatica*	341.89	3.73	186.76	4.34
American Bittersweet	*Celastrus scandens*	22.15	0.07	18.23	0.29
Black Cherry	*Prunus serotina*	13,724.74	5.69	10,402.63	41.77
Spotted Knapweed	*Centaurea stoebe*	23.08	0.15	17.15	0.42
Balsam Poplar	*Populus balsamifera*	15.43	0.07	12.98	0.15
European Fly Honeysuckle	*Lonicera xylosteum*	10.34	0.11	6.92	0.33
Red Osier Dogwood	*Cornus stolonifera*	40.33	0.12	34.04	0.42
Bigtooth Aspen	*Populus grandidentata*	1,095.75	0.20	1,066.90	1.39
American Chestnut	*Castanea dentata*	73.19	0.05	75.42	0.13
Quack Grass	*Elymus repens*	2.18	0.01	2.32	0.02
Reed Canary Grass	*Phalaris arundinacea*	19.37	0.13	24.67	0.34
Sand Dune Willow	*Salix cordata*	7.85	0.04	8.14	0.00
Buttonbush	*Cephalanthus occidentalis*	15.34	0.12	21.84	0.45
Poison Ivy	*Toxicodendron radicans*	412.49	1.32	530.90	5.19
American Crab Tree	*Malus coronaria*	104.68	0.39	112.07	0.58
Frost Grape	*Vitis vulpina*	0.05	0.00	0.08	0.01
Swamp Dewberry	*Rubus hispidus*	0.29	0.01	0.54	0.04
Summer Grape	*Vitis aestivalis*	17.91	0.23	25.02	0.91
Sweet Gale	*Myrica gale*	3.06	0.16	5.42	0.68
Narrow-leaved Cattail	*Typha angustifolia*	0.34	0.00	0.36	0.00
Thicket Creeper	*Parthenocissus inserta*	6.94	0.05	7.68	0.09
Quaking Aspen	*Populus tremuloides*	216.28	1.28	327.95	7.37
Bristly Blackberry	*Rubus setosus*	7.40	0.17	14.94	0.41
Riverbank Grape	*Vitis riparia*	51.71	1.14	111.77	3.37
Black Locust	*Robinia pseudoacacia*	235.43	1.25	428.15	4.05

Table 4 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much Increased	
		mean	95% CI	mean	95% CI
Shrubby Cinquefoil	*Dasiphora fruticosa*	3.65	0.07	6.47	0.25
Mountain Holly	*Nemopanthus mucronatus*	0.60	0.01	0.65	0.00
Virginia Creeper	*Parthenocissus quinquefolia*	293.28	2.78	531.92	9.21
Common Lilac	*Syringa vulgaris*	1.89	0.04	2.33	0.05
Common Cattail	*Typha latifolia*	1.06	0.14	3.72	0.72
Yucca	*Yucca filamentosa*	0.90	0.04	1.72	0.10
Tree of Heaven	*Ailanthus altissima*	92.11	0.91	122.85	1.39
Phragmites (exotic)	*Phragmites australis*	15.49	0.38	18.68	0.17
Chokeberry	*Aronia prunifolia*	41.46	0.89	82.25	3.98
Ninebark	*Physocarpus opulifolius*	2.30	0.47	13.85	2.32
Beach Heath	*Hudsonia tomentosa*	15.91	1.11	41.91	2.63
Silky Dogwood	*Cornus amomum*	61.71	1.72	173.82	7.42
Broom Sedge	*Carex scoparia*	1.68	0.09	4.26	0.41
Timothy	*Phleum pratense*	9.35	0.15	17.40	0.59
Wahoo/Burning Bush	*Euonymus alatus*	0.00	0.00	4.04	1.19
Pussy Willow	*Salix discolor*	0.20	0.15	13.86	1.75
Winged Wahoo	*Euonymus alata*	112.42	1.86	237.62	11.40
Round Leaved Dogwood	*Cornus rugosa*	30.60	0.55	71.70	2.86
Spindle Tree	*Euonymus europaea*	8.02	0.20	10.08	0.11
Shrubby St. John's Wort	*Hypericum prolificum*	2.30	0.07	13.73	1.28
Running Strawberry Bush	*Euonymus obovata*	0.66	0.31	9.15	0.23
Bladdernut	*Staphylea trifolia*	14.44	0.71	39.06	1.45
Fly Honeysuckle	*Lonicera villosa*	14.17	2.15	115.77	1.75
Poison Sumac	*Toxicodendron vernix*	6.36	0.52	24.95	0.76

Table 4 (cont'd)

Common Name	Scientific Name	Business-as-usual mean	Business-as-usual 95% CI	Much Increased mean	Much Increased 95% CI
Labrador Tea	*Ledum groenlandicum*	2.40	0.95	18.96	2.35
Eastern Redbud	*Cercis canadensis*	3.74	0.19	6.58	0.57
Black Walnut	*Juglans nigra*	90.03	0.72	544.08	19.30
Prickly Ash	*Zanthoxylum americanum*	124.05	6.78	1,340.99	30.65
Smooth Highbush Blueberry	*Vaccinium corymbosum*	37.25	1.21	297.01	14.92
Pennsylvania Blackberry	*Rubus pensylvanicus*	0.29	0.54	10.21	0.17
Swamp White Oak	*Quercus bicolor*	46.66	4.86	540.27	18.00
Soapberry	*Shepherdia canadensis*	0.00	0.00	2.37	0.63
Bristly Locust	*Robinia hispida*	0.06	0.11	6.18	0.61
Orchard Grass	*Dactylis glomerata*	0.00	0.00	0.71	0.13
Buffalo Berry	*Shepherdia argentea*	0.00	0.00	0.77	0.31
Hop Tree	*Ptelea trifoliata*	0.05	0.10	67.40	9.27
Black Raspberry	*Rubus occidentalis*	23.72	2.33	1,482.59	29.56
Common Blackberry	*Rubus allegheniensis*	36.74	6.54	1,338.74	28.19
Wild Red Raspberry	*Rubus strigosus*	1.33	0.69	110.49	5.99
Pin Cherry	*Prunus pensylvanica*	1.43	1.17	150.37	8.75
Bush Honeysuckle	*Diervilla lonicera*	0.00	0.00	0.29	0.07
Wintergreen	*Gaultheria procumbens*	0.25	0.46	44.40	0.48
Japanese Bush Clover	*Lespedeza thunbergii*	0.00	0.00	8.64	2.25
Staghorn Sumac	*Rhus typhina*	8.05	1.96	154.11	7.17
Shellbark Hickory	*Carya laciniosa*	0.31	0.53	158.29	13.56
Fragrant Sumac	*Rhus aromatica*	1.99	0.58	7.29	0.22
Shingle Oak	*Quercus imbricaria*	0.00	0.00	2.55	0.63
Common Green Brier	*Smilax rotundifolia*	1.54	1.10	507.63	60.92

Table 4 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much Increased	
		mean	95% CI	mean	95% CI
Shrubby Lespedeza	*Lespedeza bicolor*	0.00	0.00	18.43	10.17
Smooth Blackberry	*Rubus canadensis*	0.00	0.00	107.19	13.64
Choke Cherry	*Prunus virginiana*	51.27	3.96	1,464.60	81.48
Pignut Hickory	*Carya glabra*	189.81	19.64	4,663.84	116.52
Poverty Grass	*Danthonia spicata*	0.00	0.00	16.77	2.37
Hillside Blueberry	*Vaccinium pallidum*	0.02	0.03	5.02	1.01
Tag Alder	*Alnus rugosa*	0.23	0.29	373.07	38.62
Northern Pin Oak	*Quercus ellipsoidalis*	1.76	1.06	764.93	40.39
Common Lowbush Blueberry	*Vaccinium angustifolium*	8.18	3.77	338.61	23.55
Huckleberry	*Gaylussacia baccata*	19.95	3.55	827.78	38.63
Canada Blueberry	*Vaccinium myrtilloides*	0.00	0.00	41.13	8.98
Gray Dogwood	*Cornus foemina*	40.57	6.10	2,567.62	91.00
Smooth Sumac	*Rhus glabra*	0.11	0.08	54.50	5.01
Sweet Fern	*Comptonia peregrina*	0.22	0.33	16.01	2.50
White Oak	*Quercus alba*	112.50	17.52	12,883.73	347.78
Bracken Fern	*Pteridium aquilinum*	8.88	5.39	677.76	6.64
Shagbark Hickory	*Carya ovata*	36.66	7.38	2,280.09	49.58
Alleghany Plum	*Prunus alleghaniensis*	0.00	0.00	3.01	0.48
Meadowsweet	*Spiraea alba*	0.00	0.00	22.14	1.67
Winged Sumac	*Rhus copallina*	0.94	0.89	17.50	4.14
Sand Cherry	*Prunus pumila*	0.00	0.00	26.44	4.02
Northern Dewberry	*Rubus flagellaris*	1.21	1.09	89.91	6.80
Hazelnut (American)	*Corylus americana*	7.95	3.72	708.12	26.00
Black Oak	*Quercus velutina*	56.72	10.66	7,160.10	198.68
Bluejoint Reed Grass	*Calamagrostis canadensis*	0.00	0.00	14.42	0.28
Pasture Rose	*Rosa carolina*	6.59	6.74	95.61	5.11

Table 4 (cont'd)

Common Name	Scientific Name	Business-as-usual		Much Increased	
		mean	95% CI	mean	95% CI
Flowering Spurge	*Euphoria corollata*	0.00	0.00	10.75	3.22
Bur Oak	*Quercus macrocarpa*	3.53	3.64	352.35	19.15
Little Bluestem	*Schizachyrium scoparium*	0.34	0.42	66.06	6.49
Sassafras	*Sassafras albidum*	74.69	13.48	11,793.26	358.47
New Jersey Tea	*Ceanothus americanus*	0.66	0.81	18.74	1.75
Indian Grass	*Sorghastrum nutans*	0.80	1.50	16.02	0.42
Switchgrass	*Panicum virgatum*	3.51	2.89	98.32	3.32

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Chapter 3: Wildness is self-organization: Complexity science and the diversity, resilience, and adaptive capacity of social-ecological systems

> *"…and what I have been preparing to say is, that in Wildness is the preservation of the World. Every tree sends its fibers forth in search of the Wild. The cities import it at any price. Men plow and sail for it. From the forest and wilderness come the tonics and barks which brace mankind."* (Thoreau 1862)

ABSTRACT

Wildness is an implicit value of wildlife and wild lands management. In an era of pervasive human influence, the contemporary conservation of wild organisms requires an interdisciplinary scientific definition of wildness. Wildness can be defined in several ways. Here, the definition of wildness as self-willed organisms or ecological communities is extended to include any self-organizing complex adaptive system, and thus including certain social-ecological systems. The dilemma of using prescribed fire to manage fire-dependent oak ecosystems is a useful case study in wildness. Oak ecosystems in eastern North America exist only because they were intensively managed for millennia, and lack of recent management is resulting in transformation, and potentially simplification, of a complex system. How can intensive management be reintroduced in a way that leads to greater complexity, resilience, and adaptive capacity? I apply insights from complexity science and the phenomenon of self-organizing systems to self-willed ecosystems. I extend the approach to social-ecological systems, and I use oak habitat management on state game areas in southern Michigan to

illustrate how these complex systems might be managed in ways that increase or decrease their self-organization and thus their wildness.

Wildness of Oak Social-Ecological Systems

If managers do not explicitly manage for keeping wildlife and wild places wild, do they risk domestication of wildlife and wild places that they try to manage (Wuerthner et al. 2014)? Is wildness now over (Wapner 2020)? Do managers simply need to come to grips with gardening a post-wild world (Marris 2011)? Is wildness just a concern for impractical idealists, nostalgic for a pristine nature that may, or may not have ever have existed? Wildness is an implicit value of all wildlife and wild lands management; wildness is what has distinguished wildlife management from management of life in other contexts, such as medicine or agriculture. Wildness is assumed, but in an era of pervasive human influence, the contemporary conservation of wild organisms requires a clear and explicit definition of wildness that is social-ecological. In this synthesis, I focus on wildness as a system property of autonomy or self-will, which can be described and measured, rather than absence of human influence, which is more subjective and value laden. Much scholarship exists on the value of wildness (Nash 1982, Turner 1996, Cole et al. 2010, Caro et al. 2014, Wapner 2020); this chapter extends that previous work by focusing on wildness as a property of complex adaptive systems. Describing wildness in scientific language complements other approaches and provides wildlife managers with new approaches and tools to manage wildlife and wild places in the context of social-ecological systems.

Concerns about human domination of the natural world are not new, though each generation of conservationists seems to rediscover the magnitude of human influence on the biosphere (Meine 2014). Nevertheless, humans passed key planetary boundaries in recent decades (Liu et al. 2015, Steffen et al. 2015); most bird and mammal biomass on earth is now domestic animals (Bar-On et al. 2018), and more than 70% of the global, ice-free land surface of the planet is directly affected by anthropogenic land use (IPCC 2019). Conceptually, the problem is old, but the scale is such that restoration of wildlife and wild places to goals and metrics grounded in past conditions and climates is increasingly expensive and sometimes impossible (Aplet and Cole 2010, Schuurman et al. 2020, Thompson et al. 2021). Any definition of wildness needs to include the idea of restoration and preservation but be general enough that it can be applied to systems that are constantly changing or new.

I focus on oak ecosystems as a case study because these ecosystems represent a contradiction inherent in wildlife and wildland management more broadly. To the degree that more management of a wild thing makes it less wild (Turner 1996), oak ecosystems in eastern North America could not have existed without frequent and intentional management (Nowacki and Abrams 2008, Abrams and Nowacki 2019). That this management was conducted by non-European nations makes it no less anthropogenic (Stewart 2002). Recent management of oak systems is more "hands-off"; these oak systems have been allowed to grow from abandoned agricultural land for decades, with little management (Brose et al. 2014), but despite this lack of management, these ecosystems are transforming into non-oak dominated mesic hardwoods (Nowacki and Abrams 2008, Abrams and Nowacki 2015). Oak ecosystems represent several distinctions that are important to contemporary conservation. If we consider wildness as about self-will more than a myth of a pristine past, then wildness is less about the absence of people from ecosystems and more about how social systems and ecosystems interact.

There are many ways to define ecosystem health (Döring et al. 2014, Harrison et al. 2019), but for the purposes of this essay I adopt one used by Aldo Leopold, "Health is the capacity of the land for self-renewal. Conservation is our effort to understand and preserve this capacity." (Leopold 1949) Typical wildlife and wildland conservation often involves management, which is imposing human will on the land, which runs contrary to allowing ecosystems to self-renew. This contradiction is partly about how and where one draws the boundaries of the system: there are social system and ecological systems. However, both sides of the New Conservation debate insist that humans be considered as part of the system (Marris 2011, Meine 2014). If neither system can be fully understood in isolation, then boundaries of the model need to be expanded to include a social-ecological system. A social-ecological system is distinct from two linked systems because the behavior of the larger system cannot be reduced to functioning of either system by itself (Norberg and Cumming 2014). This definition of ecosystem health as capacity for self-renewal has already been extended to social-ecological systems (Berkes et al. 2012), but not yet linked explicitly to wildness.

Oak social-ecological systems in Michigan, as in eastern North America, are changing; they are not self-renewing (Nowacki and Abrams 2008, Abrams and Nowacki 2015). The social system is disengaged from the ecological system, such that oaks and rare species associated with oak ecosystems are not regenerating and people are rarely interacting with ecosystems through disturbance regimes like fire (Knoot et al. 2009, 2015). Some prescribed fires occur, but the frequency is such that the managed system does not allow feedbacks that result in self-organizing disturbances. The current social-ecological system not only excludes fire, but several primarily social-based feedbacks reinforce both fire exclusion and fire suppression (Chapter 1, Chapter 2). In an era in which human influence on the global scale is becoming pervasive (Corlett 2015, Folke et al. 2016), and ecosystems are needing to adapt to pervasive changes in drivers like climate (Staudinger et al. 2013, Williams et al. 2019) and mesophication (Frelich 2017, Hanberry et al. 2020), the question about what type of wildlife or wildland management confers or degrades capacity for self-renewal is increasingly urgent (Levin 1999, Zavaleta and Chapin 2010, Cumming et al. 2017).

My objectives in this paper are to synthesize insights from complex adaptive systems and the concept of wildness in an era of human domination, to integrate these ideas with recent frameworks on managing wildness in ecosystems subject to transformation driven by climate change, and to explore how management of oak systems for wildlife habitat might be different in the context of creating an environment where complex adaptive systems could increase in self-organization and wildness. This information can be used to cultivate greater adaptive capacity of social-ecological systems in an era of increasingly disruptive change. This approach has the potential to foster increasing diversity and complexity, both biologically and culturally.

Complex Adaptive Systems are Wild

Complexity is an interdisciplinary science that developed in the 1980s when scientists in diverse fields including physics, economics, and biology realized that they were struggling with a similar problem: patterns that emerged in systems that could not be described adequately via reducing the system to its component parts (Waldorp 1992, Mitchell 2009, Holland 2014). As a conceptual framework, complexity is only beginning to be applied to forest ecosystems (Puettmann et al. 2009, Messier et al. 2015) and wildlife management (Salomon et al. 2019, Ibarra et al. 2020). Complexity can inform generating and sustaining diversity and heterogeneity, while also affecting resilience and adaptive capacity (Levin 1999, Levin et al. 2013). By focusing on conditions that allow emergence and self-organization in complex adaptive systems (Alexander 2011), these frameworks can be extended to include wildness explicitly. As such, it has potential to be useful for guiding and assessing management of wild organisms, places, ecosystems, and social-ecological systems. This application to wildness, while mentioned by Turner (1996), has not been developed in a natural resources management context.

A challenge of interdisciplinary work is translating terms and concepts from one discipline to another. Thus, there are many ways to describe a complex system (Miller and Page 2009, Mitchell 2009, Norberg and Cumming 2014). For the purposes of drawing parallels with wild social-ecological systems, I define a complex adaptive system as an **emergent, self-organizing, non-equilibrium system with moderately fluid memory, such that information accumulates over time** (Mitchell 2009, Holland 2014, Hidalgo 2015)**.** There are several terms and concepts used above that are not part of standard training for ecologists, and I will first walk through what each means, and then apply them to the problem of managing oaks toward increased wildness.

Emergent Self-organizing Systems

One of the central tenets of complexity theory is that simple rules followed by large numbers of entities or agents are sufficient to describe surprisingly complex patterns and phenomena that otherwise appear designed (Sole and Goodwin 2001, Holland 2014). In complex systems, these patterns or phenomena emerge from feedbacks and heterogeneities within the system itself. Complex adaptive systems maintain their patterns or phenomena even when disturbed and change in ways that adapt to those disturbances (Holling 2001, Gunderson 2009). The adaptive capacity of these systems is sometimes referred to as resilience (Allen et al. 2011, Messier et al. 2015), which Leopold (1949) might have called it the capacity for self-renewal (Berkes et al. 2012).

Ant colonies are a common example of complex adaptive systems (Holland 1998, Sole and Goodwin 2001, Mitchell 2009, Hidalgo 2015). Contrary to popular belief, the queen ant does not exercise authority by directing the behavior of ants in an ant colony. Instead, each ant obeys simple rules based on a set of pheromones and encounters with other ants. These simple rules can result in complex structures with dedicated areas for brood rearing, food storage, waste, and even dedicated areas to dispose of dead ants. Complex systems are often associated with natural or social systems (Miller and Page 2009, Holland 2014), and thus do not emerge unless the conditions necessary to foster complex systems (e.g., feedbacks, memory, freedom to self-organize, etc.) are present.

The properties of self-organization and adaptive capacity of a system cannot be manipulated directly from outside the system. If they were, then the system would no longer be self-organized; it would be organized by another. A self-organized system has a basic form of self-will. I use "will," not to indicate consciousness, but rather in the sense that a self-organizing system wants to organize in a way that emerges from its own self (Alexander 2011). Thus, a complex adaptive system is self-willed, is wild, to the degree that it is self-organizing.

The focus of management to increase wildness then is not on influencing wildness directly so much as in creating or maintaining the conditions that allow self-organization to emerge. Wildness of pristine past ecosystems is a special case of a broader class of emergent phenomena in complex adaptive systems, and the concept can be extended from ecological systems to social-ecological systems. A social-ecological system has the capacity to adapt and self-organize to the degree that it retains both its wildness and its capacity for self-renewal (Leopold 1949, Allen et al. 2011). Thus, we can paraphrase Leopold (1949) and say that conservation is our effort to understand and preserve (or restore or create) conditions that make complex adaptive systems complex and adaptive.

Open Non-equilibrium Systems

One of the conditions necessary for self-organization and thus self-renewal is that complex systems are far from energy equilibrium. Put another way, they are open systems in which energy is not bounded within the system. Complex adaptive systems are inherently dynamic. Systems that are closer to an energy equilibrium tend to be less complex (Nicolis and Prigogine 1989). The complexity of cities (Bettencourt 2013) and organisms (West and Brown 2005) can be described in part by their metabolism, the throughput of energy through the system (West 2017). At global geographic scales, cultural and biological diversity can be described by greater solar flux at lower latitudes (Hamilton et al. 2020).

In adaptive systems, one must consider the energy flux to which the system has become adapted. Many ecosystems, including oak ecosystems, require frequent disturbance that allows energy flux near and below the soil surface. Other ecosystems, such as tropical rainforests, are adapted to energy flux in the tree canopy and away from the soil. Nevertheless, sudden changes to the patterns of energy flux (e.g., via herbivory, microbial decomposition, or frequent fire) can have disproportionate effects on the conditions that allow or discourage self-organization and self-renewal.

Moderately Ephemeral Memory

Complex systems are adaptive when they have memory, although memory may take forms different than models encoded into the neurons of human brains by external stimuli. Genes provide memory for species as they adapt to changes in their environment via natural selection. Pheromone trails left by foraging ants are another example. As more ants successfully forage along a trail, more pheromones are deposited and the trail becomes a stronger stimulus, giving the colony a spatial memory of a food source. When the food source is exhausted, fewer ants return along the same path, and the memory fades (Sole and Goodwin 2001, Holland 2014).

The ephemeral tendency of memory is critical to adaptation (Hidalgo 2015). If signals cannot be forgotten, then the system will not adapt; it becomes unchanging. If signals are too ephemeral, the system will “forget” before it can change in an adaptive way. Rates at which memories persist or fade must be congruent with the rate of change in the environment. A food source that moves faster than the rate at which ant pheromones build up will not generate memory for an ant colony. Similarly, changes in the environment of a species that are faster than generation time of the species will require adaptation in behavior or physiology. Some change is too fast for an evolved response. Thus, there is a range of scales of change over which complex systems can adapt, and that scale is tied to nature of their memory, specifically how well they remember and how quickly they forget. In complex social-ecological systems, the rates at which ecological systems are disturbed should match evolutionary and ecological rates at which the system operates. Similarly, social systems of managers working with ecological systems need to craft monitoring systems that both learn and forget, and that are not tied exclusively to budget cycles or the time managers spend in job positions. In other words, institutional memory should be congruent in scale and ephemerality with ecological memory, not the mental memory of specific human actors in the social-ecological system.

Heterogeneity as Information Accumulation

Complex adaptive systems require heterogeneous agents and heterogeneous stimuli. If a complex system is comprised of homogenous agents, then there is no variation on which selection can act. The term “agent” in complex adaptive systems refers to any entity following a rule set. Agents need not be conscious entities making complicated choices (Miller and Page 2009, Holland 2014). An organismal cell reacting to a chemical gradient could be an agent. If all cells were the same, or there were no chemical gradients, the system would not behave in complex adaptive ways.

In ecological systems, the need to preserve biodiversity, which is heterogeneity at the level of species agents, is well appreciated (Folke et al. 2004, Dirzo et al. 2014, Díaz et al. 2019). However, complexity science would suggest that heterogeneity at other levels of organization is also important, both in terms of agents and in terms of disturbance regimes. A landscape with a variety of fire return intervals (i.e., pyrodiversity) will support some prairies and some rarely burned mesic forests, along with a range of woodlands and savannas (Packard and Mutel 1997, Kelly et al. 2015). However, much work remains to research the relationship between disturbance variability as it relates to system memory.

Heterogeneity and Pattern

Heterogeneity in complex systems is closely related to several other concepts that are important to wildlife and wild systems: diversity, uncertainty, legibility, and control (Holling and Meffe 1996, Puettmann et al. 2009). If community structure (i.e., species present and their relative abundances) is a pattern that emerges from disturbance regimes and ecological processes, then biodiversity metrics are a subset of measures of heterogeneity and thus one way to measure the complexity of an ecosystem.

Heterogeneity and diversity are measures of uncertainty. If we randomly pull an organism from a homogenous ecosystem, a system dominated by one species, then there is little uncertainty as to the species of the organism. If the system is highly diverse, then there is greater uncertainty as to the species identity of a randomly selected agent (i.e., individual organism). Similarly, if the return interval of a given disturbance is uncertain, then the disturbance regime is going to be more heterogeneous in time. The same holds for the size or shape of a disturbance; the more heterogeneous, the more uncertain the disturbance regime at any given location. Uncertainty is also an attribute commonly ascribed to wildness.

Uncertainty can take many forms between entirely random and entirely ordered, and complex systems tend to create particular forms of uncertainty (Mitchell 2009). Because complex adaptive systems arise in iterative systems, in which rules are repeated over heterogeneous environments and sets of agents, two distinctive types of patterns are produced depending on the discrete or continuous nature of system processes. Sometimes iteration results in power law statistical distributions; sometimes it results in fractal geometric shapes. In both patterns, there is no typical scale for a pattern, which repeats more or less continuously across a range of scales (Mandelbrot 1983, Falconer 2013, West 2017). In other complex systems, the physics of signals across boundaries leads to hierarchies or levels of organization, such as cells, tissues, and organisms (Holling 2001, Allen et al. 2014). Ecological communities are something between these patterns of complexity: they are patterned by biological agents with distinct scales and by disturbances that are often scale-free or fractal across a range of spatial and temporal scales.

The Complex Whole

A complex adaptive system is an **emergent, self-organizing, non-equilibrium system with moderately fluid memory, such that information accumulates over time** (Mitchell 2009, Holland 2014, Hidalgo 2015)**.** Describing social-ecological systems as complex adaptive systems usually describes the complexity of relationships in such systems, and particularly to identify useful points of intervention that are non-obvious without a formal, scientific description of the system (Liu et al. 2007*a*, Ostrom 2009). My intentions are slightly different, and thus the properties that I emphasize in the Mitchell (2009), Holland (2014), and Hidalgo (2015) definition are slightly different: what does it mean to be self-organizing, and what attributes of complex adaptive self-organization are relevant to intervening in systems while preserving or enhancing their capacity to self-organize.

Case Study: Wildness, Transformation, and Oak Mesophication

Increasing wildness, and thus self-renewal capacity of social-ecological systems, can be illustrated in oak ecosystem management on public lands managed specifically for wildness: wildlife management areas. Oak ecosystems provide valuable habitat for wild turkey (*Meleagris gallopavo*), white-tailed deer (*Odocoileus virginianus*), and small game species such as squirrels

(*Sciurus* spp.) and ruffed grouse (*Bonasa umbellus*). Oak ecosystems are also habitat for a disproportionate number of rare plant and animal species. Oak dominated forests (i.e., dry-mesic southern forests communities) are habitat for 14 state threatened and endangered species; oak barrens contain 40 state threatened and endangered species and 7 species that have been extirpated from the state (Cohen et al. 2020). Oak ecosystems provide habitats to two federally listed species: the endangered Karner blue butterfly (*Lycaeides melissa samuelis*) and threatened eastern Massasauga rattlesnake (*Sistrurus catenatus*).

Oak trees are failing to regenerate in oak forests because disturbance frequency, extent, timing, and intensity are changing. This change in disturbance is often framed as a change in fire frequency (Nowacki and Abrams 2008, McLauchlan et al. 2020), but other disturbances are also changing (Arthur et al. 2012), including drought frequency (Crausbay et al. 2017), herbivory (Royo et al. 2010, Redick and Jacobs 2020), and invasive species competition (MacDougall and Turkington 2005). The ecological interactions in such systems are also different than conditions a century or more ago in which these oak forests originated (Abrams and Nowacki 2019), and these drivers are all undergoing changes. Oaks are failing to regenerate and there is concern that habitat for a broad array of game and rare wildlife species is being lost (McShea et al. 2007, Lee and Kost 2008).

Oaks are adapted to frequent disturbance, and the same can be said for the suite of prairie and savanna species associated with oaks (Packard and Mutel 1997, Johnson et al. 2009). Typically, these species are adapted to full or part sun conditions with less than 100% canopy coverage. Whereas forest tree species grow taller to gain a competitive advantage and access to more sunlight, trees more common in oak ecosystems divide their energy stores between above and below ground to hedge against above ground disturbances like fire or herbivory. In the absence of intermittent disturbances that create openings in the forest canopy, oak dominated ecosystems do not thrive. Put in the context of complex systems and resilience (Berkes et al. 2012), they lose the ability to self-renew. In the absence of disturbance, oak ecosystems transform into a different and novel mix of species, either through conversion to red maple (*Acer rubrum*) dominated forest or invasive species thickets that do not reflect the goals of wildlife biologists managing the land.

With this context, we can see that wildness cannot mean leaving oak ecosystems alone because oak ecosystems cannot persist as complex and biodiverse ecosystems apart from interactions with social systems that perpetuate a disturbance regime. The self that is wild in this context is a social-ecological system, not just the ecological system. The self-renewal of this system requires that the social and ecological systems be brought back into conditions that support self-organization. The scale, frequency and variability of anthropogenic disturbance have changed; this is simplifying the system. Much of the management of these systems is currently done with the idea of regenerating the existing closed canopy oak forest with essentially the same (or a very similar mix) species, dominated by oaks, to optimize food sources for white-tailed deer (*Odocoileus virginianus*), wild turkey (*Meleagris gallopavo*), and squirrels.

The social and ecological contexts for this social-ecological system have changed such that large landscape scale fires are no longer safe. Even if they were, fuels in the contemporary landscape of road networks and human settlements are less contiguous than they were two or three centuries ago. Laws and cultural norms have changed. Passenger pigeons (*Ectopistes* *migratorius*) are gone. New invasive species complicate land management. One thing that has not changed significantly from the perspective of oaks is climate suitability; this system is located at the northern end of the central hardwoods region, and climate will only become more supportive of the most common species in the system (black oak *Quercus velutina*, white oak *Quercus alba*, and red maple). Specific species, such as the Karner blue butterfly, are threatened by climate change (Hoving et al. 2013, Thurman et al. 2020), but the most common species in the system show relatively little sensitivity to climate change in this study area (Iverson et al. 2019).

Climate change has motivated scientists and practitioners to develop frameworks to assist managers grappling with ecological transformation, and these frameworks can be useful to understand transformation caused by other anthropogenic drivers, such as fire exclusion and suppression (Abrams and Nowacki 2015, Hanberry et al. 2020) or loss of disturbance from passenger pigeon feeding and roosting (Ellsworth and McComb 2003, Buchanan and Hart 2012). Fire and passenger pigeon disturbances have not and will not occur in the foreseeable future at the extent, intensity, or frequency that created the extensive mature oak forests that are extant today. Although specific stands or areas can be managed intensively to regenerate the current canopy, it is unlikely that those resources will be available at the extent necessary to influence the trajectory of most of the study area. Are transforming ecosystems simply low-quality areas neglected as managers focus on other sites? Or can they be managed such that systems grow in complexity, capacity for self-renewal, and wildness?

The Resist-Accept-Direct (RAD) framework was developed to guide management as global change moves ecosystems away from historical baselines and toward ecological transformation (Aplet and Cole 2010, Schuurman et al. 2020, Thompson et al. 2021). Under this framework land managers can choose from among three broad categories of actions. Managers can **resist** ecological change, either by intervening to restore ecosystems or to protect existing ecosystem services. Managers can **accept** that change is happening and choose not to intervene. Or managers can intervene in the system to **direct** it toward a desirable state (Figure 22).

From a manager's perspective, the goal, or desired future condition, of land management has been, by default, to resist change wherever possible. This is a worthy goal when change is accompanied by steep losses in biodiversity, or when change threatens other valuable ecosystem services. However, in the RAD framework, resist-based approaches will become more expensive and less predictable as system drivers (e.g., climate change, fire exclusion, etc) continue to push ecosystems further from historical conditions. Managers already focus resist-based efforts on the highest quality, most biodiverse parts of the landscape. That work is important, but the landscape is transforming. The canopy may be dominated by oaks (Figure 17), but the subcanopy is dominated by red maple, black cherry, and autumn olive (*Elaeagnus umbellata*). Managers need guidance relevant to a transforming landscape where Resist strategies are no longer viable, where the system is trying to self-organize into something new. Therefore, I want to elaborate on how managers might foster conditions that support self-organization and thus wildness and complexity in these increasingly novel systems (Schuurman et al. 2020, Thompson et al. 2021) with regard to Accept and Direct.

A great deal of research exists to guide managers regenerating oak forests via oak focused silviculture (Johnson et al. 2009, Knopp and Stout 2014) or restoration of oak ecological communities (Packard and Mutel 1997, Knoot et al. 2010*b*, Frelich 2017). Less research exists on integrating Accept and Direct approaches, and creating the conditions for complexity, wildness, and self-renewal in the context of ecological or social-ecological transformation. I will focus on Accept and Direct change approaches as they could be applied in the context of managing transforming oak habitats on state game areas in southern Michigan. These approaches exist along continua in two orthogonal directions (Figure 23). I will attempt to sketch out management at the extreme corners. As such, these use cases are illustrative. They are not descriptions of plans, prescriptions, or recommendations.

To accept change in a system that is transforming is to treat the system as ecological, and to exclude, as much as possible, any management toward a specific goal. Prescribed fire would not occur, and lightning or natural origin fire in this region is rare enough that the system would likely continue to tend, slowly, toward mesic hardwoods as the existing oak canopy died and existing subcanopy trees advanced to the canopy. Invasive species control would also not occur under this scenario. Shade tolerant invasive species would likely persist, and invasive trees would likely comprise a part of the canopy.

This approach is similar, and yet distinct, from the status quo. No state lands are currently designated with this type of protection. Prescribed fire and invasive species management is allowed in state designated wildernesses and natural areas, and thus, these activities could occur on any oak forest stand on state lands. In an area truly managed as Accept Change, prescribed fire, invasive species control, and any intentional management would be forbidden. These areas would be reference areas or experimental controls against which other approaches could be compared. They would also be places where new species interactions would be allowed to coevolve at time scales longer than conservation fads or a given manager's vision. Often the most restrictive protections are applied to areas with the highest native biodiversity; in this approach restrictive protections would be applied to the opposite because areas with the highest novel and invasive species biodiversity would have the most coevolutionary potential.

Accepting change appears at first to be passive, to be the option with the least resistance. However, the ability for a landowner, organization, or agency to restrain themselves from ever intervening is difficult in practice. Even in remote wilderness areas, such as Isle Royale National Park, managers have a difficult time resisting the temptation to engage in climate adaptation to restore keystone species, such as wolves (*Canis lupus*; Fisichelli et al. 2013).

Accept Change toward a Complex System: Disturb

Oak ecosystems are complex because they are subjected to irregular yet frequent disturbance. A disturbance regime that varies in time and space is key to the complexity of the system. Pyrodiveristy begets complexity, which can be measured in terms of biodiversity. Under this scenario, humans generate disturbances, but without adapting the disturbance regime to any set of goals. Those disturbances would likely be fire, or they might be grazing or mowing or something else entirely. Because a complex disturbance regime would occur over a wide range of scales, the disturbance regime should be one that can be done relatively cheaply over large extents. Prescribed fire and commercial forest harvest might meet this criterion, work by tractor or by hand may not.

Prescribed fire without a management objective might be an innovation to many managers. To say one wants a fire to be lit, but one does not care if it burns the area within the burn breaks entirely or not at all, would confuse most fire specialists. Fire specialists pride themselves on meeting the goals (the will) of the land manager who seeks to use fire. Lighting fire for fire's sake alone seems almost like a wildfire. Indeed, adding wildness to systems involving fire should be done carefully such that it still occurs within the broad parameters of burn breaks and fire weather without undue risk to safety or property.

Again, accept change first seems passive, but in practice accepting change in wild systems shares some of the restrictions, and thus the challenges, of a wilderness or natural area. Goal-oriented management would not be allowed. This is intentional and would allow complex interactions among native and exotic species to emerge; in biological terms, it allows new species assemblages to coevolve to each other within the context of a complex disturbance regime. Unlike wilderness, in which pristineness is a core value, humans can intervene in a wild social-ecological system, but they cannot do it in a goal-oriented way lest it become a social system directing an ecological system. This may be even more difficult than the simple protection of a simple system; it calls managers to grapple with what it means to work as within a wild social-ecological system.

Direct Change toward a Simple System: Optimize

This approach is similar to focusing effort on regenerating oak to meet a narrow, single species management goal. However, rather than focusing on replacing the current canopy, the focus shifts to managing for other species in the system to maximize benefits to those species. As a direct change approach, these would be species that are expected to benefit from the drivers that are transforming the system, in the current example fire exclusion and climate change.

In the current example few plausible scenarios exist for this approach. Most of the game animals and rare species that are the focus of management in this region (deer, turkey, small game, rare butterflies, and rare herpetofauna) would benefit more from resist strategies of oak regeneration and restoration of oak ecosystems. This is one reason that oak regeneration and restoration are such a focus of managers both within and outside the case study area.

The management approach from the middle of the twentieth century in this region is probably the best example of this approach, and that was the extensive planting of exotic shrubs for upland birds. The management focus was narrow: to plant shrubs in numbers and patterns to create a landscape dominated by autumn olive (*Elaeagnus umbellata*), Eurasian honeysuckles (*Lonicera* spp.), and multiflora rose (*Rosa multiflora*). This approach met its narrow goals, but the unintended negative consequences of ecological invasion were larger than the benefit. Removal of invasive shrubs is expensive, and for several decades it has been a major barrier to oak regeneration and restoration of oak ecosystems.

Directing transformation toward simple goals that originate solely in the social part of a social ecological system (or treating social and ecological systems as separate, in which one is simply a resource for the other) is easy to justify, but fraught with potential unintended consequences. Directed transformation, by contrast, is predicated on a detailed understanding of the social systems, ecological systems, and the social-ecological systems. However, taking a systems approach would be more in keeping with the last approach of directing transformation toward complexity rather than optimizing toward one goal.

Direct Change toward a Complex System: Diversify

There are two distinct scenarios that I want to highlight here because they illustrate the difference between the ways wildness and complexity interact in practice, and how they can exist in an apparent paradox with a directed approach. The first is a variation on rewilding, and the second is square mile sandboxes.

Trophic rewilding is the reintroduction of megafauna to restore disturbance dynamics associated with large herbivores and carnivores (Marris 2011, Svenning et al. 2016). Shifting concentrations of predators in space and time are expected to create shifting mosaics of herbivory intensity as herbivores change their movement behaviors. To the degree that the wildness literature includes management recommendations, it often includes some form of trophic rewilding (Fuhlendorf et al. 2009, Svenning et al. 2016). To the degree that rewilding sometimes sets a historic baseline involving restoring species absent from the landscape for a couple centuries or less (Svenning and Faurby 2017), this would fall under the restoration approach. However, if species that fill key roles in the ecosystems are extinct, then extant exotic species with similar life history might be advocated for introduction (Seddon et al. 2014). This would be rewilding under a directed transformation approach.

In southern Michigan, this might look like introducing plains bison (*Bison bison bison*) to replace woodland bison (*Bison bison athabascae*), Rocky Mountain elk (*Cervus canadensis nelsoni*) to replace eastern elk subspecies (*Cervus canadensis canadensis*), and at some future date, if possible, a hybrid passenger pigeon (*Ectopistes migratorius* X *Patagioenas fasciata*). Predators such as wolves (*Canis lupus*) and cougars (*Puma concolor*) could be reintroduced.

Elephants might even be introduced, to mimic the effects of mastodons. This would create a new complex patterning of disturbance on the landscape.

While it is an interesting thought experiment of what a purely ecological rewilded system might entail, the social system is unlikely to support it. Predator reintroductions and de-extinction programs tend to be highly controversial. The landscape around state game areas is largely agricultural and privately owned, and the potential wildlife damage caused by large herbivores would be intolerable. The reintroduction of a small population of an herbivore, such as bison, might be tolerated if it was done within a fenced area, but the wildness of fenced wildlife is questionable, even when the fenced areas are very large (Child et al. 2019). Thus, rewilding in this landscape might increase ecological wildness, but it would not be consistent with a self-willed social system. An external authority would have to impose and maintain the ecological rewilding over the objections of the local social system.

In wild social-ecological system, the social and ecological components reinforce each other. This might look like modifications to the landscape and to the social system that allow fire; it might also look like modifications to disturbance regimes like winter mowing to create open canopy conditions with return intervals that allow more oak to mix with a red maple and invasive dominated system. It would be different than the status quo in that there would be more disturbance and more management outside of restorable areas. It would be directed in that oak and early successional habitats would be the focus, but it would not be in a simple optimizing on one goal. Rather, the focus would be on disturbance regimes that result in diversity and heterogeneity that provide a range of ecosystems services, for example, hunting and foraging opportunities, bird and butterfly watching, or basking sites for rare herpetofauna.

One approach would be to create larger permanent burn buffers around certain remote units. These might be a mix of existing barriers to fire like rivers or lakes, and developed barriers, such as long linear strips of corn or soybean that is plowed each fall to create a dormant season burn break. These bare soil and water barriers would be much wider than those typically used for prescribed fire, which is only safe over a narrow range of weather conditions as outlined in the fire prescription. Conceptually, this would create a relatively safe area in which wildlife managers could experiment with fire. This would likely require substantial changes in the social systems of plans and policies that permit adaptive learning, but still regulated to the point that it is safe for those igniting the fire. The change would be that suppression equipment would be limited to that needed for employee safety and not to keep the fire from spreading outside the burn unit.

At the opposite end of the spectrum, it might look like developing the expertise and equipment to do many very small fires at scales of 0.01 to 0.1 ha. Fires at this scale can be used to create patches of highly nutritious vegetation that could concentrate browse and activity in some areas and away from others. This could be used to create recreation opportunities akin to food plots during hunting season, or to draw browse away from regenerating oaks during seasons when they are particularly susceptible to browsing. It could even be used to focus browse on problematic vegetation (red maple, or invasive shrubs) when those species are most susceptible to browse. And by shifting this use of fire in space and season and frequency, and much more heterogeneous fire effect can be created relative to burning a large area to a uniform burn intensity.

If fire is simply not an option in an area, one could disturb ecosystems using a random mowing pattern with heavy equipment (e.g., excavators with hydro ax attachments) to clear all woody vegetation. This is already done in a restoration context to create habitat for Karner blue butterflies. It is done over snow in the winter to avoid impacting butterflies and other wildlife. If a different random path were followed each year, eventually the landscape would be a heterogeneous patchwork of frequently and infrequency mowed areas.

What these approaches have in common is that they are directed toward high level goals, and that they are carried out in ways that create shifting mosaics and heterogeneous landscapes in which parts of the landscape may escape disturbance on rotations longer than is typical in the status quo, but other parts of the landscape will be disturbed on rotations much shorter than the status quo. The disturbance regime is within the control of the manager, and they can increase or decrease or modify it to meet different needs of the social system. Thus, there is human goal-oriented behavior that is inconsistent with ecological wildness, but there is also a randomness and heterogeneity, a respect for the self-organization of the ecological system that is inconsistent with allowing the social system to obey only its self-will, to optimize only on ecosystem services, for example. However, the larger social-ecological system reinforces to create something greater than what can be reduced to the wildness of the ecological system itself or the social system itself. The social-ecological system emerges with its own self-organization, it self-renews, exhibits self-will, and thus is a wild.

APPENDIX

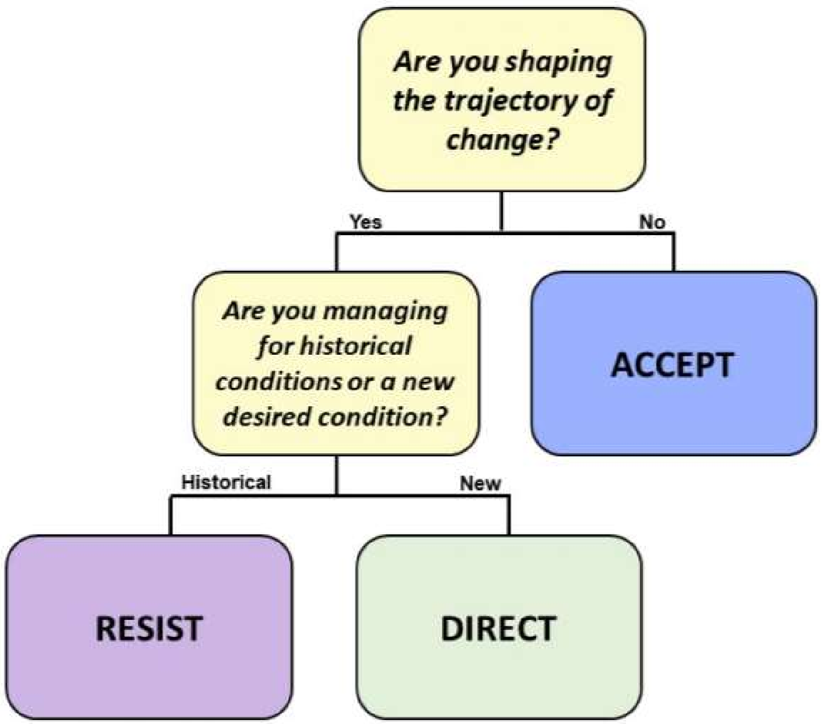

Figure 22: The Resist-Accept-Direct framework from Schuurman et al. (2020).

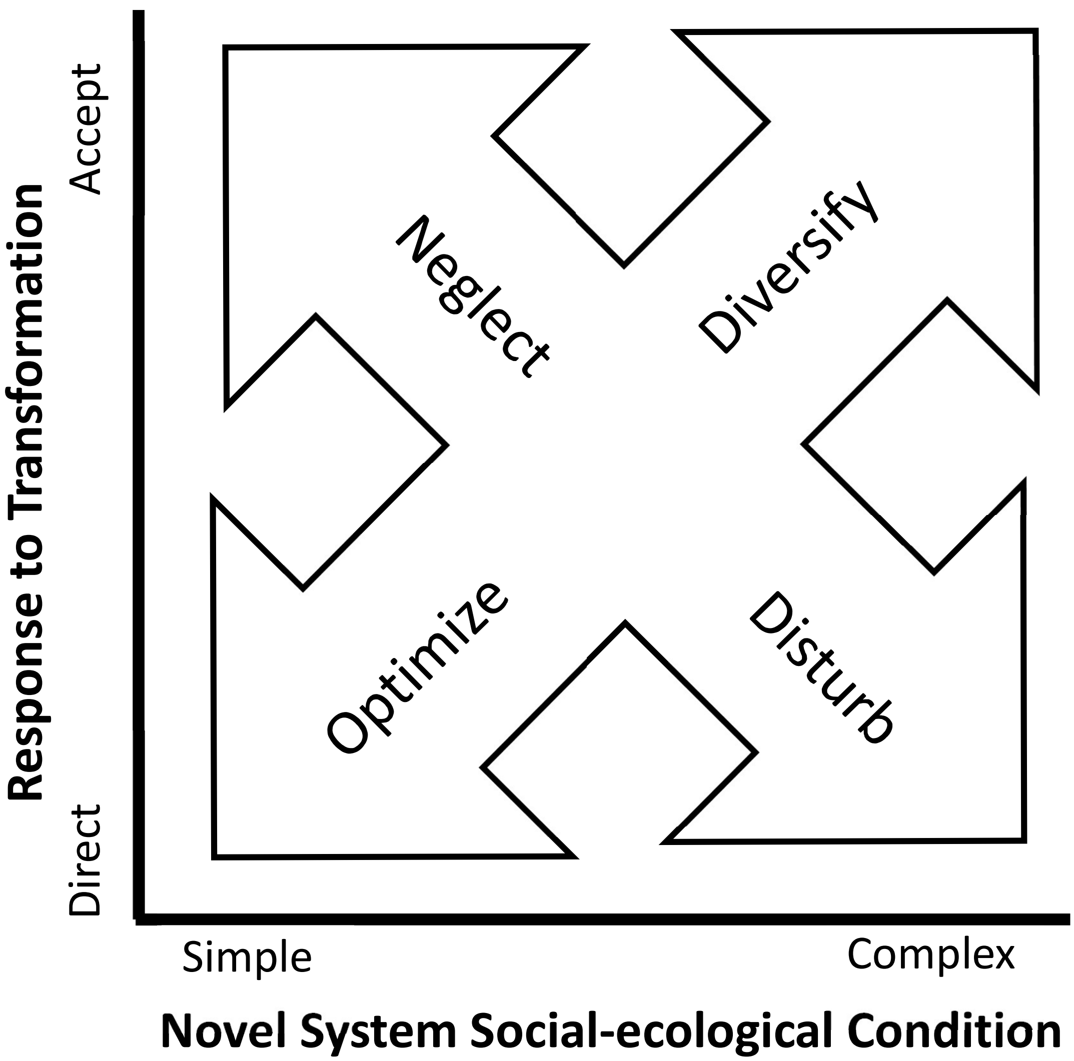

Figure 23. Management approaches integrating transformation approaches and complexity.

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Dissertation Conclusion

One of the main findings from the first chapter was that the results of the interviews and the models that originated from the qualitative analysis of the interviews were both different and more nuanced than expected. I have worked as a professional within this system for nearly two decades, and I felt that I had a great deal of expertise on the prescribed fire program within the Michigan Department of Natural Resources (MDNR). However, the interviews taught me to appreciate that social systems, even in relatively structured organizations like a state agency, are multifaceted and complicated. The ecological oak mesophication hypothesis can be extended in many different social system directions, including procedural fairness, risk perception, telecoupling to fire suppression outside the study area, and ecological transformation perceived as regeneration failure. Consideration of mesophication as both social and ecological elucidated new opportunities to intervene to address oak mesophication. These hypotheses have utility in extending an existing ecological hypothesis, but the causal loop diagrams remain untested hypotheses. They describe perceptions of the system in a qualitative way, but they have not been checked for logical consistency or compared to independent data in ways that would test the hypotheses.

The second chapter was rich in management relevant insight and provided a valuable example of emergent adaptation in a real natural resource management system. However, the core model output was analogous to a negative experimental finding. Managers did not adapt, and thus the effects of adaptation on reversing mesophication were negligible. The reason manager agents did not adapt was that climate change did not affect the seasonality in prescribed fire weather as expected. That is valuable knowledge, and the finding that mesophication swamped fire effects when fires are not repeated on the same sites was also a valuable insight. Thus, the model was useful to generate insight, even if the inferences about adaptation that can be drawn from the agent-based model are limited. It is difficult to validate agent-based models, especially those that simulate agent behavior decades in the future. The model was more useful as a digital laboratory in which what-if scenarios can be played out. There are several extensions of the model that could be useful. It could be applied to other jurisdictions, such as the state forest system in northern Michigan. The ecological portions of the model could be developed and validated to better capture disturbance processes other than fire (similar to the LANDIS-II model). Finally, agents could be given the ability to choose stands and to repeat disturbances until stands met some agent-specific criteria.

While the second chapter focused on oaks and maples in the context of mesophication, it also revealed that the third most common subcanopy species was autumn olive (*Elaeagnus umbellata*), a non-native invasive plant. When considered in conjunction with the insights regarding regeneration failures and ecological transformation, I saw a need for synthesis. Systems are changing and managers can either resist change and try to restore past ecosystems, or they can work with novel ecosystems. I argue that the manager-red-maple-autumn-olive social ecological system should be treated as a complex adaptive self-organizing system. In this context, self-organizing systems are wild social-ecological systems that can and should have some human-mediated disturbances. This is a different way of considering wildness because it includes human activity and disturbance as part of the wild system. In this paradigm, absence of human influence is a special case of wildness as self-organization of a complex adaptive system; but some special cases of human influence in social-ecological systems can be consistent with wildness. To maintain the wildness of self-organizing social-ecological systems, the whole system should be approached as an organic system based on respect and interdependent relationships rather than as mechanical systems based on prediction and control.

Insights could be gained from extending the existing ABM to incorporate ecological transformation and self-organization within the social system, within the ecological system, and within the social-ecological system. Emergence of self-organization is difficult to simulate. Nevertheless, attempting to model it in a computation model like an agent-based model helped ground the philosophical synthesis of the third chapter in a real example. The system of state game areas in southern Michigan, with their ongoing transformation, and their interactions between wildlife biologist managers and wild species, were a good case study for the wider transformations occurring globally. In the Anthropocene those who are entrusted to manage wildlife must be intentional about conserving wildness of wildlife, wild lands, and wild social-ecological systems.