Title
Assessing and Monitoring Landscape-Scale Restoration in Water-Limited, Disturbance-Prone Systems

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Publication Date
2022

Peer reviewed|Thesis/dissertation

UNIVERSITY OF CALIFORNIA
RIVERSIDE

Assessing and Monitoring Landscape-Scale Restoration in Water-Limited, Disturbance-Prone Systems

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

Doctor of Philosophy

in

Plant Biology

by

Mystyn W. Mills

September 2022

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Acknowledgements

This research and dissertation have been supported through many forms of academic and personal support. First, I must thank the Vulcan Materials company for their support and the many meetings that inspired chapter 1. A big thank you to Drs. Gary Adest and Barbara Brydolf, for opening their home to me, their inspiring commitment to the conservation and restoration of California oak woodlands, for sharing their knowledge, and the countless other ways they have supported me and my research. Thank you to Dr. Paul Laris. He's been instrumental to my career as an academic. I will be forever grateful for his mentorship and inspiration. Dr. Janet Franklin, my advisor, was encouraging, tireless, and always knew how to keep me motivated throughout the many iterations of my research, even in the face of a global pandemic. I cannot thank her enough. Thank you to the rest of my committee, Dr. Loralee Larios and Dr. Kurt Anderson. Their insights and guidance have been invaluable. Finally, thank you to my family. They have all sacrificed much to help me get to where I am today, especially my father Randy Mills (from the beginning), my partner Chris Bradford (giving me the thumbs up when it mattered most), and my son Allister Bradford (joining me in the field and understanding that "mommy" needs to write). And to my mother-in-law Luanne, who stepped in countless times with Allister so that the many distraction free hours dedicated to this research were possible.

Dedication

To Allister. That he and Riff may join me for many more explorations of oak woodlands and summer storms with the van door wide open. And that he may find a more beautiful question. Stay curious, my love.

ABSTRACT OF THE DISSERTATION

Assessing and Monitoring Landscape-Scale Restoration in Water-Limited, Disturbance-Prone Systems

by

Mystyn W. Mills

Doctorate of Philosophy, Graduate Program in Plant Biology
University of California, Riverside, September 2022
Dr. Janet Franklin, Chairperson

Most restoration projects to date have focused on a specific system in a specific location and are rarely greater than 5,000 hectares in extent. With the U.N.'s declaration of the decade of restoration (2020-2030), calls for restoration at broader scales, in more diverse ecosystems, and to shift in focus to the restoration of complexity are growing. However, ecological restoration at broader scales, encompassing multiple habitat types, management goals, and complex cross-scale interactions presents its own suite of challenges. To address these challenges, I first developed a long-term adaptive ecological monitoring framework for landscape restoration projects. Then I assessed changing fire regimes in California woodlands at regional scales and how this may influence current vegetation structure. I found that woodlands in California are experiencing increases in fire extents and severities. I also evaluated the current conditions and restoration needs of California's iconic Blue oak woodlands and savannas. These systems have experienced extensive conversion to nonnative grasslands and to early successional stages.

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Introduction

The United Nations has declared 2020-2030 the decade of ecological restoration (Cooke, Bennett, and Jones 2019). Calls for a broader and more systematic approach to ecological restoration are not new; however, these calls have become ever more imperative as the world faces increased broad-scale pressures associated with climate change and a growing human population with its inherent needs for resources and land (Holl, Crone, and Schultz 2003). To date, most restoration projects have focused on a specific system in a particular location and are rarely greater than 5,000 hectares in extent (Haugo et al. 2015; Cooke, Bennett, and Jones 2019). However, ecological restoration at broader scales, encompassing multiple habitat types, management goals, and complex cross-scale interactions presents its own suite of challenges.

Although the 'carbon copy' approach to restoration, which is based on the assumption that ecosystems develop predictably toward specific endpoints, has received its share of criticism, it is still commonly practiced (Hilderbrand, Watts, and Randle 2005). Often this is manifested in goals to protect a particular species or achieve a specific cover of vegetation. In many systems, research is showing that there are multiple possible endpoints, indeed, also multiple trajectories with non-linear progressions through multiple states (Westoby and Walker 1989; Bestelmeyer, Ash, and Brown 2017).

State and transition models, proposed by Westoby (1989), are one way to conceptualize complex ecosystem dynamics. State and transition models (STMs) are graphical representations that describe current understanding of the fundamental principles and processes of a system and the relationships among its parts (Heemskerk, Wilson, and Pavao-Zuckerman 2003). In recognition that many ecosystems are heterogeneous and non-linear, STMs provide for multiple successional pathways, multiple steady states, and multiple thresholds for transitions between states. Initially, these models were primarily used in rangeland (livestock grazing) systems to describe the dynamics between shrublands and grasslands, however, their usefulness for describing other ecosystems is beginning to be acknowledged (Bestelmeyer, Ash, and Brown 2017; Likens and Lindenmayer 2018; Blankenship et al. 2021). And although they have been criticized for being descriptive and not quantitative, there are exceptions (Phelps and Bosch 2002; Bashari, Smith, and Bosch 2008; Blankenship, Frid, and Smith 2015).

Bullock et al. (2022) have emphasized the need to move beyond the idea that restoration is the sum of activities at a particular site by proposing to refocus restoration on restoring ecological complexity. They call for a shift in the aims of restoration to restore whole system functionality across multiple scales. Their approach recognizes that for ecosystems to remain resilient for the long-term, the landscape and regional contexts and processes must be considered, and that landscape heterogeneity is key to ecosystem persistence in the face of uncertainties.

The difficulties in restoring complex ecosystems have also been met with a call for adaptive management and monitoring approaches that can be responsive to and accommodate new information and technologies when they become available (Lindenmayer and Likens 2010). Lindenmayer and Likens (2018) argue that an adaptive approach to restoration monitoring allows for findings from well-designed experiments to be used to update management approaches and goals, to modify or generate new questions regarding efficacy of management, and improve overall understanding of systems and how to manage them effectively. And while many agree on the benefits of this approach, there is relatively little guidance for, or examples of, implementation (Likens and Lindenmayer 2018).

Landscape and regional-scale restoration and conservation projects – spanning more than a single site – not only comprise multiple habitat types, but they typically must also accommodate multiple land uses and multiple management goals (Stringham, Krueger, and Shaver 2003; Likens and Lindenmayer 2018). Thus, it can be challenging to determine what strategies and methods will be effective, where they will be effective and why (Méndez-Toribio, Martínez-Garza, and Ceccon 2021). Moreover, the long-term monitoring of restoration sites, evaluation of restoration outcomes, and the communication of findings, have proven challenging to implement for landscape-level projects (Likens and Lindenmayer 2018). This has resulted in the underutilization of large-scale restoration projects as experiments to inform ecological theory (Bullock et al. 2022). Additionally, it has limited the application of new findings and the updating or adoption of new methods and technologies (Likens and Lindenmayer 2018).

Along with the need to scale up restoration, Temperton et al. (2019) and others note, there is the need for landscape ecological restoration to be more inclusive (Chazdon and Laestadius 2016). With the rise of carbon sequestration programs emphasizing tree planting as a tool to mitigate climate change, the majority of larger-scale restoration projects have thus far been in forested systems. Broad-scale restoration of other at-risk systems is less often implemented, despite the importance of these systems to the maintenance of biodiversity, ecosystem services, and the mitigation of climate and global change (Temperton et al. 2019).

My research takes a landscape to regional perspective on ecological restoration and follows Bullock et al (2021) in that it encompasses the need to restore ecological complexity. This includes the restoration of vegetation structural heterogeneity and also ecosystem process heterogeneity, specifically disturbance processes. I also focus on non-forested systems – oak woodlands and savannas and alluvial fan shrublands. I incorporate the use of state and transition models, both as conceptual and as quantitative simulation tools, to communicate ecological complexity and to assess the potential impacts of ecosystem change and management actions.

First, I developed a long-term adaptive ecological monitoring framework for landscape restoration. The framework is a systematic holistic approach to developing long-term ecological adaptive monitoring and management (L-TEAM). After discussing some of the challenges to long-term ecological monitoring, I demonstrate how L-TEAM can mitigate some of these challenges, namely by establishing clearly defined goals and indicators, well designed experimental tests of management efficacy, and effective communication among stakeholders. This framework builds on four conceptual and methodological developments that can support long-term ecological monitoring and management - adaptive ecological monitoring, state-and-transition models, objective oriented goal development, and decision support tools. I demonstrate the use of this framework applied to case studies in a Southern California riparian shrubland and a central California foothill woodland.

Second, I assessed changing fire regimes in California woodlands at regional scales and how this may influence current vegetation structure. Restoring complexity is more than just restoring different habitat types. It is important to recognize that a variety of processes influence habitat development, heterogeneity, and resilience. Throughout much of California, fire is a key driver of ecosystem change. But fire itself is a heterogenous process, varying in severity, frequency, and extent. I compared the historical range of variation in fire severity extent and proportions and compared them to contemporary fire severity distributions. Understanding differences in modern and historical fire severity distributions and its potential influence on vegetation structure can inform current management practices and help us understand how foothill woodland systems may respond to future changes.

Finally, I evaluated the current conditions and restoration needs of California's Blue oak woodlands and savannas. Blue oak woodlands and savannas are one of California's most extensive and biodiverse vegetation communities (Bernhardt and Swiecki 2001). They are also one of the states most degraded (Thorne et al. 2018). I assessed how blue oak woodlands and savannas differ in rates of conversion and departure from historical structural variation by ecoregion in the three major ecoregions where it occurs. I then used that assessment to identify restoration needs based on departures from historical patterns. And then I demonstrated the usefulness of state and transition simulation modeling to help determine the level of restoration treatments necessary to restore the historical structural variation of Blue oak woodlands and savannas.

Ecological restoration, like conservation biology, could be called a crisis science (Soulé 1985; Meine, Soulé, and Noss 2006). Scientists and practitioners are often on scene after the damage has been done. Moreover, timelines are often constricted, with results expected on timescales relevant to humans (i.e. a few years to a decade) as opposed to ecosystems (i.e. decades to centuries) (Likens and Lindenmayer 2018). While challenging, this does not negate the need for management of these projects to be strategically and scientifically informed and implemented. Nor does it mean we should not make the effort to use approaches that will give us the ability to learn how management actions and other anthropogenic actions influence ecosystem heterogeneity, resilience, and persistence. My research aims to address these challenges.

Works Cited

Bashari, H., C. Smith, and O. J. H. Bosch. 2008. “Developing Decision Support Tools for Rangeland Management by Combining State and Transition Models and Bayesian Belief Networks.” *Agricultural Systems* 99 (1): 23–34.

Bernhardt, and Swiecki. 2001. “Ecological Importance of California Oak Woodlands.” *Restoring Oak Woodlands in California: Theory*.

Bestelmeyer, B. T., A. Ash, and J. R. Brown. 2017. “State and Transition Models: Theory, Applications, and Challenges.” *Rangelands*.

Blankenship, Frid, and Smith. 2015. “A State-and-Transition Simulation Modeling Approach for Estimating the Historical Range of Variability.” *AIMS Environmental Science*.

Blankenship, Kori, Randy Swaty, Kimberly R. Hall, Sarah Hagen, Kelly Pohl, Ayn Shlisky Hunt, Jeannie Patton, Leonardo Frid, and Jim Smith. 2021. “Vegetation Dynamics Models: A Comprehensive Set for Natural Resource Assessment and Planning in the United States.” *Ecosphere* 12 (4). https://doi.org/10.1002/ecs2.3484.

Bullock, James M., Elisa Fuentes-Montemayor, Ben McCarthy, Kirsty Park, Rosie S. Hails, Ben A. Woodcock, Kevin Watts, Ron Corstanje, and Jim Harris. 2022. “Future Restoration Should Enhance Ecological Complexity and Emergent Properties at Multiple Scales.” *Ecography* 2022 (4). https://doi.org/10.1111/ecog.05780.

Chazdon, Robin L., and Lars Laestadius. 2016. “Forest and Landscape Restoration: Toward a Shared Vision and Vocabulary.” *American Journal of Botany* 103 (11): 1869–71.

Cooke, Steven J., Joseph R. Bennett, and Holly P. Jones. 2019. “We Have a Long Way to Go If We Want to Realize the Promise of the ‘Decade on Ecosystem Restoration.’” *Conservation Science and Practice* 1 (12). https://doi.org/10.1111/csp2.129.

Haugo, Ryan, Chris Zanger, Tom DeMeo, Chris Ringo, Ayn Shlisky, Kori Blankenship, Mike Simpson, Kim Mellen-McLean, Jane Kertis, and Mark Stern. 2015. “A New Approach to Evaluate Forest Structure Restoration Needs across Oregon and Washington, USA.” *Forest Ecology and Management* 335 (January): 37–50.

Heemskerk, Marieke, Karen Wilson, and Mitchell Pavao-Zuckerman. 2003. “Conceptual Models as Tools for Communication Across Disciplines.” *Conservation Ecology* 7 (3).

Hilderbrand, Robert H., Adam C. Watts, and April M. Randle. 2005. "The Myths of Restoration Ecology." *Ecology and Society* 10 (1).

Holl, Karen D., Elizabeth E. Crone, and Cheryl B. Schultz. 2003. "Landscape Restoration: Moving from Generalities to Methodologies." *Bioscience* 53 (5): 491–502.

Likens, G., and D. Lindenmayer. 2018. "Effective Ecological Monitoring."

Lindenmayer, David B., and Gene E. Likens. 2010. "The Science and Application of Ecological Monitoring." *Biological Conservation* 143 (6): 1317–28.

Meine, Curt, Michael Soulé, and Reed E. Noss. 2006. "'A Mission-Driven Discipline': The Growth of Conservation Biology." *Conservation Biology: The Journal of the Society for Conservation Biology* 20 (3): 631–51.

Méndez-Toribio, Moisés, Cristina Martínez-Garza, and Eliane Ceccon. 2021. "Challenges during the Execution, Results, and Monitoring Phases of Ecological Restoration: Learning from a Country-Wide Assessment." *PloS One* 16 (4): e0249573.

Phelps, D. G., and O. J. H. Bosch. 2002. "A Quantitative State and Transition Model for the Mitchell Grasslands of Central Western Queensland." *Rangeland Journal* 24 (2): 242–67.

Soulé. 1985. "What Is Conservation Biology?" *Bioscience*.

Stringham, Tamzen K., William C. Krueger, and Patrick L. Shaver. 2003. "State and Transition Modeling: An Ecological Process Approach." *Rangeland Ecology & Management/Journal of Range Management Archives* 56 (2): 106–13.

Temperton, Vicky M., Nina Buchmann, Elise Buisson, Giselda Durigan, Lukasz Kazmierczak, Michael P. Perring, Michele Sá Dechoum, Joseph W. Veldman, and Gerhard E. Overbeck. 2019. "Step Back from the Forest and Step up to the Bonn Challenge: How a Broad Ecological Perspective Can Promote Successful Landscape Restoration." *Restoration Ecology*, no. rec.12989 (May). https://doi.org/10.1111/rec.12989.

Thorne, James H., Hyeyeong Choe, Peter A. Stine, Jeanne C. Chambers, Andrew Holguin, Amber C. Kerr, and Mark W. Schwartz. 2018. "Climate Change Vulnerability Assessment of Forests in the Southwest USA." *Climatic Change* 148 (3): 387–402.

Westoby, and Walker. 1989. “Range Management on the Basis of a Model Which Does Not Seek to Establish Equilibrium.” *Journal of Arid Environments*.

Ch. 1: LTEAM: A framework for long-term ecological adaptive monitoring, management, and evaluation of landscape restoration

Abstract

The United Nations declared 2021-2030 the Decade of Ecological Restoration. Billions of dollars are spent on restoration projects every year. However, monitoring, assessing, and managing these efforts for the long-term has been difficult to implement, especially at the landscape-level. We review some of the challenges faced in the long-term management and monitoring of landscape-scale restoration and discuss four different but promising conceptual and methodological developments that have the potential to support long-term ecological monitoring and management. Then, we propose L-TEAM, a long - term ecological adaptive monitoring and management framework that is based on a conceptual model of a system, clearly defined and measurable goals, rigorous experimentation, and decision support tools. As our case-studies demonstrate, L-TEAM can be applied to a variety of restoration projects and habitat types with a range of restoration and management goals. Use of the framework not only informs management decisions and ensures the implementation of scientifically informed long-term monitoring but also has the potential to improve our understanding of ecosystem function and response to disturbance and management actions.

Introduction

Billions of dollars are being spent on restoration projects every year (BenDor et al. 2015). With the United Nations's Declaration of The Decade of Restoration (2021-2022; Aronson et al. 2020) this amount is only projected to increase. While commendable, how is the success of these projects being monitored and assessed? Are the outcomes being used to learn, inform and guide future management – to improve our approaches to ecological restoration and our success rates?

Landscape-scale restoration and conservation projects, spanning more than a single site, present especially challenging circumstances for ecological management and monitoring (Stringham et al. 2003). Often these projects comprise complex and non-equilibrium systems, are made up of multiple habitat types, and accommodate multiple uses and management goals. Long-term ecological monitoring to determine what strategies and methods are effective and why, is crucial to our understanding of how to better conserve and restore these systems (Méndez-Toribio, Martínez-Garza, and Ceccon 2021). The benefits of long-term ecological monitoring are many (see table 1). However, due to the complex nature of these systems (Berkes, Folke, and Colding 2000; Walker et al. 2004; Berkes, Colding, and Folke 2008), long-term ecological monitoring, regular evaluation of restoration and conservation outcomes, communication of findings, and adaptive management, have proven challenging to implement for landscape-level projects (Likens and Lindenmayer 2018).

Many questions confront land managers looking to assess and monitor conservation and restoration projects including but not limited to: what, how, where and when to assess (Vos, Meelis, and Ter Keurs 2000). Because management and monitoring of landscapes must accommodate a variety of soil and vegetation complexes, as well as seasonal and annual patterns of precipitation, and variation in disturbance regimes (Holl, Crone, and Schultz 2003; Hood et al. 2021) there are often multiple answers to these questions. These projects must also balance management for human use and resource procurement with restoration and conservation goals (Eastburn et al. 2017; Burger et al. 2019; Western et al. 2020). Thus, restoration and conservation projects at the landscape-level require both spatially and temporally explicit approaches for implementing and monitoring management practices.

Table 1.1. Some of the benefits of well-designed ecological monitoring programs. Adapted from (D. B. Lindenmayer and Likens 2010).

Benefit	**Study Exemplifying Benefit**
Documents & Establishes baselines	Keeling et al. 1995, 1996
Ability to detect change in baselines & ecosystem function	Krebs et al. 2001; Danell et al. 2006
Generate new questions	Persson et al. 2009
Ability to evaluate response to disturbance/management actions	Schindler et al.1985
Identify Surprises	Zhan et al. 2006
Test theory with empirical data	ShraderFrechette & McCoy 1993
Provide data for development of simulation models	Burgman et al. 1993

Successful management also depends on diverse knowledge sources to understand and articulate system dynamics and ecological processes and mechanisms (Derner et al. 2021). Often, this knowledge is spread across projects, stakeholders, and documents. It can be difficult to communicate among stakeholders, with the public and, importantly, it can be challenging to update management plans and practices when new information or technologies become available. Hence, there is a need for a clear, concise and systematic way to communicate between stakeholders that can both identify where new information is needed and be updated once it is obtained.

Here I present a systematic holistic approach to developing long-term ecological adaptive monitoring and management (L-TEAM). We first describe some key challenges to long-term ecological adaptive monitoring and management (L-TEAM), namely clearly defined goals and indicators, well designed experimental tests of management efficacy, and effective communication. Next, we present four promising conceptual and methodological developments that can support long-term ecological monitoring and management - adaptive ecological monitoring, state-and-transition models, objective oriented goal development, and decision support tools. Then, we build upon these four tools to present a framework we developed for applying principles from long-term ecological monitoring and management to the practice of restoration and conservation in heterogeneous landscapes. Finally, we demonstrate the use of this framework applied to case studies in a Southern California riparian shrubland, and central California foothill woodland.

Challenges and Four Tools for Addressing Them

While long-term monitoring has been acknowledged as a crucial part of restoration and conservation actions, several factors make it difficult to implement and maintain a long-term ecological monitoring program. Many of these problems stem from the lack of clearly articulated questions and goals at the outset (Lindenmayer and Likens 2009). Data may be collected, even in copious amounts, but without driving questions and goals, the data may be collected haphazardly. This often includes poorly designed treatment and control sites (if there are any at all), a lack of statistical consideration of the power to detect trends, and inconsistent or poorly communicated management and data collection protocols. Moreover, without clearly defined questions and goals, there is often disagreement over what should actually be monitored, which can lead to the monitoring of many things poorly instead of a few things well. Or, if an indicator species or other proxy(ies) is(are) chosen, there tends to be a lack of a quantified relationship between the entity and the process(es) for which they are surrogates (Likens and Lindenmayer 2018; Calvache et al. 2021).

Projects often also face constraints related to funding and time. Research, when it is conducted, is often done on timescales related to graduate programs and grant duration (i.e. 3-5 years). And while funding for monitoring may be included in project and research proposals, by the time monitoring is implemented funds may be sparse. This often results in an ad hoc strategy with a focus on easy to assess, short-term ecological indicators that may or may not fully relate to the recovery and function of an ecosystem let alone answer outstanding questions about management actions and long-term system recovery and function (Likens and Lindenmayer 2018; Méndez-Toribio, Martínez-Garza, and Ceccon 2021).

Ultimately, the management of ecological systems should be linked to scientifically informed monitoring (Lindenmayer and Likens 2010; Derner et al. 2021). This requires clearly defined questions regarding specific management actions, and restoration and conservation goals that are tractable. These questions and goals should be established early on, and they should help guide what is being monitored and why (i.e. indicators) as well as clearly articulate how success is measured (Herrick et al. 2012). Both management and monitoring should be adaptive, in that as new information or technologies are acquired or new questions arise, monitoring and management actions can be updated accordingly. Moreover, a system should be in place for communicating understanding of system dynamics as this understanding evolves over time; this requires clear and accessible records of past experiments and findings so that new stakeholders do not "reinvent the wheel" or succumb to past mistakes. Thus, the challenge is how to provide restoration and conservation scientists, land managers, and other stakeholders with a systematic approach to collaboratively design and implement a long-term, adaptive ecological monitoring (L-TEAM) program.

The following sections review four conceptual and methodological developments that can address the challenges of defining goals, targets and indicators, and designing good experiments for testing and improving long-term ecological monitoring and management of heterogeneous landscapes.

Adaptive Ecological Monitoring

In their extensive survey of long-term ecological monitoring programs, Lindenmayer and Likens (2018) identified key factors that contributed to successful projects which included 1) clear management goals and questions linking monitoring to those goals, 2) detailed conceptual models, 3) sound experimental designs, and 4) relevance to management objectives and targets. In light of their review, the authors argue for an adaptive monitoring approach where question setting, experimental design, data collection, analysis and interpretation take place iteratively. Findings from well-designed experiments are used to update management approaches and even goals, modify or generate new questions regarding efficacy of management, and improve overall understanding of systems and how to manage them effectively.

While the benefits of adaptive ecological monitoring are evident (McIntosh et al. 2018; Lindenmayer and Likens 2009), there has yet to be clear guidance on how to implement an adaptive monitoring program using a standardized approach – especially at the landscape level. Even among the successful projects reviewed by Lindenmayer and Likens (2018) there is considerable disparity among approaches. Thus, we propose three techniques that address key components to a successful adaptive management plan. We then build on these techniques and integrate them into a framework for a transparent and standardized approach to implementing adaptive ecological monitoring and management of landscape restoration and conservation projects.

State and Transition Models

Lindenmayer & Likens (2009; 2019) along with others consider conceptual models essential to successful adaptive management (Heemskerk et al. 2003; Keenelyside et al. 2012; Derner et al. 2021). Conceptual models are typically graphical representations of concepts that describe a current understanding of the fundamental principles and processes of a system and the relationships among its parts (Heemskerk, Wilson, and Pavao-Zuckerman 2003). Ideally, conceptual models should enhance understanding of the system and facilitate communication amongst stakeholders (for an example from conservation monitoring see (Franklin et al. 2011). Importantly, conceptual models provide organizing frameworks for planning in complex systems that can be updated when new information arises.

State and transition models (STMs) are one approach to conceptualize representations of complex systems. Traditional views on plant successional theories that centered around single climax communities have proven inadequate to describe many restoration settings - especially in semi-arid systems (Westoby, Walker, and Noy-Meir 1989; Huntsinger and Bartolome 1992; Bestelmeyer, Ash, and Brown 2017). Westoby (1989) pioneered the use of state-and-transition models. These models provide for multiple successional pathways, multiple steady states, and multiple thresholds for transitions between states.

Stringham et al. (2003) noted the need for consistent terminology and components when using STMs because lack of consistency has led to criticism that they are difficult to compare and to communicate across projects. Their proposed definitions help clarify what each component of the model should represent and assist with communicating the model results. *States* are recognizable, resistant and resilient complexes of soil and vegetation structure (See Fig. 1 large boxes). *Thresholds* are points in space and time where, once crossed, the key ecological processes responsible for a system's identity degrade past a point of self-repair and active restoration is now required to restore the previous state (See Fig. 1 line bisecting the arrows between the boxes). *Transitions* are pathways between states (See Fig. 1 arrows). Transitions are often triggered by natural or human-caused disturbances which may occur quickly, as with a fire or flood, or more slowly in response to repeated stress such as grazing or drought. *Alternate states* are the long-term persistence of different plant and soil complexes on an alternative trajectory than the state of interest.

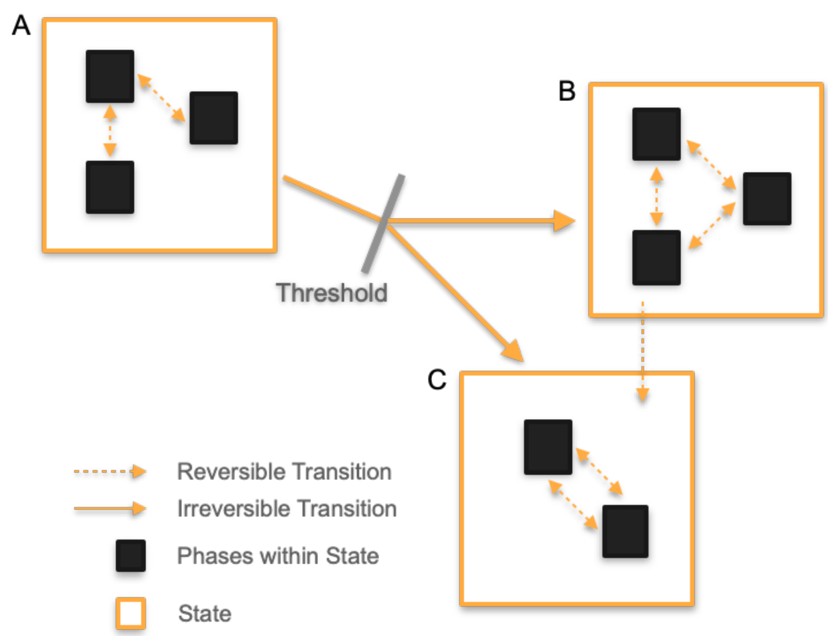


Fig. 1.1. A simplified conceptual state and transition model. The large boxes represent states. A is the original state and B and C are alternative states. The smaller solid boxes represent phases within a state, and the arrows indicate directional transitions between states with known or hypothesized drivers. The line bisecting the arrows between the larger boxes represents a threshold that once crossed often requires active restoration approaches to return the system to the desired state. Inspired by (Stringham et al. 2001).

Inherently stable, state changes are only possible when a threshold has been crossed. Within each vegetation state there is often the potential for large variation in species composition which is accommodated in the concept of "phase-shifts" which are defined as plant community dynamics within a state (Stringham, Krueger, and Shaver 2003). For a true state change, the system must cross a boundary or threshold that results in changes in a site's "identity" (its underlying processes; moving from A to B in fig.1) resulting in different sets of potential plant communities - for example, from grasslands to shrublands (Stringham, Krueger, and Shaver 2003). With the addition of phase-shifts, successional trajectories can be recognized and incorporated into these models (See Fig.1 smaller, solid boxes). Unlike state changes, phase-shifts align with a site's natural (successional) trajectory - they fall within the site's "identity".

In landscape restoration and conservation, the difference between phases and states is key and likely necessitates different management approaches for each. For example, restoration objectives may dictate that all phases in a target state are present, especially if species of concern are associated with specific phases. Moreover, if natural disturbances (for example, a natural flooding regime) are no longer operating to drive shifts between phases, management may be called upon to implement actions that initiate or inhibit phase shifts. This emphasizes the need for spatially explicit consideration of heterogeneous landscapes as well as for a robust conceptual model to clearly communicate this information.

STMs allow scientists and managers to synthesize scientific information and clearly communicate among stakeholders what is known about a system, its states, its phases, its thresholds and its hypothesized drivers of change. They can incorporate spatial heterogeneity and both natural and anthropogenic disturbances. STMs also help identify where knowledge is weak or lacking. Although STMs have been used frequently in rangeland sciences, their application for management in other ecological systems has been limited (Bestelmeyer, Ash, and Brown 2017; Likens and Lindenmayer 2018). While they have at times been criticized for not being quantitative (although there is potential, see (Phelps and Bosch 2002; Bashari, Smith, and Bosch 2008), we argue that given their ability to structure and communicate information about system states, disturbances, and management responses, and to generate hypotheses about drivers and change, STMs are well suited to the conceptual representation of systems undergoing long term management.

Establishing Measurable Objectives

Lindenmeyer and Likens (2018) and others have also identified the importance of well-defined objectives, goals and questions to guide management and monitoring of ecological systems (McIntosh et al. 2018; Calvache et al. 2021; Derner et al. 2021). And while STMs help consolidate knowledge into a conceptual representation of a system, they do not necessarily articulate management objectives and questions that can drive a sampling design for monitoring to determine if the restoration activities are achieving desired goals. Nor do STMs tend to make goals quantifiable.

Typically, goals are statements of intent which are then further developed into clear, and ideally measurable, outcomes (Keenelyside et al. 2012). Often there may be questions regarding how to obtain outcomes (e.g., best methods for nonnative plant species removal, effectiveness of planting or thinning treatments to achieve target plant densities, etc.) which require the establishment of monitoring and assessment regimes. Objectives and questions should be collaboratively developed among stakeholders, including scientists, statisticians, natural resource managers, etc. They should be germane to management goals and should help inform monitoring of specific management actions. Working to establish clear objectives is especially important when there are multiple, often competing goals for a restoration or conservation landscape – for example, allowing for recreational use while protecting and restoring wildlife interactions. In these cases, tradeoffs must often be assessed, and their consequences monitored (Keenelyside et al. 2012). However, limited tools are available to assist in 1) identifying target outcomes and questions, 2) identifying the specific (preferably quantifiable) metrics to monitor in order to assess progress toward outcomes, and 3) allowing management actions to be modified in response to information acquired during monitoring in order to improve the probability of attaining outcomes (Derner et al. 2021). For the management of rangelands, Derner et al (2021) propose a management tool that aims to address this challenge. Their approach is adaptive and outcome-focused and emphasizes the need for establishing clear, specific desired outcomes that are quantifiable whenever possible as well as science-driven monitoring to inform decision making. This facilitates sound data collection that is relevant to management goals. Although focused on grazing management, with modifications their approach has potential applications in landscape restoration monitoring and management.

Decision Support Tools

While STMs can incorporate landscape heterogeneity, because environmental variation can so strongly influence management action's success and cost-effectiveness, decision support tools (DSTs) help clarify where, when and how management actions should be carried out on the ground (Fig.2; Spiegal et al. 2014; Spiegal, Bartolome, and White 2016; Roesch-McNally et al. 2021). They further explain drivers of system change (between states and phases). They can be used to operationalize STMs and management objectives, articulating the expected outcomes of management actions, and to help identify where experiments are necessary to gather needed information. DSTs should include triggers or thresholds that initiate management actions. When possible, these triggers and thresholds should be quantifiable (see criticism of STMs above). Thus, DSTs can be used to directly guide management actions, can ensure management continuity when personnel changes occur, and importantly, they can be adapted when predicted outcomes are shown to be inaccurate, or when new technologies, information, or strategies are obtained.

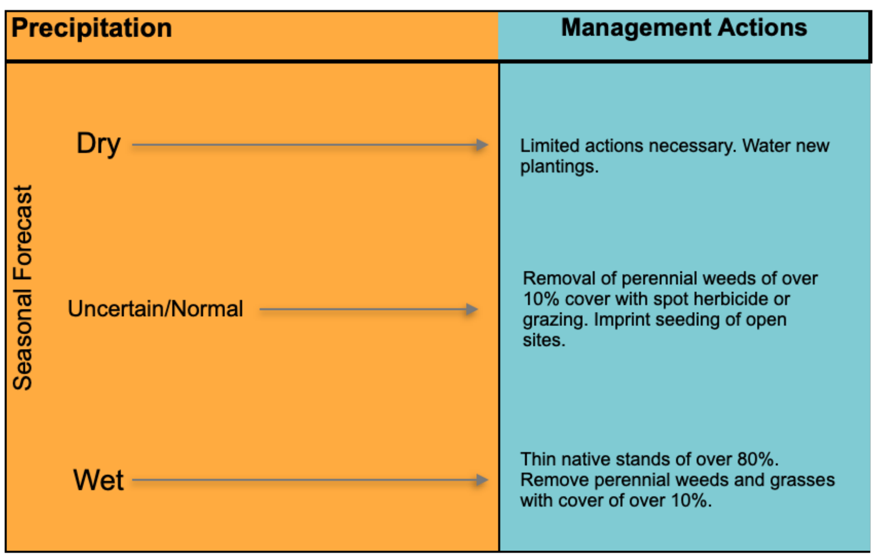


Fig.1.2. A simplified example of a decision support tool. The left column identifies the conditions under consideration, the right the appropriate management actions given those conditions.

Each of the approaches discussed above have strengths and limitations. When used together under one systematic framework, we believe they are complementary and have the potential to facilitate not only the long-term monitoring and management of conservation and restoration in heterogeneous landscapes but also ultimately a better understanding of how these systems function.

Methods: The Framework

We propose a new framework, L-TEAM (Long-Term Ecological Adaptive Monitoring), for the long-term adaptive monitoring, management, and evaluation of ecological restoration and conservation efforts at the landscape level (Fig. 3). Our framework links methods that can operationalize Lindenmayer and Likens's (2018) long-term adaptive monitoring approach based on their findings of what makes for successful monitoring projects. L-TEAM integrates STMs, objective oriented goals, and decision support tools as elements that represent system dynamics, define management goals and guide management actions. The framework helps develop questions and experiments that can test whether goals are met, adapt in light of new information, and support management decisions.

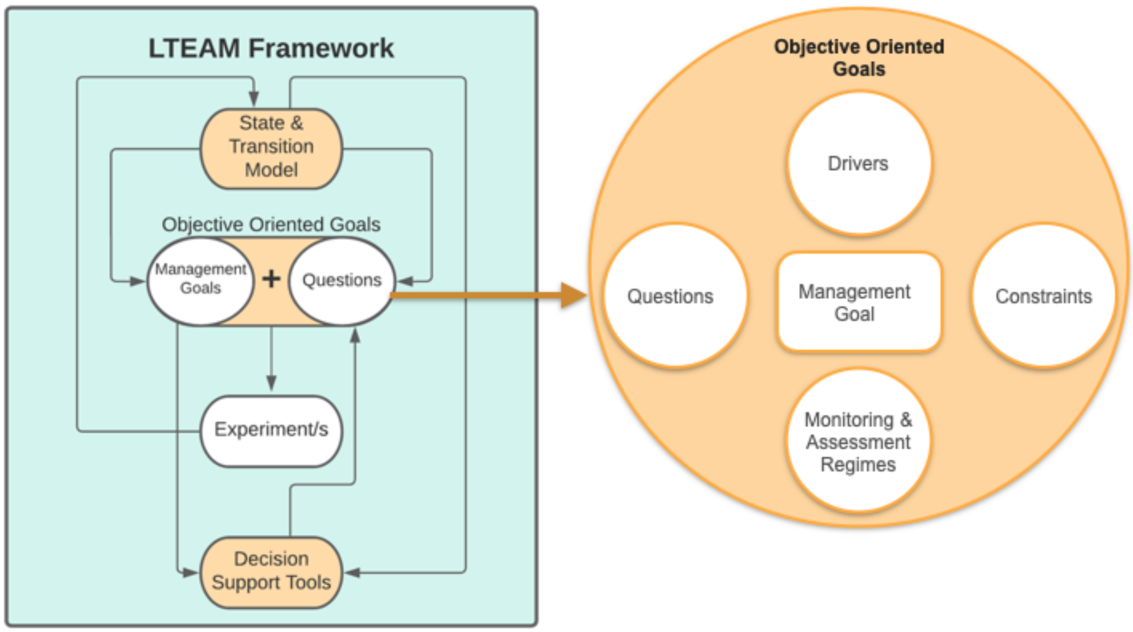


Fig. 1.3. A conceptual representation of the Long-Term Ecological Adaptive Monitoring (L-TEAM) framework. The conceptual model is an STM, which lends itself well to the representation of complex landscapes. Management and restoration goals are coupled with questions to create outcome-oriented goals (OOGs). OOGs articulate the drivers and constraints identified in the STM that directly affect the management goal of concern, they help turn those management goals into questions, and finally they assist in identifying what we need to monitor to answer our questions. This informs the design of rigorous experiments the results of which can be incorporated back into the STM and then used to inform the development of decision support tools.

In L-TEAM, the system is represented conceptually as an STM that is developed with input from stakeholders. The system's states and known and hypothesized drivers and transitions are described at the outset of the project. The STM can be updated if new information is obtained. The STM is then used by stakeholders to inform the development of Objective Oriented Goals (OOGs; see right side of fig. 3). OOGs couple management goals with outstanding questions that need to be answered in order to determine whether goals are being met. The STM also helps identify the specific drivers and constraints to achieving the management goal, which are incorporated into the OOGs. And finally, the OOGs help identify what needs to be monitored and assessed to answer the outstanding questions and to determine if a goal has been achieved. This leads to scientifically sound experiments designed to answer specific questions and determine if goals have been met.

The results of experiments are incorporated back into STMs and OOGs, informing the development of Decision Support Tools that provide a clear understanding or hypothesis of which management actions are effective and when and where they are effective. The decision support tools include thresholds or triggers (either known or hypothesized) that initiate specific management actions.

The framework is adaptive in that any new information or findings from experiments or new technologies can be incorporated back into the STM, used to modify or identify new OOGs, and to update the Decision Support Tools.

Results: Applying the L-TEAM framework

To illustrate the strengths of the L-TEAM framework, we present some of its tools developed for two case studies: The Cajon Creek Conservation Area and The River Ridge Ranch. Both are restoration projects located in California, USA. Cajon Creek represents a conservation bank-approach to restoration. A conservation bank consists of a parcel or parcels of private property managed in perpetuity for the protection of endangered species. The owner/s of the property/ies are granted credits through state and/or federal agencies for the value of the species and habitat being protected which they can then use, bank for the future, or sell to offset development (Fox and Nino-Murcia 2005). River Ridge Ranch represents a case of conservation land on private working landscapes where managers balance recreational use and restoration goals on rangeland historically used for cattle grazing.

Cajon Creek Conservation Area

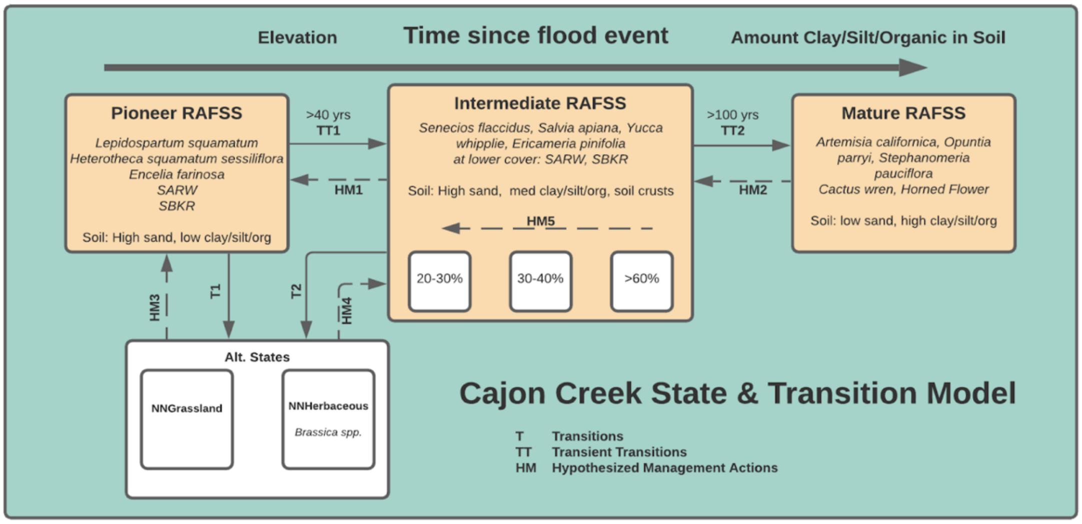
The Cajon Creek Conservation Area (CCCA) is located in Riverside County in Southern California, a semiarid region with a Mediterranean-type climate (Lockhart and Sprauge 1999). Cajon Creek vegetation consists primarily of Riversidian Alluvial Fan Sage Scrub ("RAFSS") which is a rare Southern Californian alluvial floodplain ecosystem. CCCA hosts about 45 species of conservation concern including the small mammal *Dipodomys merriami parvus* (San Bernardino Kangaroo Rat; SBKR) and *Eriastrum densifolium* ssp. *sanctorum* (Santa Ana River Woolly Star), a perennial herb. The RAFFS ecosystem occurs in a highly dynamic alluvial plain of rivers with seasonal flow and these ecosystems are threatened by development, illegal dumping, invasive species, hydrological modification and other anthropogenic modifications (Hanes, Friesen, and Keane 1989; Lockhart and Sprauge 1999).

In Cajon Creek, sand and gravel mining by the Vulcan Materials Company has also had substantial impacts on the system. In 1998, an agreement was established between Vulcan Materials and federal and state agencies to establish the conservation bank and restore and conserve portions of the Cajon Creek property. The Conservation Area now consists of over 1,200 acres of pioneer, intermediate, and mature successional phases of RAFSS, mule fat scrub and buckwheat scrub plant communities (Hanes, Friesen, and Keane 1989; Lockhart and Sprauge 1999). Restoration efforts to date include various approaches to removing nonnative plants, revegetating by imprinting seeding of native plants, and translocating SBKR. Monitoring habitat conditions through established ground-based transects and photo points has been implemented and annual Unmanned Aerial Vehicle (UAV) flights were recently initiated to collect imagery. A trapping grid is used to monitor SBKR populations.

River Ridge Ranch

River Ridge Ranch is a 722 acre property located in the Southern Sierra Nevada foothills just outside of Springville, California. In California, rangelands are the largest land cover by area, covering over half of the state. California annual grassland and hardwood woodlands are characterized as savanna with an over-story typically dominated by oaks *(Quercus* sp.). They provide more than two-thirds of forage for domestic livestock.

River Ridge Ranch was operated as a cattle ranch for over 100 years. Decades of land cover manipulation have resulted in a gradation of savanna landscapes from herbaceous dominated pastures, to savannas, to a patchwork of woodlands. It includes a riparian corridor of the North Fork of the Tule River, an irrigated lowland pasture and a larger unirrigated upland pasture. It is now managed under a conservation easement, preventing non-agricultural development. Landowners Gary Adest and Barbara Brydolf established this easement and have managed the land for the past 20 years with the intent of restoring habitat and establishing a sustainable system of land use. After experimenting with several approaches to habitat improvement such as fencing riparian corridors and implementing a rotational grazing system, the landowners now intend to try more innovative strategies to improve ecosystem services and to develop sustainable income streams. Cattle were removed from the land in June of 2019 and a new management plan is being developed.



T (transitions):

T1: transition to nonnative grasslands. Clearing/ mining. Roads. Unauthorized access (humans & horses). Fire. Introduction of invasive spp.

T2: transition to nonnative herbs. Clearing/Mining. Roads. Introduction of nonnative spp. Precipitation.

TT (transient transitions):

TT1: transition from pioneer RAFFS to intermediate RAFFS. Time since flood and elevation.

TT2: transition from intermediate to mature RAFFS. Time since flood and elevation.

HM (hypothesized management actions expected to initiate transitions):

HM 1: Management actions to convert nonnative grasslands/herbaceous to pioneer RAFFS. Restrict access. Herbicide. Grazing .

HM 2: Management actions to remove nonnative grasslands/herbaceous from intermediate RAFFS. Restrict access. Herbicide. Grazing.

HM 3: Management actions to transition mature RAFFS to intermediate RAFFS. Manual removal. Herbicide, Hydrological manipulation.

HM 4: Management actions to maintain plant cover of intermediate RAFFS below 30%. Manual Removal. Herbicide. Hydrological manipulation

HM 5: Management actions to transition intermediate RAFFS to Pioneer. Scour via mechanical removal to hydrological manipulation.

Fig. 1.4. This is the schematic portion of the State and Transition Model developed for the Cajon Creek Conservation Area. One of the main drivers of the system are flood events and time since the last event. Elevation and soil are also key factors. Within the intermediate RAFSS (Riversidian Alluvial Fan Sage Scrub) habitat type, three cover levels related to management goals were identified. Two alternative vegetation states were also identified, nonnative grasslands and nonnative herbaceous cover. Transitions and transient transitions were identified as well as the hypothesized management actions necessary to reverse, initiate, or prevent transitions. SAWR: Santa Ana River Wooly Star; SBKR: San Bernardino Kangaroo Rat – the two focal conservation target species.

State and Transition Models

To illustrate STM development within the L-TEAM framework, we present the STM for the Cajon Creek Conservation Area. In consultation with the Cajon Creek restoration ecologists and managers and based on the existing literature on RAFFS ecology, we constructed an STM for the Cajon Creek system (Fig 4). Main drivers of alluvial floodplain ecosystem dynamics are flood events and time since the last flood event. Floods scour existing vegetation and soil, transitioning the system back to an earlier phase. Elevation and soil are also key factors – typically higher elevations have more developed soils and vegetation phases. Within the intermediate RAFSS phase we identified three shrub cover levels related to management goals (lower cover has higher habitat quality for SBKR). We also identified two (undesirable) alternative vegetation states, nonnative grasslands cover consisting primarily of *Avena and Bromus* spp. and nonnative herbaceous cover consisting primarily of *Brassica* spp. We identified state transitions and their drivers, and transient states between states and phases, as well as the hypothesized management actions (e.g., conservation grazing) necessary to reverse, initiate, or prevent undesirable transitions (Fig. 4 lower box).

Objective Oriented Goals

The strength and flexibility of OOGs in their ability to address both more traditional restoration and conservation goals as well as goals that include resource use and sustainability is demonstrated by contrasting OOGs for Cajon Creek and River

Ridge. First, we describe these OOGs and then we outline the experiments that have been designed to answer the questions identified and assess goal achievements.

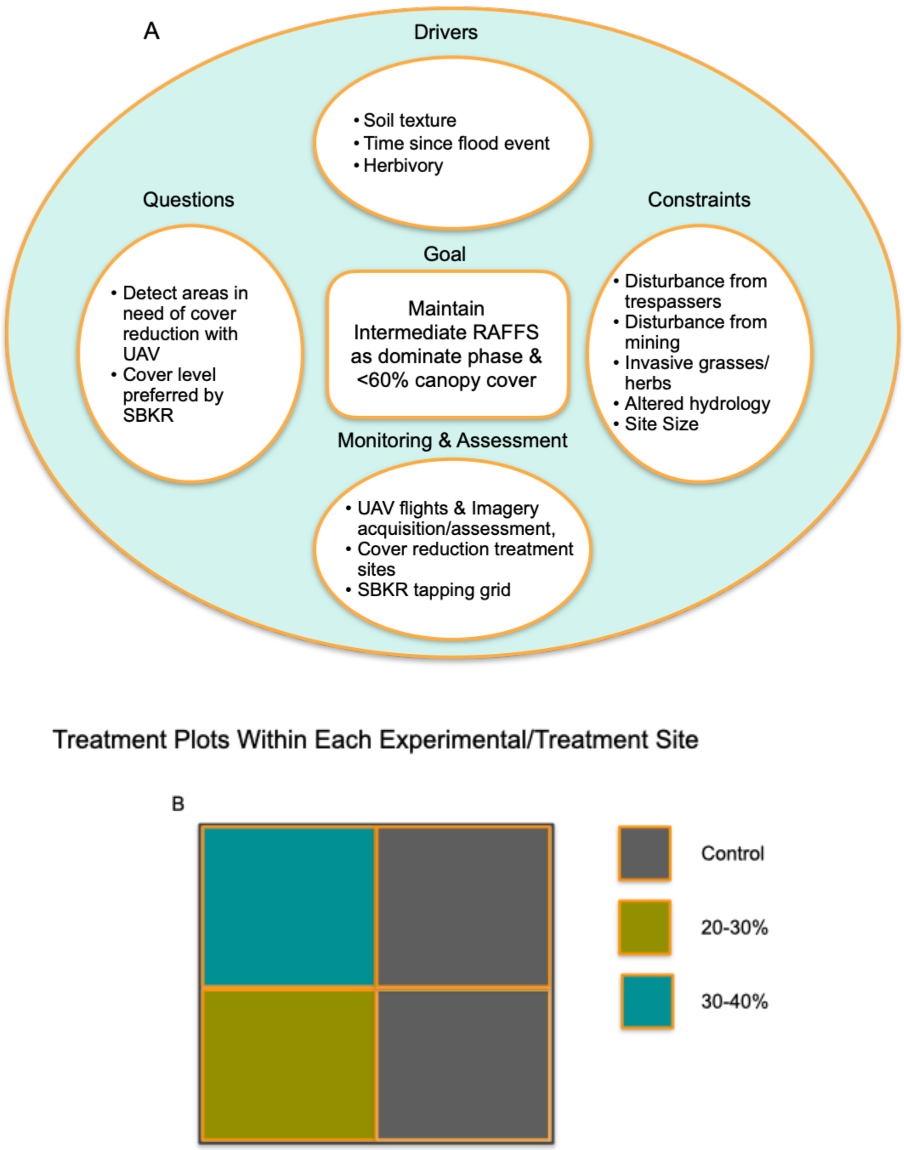


Fig. 1.5. One of the Objective Oriented Goals (A) and a schematic of the thinning treatment sites (B) developed with stakeholders for the Cajon Creek Conservation Area. Studies indicate that SBKR prefers intermediate RAFFS vegetation with more open canopies. To maintain vegetation in this phase the site must be monitored and thinning actions performed when necessary. Use of UAVs has the potential to support comprehensive, efficient monitoring if it is possible to detect RAFFS phases and cover amounts. Additionally, it will be important to determine which thinning level is most appropriate, resulting in increases in SBKR habitat use while still inhibiting invasion by nonnative grasses/forbs. Experimental thinning sites were established. At each site, subplots consisted of controls and subplots manually thinned to 20-30% and 30-40% cover.

Cajon Creek OOG

For Cajon Creek an overarching goal is the restoration of habitat for species of concern. Studies have shown that both SBKR and wooly star prefer more open shrub habitat, with some studies suggesting over 60% is too dense and cover of 30% or less as ideal (Chock et al. 2020). However, there is some concern that reducing cover too low may invite invasion by nonnative plant species. The drivers related to this habitat conservation goal (Fig. 5) identified in the STM (Fig. 4) are time since flood event, soil texture, and herbivory. Some of the constraints to both monitoring and attaining this goal are the size of the site (large sites may be too costly to monitor through ground based transects), the limited ability to manipulate hydrology and soil texture, and potential invasion by nonnative species. Thus, one of the questions that may help address the constraint of site size is, can the level of cover of intermediate RAFFS be determined by imagery obtained by UAV? Managers are also interested in how much cover should be reduced (20-30% or 30-40%) in order to promote occupancy by the species of concern.

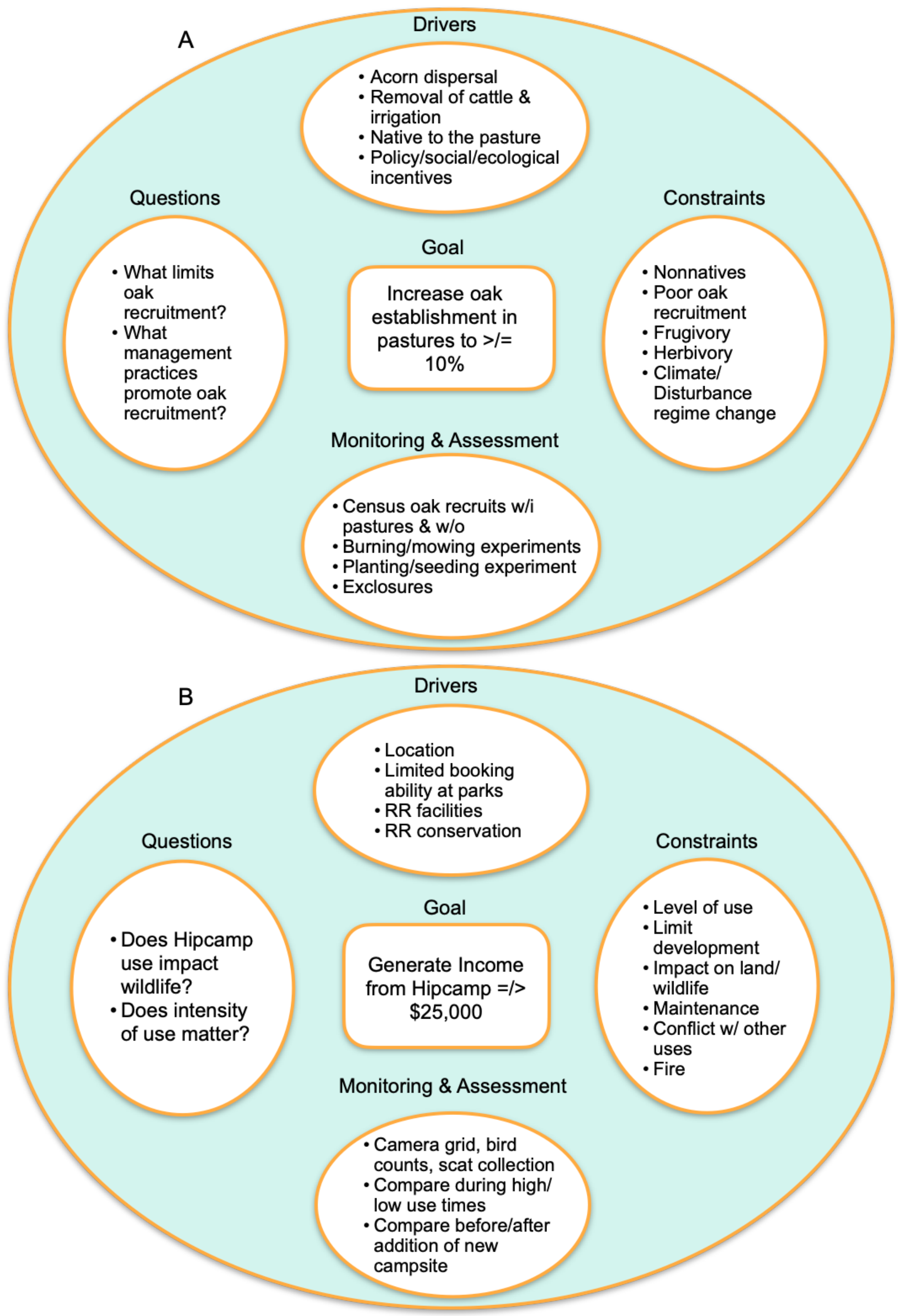


Fig. 1.6. Two Objective Oriented Goals developed with stakeholders for River Ridge Ranch. A) represents a more traditional-type restoration goal with its focus on the reestablishment of native species and understanding the limitations to reestablishment. B) demonstrates the L-TEAM framework's ability to incorporate goals related to the reconciliation of sustainable land use with restoration and conservation goals with a focus on income procurement through Hipcamp (a service for booking campsites on private lands; https://www.hipcamp.com/en-US) while balancing impacts to wildlife and habitat.

River Ridge OOGs

With River Ridge Ranch we present two OOGs, one that relates to more traditional restoration goals and one that balances income procurement with restoration goals. As part of the state's iconic oak woodland savanna ecosystem a key goal of the Ranch is the conservation and restoration of oaks - especially in the pasture areas where they were removed. These areas likely supported open oak savannas with an understory of native grasses and forbs (Whipple, Grossinger, and Davis 2011). Working with the land manager and researchers from California State University Long Beach, University of California Riverside and the California Native Plant Society, one goal on the Ranch is to increase oak establishment to at least 10% (A in Fig. 6). Along with the indication that oaks were present in the pasture lands (habitat is suitable), other drivers include statewide initiatives to restore and protect native oak trees, the removal of cattle from the ranch and cessation of irrigation to the lower pastures, active acorn dispersal by wildlife and the social/aesthetic desirability of oak restoration. Constraints include the presence of nonnative grasses and forbs, poor recruitment, predation by frugivores and herbivores, and climate and disturbance regime changes. The questions identified are: 1) What is limiting oak recruitment? and 2) What management practices promote recruitment?

While restoration is a main goal at River Ridge Ranch, sustainable income streams are also important. This second OOG in particular demonstrates how the L-TEAM framework can be used to help balance the need for sustainable use with conservation and restoration goals in a working landscape. One income stream that has proven profitable at River Ridge Ranch is Hipcamp, a booking company (hipcamp.com) that connects users with tent and RV camping, cabins, and “glamping” (glamor camping) in the United States, Australia, and Canada. Working with multiple stakeholders to develop this OOG, a goal identified is for Hipcamp bookings to generate an income of at least $25,000 (B in Fig. 6). The drivers include River Ridge’s location close to iconic state and national parks (where campgrounds are often overbooked and unavailable in peak season), the facilities, both natural (river access, extensive hiking trails, natural scenery and wildlife) and constructed (easily accessible by vehicle, electricity, wifi, kitchen, showers, fire pits, cabins, restrooms), and the conservation and restoration work being done at River Ridge Ranch. The constraints include the need to limit the level of use (so as not to adversely impact the wildlife and ecosystems), limits to infrastructure development as stipulated under the easement, required maintenance, potential conflicts with other uses, and the potential for wildfires. The questions identified are: 1) Does Hipcamp use impact wildlife and 2) does intensity of use by Hipcampers matter?

Experiments

The OOGs helped determine what needs to be monitored to answer the questions and thus helped guide experiments designed to gather those data. We illustrate this with examples for each case study.

Based on the Cajon Creek OOG (Fig. 5), experiments were designed to assess whether UAV monitoring can help identify the different cover ranges in the intermediate RAFFS cover phase of the RAFFS state and whether cover reduction by manual removal promotes SBKR occupancy. UAV flights encompassing restoration sites were established and are to be flown annually. The UAV flight path includes treatment sites that were established in intermediate RAFSS habitats consisting of two levels of cover reduction by manual removal – to 20-30% and 30-40% – and control plots (Fig. 5). The UAV imagery includes visible (red-green-blue) and near-infrared (NIR) wavelength bands in order to calculate the Normalized Difference Vegetation Index (NDVI) from red and NIR reflectance, and enhance detection of green vegetation (Tucker 1979). Imagery is being assessed to determine which spectral information is best able to discriminate the different cover amounts (Fig. 4). Furthermore, taking advantage of a previously established trapping grid for SBKR, treatment sites were located to incorporate portions of the trapping grid -- thus, SBKR occupancy can be compared between treatment plots, and to control (untreated) areas located outside of the treatment sites.

For the first River Ridge OOG (Fig. 6A) aimed at the reestablishment of oaks, oak recruits will be censused both within pasture areas and in nearby oak savanna/woodlands along established transects to understand what may be inhibiting the recruitment of oaks and what management practices would improve reestablishment. Experimental plots are being established where treatments consisting of mowing, burning, and control subplots will help assess techniques to improve establishment. Another set of experimental plots will include the planting of seedlings, sowing of seeds, and control subplots, along with random assignment of herbivore exclosures. Transects and plots will be censused annually.

To assess the second River Ridge Ranch OOG (Fig. 6B), which is to generate income through Hipcamp with minimal impact to wildlife and habitat, we were able to incorporate previously established monitoring efforts. These include: A camera trap grid, annual bird counts, and systematic seasonal scat collection to support dietary assessments of canids. The COVID-19 pandemic provided a unique opportunity, as data collection continued while the Ranch was closed to Hipcampers for several months, and then reopened at reduced capacity. This will allow for comparison of wildlife use during high and low bookings by Hipcampers. We also have some data prior to the addition of a new campsite and bunkhouses which we can compare with data to be collected in the future. We have detailed booking and income information through the Hipcamp platform and Ranch owner's records. Thus, we can determine if there are changes in wildlife use of the site with changes in Hipcamp booking patterns and infrastructure development.

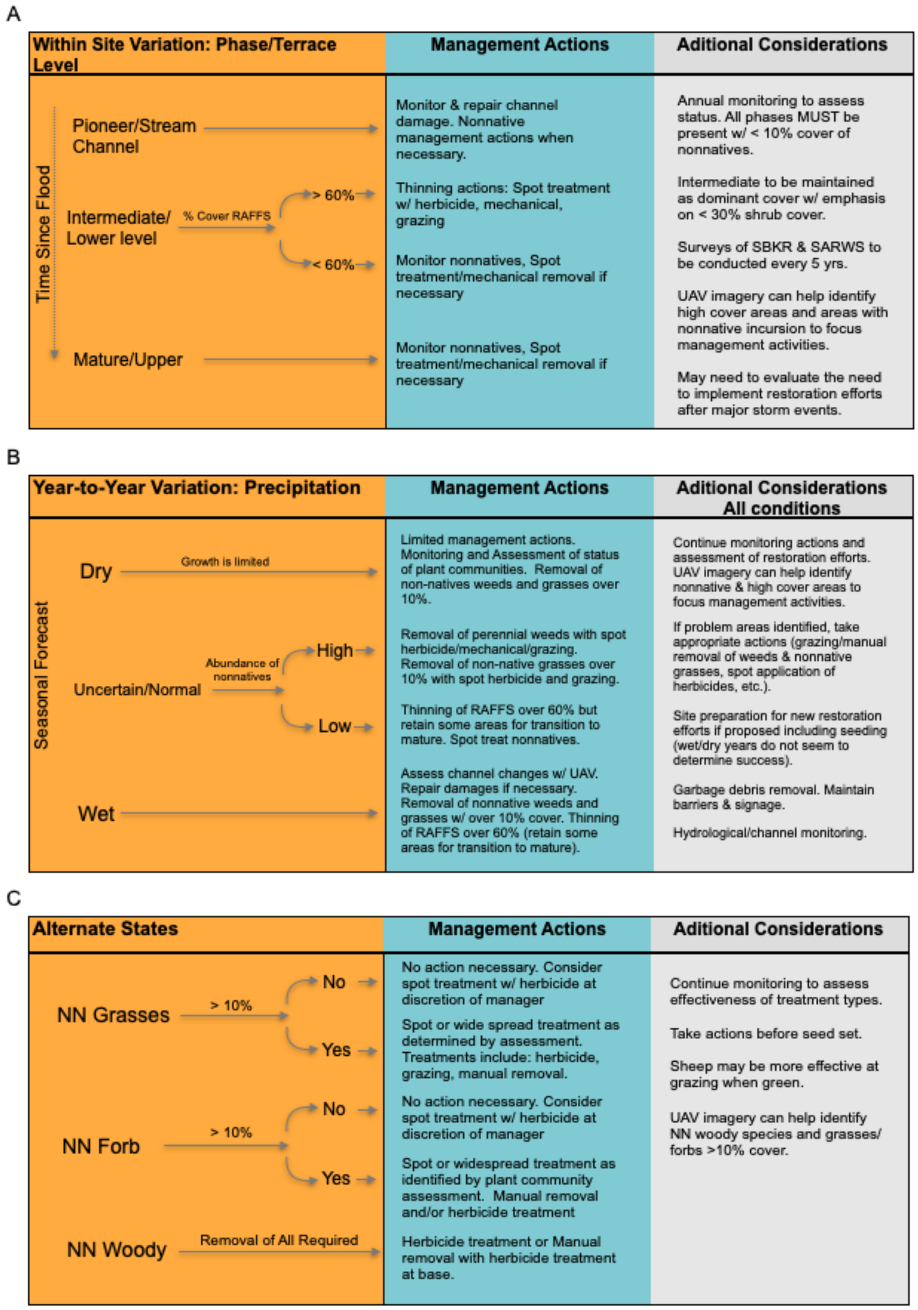


Fig. 1.7. A Decision support tool developed with stakeholders for the Cajon Creek Conservation Area. Management goals and the system's STM informed the design of the Decision Support Tool to help guide management actions and ensure management continuity. Thresholds are included that trigger management actions. The tool describes the different management actions necessary for (A) the particular phase/habitat under consideration, (B) responding to variation in annual precipitation (as precipitation and flood events are system drivers), and (C) the functional group of nonnatives present.

Decision Support Tools

To demonstrate the applicability of Decision Support Tools and their ability to support management decisions and identify triggers, we present a Decision Support Tool developed for the Cajon Creek Conservation Area. The STM (Fig. 4) and objective oriented management goals (Fig. 5) framed Decision Support Tool development to help management actions and ensure management continuity (Fig. 7). Thresholds were identified that trigger management actions (e.g. nonnative cover of greater than 10%). The three parts of this Decision Support Tool correspond to goals, drivers and constraints defined in the OOG: 1) the particular RAFFS phase for the habitat under consideration; 2) the annual precipitation (as precipitation and flood events are system drivers); and 3) the functional group of nonnative herbaceous plants present (broadleaf forbs versus graminoids). For example, if intermediate RAFFS has more than 60% cover, thinning actions are to be implemented. During a wet year, the channel should be assessed, UAV imagery should be used to identify flushes of nonnative herbaceous vegetation with a cover of over 10%, and nonnative removal should be initiated before seed set. As new information is obtained the decision support tool can be updated, thus allowing for data collected through long-term monitoring to directly influence management actions.

Discussion

Long-term management and monitoring of restoration and conservation at the landscape-level is challenging, and L-TEAM helps address challenges related to defining goals, formulating questions, designing monitoring experiments and adapting to change. As demonstrated with the case studies, the L-TEAM framework can be applied to projects of various sizes and locations, with heterogeneous habitats, various stakeholders, and multiple management goals. The L-TEAM framework: 1) clarifies understanding of ecosystem function and drivers through the use of STMs, 2) assists stakeholders, through the use of OOGs, in establishing clear, actionable goals that can be assessed through the development of rigorous experiments, the results of which inform further understanding of system function and drivers; these can then be used to, 3) develop DSTs to support management decisions and actions and ensure continuity. However, I will address some challenges that remain and future applications and improvements of the L-TEAM framework should consider ways to mitigate these challenges.

It is essential that communication among stakeholders is maintained -- especially among land managers and researchers. Input from landowners/managers must inform both the development and implementation of the framework (Sterling et al. 2017; Dale et al. 2019). The process should be iterative to assure that goals are clear and agreed upon, implementation of management actions and experiments are accurate, and results are understood. While L-TEAM is designed with the facilitation of clear, open, and iterative communication in mind, it remains the responsibility of the stakeholders to put this communication into practice. This requires regular interactions, especially during development. However, as time passes, it is also important that communication channels remain open, especially when there is turnover in individuals in an organization. Easy, live access to L-TEAM documentation and regular review may further ensure clear communication.

While L-TEAM is adaptive, uncertainty still looms large. Project goals may change radically in the face of unexpected disturbance, climate change, or sale of the land and/or management change. Although L-TEAM is designed to be updated and adapted to reflect these changes, it has yet to be proven under such circumstances. However, these uncertainties also bring with them opportunities for further understanding of how to manage and monitor conservation and restoration in heterogeneous landscapes under uncertainty. The L-TEAM framework, with STM and OOG development, would be well suited to help in reaching this understanding.

The development of new technologies for monitoring and management is both exciting and challenging. It is important when considering the adoption of new technologies to be cognizant of how this will affect the continuity of data collection and analyses moving forward. While it may be tempting to implement the latest innovations (e.g. UAV-borne imagery, field-based environmental sensors, machine learning for data mining), maintaining the integrity of the monitoring and assessment program should take precedence. Furthermore, new technologies can often be expensive and require new infrastructure and training. The L-TEAM framework's emphasis on objective oriented goals and the identification of key questions and design of experiments to answer questions and assess goal attainment should help in determining when, and if, new technologies should be adopted, and in fact, can help determine if they are beneficial.

Related to issues with technological advancements are those concerning data management, storage and curation (Michener 1997; Likens and Lindenmayer 2018). Long-term management and monitoring can result in copious amounts of data. While L-

TEAM advocates for data collection, it does not necessarily include provisions for how these data should be managed. The ability to organize, store and readily access large amounts of data is improving and becoming more affordable, however it can still prove prohibitive for some projects and organizations. Moreover, special training may be required and not all organizations have the team members or resources for such training. Best practices are available (Borghi et al. 2018; Aubin et al. 2020) and, when possible, at least one project member should be familiar with these practices. This problem is not unique to restoration and conservation in heterogeneous landscapes. However, L-TEAM's emphasis on clearly identifying specific factors to monitor should help - to some extent - by tempering the tendency to monitor a "blizzard of details" (Likens and Lindenmayer 2018) thus generating an abundance of irrelevant data. Additionally, L-TEAM's documentation of experiments should ensure that protocols for data collection maintain integrity unless stakeholders agree upon changes. Moving forward, the L-TEAM framework may be improved by including consideration for data management and a way to clearly articulate standards for data collection, curation, and best practices.

Despite these challenges, L-TEAM remains a promising step toward a more systematic approach to the long-term management and monitoring of restoration and conservation at the landscape-level. L-TEAM takes an adaptive approach and combines STMs, objective oriented goals, and decision support tools into a framework that can help scientists and managers design a long-term monitoring and management plan for landscape restoration projects. Future evaluation of L-TEAM should include its application in different systems. L-TEAM also lends itself well to the incorporation of diverse stakeholder knowledge and management strategies - for example it has the potential to include indigenous knowledge and management practices (Anderson and Barbour 2003; Reyes-García et al. 2019). It is especially well suited to capture the complexity inherent in landscape-level projects, which includes habitat heterogeneity, multiple land uses and land use histories, and multiple management goals. L-TEAM guides the establishment of sound experiments to answer well defined questions that will help not only in improving management of complex systems but in our overall understanding of how these systems function and their responses to management and restoration actions.

Works Cited

Anderson, M. Kat, and Michael G. Barbour. 2003. "Simulated Indigenous Management: A New Model for Ecological Restoration in National Parks." *Ecological Restoration* 21 (4): 269–77.

Aronson, James, Neva Goodwin, Laura Orlando, Cristina Eisenberg, and Adam T. Cross. 2020. "A World of Possibilities: Six Restoration Strategies to Support the United Nation's Decade on Ecosystem Restoration." *Restoration Ecology* 28 (4): 730–36.

Aubin, Isabelle, Françoise Cardou, Laura Boisvert-Marsh, Eric Garnier, Manuella Strukelj, and Alison D. Munson. 2020. "Managing Data Locally to Answer Questions Globally: The Role of Collaborative Science in Ecology." *Journal of Vegetation Science: Official Organ of the International Association for Vegetation Science* 31 (3): 509–17.

Bashari, H., C. Smith, and O. J. H. Bosch. 2008. "Developing Decision Support Tools for Rangeland Management by Combining State and Transition Models and Bayesian Belief Networks." *Agricultural Systems* 99 (1): 23–34.

BenDor, Todd, T. William Lester, Avery Livengood, Adam Davis, and Logan Yonavjak. 2015. "Estimating the Size and Impact of the Ecological Restoration Economy." *PloS One* 10 (6): e0128339.

Berkes, F., J. Colding, and C. Folke. 2008. "Navigating Social-Ecological Systems: Building Resilience for Complexity and Change."

Berkes, F., C. Folke, and J. Colding. 2000. "Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience."

Bestelmeyer, B. T., A. Ash, and J. R. Brown. 2017. "State and Transition Models: Theory, Applications, and Challenges." *Rangelands*.

Borghi, John, Stephen Abrams, Daniella Lowenberg, Stephanie Simms, and John Chodacki. 2018. "Support Your Data: A Research Data Management Guide for Researchers." *Research Ideas and Outcomes* 4 (May): e26439.

Burger, L. Wes, Jr., Kristine O. Evans, Mark D. McConnell, and Leslie M. Burger. 2019. "Private Lands Conservation: A Vision for the Future." *Wildlife Society Bulletin* 43 (3): 398–407.

Chock, Rachel Y., Sarah McCullough Hennessy, Thea B. Wang, Emily Gray, and Debra M. Shier. 2020. "A Multi-Model Approach to Guide Habitat Conservation and Restoration for the Endangered San Bernardino Kangaroo Rat." *Global Ecology and Conservation* 21 (March): e00881.

Clancy, Niall G., John P. Draper, J. Marshall Wolf, Umarfarooq A. Abdulwahab, Maya C. Pendleton, Soren Brothers, Janice Brahney, Jennifer Weathered, Edd Hammill, and Trisha B. Atwood. 2020. "Protecting Endangered Species in the USA Requires Both Public and Private Land Conservation." *Scientific Reports* 10 (1): 11925.

Dale, Virginia H., Keith L. Kline, Esther S. Parish, and Sarah E. Eichler. 2019. "Engaging Stakeholders to Assess Landscape Sustainability." *Landscape Ecology* 34 (6): 1199–1218.

Derner, Justin D., Bob Budd, Grady Grissom, Emily J. Kachergis, David J. Augustine, Hailey Wilmer, J. Derek Scasta, and John P. Ritten. 2021. "Adaptive Grazing Management in Semiarid Rangelands: An Outcome-Driven Focus." *Rangelands*, April. https://doi.org/10.1016/j.rala.2021.02.004.

Eastburn, Danny J., Anthony T. O'Geen, Kenneth W. Tate, and Leslie M. Roche. 2017. "Multiple Ecosystem Services in a Working Landscape." *PloS One* 12 (3): e0166595.

Franklin, Janet, Helen M. Regan, Lauren A. Hierl, Douglas H. Deutschman, Brenda S. Johnson, and Clark S. Winchell. 2011. "Planning, Implementing, and Monitoring Multiple-Species Habitat Conservation Plans." *American Journal of Botany* 98 (3): 559–71.

Hanes, Ted L., Richard D. Friesen, and Kathy Keane. 1989. "ALLUVIAL SCRUB VEGETATION IN COASTAL SOUTHERN CALIFORNIA1." In *CALIFORNIA RIPARIAN SYSTEMS CONFERENCE*, 187.

Heemskerk, Marieke, Karen Wilson, and Mitchell Pavao-Zuckerman. 2003. "Conceptual Models as Tools for Communication Across Disciplines." *Conservation Ecology* 7 (3).

Herrick, Jeffrey E., Michael C. Duniway, David A. Pyke, Brandon T. Bestelmeyer, Skye A. Wills, Joel R. Brown, Jason W. Karl, and Kris M. Havstad. 2012. "A Holistic Strategy for Adaptive Land Management." *Journal of Soil and Water Conservation* 67 (4): 105A – 113A.

Holl, Karen D., Elizabeth E. Crone, and Cheryl B. Schultz. 2003. "Landscape Restoration: Moving from Generalities to Methodologies." *Bioscience* 53 (5): 491–502.

Hood, W. G., K. Blauvelt, D. L. Bottom, J. M. Castro, G. E. Johnson, K. K. Jones, K. L. Krueger, R. M. Thom, and A. Wilson. 2021. "Using Landscape Ecology Principles to Prioritize Habitat Restoration Projects across the Columbia River Estuary." *Restoration Ecology*, no. rec.13519 (August). https://doi.org/10.1111/rec.13519.

Huntsinger, L., and J. W. Bartolome. 1992. “Ecological Dynamics of Quercus Dominated Woodlands in California and Southern Spain: A State-Transition Model.” *Vegetatio* 99-100 (1): 299–305.

Huntsinger, L., and N. F. Sayre. 2007. “Introduction: The Working Landscapes Special Issue.” *Rangelands*.

Likens, G., and D. Lindenmayer. 2018. “Effective Ecological Monitoring.”

Lindenmayer, David B., and Gene E. Likens. 2009. “Adaptive Monitoring: A New Paradigm for Long-Term Research and Monitoring.” *Trends in Ecology & Evolution* 24 (9): 482–86.

Lindenmayer, D. B., and G. E. Likens. 2010. “The Science and Application of Ecological Monitoring.” *Biological Conservation*.

Lockhart, Sharon, and Douglas W. Sprauge. 1999. “Case Study: Cajon Creek Habitat Conservation Management Area.” *Transactions of the Western Section of the Wildlife Society* 35: 57–62.

Méndez-Toribio, Moisés, Cristina Martínez-Garza, and Eliane Ceccon. 2021. “Challenges during the Execution, Results, and Monitoring Phases of Ecological Restoration: Learning from a Country-Wide Assessment.” *PloS One* 16 (4): e0249573.

Merenlender, A. M., L. Huntsinger, G. Guthey, and S. K. Fairfax. 2004. “Land Trusts and Conservation Easements: Who Is Conserving What for Whom?” *Conservation Biology: The Journal of the Society for Conservation Biology* 18 (1): 65–76.

Michener, William K. 1997. “Quantitatively Evaluating Restoration Experiments: Research Design, Statistical Analysis, and Data Management Considerations.” *Restoration Ecology* 5 (4): 324–37.

Owley, Jessica, and Adena R. Rissman. 2016. “Trends in Private Land Conservation: Increasing Complexity, Shifting Conservation Purposes and Allowable Private Land Uses.” *Land Use Policy* 51 (February): 76–84.

Phelps, D. G., and O. J. H. Bosch. 2002. “A Quantitative State and Transition Model for the Mitchell Grasslands of Central Western Queensland.” *Rangeland Journal* 24 (2): 242–67.

Reyes-García, Victoria, Álvaro Fernández-Llamazares, Pamela McElwee, Zsolt Molnár, Kinga Öllerer, Sarah J. Wilson, and Eduardo S. Brondizio. 2019. “The Contributions of Indigenous Peoples and Local Communities to Ecological Restoration." *Restoration Ecology* 27 (1): 3–8.

Roesch-McNally, Gabrielle E., Sarah Wiener, Julian Reyes, Caitlin M. Rottler, Jennifer Balachowski, and Rachel E. Schattman. 2021. "Supporting Practitioners in Developing Effective Decision Support Tools for Natural Resource Managers." *Journal of Soil and Water Conservation* 76 (4): 69A – 74A.

Spiegal, Sheri, James W. Bartolome, and Michael D. White. 2016. "Applying Ecological Site Concepts to Adaptive Conservation Management on an Iconic Californian Landscape." *Rangelands* 38 (6): 365–70.

Spiegal, Sheri, Loralee Larios, James W. Bartolome, and Katharine N. Suding. 2014. *Restoration Management for Spatially and Temporally Complex Californian Grassland*. escholarship.org.

Sterling, Eleanor J., Erin Betley, Amanda Sigouin, Andres Gomez, Anne Toomey, Georgina Cullman, Cynthia Malone, et al. 2017. "Assessing the Evidence for Stakeholder Engagement in Biodiversity Conservation." *Biological Conservation* 209 (May): 159–71.

Stringham, Tamzen K., William Clement Krueger, Patrick L. Shaver, and Others. 2001. "States, Transitions, and Thresholds: Further Refinement for Rangeland Applications."

Stringham, Tamzen K., William C. Krueger, and Patrick L. Shaver. 2003. "State and Transition Modeling: An Ecological Process Approach." *Rangeland Ecology & Management/Journal of Range Management Archives* 56 (2): 106–13.

Tucker, C. J. 1979. "The NDVI Has Been Proven to Be Well Correlated with Various Vegetation Parameters, such as Green Biomass." *Rouse et Al*, 605–790.

Vos, P., E. Meelis, and W. J. Ter Keurs. 2000. "A Framework for the Design of Ecological Monitoring Programs as a Tool for Environmental and Nature Management." *Environmental Monitoring and Assessment* 61 (3): 317–44.

Walker, Brian, C. S. Holling, Stephen R. Carpenter, and Ann Kinzig. 2004. "Resilience, Adaptability and Transformability in Social–ecological Systems." *Ecology and Society* 9 (2).

Western, David, Peter Tyrrell, Peadar Brehony, Samantha Russell, Guy Western, and John Kamanga. 2020. "Conservation from the inside-out: Winning Space and a Place for Wildlife in Working Landscapes." *People and Nature* 2 (2): 279–91.

Westoby, Mark, Brian Walker, and Imanuel Noy-Meir. 1989. "Range Management on the Basis of a Model Which Does Not Seek to Establish Equilibrium." *Journal of Arid Environments* 17 (2): 235–39.

Whipple, A. A., R. M. Grossinger, and F. W. Davis. 2011. "Shifting Baselines in a California Oak Savanna: Nineteenth Century Data to Inform Restoration Scenarios." *Restoration Ecology*. https://doi.org/10.1111/j.1526-100X.2009.00633.x.

Ch.2 A regional assessment of California woodlands historical and modern fire severity and vegetation trends

Abstract: Fire, both natural and anthropogenic, is a key driver in ecosystems worldwide. In much of the arid Western United States, fire frequency, size, and intensity have increased in the 21st century and have become devastating economically, socially, and ecologically. In California forests, research has shown that decades of fire suppression since European settlement have resulted in shifts from historic fire regimes of frequent, small, low-intensity burns to high severity crown fires fueled by the accumulation of ladder fuels during fire suppression. This has led to large, high-severity crown fires fueled by those ladder fuels. However, how post-European settlement fire management practices have impacted fire regimes in California's lower-elevation woodlands, some of the most extensive and iconic of California's landscapes, has received less attention. Prior to European settlement, lower-elevation woodlands are believed to have also been shaped by frequent, low-intensity surface-fires that reduced shrub and sapling cover, rejuvenated herbaceous plants, and maintained the open canopies characteristic of these landscapes. My research aimed to determine if, and how far, modern burn severity distributions have departed from historical distributions. Additionally, I determined rates of land cover conversion and shifts in vegetation successional states. I show that much of the state's woodland regions are outside the range of their historic (pre-European) fire regimes but that this varies by region, fire severity class, and whether we considered extent or proportion of vegetation types and age classes. Vegetation has also shifted, dominated perhaps not surprisingly, by the establishment of exotic annual plant species in the understory, but also by early successional states. The once-dominant later and more open-

canopy states have decreased. However, this varies by region. Conservation and restoration of these landscapes, as well as the safety of millions of people living within them, requires a re-evaluation of current fire management practices with a focus on restoring frequent, low intensity surface-fires and/or thinning activities.

Introduction

Fire is a key component shaping many landscapes. Variation in fire severities and extent result in shifting mosaics of vegetation in different successional stages and cover classes. This heterogeneity is essential to landscape and ecosystem identity and resilience (Bond and Keeley 2005; Safford and Stevens 2017; He, Lamont, and Pausas 2019). Changes in fire regimes driven by anthropogenic practices of fire suppression and altered ignition patterns as well as land use that alters vegetation, often result in changes in these patterns. These changes degrade ecosystem integrity resulting in economic, social, and ecological consequences (Krofcheck et al. 2017; Syphard, Keeley, and Abatzoglou 2017; He, Lamont, and Pausas 2019). While research has been directed at understanding the extent and consequences of these changes in forest ecosystems, less has focused on woodland systems (Stahle et al. 2013; Temperton et al. 2019). In California, lower elevation (foothill) woodland systems comprise approximately 20 million acres, are home to millions of people, and to many iconic and endemic species (Tietje and Vreeland 1997; Zack et al. 2005; George and Alonso 2008; Stahle et al. 2013). Understanding shifts in fire regimes and the potential shifts in landscape composition in foothill regions is essential to improve the management and resilience of these systems under the pressures of further development, land use change, and a changing climate.

Anthropogenic changes in fire regimes, including changes to fuel structure and ignition patterns, and to timing, size, location and frequency via fire suppression, present challenges to the management and provision of ecosystem goods and services and to the conservation and restoration of biodiversity (Noss et al. n.d.; D'Antonio and Vitousek 1992; Syphard, Keeley, and Abatzoglou 2017). Fire regimes describe the spatial, temporal, and magnitudinal characteristics of fire that are particular to a landscape (Minnich 1983; R. D. Haugo et al. 2019). Altered fire regimes often result in changes in ecological processes and cycles, such as carbon and nutrient cycling, changes in vegetation patterns, structure, and composition, and to overall biodiversity (Krofcheck et al. 2017; He, Lamont, and Pausas 2019; Falk et al. 2022). Moreover, landscapes already experiencing stress from altered fire regimes are likely more vulnerable to climate-driven changes (Hessburg et al. 2019; Das et al. 2020).

Fire regimes tend to be landscape-scale phenomena – they have broad spatial and temporal effects on ecosystems. Because restoration and conservation of fire-prone landscapes requires us to consider changes to historic fire regimes, taking a landscape to regional perspective is necessary (Van de Water and Safford 2011). This allows us to evaluate the overall magnitude of changes in fire regimes, the effects of these changes on broad-scale vegetation patterns, and to assess implications for future management (Haugo 2019).

Some argue that, in the face of climate and other anthropogenic changes, trying to restore or conserve historic conditions may be impossible or even potentially undesirable (Millar, Stephenson, and Stephens 2007; Shive et al. 2018). However, historic disturbance regimes shaped current systems and provided a range of conditions that resulted in system heterogeneity and resilience. Identifying the historic variation in these drivers and the system's responses helps identify what a particular system is adapted to, the inherent variability of the system, and how it may respond to future changes (McGarigal et al. 2018; Safford and Stevens 2017; Keane and Loehman 2019; Swaty et al. 2021). Quantifying the difference between current and pre-settlement fire extent and severities can help us identify areas at risk of conversion (changes from one vegetation type to another) and other vegetational shifts due to altered fire regimes (Van de Water and Safford 2011; McGarigal et al. 2018; Keane and Loehman 2019).

In fire-prone forests, where low-severity, surface fires were frequent, there is concern that modern fire suppression practices and the increase in the size and frequency of more severe wildfires may decrease forest resilience and increase the rates forests are transitioning to non-forest ecosystems (Hessburg et al. 2019; Tepley et al. 2018; Serra-Diaz et al. 2018; R. D. Haugo et al. 2019). Like forests, fire plays a major role in the structure and composition of dry woodlands. But unlike forests, fire in woodlands may be key to the maintenance of the more open canopy structure characteristic of woodlands. Frequent, low-intensity fires tend to clear understory shrubs and younger trees, potentially reducing competition for water and resources resulting in more widely spaced, healthier, more resilient mature trees – key for systems which may be more prone to longer, dryer drought brought on by a warming climate (Crockett and Westerling 2017; Zald et al. 2022).

Studies in forests have found that with the arrival of Europeans in the western USA, the interactions of settlement, grazing, mining, logging, and fire exclusion have altered historic fire regimes, resulting in changes to the shifting mosaic of vegetation patterns (Knapp et al. 2013; Mallek et al. 2013; Stephens et al. 2015; Hagmann et al. 2021). Mallek et al (2013) found that rates of burning in Sierra Nevada forests for low to moderate severity fires are "far below" pre-settlement levels and Haugo et al. (R. Haugo et al. 2015) found that, in the forests of Oregon, later and more closed-canopy vegetation stages were overrepresented while early and more open-canopied late vegetation stages were underrepresented. But how have post-European development and management practices influenced fire regimes and vegetation patterns in California foothill woodlands?

Prior to intensive European settlement, various records indicate California foothill landscapes were composed primarily of oak woodlands with an open "park-like" structure (Allen-Diaz and Standiford 2007; Klimaszewski-Patterson et al. 2018; Klimaszewski-Patterson, Morgan, and Mensing 2021). Fire scars confirm that this was likely the result of frequent, low-intensity fires (Standiford, Phillips, and McDougald 2012). Unlike in higher elevation montane forests, lightning-caused fires are believed to be only a small component of historic fire regimes, with decades passing between events (Sugihara et al. 2006). Extensive evidence demonstrates that the small, frequent, surface-fires that shaped these landscapes were the result of Native American stewardship

(Anderson 2007; Lightfoot et al 2013; Klimaszewski-Patterson et al 2021). Burning was used to enhance the habitat for important resource species, hunting, travel, and protection (Anderson 2006; Klimaszewski-Patterson et al. 2018). Indeed, a global study has found that humans use fire as a management method in fire-prone landscapes precisely because they are fire prone (Coughlan, Magi, and Derr 2018). Frequent, low-severity surface burning reduces the potential for larger, catastrophic fires, particularly in forests (Coughlan, Magi, and Derr 2018; Bond and Keeley 2005). While it is difficult to "separate" the role of natural and anthropogenic fires to determine the natural range of variation in California woodland fire regimes, it is evident that these landscapes are well adapted to regular, low-intensity fires (R. Standiford et al. 1997; Allen-Diaz and Standiford 2007). Thus, I instead follow Keane (2009) and McGarigal (2018) and use the term historical range of variation (HRV) to acknowledge that this landscape evolved under the influence of both natural and anthropogenic fires.

To determine whether contemporary fire trends are outside of the historic ranges of variation to which California foothill woodland ecosystems are adapted, I compared modern trends in fire burn severity to expected historic fire burn severity distributions. Advances in simulation modeling (Rollins 2009; Blankenship et al. 2021) and in consistently quantifying modern burn severity (Eidenshink et al. 2007) provide an opportunity to compare modern fire regimes to the period prior to intensive European settlement (R. D. Haugo et al. 2019). Due to the extent and variation of California foothill woodlands I also explored the potential for regional differences between these patterns for two different foothill woodland regions – those of the California Sierra Nevada and the California Northern Interior Coast Ranges (See figure1). Finally, to demonstrate how changes in fire regimes influence patterns in vegetation, I hypothesized that vegetation in these regions will have shifted from middle and late vegetation stages with more open canopy cover to those with more closed canopy covers. To test this hypothesis, I determined the extent of conversion and the departure from historical ranges of each region's vegetation structure and cover. Again, I also looked at regional differences in rates of conversion and departure.

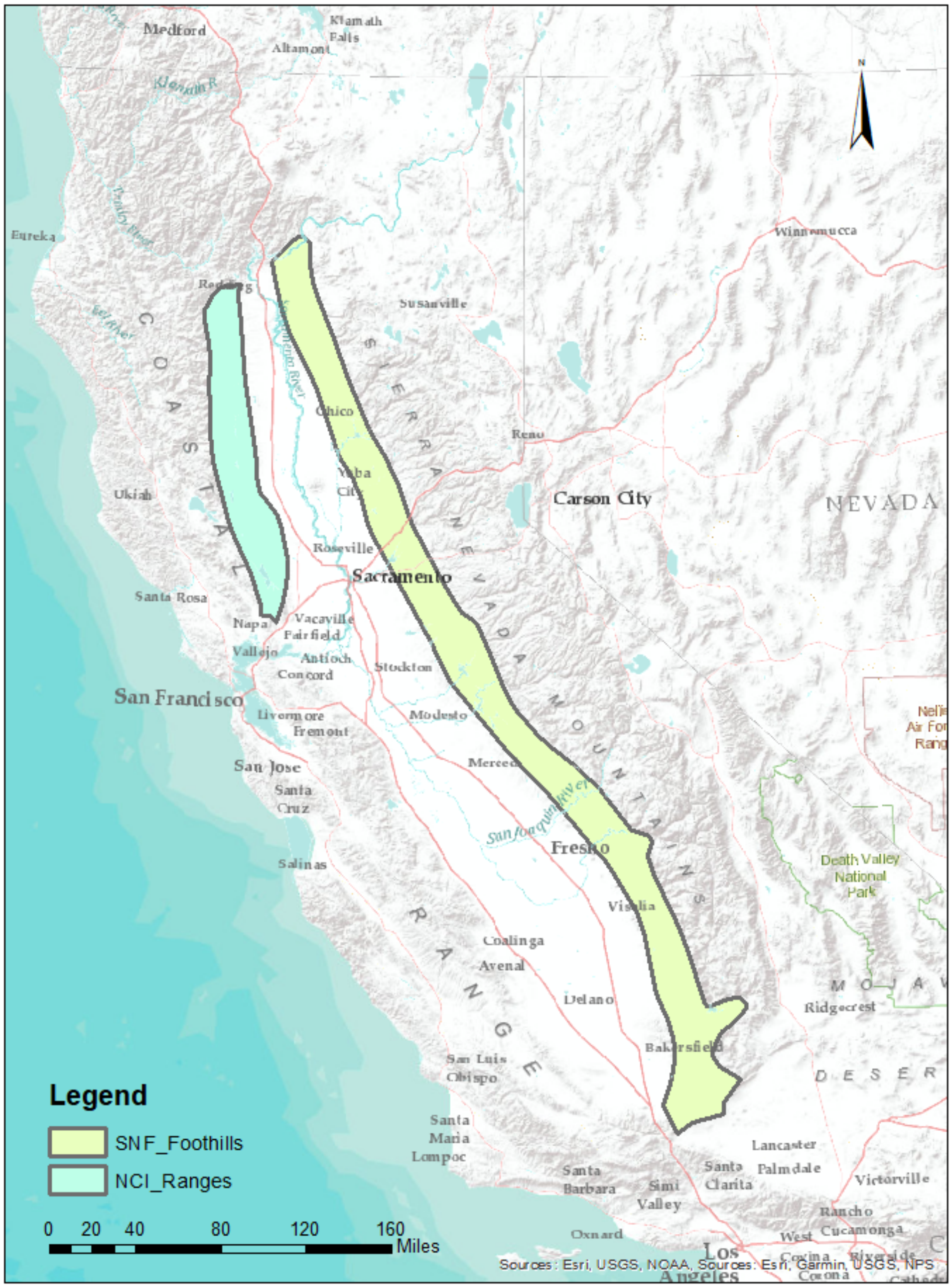


Fig. 2.1. Sierra Nevada Foothills and Northern California Interior Coast Ranges Ecoregions of California. Both regions border California's great central valley and are dominated by oak woodland systems.

Methods

I compared modern (1984-2019) burn severity trends and current vegetation cover trends for two regions of California valley foothill woodlands, the Northern California Interior Coast Ranges (NCI) and the Sierra Nevada Foothills (SNF; see figure 1). These regions correspond to Bailey's ecoregion sections and are part of the same province (Bailey 1998). California foothill woodlands are some of the most extensive landscapes in California. Elevation ranges from approximately 1,500 feet to 6,000 feet depending upon latitude. They experience California's Mediterranean climate with hot dry summers and cool, moist winters. These landscapes are home to some of the highest biodiversity in the state and house many endemic species (George and Alonso 2008). The dominant vegetation of these regions includes some of California's most iconic oaks, Blue oaks (*Quercus douglasii*) Valley oaks (*Q. lobata*), Interior Live oaks (*Q. wislizenii*), Coast live oaks (*Q. agrifolia*) and at higher elevations, Black oak (*Q. kellogii*). These landscapes have also been subject to a host of anthropogenic modifications and degradation including mining, ranching and associated housing and infrastructure development. Although there is much similarity between the regions, especially with regard to native vegetation, there are some differences.

The Sierra Nevada foothill woodlands occur along a thin strip separating the Sierra Nevada mountains in the East from the great Central Valley in the West (light green region figure 1). This region is considered one of the most imperiled regions in California, with the majority of lands in private ownership (Bernhardt and Swiecki 2001;

Zack et al. 2005) and much of the area within commuting distance of major city centers in the central valley. Pressures for further development of the region loom large with this area projected to have some of the largest development rates statewide (Alagona 2008). This threatens the vegetation and biodiversity of the region both directly through conversion, and indirectly through altered fire regimes.

The Northern California Interior Coast Ranges foothills lie West of the Central Valley and East of the Coastal Ranges (light blue region figure 1). While cattle grazing dominated the Sierra Nevada foothills, by the 1870's, in this region, sheep grazing became more prevalent (Tehama County Resource Conservation District, 2006). Active fire suppression in this region during the last century has contributed to the accumulation of fuels and trends towards larger, more devastating fires (Arno and Allison-Bunnell, 2002). This region is home to the remote Diablo Range which has been identified as a potential diversity and evolutionary hotspot (Kling et al. 2018).

Departure is a metric used as an indicator of landscape condition. Based upon reference conditions, it suggests the degree to which an observed set of classes or processes (i.e. fire severity classes or a shifting mosaic of vegetation stages) "is departed from" what one would expect given those historic conditions (Swaty et al. 2021). This allows for a quantitative evaluation of landscapes that focuses not only on the presence and extent of an ecosystem or process, but also on within system heterogeneity – a key component of ecosystem resilience (Swaty et al. 2021).

This project used datasets from two national multi-agency programs, Monitoring Trends in Burn Severity (MTBS; mtbs.gov) and LANDFIRE (landfire.gov). I obtained contemporary fire severity and extent data from the Monitoring Trends in Burn Severity (MTBS) program specific to our study areas from 1984 to 2019. MTBS is an interagency program (U.S. Geological Survey Center for Earth Resources Observation and Science, USDA Forest Service Geospatial Technology and Applications Center) established in 2005 to consistently map, document, and assess burn severities and extents of fires equal to or greater than 1,000 acres (500 acres in the east) across the United States at 30 meter resolution (Eidenshink et al. 2007). MTBS data are classified into low (surface), moderate (mixed), and high (replacement) severity classes, using NBR (Normalized Burn Ratio), dNBR (differenced Normalized Burn Ration) and RdNBR (Relative differenced Normalized Burn Ratio) data from prefire and postfire Landsat imagery assessed by an analyst with expertise in fire behavior and effects in a given ecological setting (mtbs.gov). NBR is a spectral index calculated from TM bands 4 and 7 as: (TM4 – TM7)/ (TM4 + TM7) where TM4 represents the near-infrared spectral range (0.76 μm to 0.90 μm) and TM7 the shortwave infrared spectral range (2.08 μm to 2.35 μm). Data are freely available and have been used for a variety of research and operational projects (Eidenshink et al. 2007).

I also used Biophysical settings (BPS) models to determine historical ranges in variation and spatial data layers for current vegetation cover developed by the LANDFIRE program (landfire.gov). The LANDFIRE program is a joint program managed by the U.S. Department of Agriculture Forest Service and the U.S. Department of the Interior.

BPS models represent the vegetation and disturbance processes that are believed to have been present on a particular landscape prior to European settlement. To develop BPS models, the program performed extensive literature and expert review processes as well as compiled several historical empirical data sources (e.g. pollen and charcoal in sediments, dendrochronological reconstructions, and historic survey records) (Rollins 2009). This information was then used to quantify fire regimes and vegetation patterns to estimate pre-European rates of succession and disturbance probabilities (Keane et al. 2002, 2006, 2007, Pratt et al. 2006, Rollins 2009, DeMeo et al. 2018, LANDFIRE 2018).

The BpS models are a combination of state and transition simulation models (STSM) designed to capture variability in ecosystem processes, and associated peer reviewed documents that describe the vegetation and disturbance regimes specific to the BpS and its model, including transition probabilities (Figure 2; (Rollins 2009; Blankenship et al. 2021). State and transition models, introduced by Westoby (1989), are non-equilibrium models developed to capture the complexity of vegetation dynamics by incorporating multiple successional pathways, multiple steady states, thresholds of change, and discontinuous and irreversible transitions (Westoby and Walker 1989; Bestelmeyer, Ash, and Brown 2017). STSMs operationalize state and transition models by using an adapted Markov chain approach to predict how vegetation transitions between states over time in response to interactions between succession, disturbances, and potential management actions (Daniel and Frid 2012). In STSMs time is represented in discrete steps, space as a set of discrete spatial units, change over time is represented as a stochastic process, and rates of change between states are expressed as probabilities (Daniel et al. 2016).

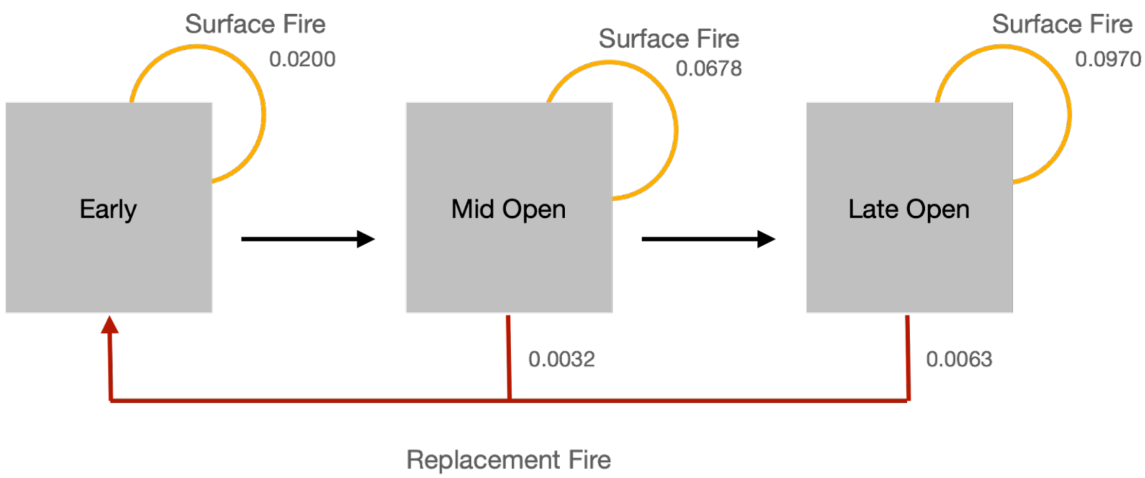


Fig. 2.2. Simplified example of a LANDFIRE Biophysical State Model. The black arrows represent time or growth. The dark red arrows Replacement fire and the orange arrows Surface fires. Probabilities are included for each state and transition. Replacement fires transition a state "back" to another state, whereas surface fires maintain a state "as-is."

I also used LANDFIRE's current succession class (sClass) layer determine conversion and departure of contemporary vegetation. LANDFIRE spatial data layers are at 30 meter resolution. They have a variety of data for vegetation and fuel collected primarily from Landsat 8 Operational Land Imager (OLI) image products (when OLI data are not available, Landsat 7 Enhanced Thematic Mapper Plus (ETM+) may be used) and from approximately 800,000 geo-referenced sampling units throughout the United States (Picotte et al., n.d.). Specifically, the sClass layer characterizes current vegetation conditions with respect to species composition, cover, and height ranges of successional states. The sClass layer also includes uncharacteristic classes, such as exotic species, uncharacteristic native species, and agriculture as well as a class for urbanization (Blankenship et al. 2021).

State and Transition Simulation Modeling

I ran state and transition simulation models in ST-Sim version 3.0 (Daniel and Frid 2012; http://www.apexrms.com/stsm). For my models, to mitigate problems of smaller simulation cell counts causing artificially inflated variation due to smaller population size, I selected the top 11 BPS models which accounted for over 96% of each ecoregion (further details below; Table 2.1). Each region and model were assigned proportional acreage during model initialization and each modeled cell represented 10 acres. Following Haugo (2019) and Blankenship (2015) each model was run for 1000 years and 100 Monte Carlo iterations. Transition multipliers based upon methods developed by Blankenship et al. (2015; see details below) were input for each BpS. Model runs were initialized with an equal distribution of simulation cells among the vegetation successional classes and stabilized within 200-250 years (Blankenship, Frid, and Smith 2015; R. D. Haugo et al. 2019). For fire severity HRV distributions, only the last 35 years were used for analysis (Haugo et al. 2019) and for vegetation HRV, the last 500 time steps (Blankenship et al. 2015; Haugo et al. 2015). Data were analyzed using the R statistical computing platform (R Core Team 2021).

Table 2.1. The top 11 BPS models for each ecoregion.

Northern California Interior Coast Ranges

BPS	Name
10300	Lower Montane Black Oak Woodland
11510	California Central Valley Riparian Woodland Shrubland
11130	California Coastal Live Oak Woodland Savanna
10290	Mixed Oak woodland
10980	California Montane Woodland & Chap
10280	Med California Mesic Mixed Conifer woodland
11520	California Montane Riparian Woodlands
10970	California Mesic Chaparral
10270	Dry-mesic Mixed conifer Woodland
11050	North & Central Dry-Mesic Chap
11140	Blue Oak Woodlands

Sierra Nevada Foothills

BPS	Model Name
10980	California Montane Woodland and Chaparral
10310	Jeffery-Pine Woodland
10280	Mediterranean California Mesic Mixed Conifer woodland
10970	California Mesic Chaparral
11540	Inter-Mountain Basins Montane Riparian Woodland
10300	Lower Montane black Oak Woodland
11520	California Montane Riparian Woodland
10270	Dry-Mesic Mixed Conifer Woodland
10290	Mixed Oak Woodland
11050	North & Central Dry-Mesic Chaparral
11140	Blue Oak Woodland

Fire

I simulated the extent and variability of low severity surface (less than 25 percent top-kill), moderate-severity mixed (25-75 percent top-kill), and high-severity replacement (greater than 75 percent top-kill) fire within a 35-yr observation window (1984-2019) for the combination of BPS models and Ecoregion. Based upon the stochastic variation in the 100 replicated model runs, I captured the minimum and maximum of the simulated occurrence (number of acres represented by model cells) of surface, mixed, and replacement fire. I represented HRV using minimum and maximum for each severity class to compensate for any potential modeling artifacts.

To capture HRV in fire severity distributions that were appropriate for my ecoregions and my contemporary observation window I modified LANDFIRE BpS model parameters to account for two sources of variability, problems with sample sizes and uncertainty regarding fire rotations, following Haugo et al (2019) and (Blankenship, Frid, and Smith 2015). First, within each ecoregion, several BpS models accounted for a very low proportion of the landscape (less than 1,000 acres). Each simulation cell in my models represented a point sample from a specific ecological combination of BpS and ecoregion (Keane et al. 2019). Thus, models with lower cell counts may result in greater overall variability. However, Haugo et al. (2019) found that across a range of BpS models cell count had a low influence on model variation when using greater than 100 cells (Haugo et al. 2019). Consequently, I chose to model the top 11 BpS models for each region which accounted for greater than 96% of the landscape and excluded all models with less than 100 cells (Table 2.1). The majority of the models were for vegetation classified as woodlands for each ecoregion and accounted for During analysis and model initiation, acreage was adjusted accordingly.

Next, to better account for uncertainty in fire rotations (Blankenship et al. 2015), I followed methods established by Blankenship et al. (2015) that vary fire transition probabilities between Monte Carlo iterations. Their methodology uses ST-Sim's transformed beta distribution fitted to the range of fire return intervals reported by LANDFIRE for each BpS model's severity class. The inputs include a mean probability multiplier, a standard deviation, and minimum and maximum multipliers. Per the methodology, the mean probability multiplier was set to 1 for all models so that across all iterations, on average, the mean fire return interval was equivalent to that reported in the LANDFIRE BpS models' source documentation. I used minimum and maximum multipliers to stretch the distribution between the minimum and maximum fire return interval reported by LANDFIRE and selected a standard deviation that maintained the widest possible distribution while maintaining an approximate bell-shaped curve (Blankenship et al. 2015).

I compared observed and HRV of fire severity based both on acres burned in each severity class and the proportion each severity class contributed to area burned. To determine whether current fire severity is outside of historical ranges, the departure of current fire severity classes from HRV for both area and proportion was calculated as the difference between the observed (contemporary) and either the minimum or maximum end of the expected range (HRV). Following Haugo et al. (2019), because the MTBS dataset was not a sampling but a census of all large wildfires for our window of observation, statistical significance levels were not assigned to these comparisons. I summed fire acreage and proportions of each severity class for my observation window and constrained my simulation data to the same temporal time-span and spatial extent. My analyses also corresponded to current standards, where modeling of climatic and fire norms typically uses three-decade time-periods to establish a baseline which is then used to assess for departure (Arguez and Vose 2011; Lutz et al. 2011; R. D. Haugo et al. 2019).

Vegetation

For my analysis to determine amount of land use conversion and whether current vegetation states and cover are departed from HRV for each ecoregion, I spatially combined the LANDFIRE BpS and Succession Class layers with spatial layers representing the Northern Interior Coast Ranges (NCI) and Sierra Nevada Foothills

(SNF) ecoregions of California in GIS software (ESRI ArcMap version 10.8) (Figure 3). I summed the amount of each ecosystem in the BpS spatial layer and converted these sums to acreage to calculate estimated historical extent which were used during model initialization.

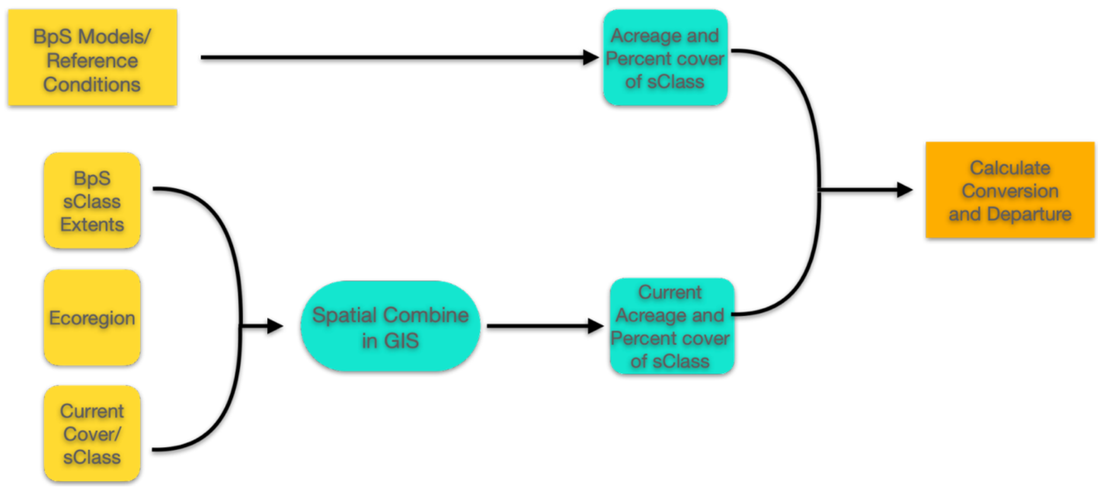


Fig. 2.3. Workflow for determining Conversion and Departure for each ecoregion using GIS. We started with the spatial layers on the left, combined them using ESRI ArcMap version 10.8 and were then able to compare historical reference conditions (Biophysical Settings; BpS) to current cover.

Next, I calculated conversion as the sum of agricultural and urban classes identified in the succession class spatial layer. I excluded acreage in the "Water" and "Barren" class. Then I calculated vegetation departure by summarizing the current amount and proportion of each stage in the succession class spatial layer. Finally, I compared these totals to the HRV determined by the last 500 time steps of simulation model runs. Similar to determining departure for fire, the departure of current sClass and cover from HRV for both area and proportion was calculated as the difference between the observed (contemporary) and either the minimum or maximum end of the expected range (HRV) for vegetation amount in each state class.

Results

Fire Departure

From 1984 to 2019 in the SNF ecoregion, Surface fires burned 662,868 acres, Mixed fires 334,643 acres, Replacement fires burned 1652,62 acres for a total of 1,162,773 acres out of the regions 5,587,353 acres. For the NCI ecoregion, Surface fires accounted for 427,662 acres, Mixed fires for 380,025 acres, and Replacement fires for 174,695 acres for a total of 982,381.6 acres out of the regions 1,707,873 acres. For the NCI ecoregion approximately 58 percent of the landscape burned during this time period, which is substantially greater than the 21 percent total area burned for the SNF ecoregion. This is likely partially due to the large fire seasons in 2015 and especially 2018 which included the Mendocino Complex Fire (459,123 acres), the largest recorded fire in California's history at that time (Figure 4).

Fig. 2.4. Modern (1984 - 2019) Fire Perimeters for Sierra Nevada Foothills and the Northern California Interior Coast Ranges Ecoregions. The largest fires recorded in California history have occurred in the Northern California Interior Coastal Ranges in the last few years.

In SNF, Mixed and Replacement fires are currently overrepresented compared to HRV both in acreage and proportionately (Figure 5). Surface fire acreage is within HRV, but it is underrepresented when we look at proportions. In NCI, all severity classes are overrepresented in acreage, but both surface and replacement fire fall within the expected proportions determined by our HRV analysis. Mixed fire makes up a greater proportion of fires than expected (Figure 6).

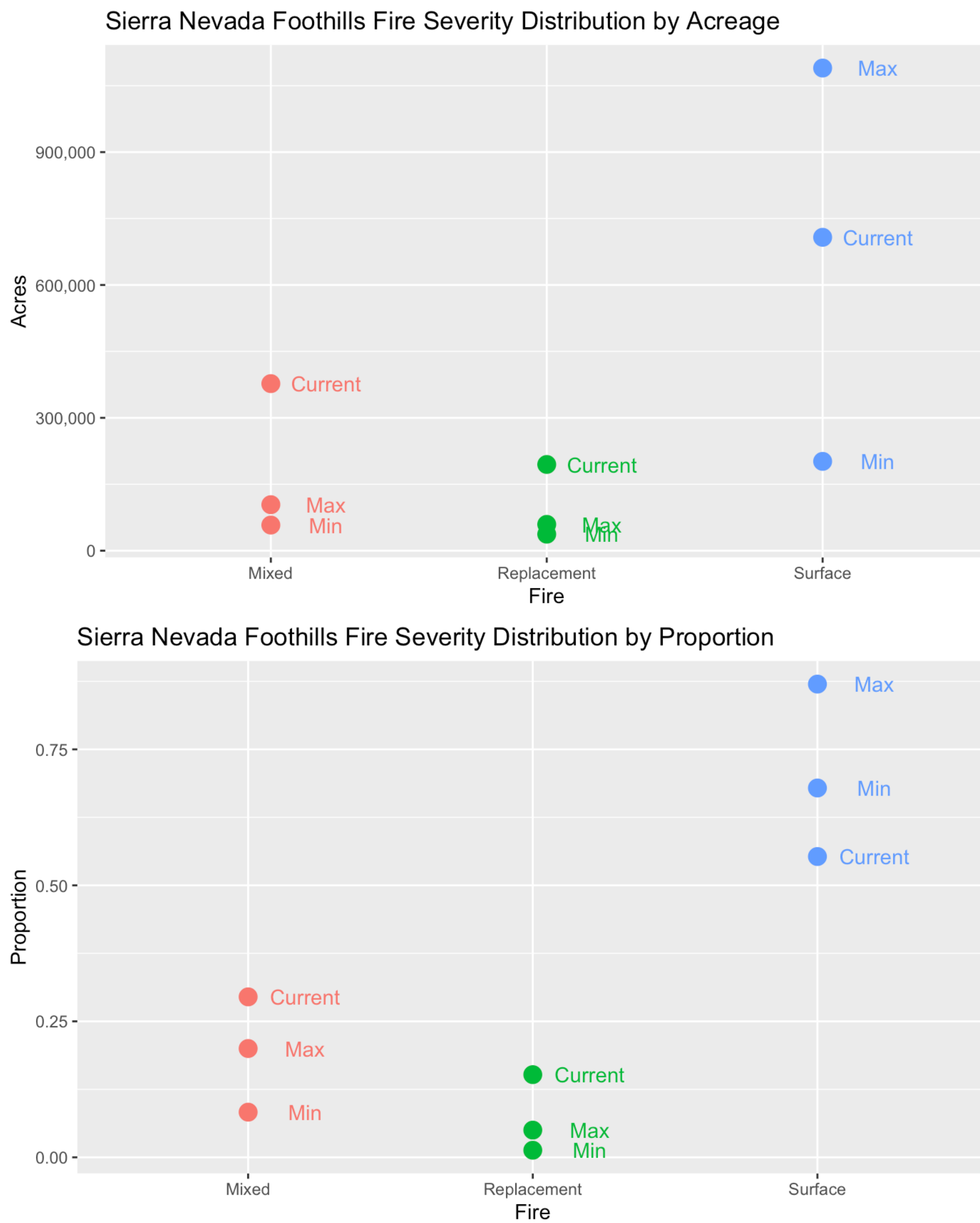


Fig. 2.5. Current and Historical Range of Variation (historical min and max) for the Sierra Nevada Foothills ecoregion's fire severity classes. Current mixed and replacement fires are burning more acreage and make up a larger proportion of the fire regime than historically projected. Surface fires are within the expected range of variation for acreage burned but are underrepresented proportionately.

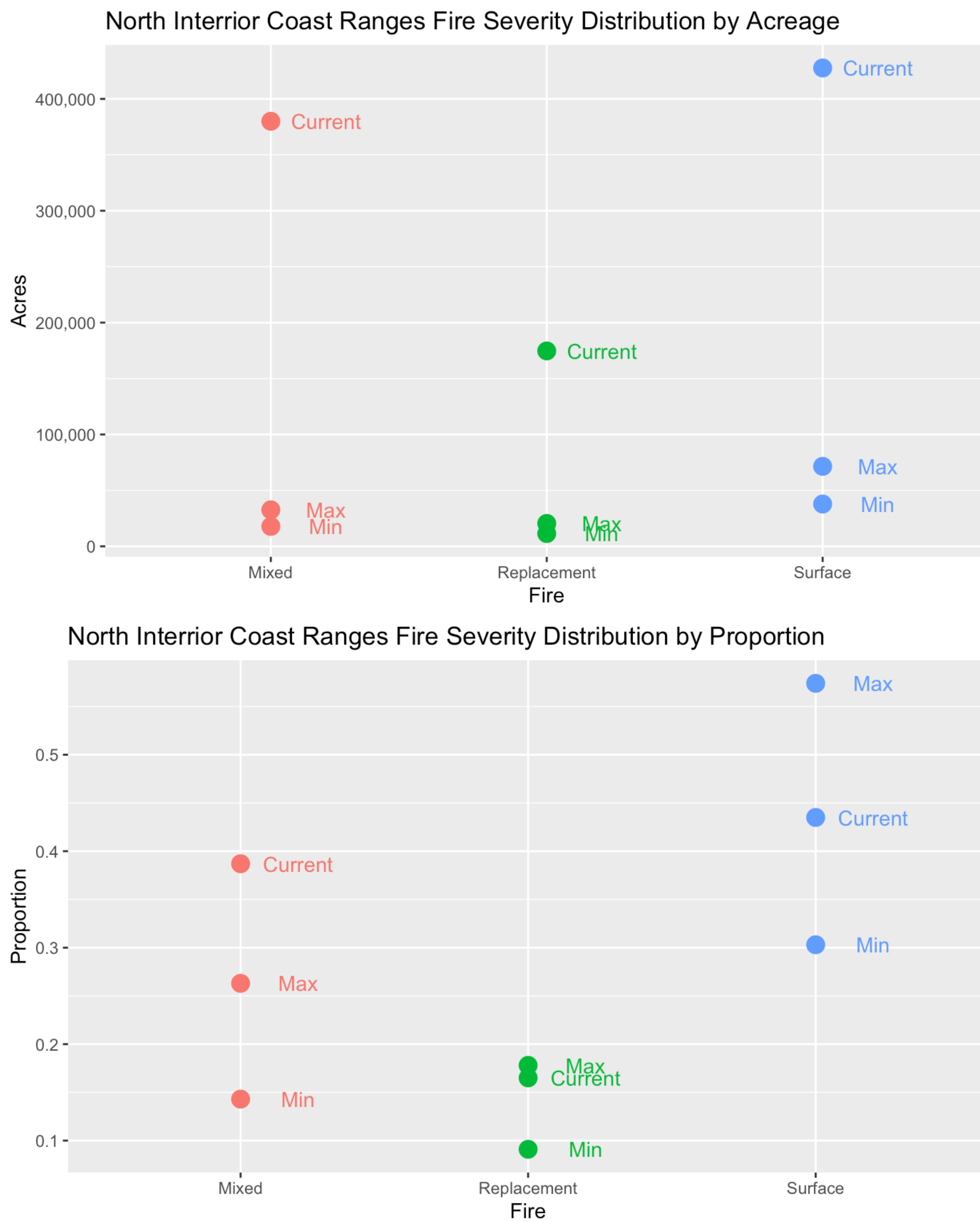


Fig. 2.6**.** Current and Historical Range of Variation (historical min and max) for the Northern California Interior Coast Range ecoregion fire severity classes. Currently, all classes are burning more acreage than historically projected. Mixed fires represent more of the modern fire regime than historically. Replacement and surface fires are within historical ranges of variation.

Vegetation Departure

Current land cover classes varied by ecoregion. Figure 7 is a map of current class coverage for each of the ecoregions. While both SNF and NCI had substantial amounts of exotic vegetation cover, it accounted for over twice as much acreage as any other class in the SNF ecoregion (2,007397 acres or 36 percent of the region; Figure 8). In the NCI ecoregion, vegetation in the Early successional cover class was the highest land cover (546,895 acres or 32 percent of the region) followed closely by exotic vegetation (512,793 acres or 30 percent of the region). Conversion to Urban and Agriculture accounted for 4 percent (66,555 acres) of the NCI region (Figure 8). SNF had a higher amount of conversion, with 7 percent of the region (363,154 acres) converted to agriculture or urban areas. When combined, SNF experienced 43 percent land cover conversion and NCI 36 percent.

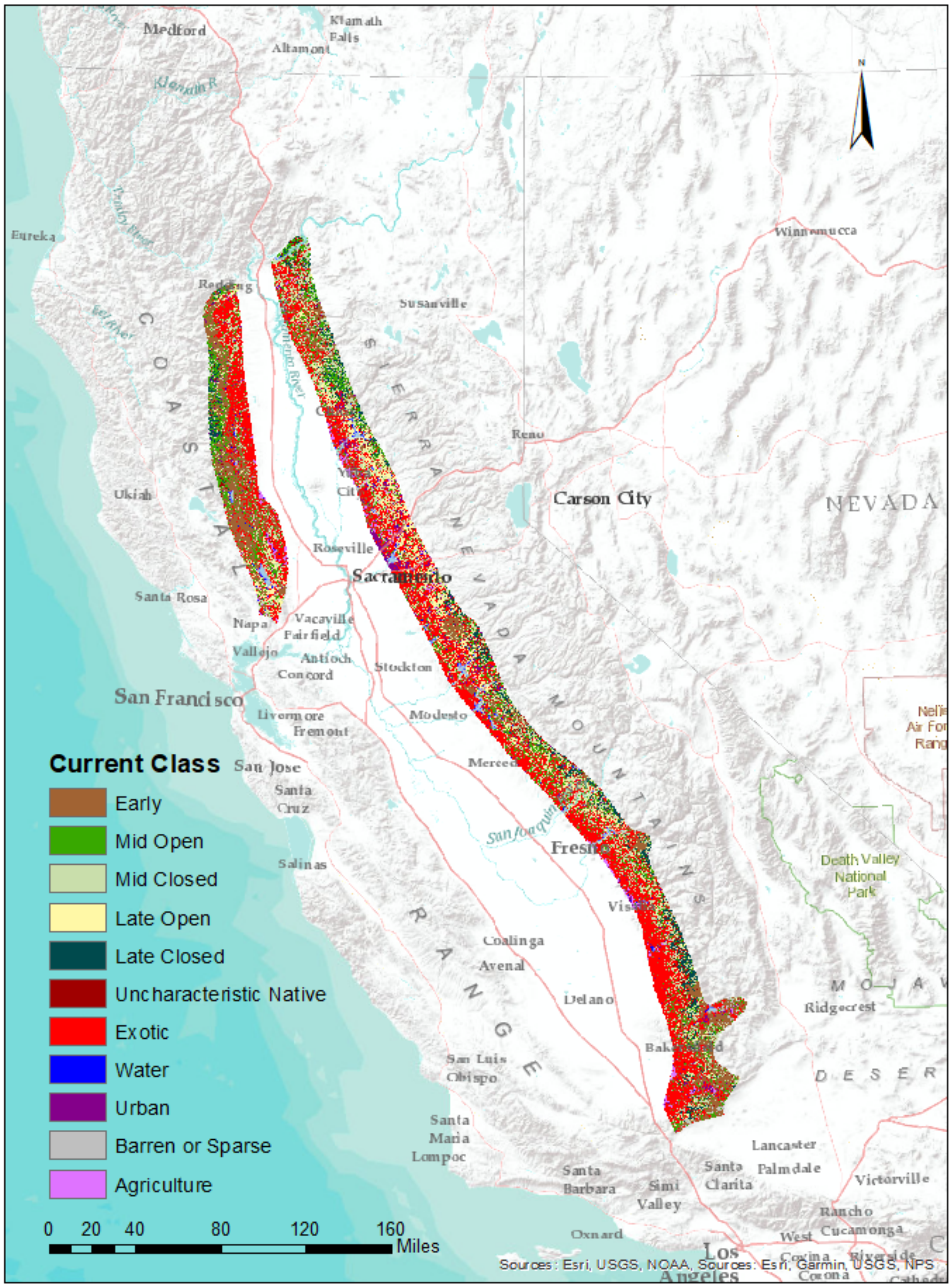
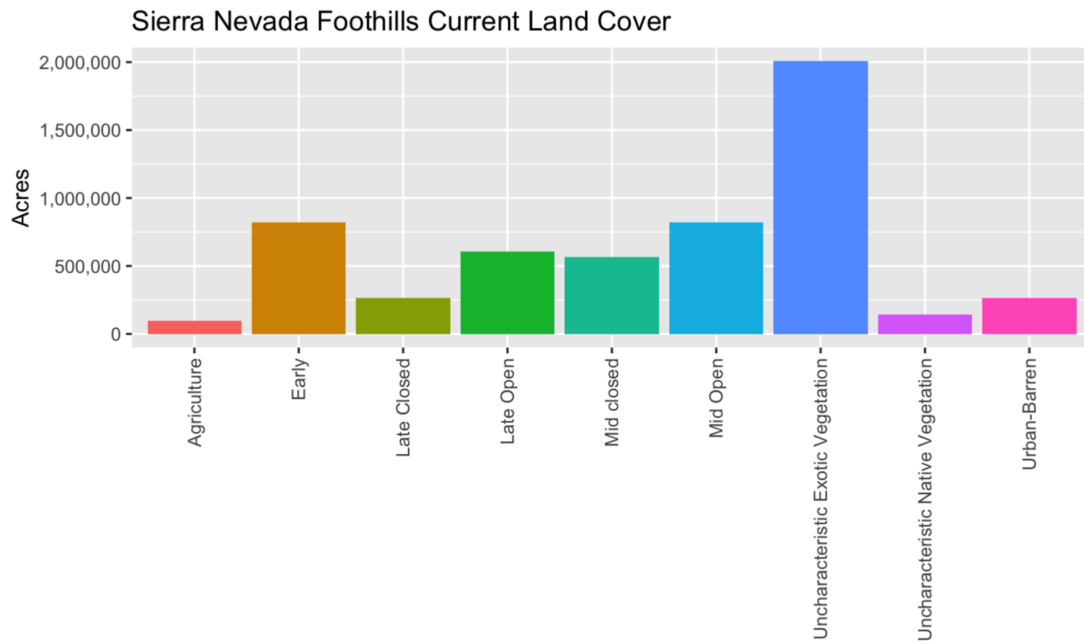


Fig. 2.7. Current Cover Classes for Each Ecoregion. Both regions had high amounts of conversion to exotic species.



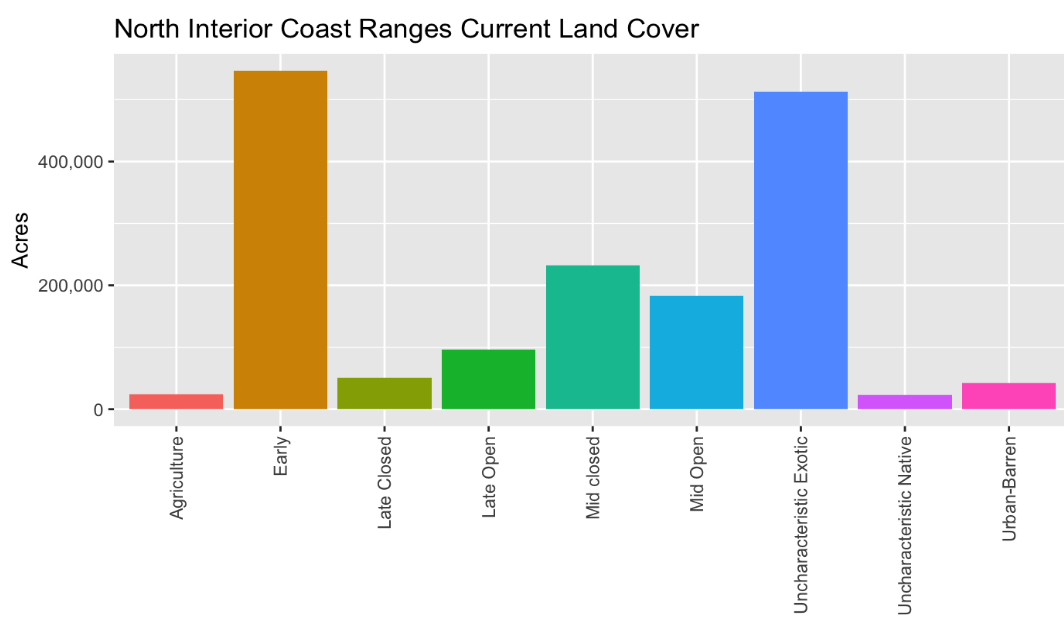


Fig 2.8. Current Land Cover for Each Ecoregion. Both Ecoregions have a substantial amount of exotic vegetation – in fact it covers twice as much acreage than any other cover class in the Sierra Nevada Foothills. In the Northern Coast Interior Ranges (NCI) vegetation in the Early successional stage covers most of the landscape followed closely by exotic vegetation.

For both regions, the Early successional sClass is greatly overrepresented both by acreage and proportion as is the Mid Closed sClass, consistent with our hypothesis (Figures 9 and 10). Historically, these classes comprised the lowest coverage, and yet currently account for some of the highest coverage for both ecoregions. However, for SNF, the Mid Open sClass when looking at acres is underrepresented but not when considering proportions where it accounts for slightly a higher proportion than expected. Conversely, the Mid Open sClass in NCI is slightly underrepresented in acreage but falls within HRV proportions. The Late Open sClass is underrepresented in both regions and for both acreage and proportions. For both regions, the Late Closed sClass is greatly underrepresented in both acreage and proportion. In NCI the Late Open and Late Closed sClasses were historically the dominant cover classes. In SNF, Mid Open accounted for the majority of historical land cover, followed closely by Late Closed and Late Open.

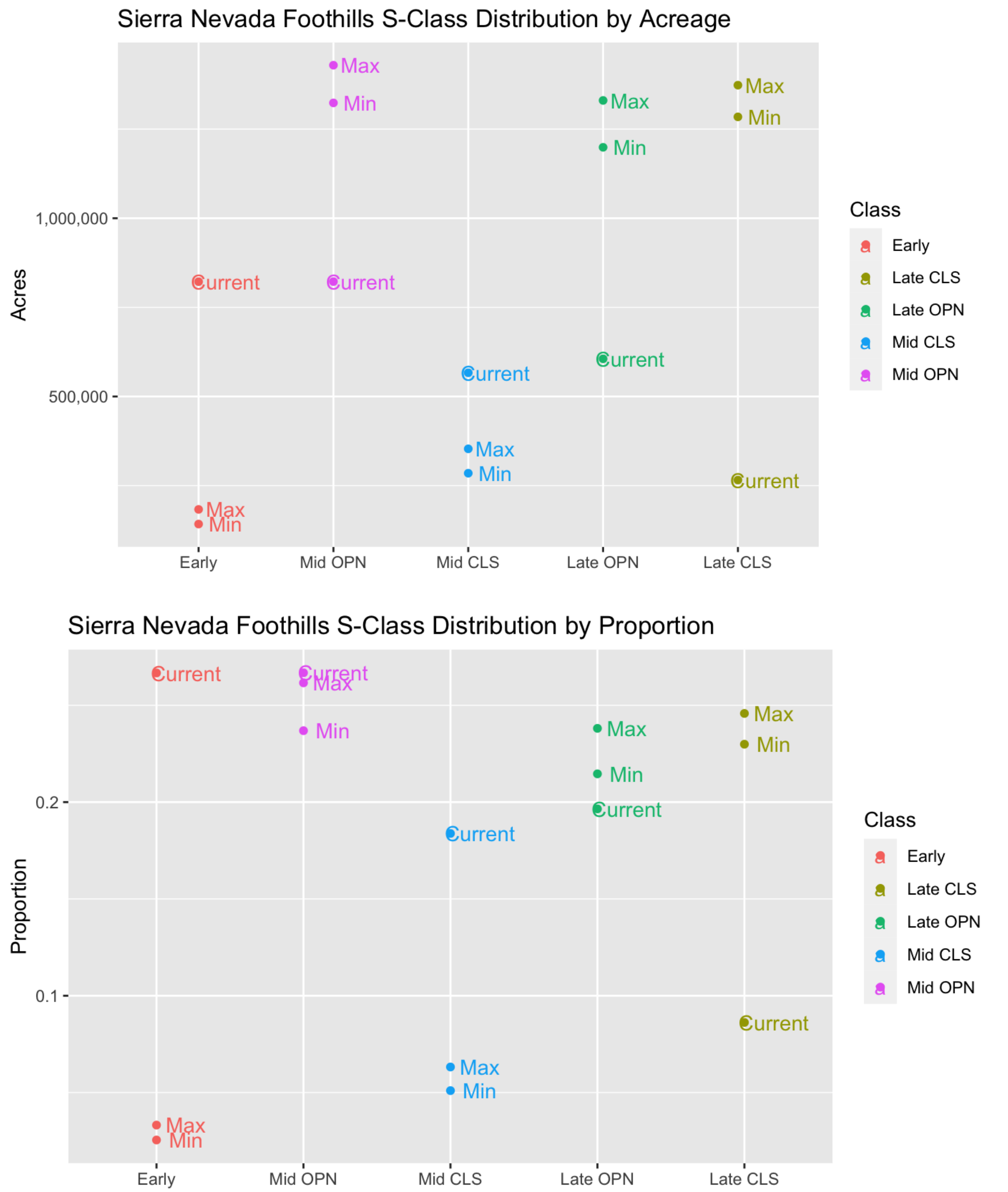


Fig. 2.9. Sierra Nevada Foothill Vegetation Current and Historical Range of Variation. The Early successional class is overrepresented both in acreage and as a proportion of total successional classes present. As is the Mid closed successional class. Late Open, Late Closed, and Mid open are underrepresented by acreage, however, mid open is slightly overrepresented as a proportion of the total successional classes present.

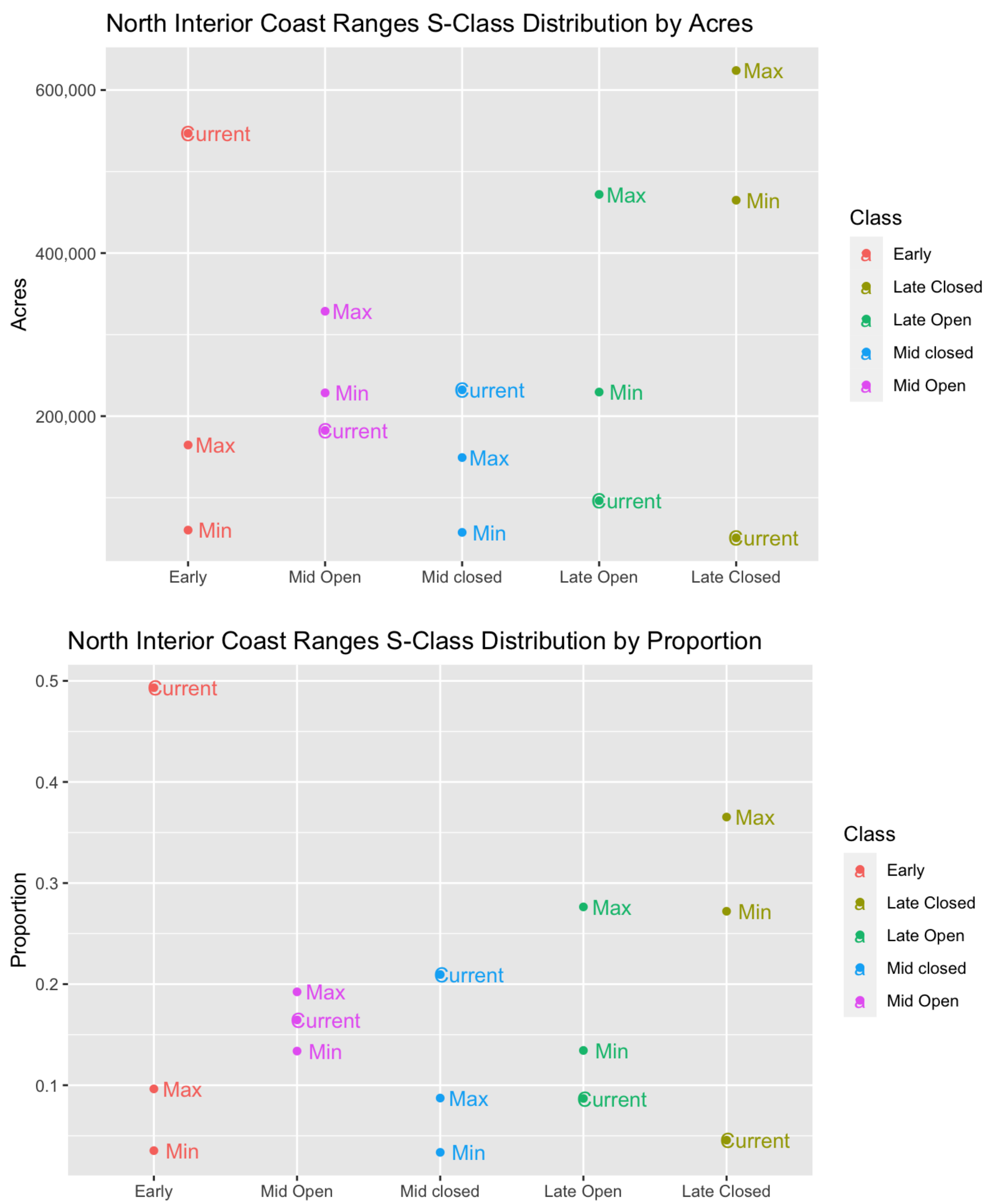


Fig. 2.10. Northern Coast Interior Ranges Vegetation Current and Historical Range of Variation. The Early successional class is greatly overrepresented both in acreage and as a proportion of total successional classes present. The Mid closed successional class is also overrepresented. Late Open, Late Closed, and Mid Open are underrepresented by acreage, however, Mid Open is within the historical range of variation for this class proportionately. The Late Closed class is greatly underrepresented.

Discussion

In contrast with recent findings in arid Western forested regions (R. D. Haugo et al. 2019) and specifically, California forested regions (Mallek et al. 2013; McGarigal et al. 2018), total contemporary acreage burned in both the SNF and NCI foothill woodland ecoregions are greater than expected. Thus, while early 20th century fire suppression may have resulted in reduced area burned during the first half of the 20th century (Mensing 1993; R. B. Standiford, Phillips, and McDougald 2012), since 1984, fires have indeed been increasing in extent in foothill woodlands. The difference between higher elevation forests and lower elevation foothill woodlands may in part be due to higher populations in these foothill ecoregions (Syphard, Keeley, and Abatzoglou 2017) and increased temperatures and drought due to the changing climate (McEvoy et al. 2019; Das et al. 2020). Lower elevations are expected to experience more severe warming and drying trends in the 21st century based on climate projections for California (Thorne et al. 2017, 2018) . Lightning strike-caused fires are rarer in foothill woodlands (Allen-Diaz and Standiford 2007; Sugihara et al. 2006) with the majority of current (and likely historical) fires being human-caused (Anderson 2006; Klimaszewski-Patterson et al. 2018). Unlike historic anthropogenic fires, which were likely most often being used as a land management tool, modern anthropogenic fires tend to be accidental, unplanned, and hence, uncontrolled (Kimmerer and Lake 2001; Anderson 2006; McGarigal et al. 2018; Hantson et al. 2022). Records indicate that traditional burning was likely done in environments and when conditions were less conducive to uncontrolled spread (i.e. not during excessively hot and windy conditions or where extensive underbrush or dead debris had built up) (Kimmerer and Lake 2001; Lightfoot and Cuthrell 2015; Codding and Bird 2013).

Fire severity in SNF is also increasing, with Mixed and Replacement fires comprising a greater proportion of area burned than expected. This is again likely due to the uncontrolled nature of these fires. Most large fires that surpass 1,000 acres are unmanaged and occur during conditions conducive to rapid spread and to hotter, more severe burning (i.e. windy, dry conditions)(Hantson et al. 2022). With climate change induced warming and drying trends, these conditions are likely to become more frequent (McEvoy et al. 2019; Das et al. 2020) with serious implications for management of these landscapes. While fire suppression activities may be necessary in some areas, it has become evident that, like advances being made in forested systems, consideration of alternative management actions is also necessary for foothill woodland systems (Steel, Safford, and Viers 2015). It is important to note that historically frequent surface fires, set by indigenous peoples (Kimmerer and Lake 2001; Anderson 2006; Lightfoot and Cuthrell 2015), likely reduced the fuels available for mixed and replacement fires (Kimmerer and Lake 2001; Codding and Bird 2013). And while we found that surface fires in recent decades were within the HRV for both acreage and proportion for the SNF region, historic variability for this severity class was greater than the others and current trends tend toward the middle of the distribution. Future management actions in this region should consider the potential of controlled surface fires as well as other fuels reduction methods.

But the story is more convoluted for the NCI ecoregion than for SNF. In that ecoregion, replacement fire proportion is within HRV, however, mixed fires comprise a greater proportion of area burned than expected. This, along with the much greater amount of total acreage burned in NCI (58 percent vs. SNF's 21 percent) may be in part because of the large fires in 2015 and, especially, 2018. Large complexes of fires appear to be on the rise in this region. Indeed, since our observation window, in 2020, the August Complex fire which burned approximately 1,032,648 acres and is the largest fire to date in the state's recorded history, burned through the region (https://www.fire.ca.gov) including approximately 191,882 acres of woodland. The dramatic increase in fire extents in this region may be due to climate induced warming and drying (McEvoy et al. 2019; Das et al. 2020). This region has also experienced less conversion to agriculture and urban areas, perhaps resulting in it taking longer to detect wildfire occurrence, less intensive responses to wildfires due to more limited firefighting resources, and less intensive fuels reduction efforts (Starrs et al. 2018). The decades of prior fire suppression may have also contributed to the buildup of fuels. Future research could look at how land ownership and firefighting resources and policies influence fire extent and fir severity distributions.

The relationship between fire regime change and vegetation patterns is complex and our hypothesis that shifts in fire regimes resulted in a shift from more open to more closed Mid and Late sClasses is only partially supported. Indeed, that the Mid Closed sClass, which historically accounted for only a small portion of land cover in both regions, has increased in dominance, supports our hypothesis. This is accompanied by the fact that the Late open sClass, which was one of the more dominant land covers for both regions, is currently underrepresented in both. However, in both SNF and NCI, the Early successional stages, despite being only a small portion of historical land cover, have become one of the most dominant.

It is difficult to tease apart the vegetation effects of changes in natural disturbance regimes, such as fire, from other anthropogenic disturbances including land cover and invasive species. Conversion of these regions to agriculture, urban development and the introduction of disturbances such as grazing by cattle, sheep, and goats have likely contributed to transitioning land cover in these regions to earlier states if not entirely to dominance by exotic annual plant species in the herbaceous layer (SNF:43 percent; NCI: 36 percent). These factors might also be contributing to the patterns in departure seen for The Mid Open sClass, an historically dominant land cover in these ecoregions. Extent in acreage of this sClass is currently underrepresented in both regions, however, it is only slightly outside of historical landscape proportions in SNF while it is within HRV of proportions for NCI. The loss in acreage could be because this sClass experienced high levels of conversion to other land uses and grazing by livestock. Conversely, the reason this land cover may be within its historical proportions of the remaining natural habitats on the landscape once conversion is accounted for, may be because use as rangelands afford it a certain amount of protection, with large swathes of land remaining undeveloped (Huntsinger et al. 2004; Cameron, Marty, and Holland 2014; Huntsinger and Oviedo 2014). Furthermore, rangelands may experience fire regimes more reflective of historical conditions as a result of fuel reduction by grazing and less intensive fire response when fires are present (Sulak and Huntsinger 2007).

The dramatic reduction in later, closed-canopy woodland successional states in both ecoregions must also be noted. Increases in Mixed severity and Replacement fires may partially account for the reduction in older, more closed canopy stands, transitioning these areas back to earlier states. Perhaps this is part of why we see such an increase in Early sClasses. However, this does not account for the increase in mid-closed canopy states. There may be a spatial part of this story that we are missing. It is also important to consider that mid-closed canopy states were not, and are still not, an extensive part of these landscapes. Timber harvesting may also need to be considered. While more common in higher elevation forests, the late-closed canopy states in the foothills woodland regions tend to occur at the higher elevations in the foothills and tend to be dominated by more mature pine and hardwood species than those of mid-closed canopy states. Thus, timber harvesting of these more mature pine and hardwood species may be a factor in the reduction of this sClass.

While our study used the longest currently available data set for contemporary trends in burn severity, some fire return intervals are longer than 35 years. The influence of some climatic events are also likely not fully captured during a 35 year time span. However, in our study regions, the fire return intervals tended to be shorter and in restricting the data used for comparison also to a 35 year window, we hope to have overcome some of these limitations. But as our record for contemporary trends in burn severity grows, future analysis should take advantage. Moreover, as climate changes, and advances are made in our modeling capabilities, including potential fire-climate-drought feedbacks will be beneficial.

The relationship between fire and vegetation is complex. Some other factors that we did not address include spatial configurations – vegetation mosaics – created by fire. While we did look at broad scale patterns of vegetation, varying fire severities and burn extents create a patchiness in the environment that affects subsequent burning patterns (Minnich 1983). Future work could focus on the finer scale implications of burn patchiness and vegetation patterns and their interactions across a specific landscape. Additionally, we considered only fire as a disturbance in our models. Other disturbances, such as grazing and timber harvest, interact with fire and influence vegetation patterns, potentially even at regional scales. In arid California foothill regions fire was historically the primary broadscale disturbance (George and Alonso 2008), however, European settlement brought with it a host of other disturbances. And while we looked at broadscale conversion of the landscape within our regions, future research could explore more precisely how these disturbances interact with fire and influence vegetation patterns across scales.

Despite these limitations, our study has implications for future land management in these regions. First, it is important to note that while the vegetation and climate in these two regions are similar, there are differences which can also be seen in our results. Thus, management must be regionally specific yet can be informed by a general understanding of fire and vegetation patterns in foothill woodland systems. Our findings suggest that fire severity has increased in both the Sierra Nevada foothills and the northern coast interior ranges, and that more acreage is burning than would typically burn prior to European settlement. This contrasts with findings in forested regions which have found current deficits in area burned (Mallek et al. 2013; McGarigal et al. 2018; R. D. Haugo et al. 2019). As large fire complexes are becoming more frequent and threatening socially, economically, and ecologically, management actions are needed to reduce the extent and severity of these fires, such as strategic prescribed burning and thinning. These management actions have the potential to not only reduce or prevent the massive, devastating fire complexes that appear to be on the rise, but also to shift current vegetation patterns back to a more natural mosaic of successional stages.

In conclusion, fire is an essential part of ecosystem processes and resilience, especially in water-limited forests and woodland systems. In California, foothill woodland systems are an extensive and iconic part of the landscape with a complex history tied inexorably to human use and management both in contemporary and pre-European settlement time-periods. Shifts in the extent and distribution of fire severities outside of historical norms in these landscapes are compounded by land use change, introduced disturbances, and a changing climate. Conversion of, and shifts in, native vegetation cover also results from complex interactions between fire and other anthropogenic disturbances. To manage these landscapes for resilience under future uncertainties it is necessary to consider actions that may help push them back into historic ranges of variation and to help maintain those that are within historic ranges. Extents may have been reduced due to conversion to urban and agriculture and while these acres may be lost, acreage converted to exotic cover may have some restoration potential.

Moreover, it is important to consider not only the restoration of a particular vegetation, but of restoring or maintaining current native vegetation cover in a heterogeneous mosaic of state classes. This may be dependent on the restoration or replication of heterogeneous disturbance regimes, where varying frequencies, extents, and intensities may be crucial to the persistence of these iconic landscapes under increasing uncertainties surrounding future climate change.

Works Cited

Alagona, Peter S. 2008. "Homes on the Range: Cooperative Conservation and Environmental Change on California's Privately Owned Hardwood Rangelands." *Environmental History* 13 (2): 325–49.

Allen-Diaz, and Standiford. 2007. "Oak Woodlands and Forests." *Vegetation of California*.

Apex Resource Management Solutions (2014). ST-Sim state-and-transition simulation model software.

Anderson, M. K. 2006. "The Use of Fire by Native Americans in California." *Fire in California's Ecosystems*, 417–30.

Arguez, and Vose. 2011. "The Definition of the Standard WMO Climate Normal: The Key to Deriving Alternative Climate Normals." *Bulletin of the American Meteorological Society*.

Bailey, Robert G. 1998. *Ecoregions of North America*. U.S. Department of Agriculture, Forest Service.

Bernhardt, and Swiecki. 2001. "Ecological Importance of California Oak Woodlands." *Restoring Oak Woodlands in California: Theory*.

Bestelmeyer, Ash, and Brown. 2017. "State and Transition Models: Theory, Applications, and Challenges." *Rangeland Ecology & Management*.

Blankenship, Frid, and Smith. 2015. "A State-and-Transition Simulation Modeling Approach for Estimating the Historical Range of Variability." *AIMS Environmental Science*.

Blankenship, Kori, Randy Swaty, Kimberly R. Hall, Sarah Hagen, Kelly Pohl, Ayn Shlisky Hunt, Jeannie Patton, Leonardo Frid, and Jim Smith. 2021. "Vegetation Dynamics Models: A Comprehensive Set for Natural Resource Assessment and Planning in the United States." *Ecosphere* 12 (4). https://doi.org/10.1002/ecs2.3484.

Bond, William J., and Jon E. Keeley. 2005. "Fire as a Global 'Herbivore': The Ecology and Evolution of Flammable Ecosystems." *Trends in Ecology & Evolution* 20 (7): 387–94.

Cameron, D. Richard, Jaymee Marty, and Robert F. Holland. 2014. "Whither the Rangeland?: Protection and Conversion in California's Rangeland Ecosystems." *PloS One* 9 (8): e103468.

Codding, Brian F., and Douglas W. Bird. 2013. "Forward: A Global Perspective on Traditional Burning in California." *California Archaeology* 5 (2): 199–208.

Coughlan, Michael R., Brian I. Magi, and Kelly M. Derr. 2018. "A Global Analysis of Hunter-Gatherers, Broadcast Fire Use, and Lightning-Fire-Prone Landscapes." *Fire* 1 (3): 41.

Crockett, J., and A. L. Westerling. 2017. "Climate-Driven Risk of Large Fire Occurrence in the Western United States, 1500 to 2003." In , 2017:GC51A – 0796. ui.adsabs.harvard.edu.

Daniel, Colin J., Leonardo Frid, Benjamin M. Sleeter, and Marie-Josée Fortin. 2016. "State-and-transition Simulation Models: A Framework for Forecasting Landscape Change." *Methods in Ecology and Evolution / British Ecological Society* 7 (11): 1413–23.

Daniel, and Frid. 2012. "Predicting Landscape Vegetation Dynamics Using State-and-Transition Simulation Models." *The First Landscape State-and-Transition Simulation …*. https://www.fs.usda.gov/treesearch/pubs/42566.

D'Antonio, Carla M., and Peter M. Vitousek. 1992. "Biological Invasions by Exotic Grasses, the Grass/Fire Cycle, and Global Change." *Annual Review of Ecology and Systematics* 23 (1): 63–87.

Das, Adrian J., Nicholas J. Ampersee, Anne H. Pfaff, Nathan L. Stephenson, Tedmund J. Swiecki, Elizabeth A. Bernhardt, Patricia K. Haggerty, and Koren R. Nydick. 2020. "Tree mortality in blue oak woodland during extreme droughtin Sequoia National Park, California." *Madrono; a West American Journal of Botany* 66 (4): 164–75.

Eidenshink, Jeff, Brian Schwind, Ken Brewer, Zhi-Liang Zhu, Brad Quayle, and Stephen Howard. 2007. "A Project for Monitoring Trends in Burn Severity." *Fire Ecology* 3 (1): 3–21.

Falk, Donald A., Philip J. van Mantgem, Jon E. Keeley, Rachel M. Gregg, Christopher H. Guiterman, Alan J. Tepley, Derek JN Young, and Laura A. Marshall. 2022. "Mechanisms of Forest Resilience." *Forest Ecology and Management* 512 (May): 120129.

George, Melvin R., and Maximo F. Alonso. 2008. "Oak Woodland Vegetation Dynamics: A State and Transition Approach." In *In: Merenlender, Adina; McCreary, Douglas; Purcell, Kathryn L., Tech. Eds. 2008. Proceedings of the Sixth California Oak Symposium: Today's Challenges, Tomorrow's Opportunities. Gen. Tech. Rep. PSW-GTR-217. Albany, CA: US Department of Agriculture, Forest Service, Pacific Southwest Research Station: Pp. 93-104*, 217:93–104. fs.usda.gov.

Hagmann, R. K., P. F. Hessburg, S. J. Prichard, N. A. Povak, P. M. Brown, P. Z. Fulé, R. E. Keane, et al. 2021. “Evidence for Widespread Changes in the Structure, Composition, and Fire Regimes of Western North American Forests.” *Ecological Applications: A Publication of the Ecological Society of America* 31 (8): e02431.

Hantson, Stijn, Niels Andela, Michael L. Goulden, and James T. Randerson. 2022. “Human-Ignited Fires Result in More Extreme Fire Behavior and Ecosystem Impacts.” *Nature Communications* 13 (1): 2717.

Haugo, Ryan D., Bryce S. Kellogg, C. Alina Cansler, Crystal A. Kolden, Kerry B. Kemp, James C. Robertson, Kerry L. Metlen, Nicole M. Vaillant, and Christina M. Restaino. 2019. “The Missing Fire: Quantifying Human Exclusion of Wildfire in Pacific Northwest Forests, USA.” *Ecosphere* 10 (4): e02702.

Haugo, Ryan, Chris Zanger, Tom DeMeo, Chris Ringo, Ayn Shlisky, Kori Blankenship, Mike Simpson, Kim Mellen-McLean, Jane Kertis, and Mark Stern. 2015. “A New Approach to Evaluate Forest Structure Restoration Needs across Oregon and Washington, USA.” *Forest Ecology and Management* 335 (January): 37–50.

Hessburg, Paul F., Carol L. Miller, Sean A. Parks, Nicholas A. Povak, Alan H. Taylor, Philip E. Higuera, Susan J. Prichard, et al. 2019. “Climate, Environment, and Disturbance History Govern Resilience of Western North American Forests.” *Frontiers in Ecology and Evolution* 7. https://doi.org/10.3389/fevo.2019.00239.

He, Tianhua, Byron B. Lamont, and Juli G. Pausas. 2019. “Fire as a Key Driver of Earth’s Biodiversity.” *Biological Reviews of the Cambridge Philosophical Society* 94 (6): 1983–2010.

Huntsinger, Lynn, Adriana Sulak, Lauren Gwin, and Tobias Plieninger. 2004. “Oak Woodland Ranchers in California and Spain: Conservation and Diversification.” *Advances in Geoecology* 37: 309–26.

Huntsinger, and Oviedo. 2014. “Ecosystem Services Are Social–ecological Services in a Traditional Pastoral System: The Case of California’s Mediterranean Rangelands.” *Ecology and Society*.

Keane, Robert E., Kathy Gray, Brett Davis, Lisa M. Holsinger, and Rachel Loehman. 2019. “Evaluating Ecological Resilience across Wildfire Suppression Levels under Climate and Fuel Treatment Scenarios Using Landscape Simulation Modelling.” *International Journal of Wildland Fire* 28 (7): 533–49.

Keane, Robert E., and Rachel Loehman. 2019. “Historical Range and Variation (HRV).” In *Encyclopedia of Wildfires and Wildland-Urban Interface (WUI) Fires*, edited by Samuel L. Manzello, 1–12. Cham: Springer International Publishing.

Kimmerer, R. W., and F. K. Lake. 2001. “The Role of Indigenous Burning in Land Management.” *Journal of Forestry* 99 (11): 36–41.

Klimaszewski-Patterson, Anna, Christopher T. Morgan, and Scott Mensing. 2021. “Identifying a Pre-Columbian Anthropocene in California.” *Annals of the Association of American Geographers. Association of American Geographers* 111 (3): 784–94.

Klimaszewski-Patterson, Anna, Peter J. Weisberg, Scott A. Mensing, and Robert M. Scheller. 2018. “Using Paleolandscape Modeling to Investigate the Impact of Native American–Set Fires on Pre-Columbian Forests in the Southern Sierra Nevada, California, USA.” *Annals of the Association of American Geographers. Association of American Geographers* 108 (2): 1–20.

Kling, Matthew M., Brent D. Mishler, Andrew H. Thornhill, Bruce G. Baldwin, and David D. Ackerly. 2018. “Facets of Phylodiversity: Evolutionary Diversification, Divergence and Survival as Conservation Targets.” *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* 374 (1763). https://doi.org/10.1098/rstb.2017.0397.

Knapp, Eric E., Carl N. Skinner, Malcolm P. North, and Becky L. Estes. 2013. “Long-Term Overstory and Understory Change Following Logging and Fire Exclusion in a Sierra Nevada Mixed-Conifer Forest.” *Forest Ecology and Management* 310 (December): 903–14.

Krofcheck, Daniel J., Matthew D. Hurteau, Robert M. Scheller, and E. Louise Loudermilk. 2017. “Restoring Surface Fire Stabilizes Forest Carbon under Extreme Fire Weather in the Sierra Nevada.” *Ecosphere* 8 (1): e01663.

Lightfoot, Kent G., and Rob Q. Cuthrell. 2015. “Anthropogenic Burning and the Anthropocene in Late-Holocene California.” *Holocene* 25 (10): 1581–87.

Lutz, J. A., C. H. Key, C. A. Kolden, and J. T. Kane. 2011. “Fire Frequency, Area Burned, and Severity: A Quantitative Approach to Defining a Normal Fire Year.” *Fire Ecology*.

Mallek, Chris, Hugh Safford, Joshua Viers, and Jay Miller. 2013. “Modern Departures in Fire Severity and Area Vary by Forest Type, Sierra Nevada and Southern Cascades, California, USA.” *Ecosphere* 4 (12): art153.

McEvoy, Daniel J., Mike Hobbins, Timothy J. Brown, Kristin VanderMolen, Tamara Wall, Justin L. Huntington, and Mark Svoboda. 2019. “Establishing Relationships between Drought Indices and Wildfire Danger Outputs: A Test Case for the California-Nevada Drought Early Warning System.” *Climate* 7 (4): 52.

McGarigal, Mallek, Estes, and Tierney. 2018. “Modeling Historical Range of Variability and Alternative Management Scenarios in the Upper Yuba River Watershed, Tahoe National Forest, California.” *Gen. Tech. Rep. RMRS*.

Mensing, S. A. 1993. “The Impact of European Settlement on Oak Woodlands and Fire: Pollen and Charcoal Evidence from the Transverse Ranges, California. Berkeley: University of California.” Ph. D. dissertation.

Millar, Constance I., Nathan L. Stephenson, and Scott L. Stephens. 2007. “Climate Change and Forests of the Future: Managing in the Face of Uncertainty.” *Ecological Applications: A Publication of the Ecological Society of America* 17 (8): 2145–51.

Minnich, R. A. 1983. “Fire Mosaics in Southern California and Northern Baja California.” *Science* 219 (4590): 1287–94.

Noss, Reed F., Jerry F. Franklin, William L. Baker, Tania Schoennagel, and Peter B. Moyle. n.d. “Ecology and Management of Fire-Prone Forests of the Western United States.” Accessed June 25, 2022.

Picotte, Long, Peterson, and Nelson. n.d. “LANDFIRE 2015 Remap – Utilization of Remotely Sensed Data to Classify Existing Vegetation Type and Structure to Support Strategic Planning and Tactical Response.” *Earthzine*.

R Core Team (2021) R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Rollins, Matthew G. 2009. “LANDFIRE: A Nationally Consistent Vegetation, Wildland Fire, and Fuel Assessment.” *International Journal of Wildland Fire* 18 (3): 235–49.

Safford, Hugh D., and Jens T. Stevens. 2017. “Natural Range of Variation for Yellow Pine and Mixed-Conifer Forests in the Sierra Nevada, Southern Cascades, and Modoc and Inyo National Forests, California, USA.” Vol. 256. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station.

Serra-Diaz, Josep M., Charles Maxwell, Melissa S. Lucash, Robert M. Scheller, Danelle M. Laflower, Adam D. Miller, Alan J. Tepley, Howard E. Epstein, Kristina J. Anderson-Teixeira, and Jonathan R. Thompson. 2018. “Disequilibrium of Fire-Prone Forests Sets the Stage for a Rapid Decline in Conifer Dominance during the 21st Century.” *Scientific Reports* 8 (1): 1–12.

Shive, Kristen L., Haiganoush K. Preisler, Kevin R. Welch, Hugh D. Safford, Ramona J. Butz, Kevin L. O'Hara, and Scott L. Stephens. 2018. "From the Stand Scale to the Landscape Scale: Predicting the Spatial Patterns of Forest Regeneration after Disturbance." *Ecological Applications: A Publication of the Ecological Society of America* 28 (6): 1626–39.

Stahle, D. W., R. D. Griffin, D. M. Meko, M. D. Therrell, J. R. Edmondson, M. K. Cleaveland, L. N. Stahle, et al. 2013. "The Ancient Blue Oak Woodlands of California: Longevity and Hydroclimatic History." *Earth Interactions* 17 (12): 1–23.

Standiford, Richard B., Ralph L. Phillips, and Neil K. McDougald. 2012. "Fire History in California's Southern Sierra Nevada Blue Oak Woodlands." *Fire Ecology* 8 (2): 163–67.

Standiford, Richard, Neil McDougald, William Frost, and Ralph Phillips. 1997. "FACTORS INFLUENCING THE PROBABILITY OF OAK REGENERATION ON SOUTHERN SIERRA NEVADA WOODLANDS IN CALIFORNIA." *Madroño* 44 (2): 170–83.

Starrs, Carlin Frances, Van Butsic, Connor Stephens, and William Stewart. 2018. "The Impact of Land Ownership, Firefighting, and Reserve Status on Fire Probability in California." *Environmental Research Letters: ERL [Web Site]* 13 (3): 034025.

Steel, Zachary L., Hugh D. Safford, and Joshua H. Viers. 2015. "The Fire Frequency-severity Relationship and the Legacy of Fire Suppression in California Forests." *Ecosphere* 6 (1): 1–23.

Stephens, Jaime L., Ian J. Ausprey, Nathaniel E. Seavy, and John D. Alexander. 2015. "Fire Severity Affects Mixed Broadleaf–conifer Forest Bird Communities: Results for 9 Years Following Fire." *The Condor* 117 (3): 430–46.

Sugihara, Neil G., Jan W. Van Wagtendonk, Joann Fites-Kaufman, Kevin E. Shaffer, and Andrea E. Thode. 2006. *Fire in California's Ecosystems*. Edited by Neil G. Sugihara, Jan W. Van Wagtendonk, Kevin E. Shaffer, Joann Fites-Kaufman, and Andrea E. Thode. Berkeley, CA: University of California Press.

Sulak, Adriana, and Lynn Huntsinger. 2007. "Public Land Grazing in California: Untapped Conservation Potential for Private Lands?" *Rangelands* 29 (3): 9–12.

Swaty, Randy, Kori Blankenship, Kimberly R. Hall, Jim Smith, Megan Dettenmaier, and Sarah Hagen. 2021. "Assessing Ecosystem Condition: Use and Customization of the Vegetation Departure Metric." *Land* 11 (1): 28.

Syphard, Alexandra D., Jon E. Keeley, and John T. Abatzoglou. 2017. “Trends and Drivers of Fire Activity Vary across California Aridland Ecosystems.” *Journal of Arid Environments* 144 (September): 110–22.

Temperton, Vicky M., Nina Buchmann, Elise Buisson, Giselda Durigan, Lukasz Kazmierczak, Michael P. Perring, Michele Sá Dechoum, Joseph W. Veldman, and Gerhard E. Overbeck. 2019. “Step Back from the Forest and Step up to the Bonn Challenge: How a Broad Ecological Perspective Can Promote Successful Landscape Restoration.” *Restoration Ecology*, no. rec.12989 (May). https://doi.org/10.1111/rec.12989.

Tepley, Alan J., Enrique Thomann, Thomas T. Veblen, George L. W. Perry, Andrés Holz, Juan Paritsis, Thomas Kitzberger, and Kristina J. Anderson-Teixeira. 2018. “Influences of Fire–vegetation Feedbacks and Post-fire Recovery Rates on Forest Landscape Vulnerability to Altered Fire Regimes.” *The Journal of Ecology* 106 (5): 1925–40.

Thorne, James H., Hyeyeong Choe, Ryan M. Boynton, Jacquelyn Bjorkman, Whitney Albright, Koren Nydick, Alan L. Flint, Lorraine E. Flint, and Mark W. Schwartz. 2017. “The Impact of Climate Change Uncertainty on California’s Vegetation and Adaptation Management.” *Ecosphere* 8 (12): e02021.

Thorne, James H., Hyeyeong Choe, Peter A. Stine, Jeanne C. Chambers, Andrew Holguin, Amber C. Kerr, and Mark W. Schwartz. 2018. “Climate Change Vulnerability Assessment of Forests in the Southwest USA.” *Climatic Change* 148 (3): 387–402.

Tietje, W., and J. Vreeland. 1997. “Vertebrates Diverse and Abundant in Well-Structured Oak Woodland.” *California Agriculture*.

Van de Water, Kip M., and Hugh D. Safford. 2011. “A Summary of Fire Frequency Estimates for California Vegetation before Euro-American Settlement.” *Fire Ecology* 7 (3): 26–58.

Westoby, and Walker. 1989. “Opportunistic Management for Rangelands Not at Equilibrium.” *Rangeland Ecology & Management*.

Zack, Steve, Mary K. Chase, Geoffrey R. Geupel, and Diana Stralberg. 2005. “The Oak Woodland Bird Conservation Plan: A Strategy for Protecting and Managing Oak Woodland Habitats and Associated Birds in California.” In *In: Ralph, C. John; Rich, Terrell D., Editors 2005. Bird Conservation Implementation and Integration in the Americas: Proceedings of the Third International Partners in Flight Conference. 2002 March 20-24; Asilomar, California, Volume 1 Gen. Tech. Rep. PSW-GTR-191.*

Albany, CA: US Dept. of Agriculture, Forest Service, Pacific Southwest Research Station: P. 174-178. Vol. 191. fs.usda.gov.

Zald, Harold S. J., Chance C. Callahan, Matthew D. Hurteau, Marissa J. Goodwin, and Malcolm P. North. 2022. “Tree Growth Responses to Extreme Drought after Mechanical Thinning and Prescribed Fire in a Sierra Nevada Mixed-Conifer Forest, USA.” *Forest Ecology and Management* 510 (April): 120107.

Ch.3 Assessing California Blue oak woodland and savanna land use conversion, structural diversity departure, and restoration needs.

Abstract: The need for widespread ecological restoration efforts has come to the forefront of worldwide efforts to protect biodiversity and ecosystem services with the declaration by the United Nations that we are now in the decade of restoration (2020-2030). Forests and reforestation efforts are on the rise, however, the restoration of some at-risk ecosystems, such as woodlands, may require actions that reduce tree cover. In California, Blue oak woodlands and savannas are one of the state's most extensive, diverse, and at-risk systems. They span more than 4.5 million hectares and are home to many of the state's iconic and endemic species. We provide an assessment of Blue oak woodlands' current status and restoration needs for the three regions in which they are most extensive and demonstrate the potential effectiveness of management actions needed for restoration. Blue oak woodlands and savannas have experienced high rates of land use conversion and fall outside of their historical ranges of structural diversity. Restoration needs vary by region suggesting the need for regionally specific restoration efforts. However, our findings overwhelmingly suggest that these landscapes need overall reductions in disturbance, and that restoration efforts require the limited reintroduction of low-severity disturbances, such as locally targeted prescribed burning, grazing, or mechanical thinning.

Introduction

The United Nations has declared 2020-2030 the decade of restoration (Cooke, Bennett, and Jones 2019). The recognition that restoration needs to happen more extensively and at broader scales (R. Haugo et al. 2015; Cooke, Bennett, and Jones 2019) has been met with a surge in research in landscape reforestation (Mansourian, Dudley, and Vallauri 2017; Temperton et al. 2019). This has resulted in a focus on broad-scale restoration and reforestation of forested ecosystems worldwide and less focus on the broadscale restoration of non-forested systems (Temerton et al. 2019).

In California, Blue oak woodlands and savannas are one of the most extensive of the state's ecosystems, covering 11 percent of the land area (4.5 million hectares) spanning multiple regions. They form an almost continuous ring around the state's great central valley, one of the most agriculturally productive regions in the world (Elena-Rosselló et al. 2013; Tyler, Kuhn, and Davis 2006). They are also one of the most biodiverse ecosystems in the state and are home to some of its most iconic species, including many endemic species of oaks (Tyler, Kuhn, and Davis 2006). Wilson et al. (Wilson, Sleeter, and Davis 2015) found that Blue oak woodlands and savannas are one of the most threatened ecosystems in California. These landscapes have experienced land use conversion and ecological degradation due to a number of anthropogenic drivers including livestock ranching, conversion to crop agriculture, mining, urban development, and invasion by exotic species. Multiple threats continue to challenge their conservation and restoration, such as further urban and exurban development through the subdivision of once extensive ranchlands, changes in disturbance regimes such as fire, and increased drought brought on by a changing climate (Wilson, Sleeter, and Davis 2015).

Journals, other written accounts, and historical photographs from early European explorers to California comment on the existence of common "park-like settings" throughout the Blue oak woodlands and savannas of California. The landscapes comprised mixed age, patchy tree communities, free from dense underbrush (Jepson 1923; Muir 1979; Lewis, Bean, and Lawton 1973; Anderson 2006). These vegetation patterns are largely attributed to frequent, low-intensity fires that consume dead debris, grasses, shrubs, and seedlings without damaging mature trees (Anderson 2006; Klimaszewski-Patterson et al. 2018; Klimaszewski-Patterson, Morgan, and Mensing 2021). Indigenous communities intentionally set low-intensity surface fires near settlements to reduce underbrush density, stimulate sprouting of useful shrubs and herbaceous species, provide forage for game and facilitate hunting, clear travel corridors, promote seed germination, manage pests and disease, and increase yields of other natural resources (Anderson 2006; Klimaszewski-Patterson and Mensing 2020).

Indeed, the evidence is mounting that the iconic California Blue oak woodlands and savannas may largely be an anthropogenic landscape. Mensing (1992) found evidence of changes in recruitment patterns with relatively continuous recruitment during the pre-European settlement period, an increase in recruitment and density initially following the period of European settlement and an almost complete lack of recruitment following increased fire suppression and livestock stocking densities beginning around 1864. Other findings suggest that oak recruitment has been in decline since at least the

1900’s (Mensing 1992; Zavaleta, Hulvey, and Fulfrost 2007; Tyler, Kuhn, and Davis 2006). This corresponds to a time period when the effects of the removal of native peoples and their management practices from the landscapes and intensive fire suppression efforts in forests by settlers would begin to influence vegetation patterns (Klimaszewski-Patterson and Mensing 2020). Millions of acres of oaks were also cleared to obtain firewood, to make way for farms and orchards, and to improve rangelands for cattle. They were often subsequently replaced by nonnative grasses and forbes, many of which were introduced as livestock forage (Pavlik, Muick, and Johnson 1993).

Many are concerned that current oak recruitment trends are not at replacement levels (Swiecki, Bernhardt, and Drake 1997; Zavaleta, Hulvey, and Fulfrost 2007; Ackerly et al. 2019). Swiecki et al. (1998) found that while seedlings are found under the canopies of mature oaks, saplings were positively associated with open canopies. Open canopies, maintained by low-intensity surface fires, may promote seedling survival and tree health as well as a diversity of understory species (Fry 2008; Hankins 2015; Das et al. 2020). Moreover, the buildup of thatch and a dense herbaceous layer (whether native or not) may reduce seedling establishment and survival due to competition and herbivory by small mammals (Tecklin, Connor, and Mccreary n.d.).

Thus, the conservation, restoration, and preservation of Blue oak woodlands and savannas, especially in their dynamic and more open states, likely requires a different approach than reforestation (Dudley et al. 2020; Temperton et al. 2019). Indeed, we may need to institute practices that reduce canopy covers, such as the re-introduction of indigenous burning practices which include more frequent, low-severity surface fires.

However, that goal may be complicated by multiple factors including safety related issues such as air quality and the protection of infrastructure and livelihoods (Syphard, Keeley, and Abatzoglou 2017). Thinning (reducing tree density) by mechanical thinning, targeted grazing, or other means is a potential complement or alternative to prescribed fires where they may be too difficult to implement. It is important to also note that some climate projections include potential increases in fire and drought severity and thinning has the potential to enhance woodland resilience to both fire and drought (Beckmann 2019; Dwomoh et al. 2021). Frequent, low-severity fires also reduce fuel loads and maintain a patchy mosaic of vegetation, thus reducing the potential for large, widespread, catastrophic fires (Keane et al. 2019).

To evaluate the current conditions and restoration needs of California's Blue oak woodlands and savannas, we first assess the land use conversion rates and current structure (successional canopy density states) of blue oak woodlands in the three different ecoregions where they are a dominant ecosystem. We hypothesize first, that because of their lower elevations, proximity to California's great valley with its extensive agriculture, changes in historic disturbance regimes, and the introduction of nonnative grazing, Blue oak woodlands have experienced high conversion rates, with higher rates of conversion to 1) exotic species than regional rates (across all vegetation types) and 2) higher rates of conversion to agriculture than regional rates. Next, we hypothesize that, due to contemporary land management practices which include a century of intensive livestock grazing and an emphasis on fire suppression, middle to late open canopy successional states – which were once the dominant structural components of these landscapes – will be underrepresented relative to historical patterns. Instead, the early successional state and the later closed canopy states of blue oak woodlands will be overrepresented relative to historical patterns. Next, we identify restoration needs based on departures from those historical patterns. And finally, we demonstrate the usefulness of state and transition simulation modeling to help determine the level of restoration treatments necessary to restore the structure of Blue oak woodlands through a single region case-study.

Methods

California Blue oak woodlands form an almost continuous belt between the California Great Central Valley and the Sierra Nevada and coast ranges (Fig.1). Elevation spans from approximately 100 meters to 1,200 meters and the climate varies regionally but is typical of a Mediterranean-type climate with hot dry summers and cool damp winters. Historically, these landscapes were dominated by open, “park-like” stands of *Quercus douglasii, Q. wislizeni, Q. agrifolia, Q. lobata, and Pinus sabiniana* with an understory of consisting of various scattered shrubs and herbaceous species, including annual forbs and perennial bunchgrasses. Canopy densities likely varied due to differences in soil-moisture regime and the natural patch dynamics of fire. Currently, extensive grazing, establishment of nonnative annual grasses as dominating the ground layer, changes to fire regimes, mining, and other land use changes have led to the conversion and degradation of these landscapes (Bernhardt and Swiecki 2001). Our study focused on the three Bailey’s Ecoregions (Bailey 1998) where these landscapes were (and are) most extensive:

The Sierra Nevada Foothills (SNF), The Northern California Interior Coast Ranges (NCI), and the Central Coast Interior Ranges (CCR) (Fig. 1).

Fig.3.1. Blue Oak Woodlands and Savannas Regional Sites. Blue oak woodlands and savannas form an almost continuous belt around California's great central valley. Delimited are the three ecoregions (Bailey, 1998) where Blue oak woodlands and savannas occur: Sierra Nevada Foothills (SNF;purple), Northern California Interior Coast Ranges (NCI; light orange), and Central Coast Interior Ranges (CCI; Green).

To determine the restoration needs of California Blue oak woodlands and savannas, I first assessed their conversion and departure from the range of historic conditions. Historical Range of Variation (HRV) is the expected natural variability of change through time of an ecological system or process (Keane and Loehman 2019). Departure is a metric used to assess landscape condition and helps determine the degree to which a landscape's various successional stages "is departed from" a measure such as HRV (Swaty et al. 2021). Together HRV and Departure allow for a quantitative evaluation of landscapes that moves beyond an assessment of whether an ecosystem is present and how extensive it is. They also allow for an assessment of within-system heterogeneity, which is a key component of ecosystem resilience (Swaty et al. 2021).

To establish HRV, I used the Biophysical settings (BpS) model for Blue oak woodlands and savannas (BpS 11140) developed by the LANDFIRE program (landfire.gov). A joint program managed by the U.S. Department of Agriculture Forest Service and the U.S. Department of the Interior, the LANDFIRE program has created over 1,000 BpS models for the United States and its territories. Each model represents the vegetation and disturbance processes believed to have existed at a particular landscape prior to European settlement. The models were developed through extensive literature and expert review processes coupled with several historical empirical data sources (e.g. pollen and charcoal in sediments, dendrochronological reconstructions, and historic survey records) (Rollins 2009).

BpS models are a combination of state and transition simulation models (STSM) and peer reviewed documents specific to each model that describe the vegetation and disturbance regimes of that model, including transition probabilities among the vegetation states (Rollins 2009; Blankenship et al. 2021) (Figure 2). Westoby (1989) introduced state and transition models as a method to capture the complexity of non-equilibrium systems. For vegetation, modeling their complex dynamics requires incorporating multiple successional pathways, as well as multiple states, thresholds, and transitions (Westoby and Walker 1989; Bestelmeyer, Ash, and Brown 2017). Using an adapted Markov chain approach, STSMs operationalize state and transition models by predicting how vegetation transitions through states in response to interactions between various disturbances, successional pathways, and management actions (Daniel and Frid 2012). Change over time in these models is a stochastic process, with time represented as discrete steps and space as a set of discrete units. Rates of change between states are expressed as probabilities (Daniel et al. 2016).

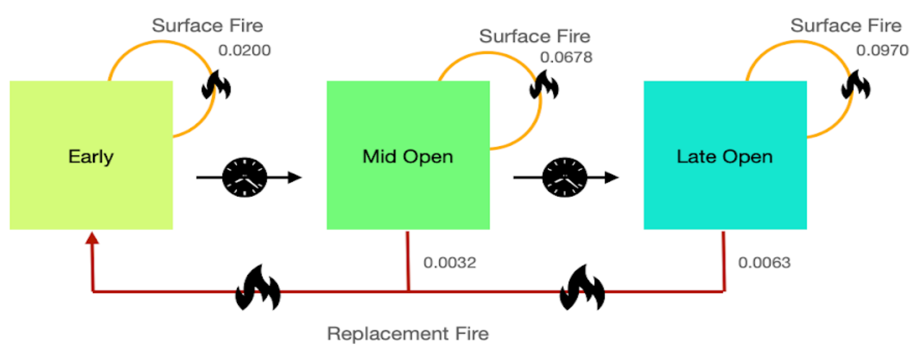


Fig. 3.2. A schematic example of a state and transition simulation model. The black arrows represent succession (time). The red arrows represent replacement fire which transitions a state back to a previous state. The orange circles represent surface fires which maintain a state within its current structure. For state and transition simulation models, transition probabilities are included and can be adjusted to represent current, historical, or projected fire return intervals.

Our STSMs were run in ST-Sim version 3.0 (Daniel and Frid 2012; http://www.apexrms.com/stsm). Ie set initial acreage to match historically projected extents of Blue oak woodland and savannas for each ecoregion. I ran each model for

1000 years and 100 Monte Carlo iterations (R. Haugo et al. 2015; Blankenship, Frid, and Smith 2015). To establish HRV for each ecoregion, model runs were initialized with an equal distribution of simulation cells among the vegetation successional classes until they stabilized. Consistent with the findings of others, the models stabilized within 200-250 years (Blankenship, Frid, and Smith 2015; R. D. Haugo et al. 2019). I used the last 500 time steps for analysis (Blankenship et al. 2015; Haugo et al. 2015). I analyzed my data using the R statistical computing platform (R Core Team 2021).

The first step to determining the amount of Blue oak woodlands and savannas that have been converted to agriculture, urban development and exotic species, and the level of departure from HRV for each ecoregion, required the use of GIS software (ESRI ArcMap version 10.8) to spatially combined the LANDFIRE BpS and Succession Class (sClass) layers representing current conditions with spatial layers representing each ecoregion. The LANDFIRE Succession Class (sClass) layer is at 30 meter resolution, collected from Landsat 8 Operational Land Imager (OLI) image products (when OLI data are not available, Landsat 7 Enhanced Thematic Mapper Plus (ETM+) is used) and georeferenced groundpoints (Picotte et al., n.d.). The sClass layer classifies the distribution of current vegetation successional states using species composition, cover, and height ranges. The sClass categories also include exotic species cover, uncharacteristic native species, agriculture and urban development (Blankenship et al. 2021).

Once these layers were combined, I was able to calculate conversion as the sum of the acreage covered by the exotic cover, agricultural, and urban classes. “Water” and

“Barren” classes were excluded from my analysis. To determine Departure, I summarized the current area and landscape proportion of each successional stage in the Succession class spatial layer. I then compared these contemporary values to the HRV established during our STSM runs. I compared both acreage and proportions. Departure was calculated as the difference between the observed (contemporary) and either the minimum or maximum of the expected range (HRV) for vegetation amount in each state class.

Restoration Needs

To determine restoration needs I used my assessment of departure to identify which successional classes (sClasses) were overrepresented and which were underrepresented. Lands that have undergone conversion may not be candidates for restoration and therefore, based on the goal of achieving an historically representative distribution of sClasses within the remaining landscape, I focused on sClasses that were not within HRV proportions (as opposed to historical area). I calculated the necessary acreage to return sClasses outside of HRV to within HRV proportions based upon the acreage currently classified as Blue oak woodlands and savannas. Then, building off Haugo et al. (2015) and LANDFIRE BpS documentation (Rollins 2009), I determined the broad treatments necessary to affect specific transitions between sClasses. These treatments were classified as ‘‘Surface fire/thinning’ (removal of small trees, shrubs, dead fuels)’, “Mixed/Harvest” (removal of small and some mature trees), “Replacement/Harvest” (removal of mature trees), ‘‘Succession’’ (allow for maturation), or ‘‘Succession with thinning’’ (allow for maturation and removal of some trees, shrubs, dead fuels) and were then assigned based upon the identity of the over- or underrepresented classes (Table 1).

Table 3.1. Restoration treatments. Specific restoration treatments to initiate transitions between successional classes of Blue oak woodlands and savannas.

Excess S-Class	Deficit S-Class	Restoration Treatment
Mid closed	Mid open	Surface Fire/thinning
Late closed	Late open	Surface Fire/thinning
Mid closed	Late open	Surface Fire/thinning
Late open	Mid open	Surface Fire/thinning
Late closed	Mid open	Mixed/Harvest
Mid closed	Early	Mixed/Harvest
Mid open	Early	Mixed/Harvest
Late open	Early	Mixed/Harvest
Late closed	Early	Replacement/Harvest
Early	Mid open	Succession with Thinning
Mid open	Late open	Succession with Thinning
Early	Mid closed	Succession
Mid closed	Late closed	Succession
Mid open	Late closed	Succession
Late open	Late closed	Succession
Mid open	Mid closed	Succession
Early	Late open	Succession with Thinning
Early	Late closed	Succession

STSM Restoration Treatment Assessment Case Study

To demonstrate the potential of STSM for assessing treatment potential to return Blue oak woodlands and savannas to more historic distribution of successional states, I used the NCI ecoregion. Almost half of this region (47 percent) is believed to have historically been Blue oak woodlands and savannas. The current conditions are outside of HRV with clear restoration needs if historic proportions are to be reestablished.

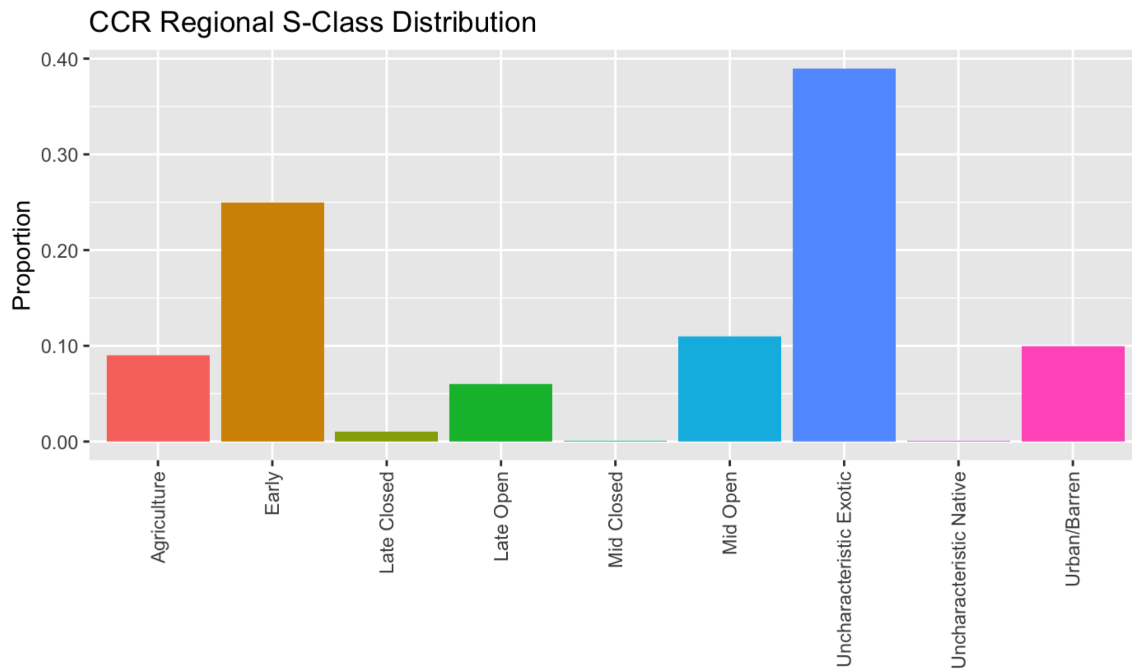
Initial conditions were set to current proportions of each sClass remaining within the NCI ecoregion and I adjusted fire return intervals to simulate modern fire return intervals. Then using my restoration needs assessment, I further adjusted fire return intervals to reduce disturbance and initiate succession. Then I instituted targeted treatments using ST-sim's transition targets for the open and closed canopy states. Transition targets allow for the specification of the number of acres per year to undergo a specific treatment (e.g. prescribed surface fire/thinning). Models were run to determine the acreage and level of treatments necessary for restoration of NCI's Blue oak woodland and savanna sClasses to HRV. Each model was run for 100 years and for 100 Monte Carlo iterations.

Results

Conversion and Departure of Blue Oak Woodland Savannas

The extent of blue oak woodland savanna ranges from about 20% to more than 45% of the three ecoregions, and a third to a half of the blue oak savanna is now dominated by exotic species (primarily nonnative, annual grasslands) across regions. Urban development is most extensive in the SNF (6%) and conversion to crop agriculture has been minimal in these foothill woodlands.

The CCR ecoregion is 7,542,856 acres. Blue oak woodlands are historically believed to make up about 22 percent (1,678,189 acres) of this region. Of that only about 50 percent (842,226) remains today. Regionally, 41 percent (3,064,246 acres) of the land has been converted to exotic species, 9 percent (684,246 acres) of the land has been converted to agriculture and 5 percent (411,147 acres) to urban development (figure 3). Blue oak woodland and savannas have lost 44 percent (739,814 acres) to the establishment of exotic species, 1 percent (24,147 acres) to agriculture, and 4 percent (51,352 acres) to urban development (figure 3).



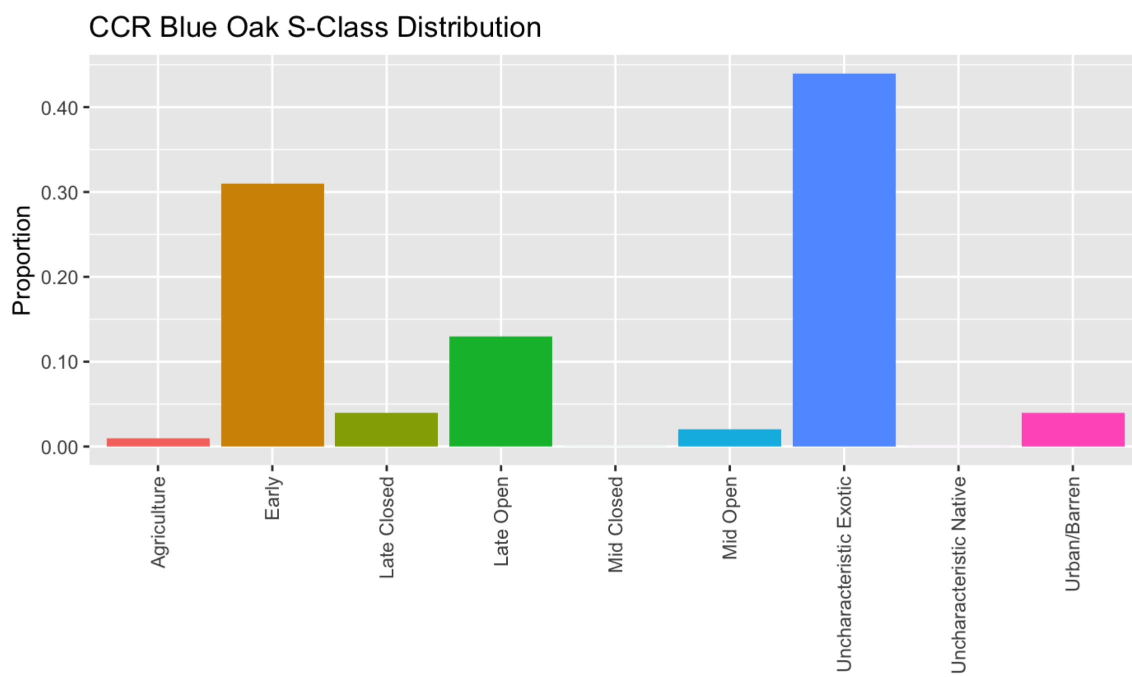
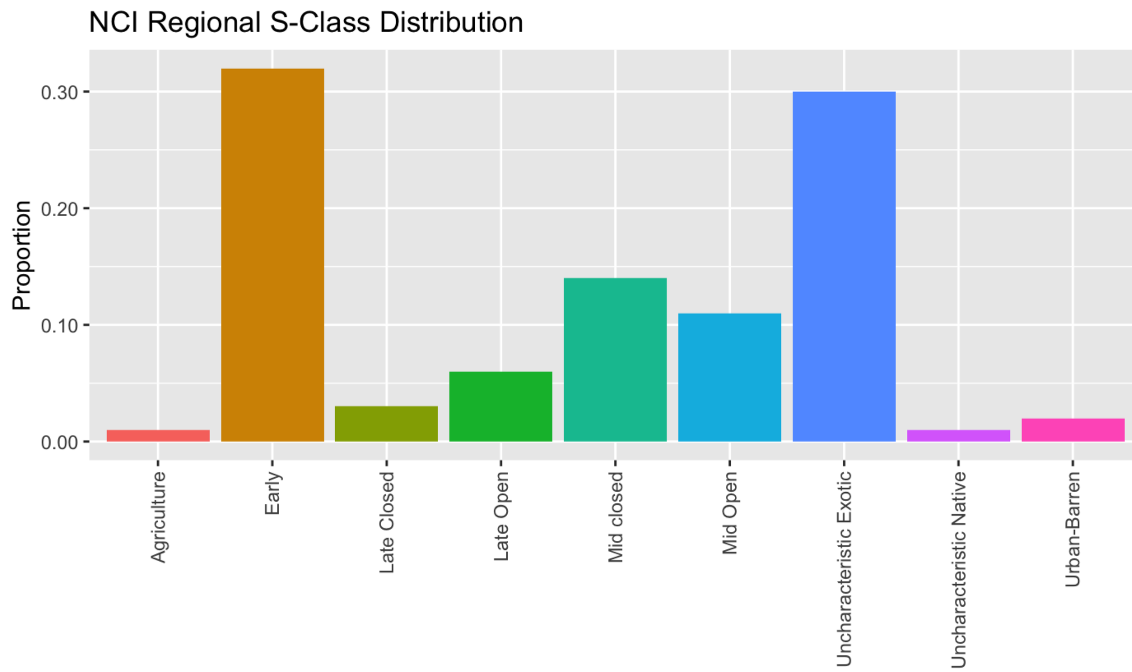


Fig. 3.3. Current land class cover for California's Central Coast Interior Ranges (CCR) as a region (top) and specific to the historical extent of the Blue oak woodlands and savannas (bottom). Nonnative-exotic species and Early successional states have come to dominate both regionally and the Blue oak woodlands and savannas. But Late Open states comprise a greater proportion of the landscape in Blue oak woodlands than regionally. Historically for Blue oak woodland and savannas, Mid Open states were the dominant successional stage.

The NCI ecoregion is the smallest of the three ecoregions encompassing 1,707,873 acres. Blue oak woodlands and savannas are historically believed to have accounted for approximately 47 percent (810,590 acres) of this ecoregion, with only 57 percent (460,085 acres) of that remaining. Forty-nine percent (512,793 acres) of the region has been converted to exotic species, only 1 percent (24,136 acres) to agriculture and 2 percent (42,418) to urban development (fig 4). Within the Oak woodlands and savannas, exotic species now cover 39 percent (317,479) of the landscape, agriculture 2 percent (16,357 acres) and urban development also 2 percent (16,451) (figure 4).



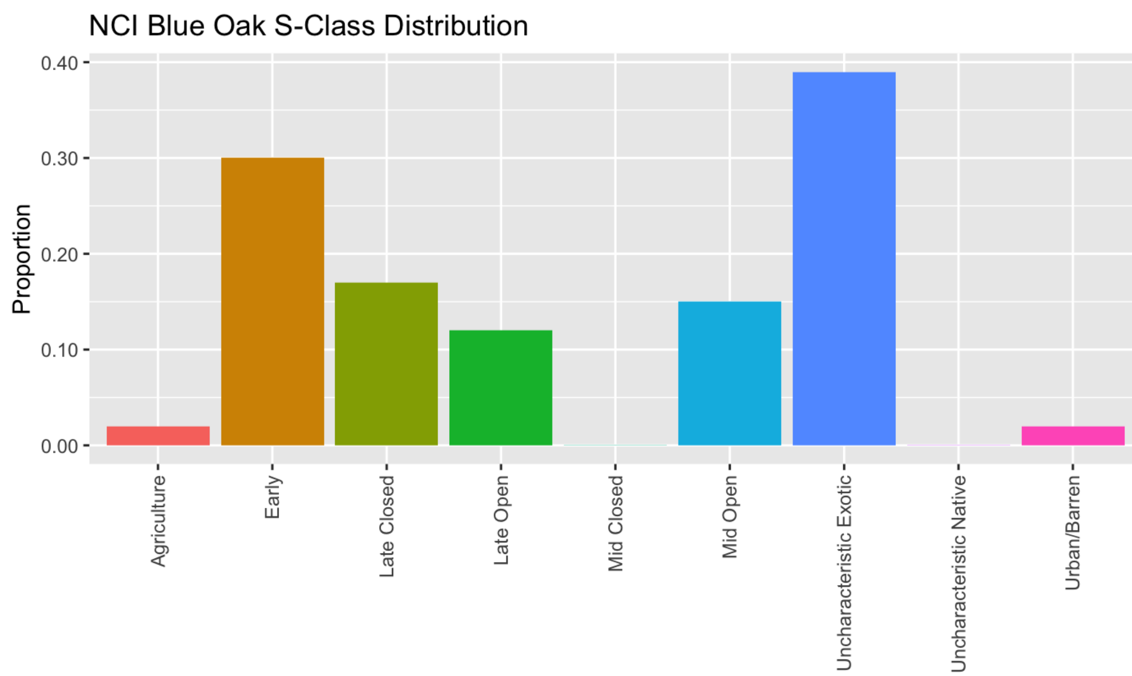
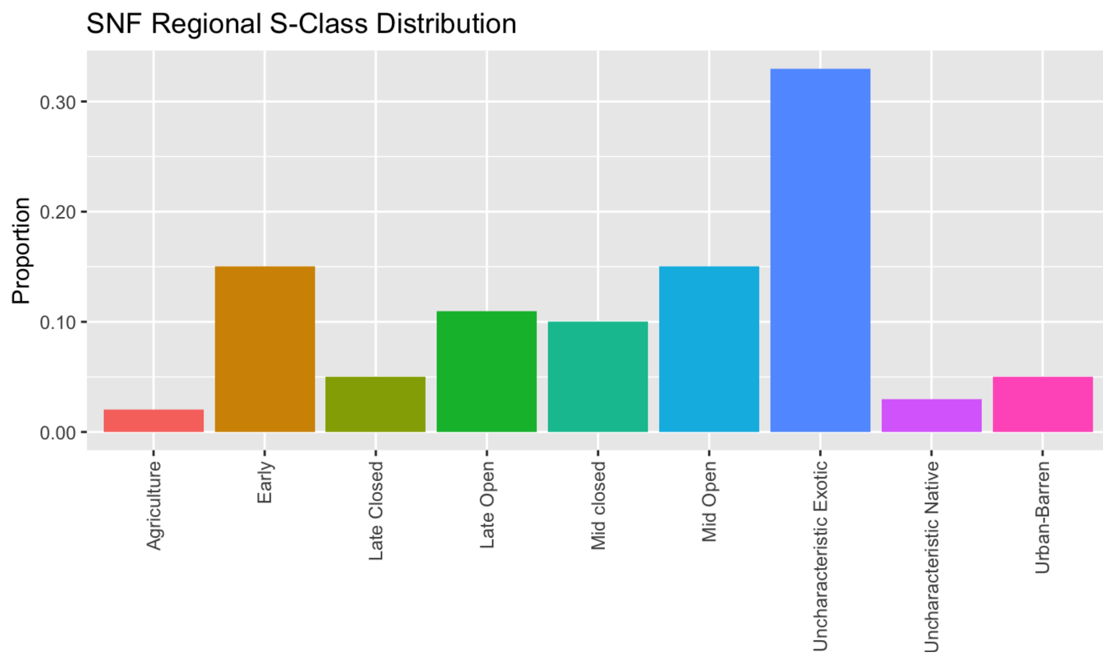


Fig. 3.4. Current land class cover for Northern California Interior Coast Ranges (NCI) as a region (top) and specific to historical extent of the Blue oak woodlands and savannas (bottom). The Early successional states dominate the region as whole, but Nonnative exotic species dominate the Blue oak woodlands and savannas. The Late Closed states comprise a greater proportion of the landscape in Blue oak woodlands than regionally.

The most extensive woodland region, the SNF region covers 5,593,909 acres. Historically, Blue oak woodland savannas are believed to have comprised about 42 percent (2,339,774 acres) of this region, of which 54 percent remains (1,271,338 acres). Regionally, 33 percent (2,007,398 acres) is now dominated by exotic species, 2 percent (98,365 acres) of the land has been converted to agriculture and 5 percent (264,790 to urban development (figure 5). Within the Blue oak woodland and savannas, 37 percent (873,561 acres) of the landscape is now covered by exotic species, 2 percent (50,031 acres) agriculture, and 6 percent (129,470 acres) urban development (figure 5).



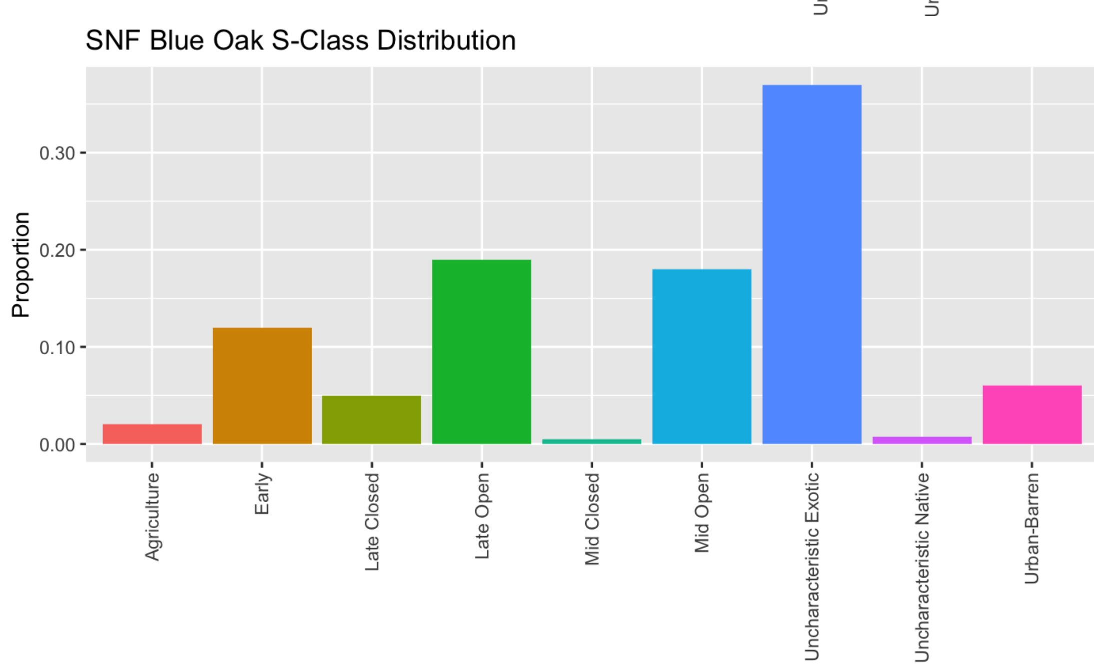


Fig. 3.5. Current land class cover for the Sierra Nevada Foothills as a region (top) and specific to historical extent of the Blue oak woodlands and savannas (bottom). Nonnative-exotic species dominate both regionally and the Blue oak woodlands and savannas. In contrast to the other ecoregions, the Mid and Late Open successional classes are still a prominent class on the landscape and account for larger proportions than the Early successional stages as they would have historically.

Blue oak woodlands and savanna HRV had a similar distribution across ecoregions. Figure 6 shows the typical distribution of sClasses for Blue oak woodland and savanna based on all three ecoregions. Historically, Mid Open and Late Open were the dominant sClasses, and the Early sClass was the least dominant.

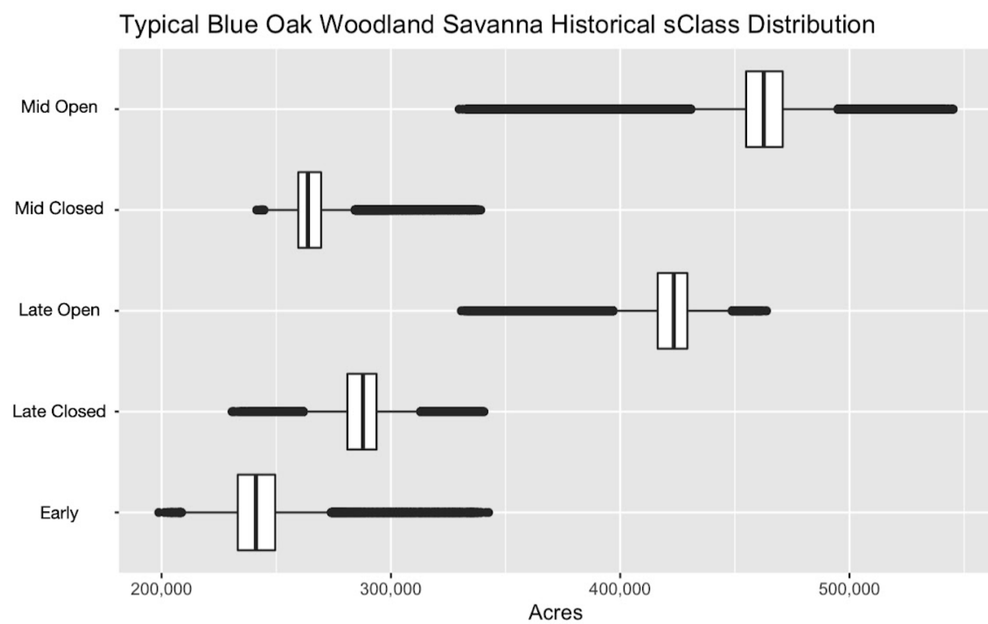


Fig. 3.6. Typical Blue oak Woodland and Savanna Historical sClass Distribution. Across all ecoregions, the historical range of variation (HRV) of Blue oak woodlands and savannas' successional class (sClass) were dominated by Mid Open and Late open sClasses, with the Early sClass making up the smallest proportion of the landscape historically. Range of values is based on 100 replications of the STSM.

Departure, however, varied by ecoregion. In the CCR ecoregion, the Early successional state is overrepresented both in acreage and proportion of the region (525,177 acres; 62 percent) relative to the predicted HRV. The Late Closed (69,254 acres; 8 percent), Mid Closed (194 acres, less than 1 percent), and Mid Open (32,587 acres; 4 percent) successional stages are all underrepresented both in acreage and proportion. The Late Open successional stage is underrepresented in acres, however it falls with HRV when looking at it proportionately (215,014 acres; 25 percent) (figure 7).

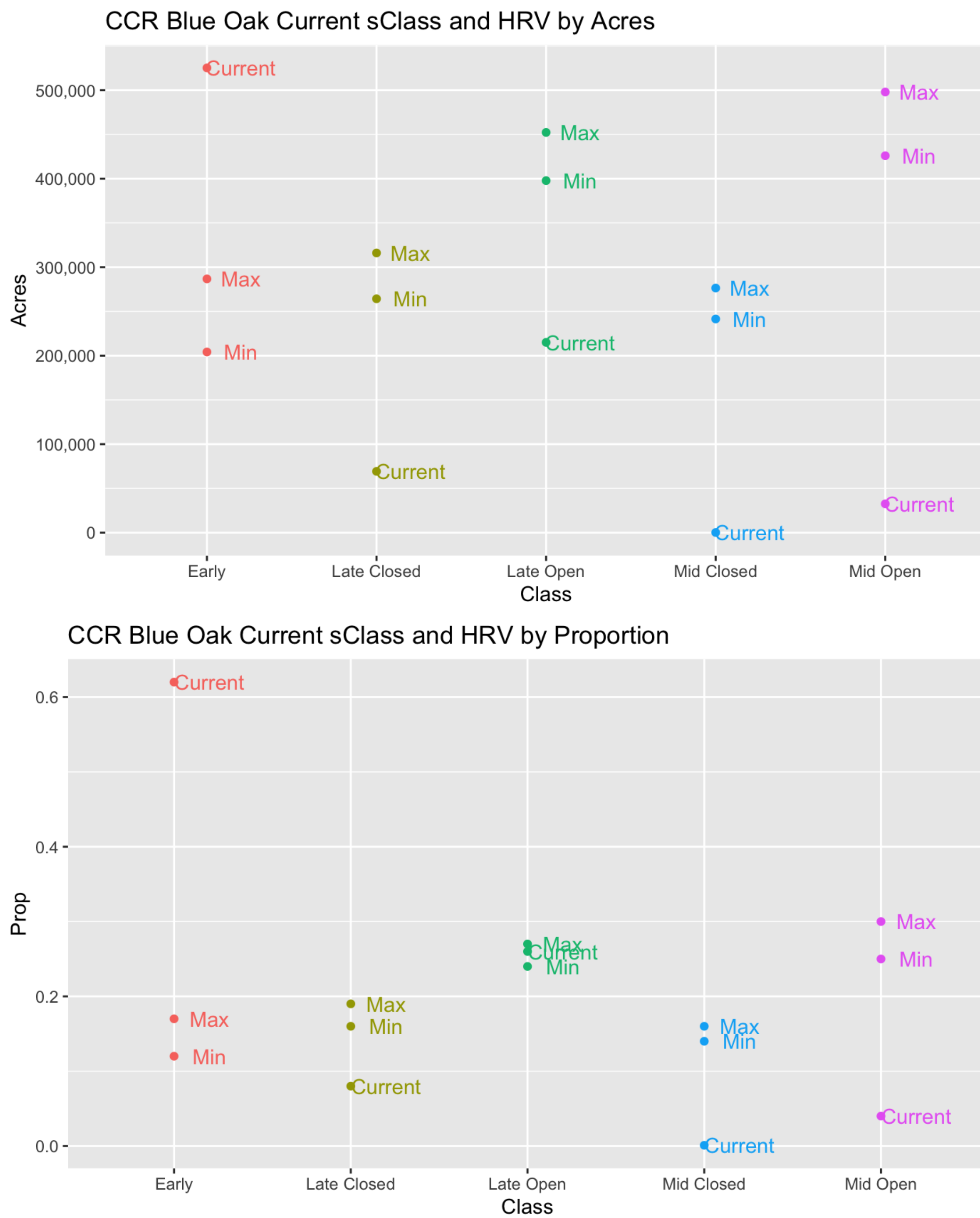


Fig.3.7. Current and historical distribution of successional classes for The Central Coast Interior Ranges (CCI) Blue oak woodlands and savannas. Currently, the Early successional class is overrepresented both in acreage and proportionately. The Late Open successional class is underrepresented in acreage but falls within historical proportions. All other successional classes are underrepresented both in acreage and proportionately.

The NCI ecoregion is similar to the CCR ecoregion in that the Early successional stage is overrepresented by both measures (226,508 acres; 49 percent) (figure 8). However, The Mid Open sClass is underrepresented in acreage (125,607 acres) but falls within HRV proportionately (27 percent). The other successional classes are underrepresented by both measures (Late Closed (13,228 acres; 3 percent), Late Open (94,534 acres; 21 percent), Mid Closed (208 acres; less than 1 percent).

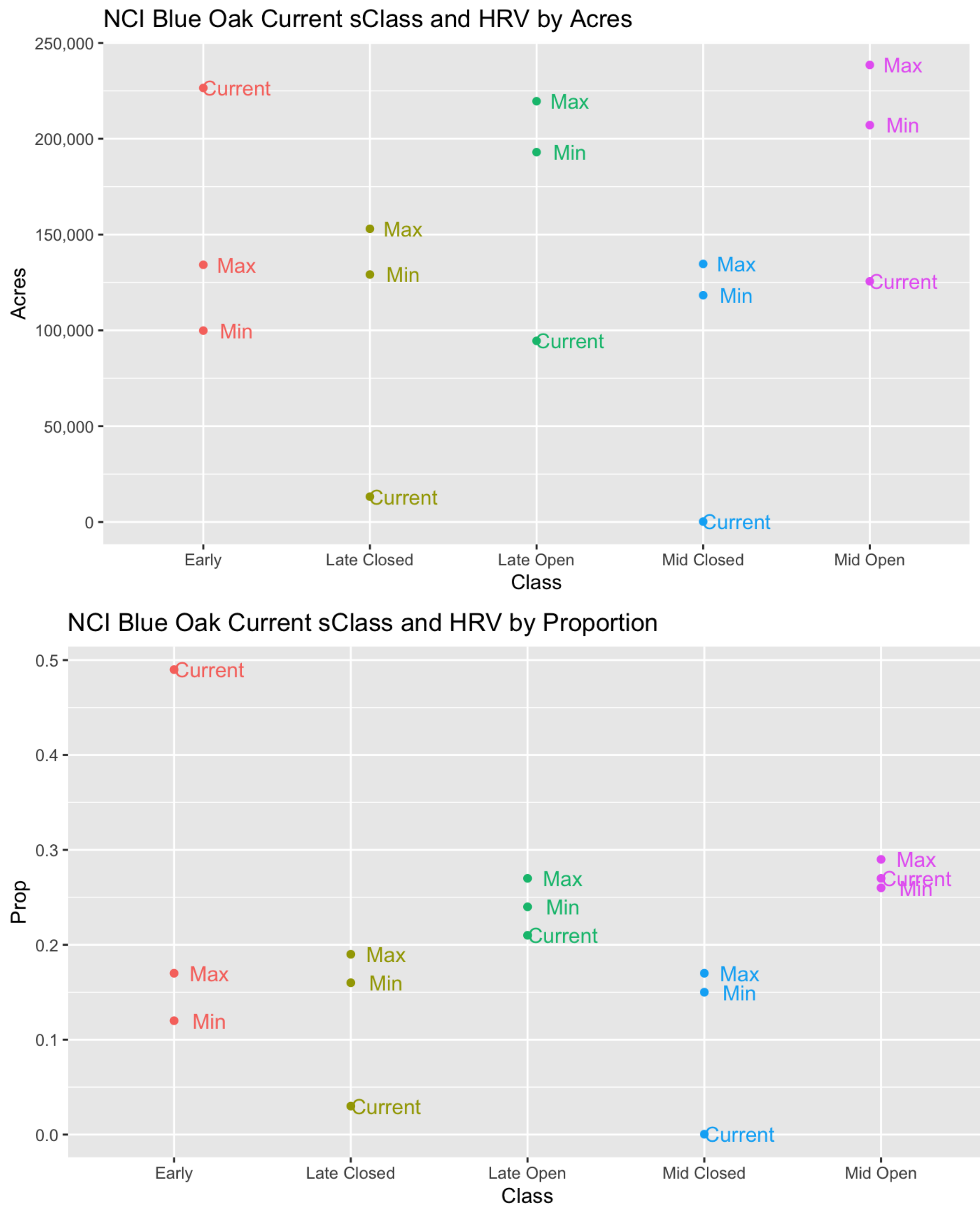
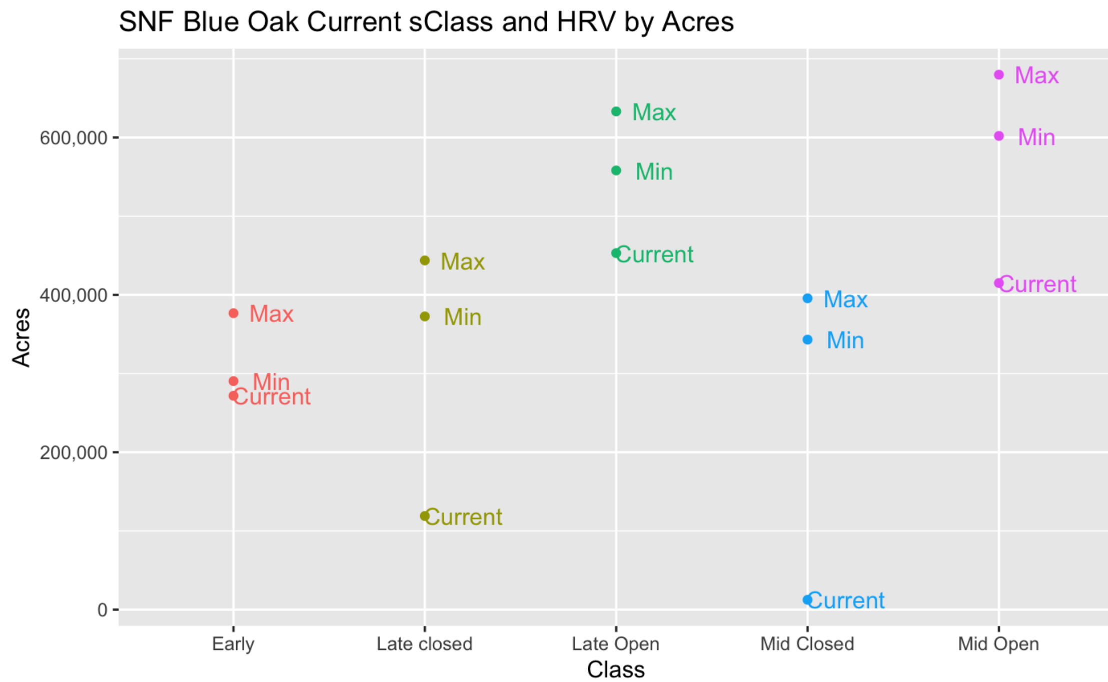


Fig. 3.8. Current and historical distribution of successional classes for Northern California Interior Ranges (NCI) Blue oak woodlands and savannas. The Early successional class is overrepresented both in acreage and proportionately. The Mid Open successional class is underrepresented in acreage but falls within historical proportions. All other successional classes are underrepresented both in acreage and proportionately.

The SNF ecoregion differs from the other regions. The Early successional stage is underrepresented by acres (271,766) but overrepresented proportionally (21 percent). The Late Closed and Mid Closed successional stages are underrepresented both in acreage (118,972 acres and 12,488 respectively) and proportions (Late Closed: 9 percent; Mid Closed: 1 percent). Both the Late Open and Mid Open are underrepresented in acreage (453,084 acres and 415,027 acres respectively) but overrepresented proportionally (Late Open: 36 percent; Mid Open:32 percent).



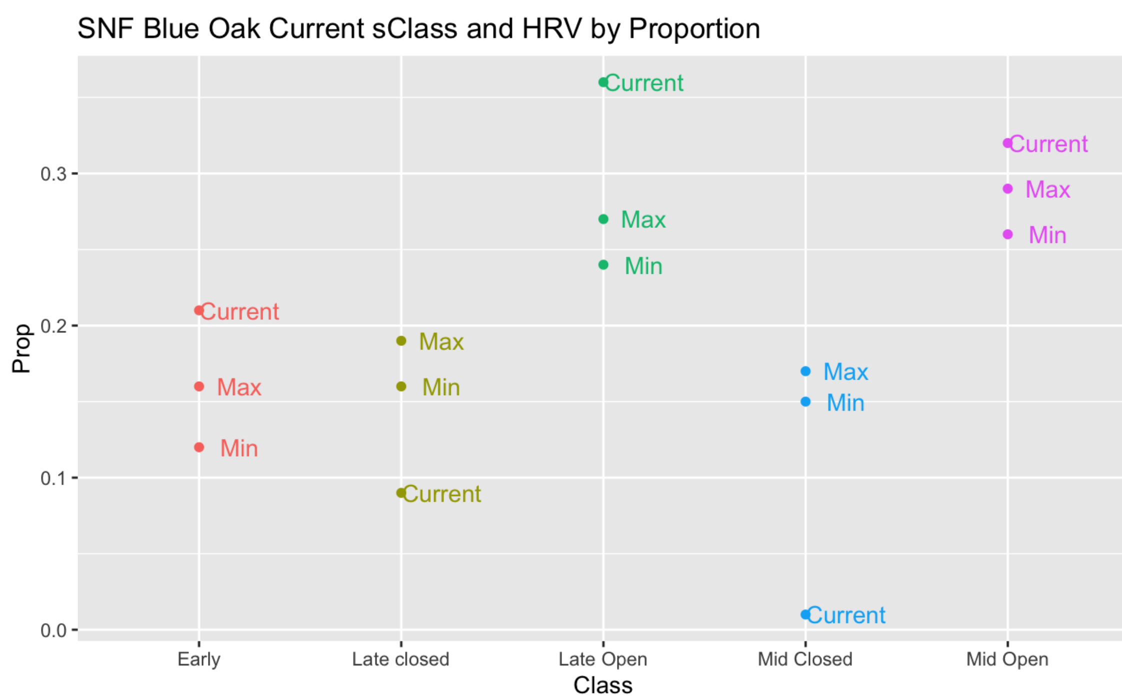


Fig. 3.9. Current and historical distribution of successional classes for The Sierra Nevada Foothills (SNF) Blue oak woodlands and savannas. Currently, all successional classes are underrepresented in acreage. The Early, Late and Mid Open successional classes are overrepresented proportionately. Both closed successional classes are underrepresented proportionately.

Restoration Needs Based on Departure Assessment

Each region's departure assessment indicated the need for restoration actions to restore Blue oak woodlands and savannas to within HRV of structural heterogeneity.

In the CCR ecoregion, the Early successional stage is greatly overrepresented and while the Late Open sClass is within HRV, all the other sClasses are underrepresented. Approximately 424,000 acres of the Early sClass are eligible for restoration treatments (Table 2; Figure 10) to encourage transition to another sClass. Selected areas of the Early sClass should receive the "Succession" only treatment allowing them to transition to the underrepresented Late Closed sClass which has approximately a 66,000 acre deficit and to the Mid Closed sClass which has 118,000 acre deficit. Approximately 178,000 acres need to be restored to return the Mid Open sClass to within HRV. Thus, portions of the Early sClass should also receive a "Succession and thinning" treatment to promote transition to the Mid Open sClass. Small portions of the Late Open sClass, which is currently within HRV, should be left undisturbed to promote succession to the Late Closed sClass.

Table 3.2. Central Coast Interior Ranges Restoration Needs Assessment.

Class	Current Acres	Current Prop	Min Acreage w/i Min HRV Prop	Acre Deficit/Surplus	Restoration Treatment
Early	525,177	0.62	101,067	+/424,110	Succession
Mid Open	32,587	0.04	210,557	-/177,970	Succession + Thinning
Late Open	215,014	0.26	202,134	w/i	Surface Fire/Thinning
Mid Closed	194	0.001	117,912	-/117,718	Succession
Late Closed	69,254	0.08	134,756	-/65,502	Succession

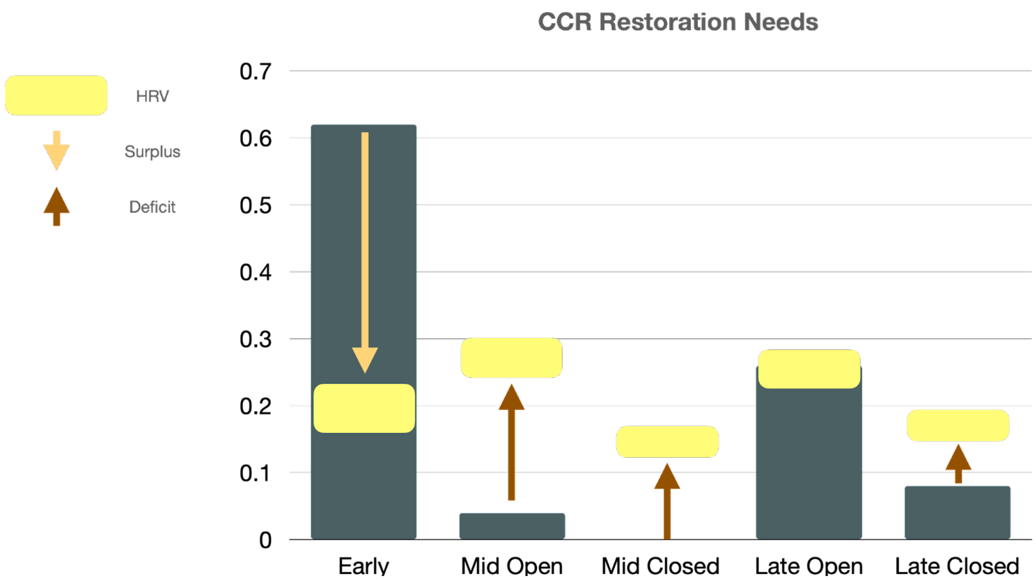


Fig. 3.10. The Central Coast Interior Ranges (CCI) Blue oak woodlands and savannas restoration needs assessment. Within CCI Blue oak woodlands, the Early successional class is overrepresented by approximately 424,110 acres thus requiring the "succession" treatment to return it to within HRV proportions. The Late Open successional class is with HRV requiring "Surface Fire/Thinning" treatment to maintain it within HRV. The Mid Open successional class is below HRV by approximately 177, 718 acres requiring a restoration treatment of "Succession + thinning" to return it to within HRV. Both closed canopy states are underrepresented with Mid Closed 117,718 acres below and Late Closed 65,502 acres below necessitating the "Succession" treatment be implemented.

The NCI ecoregion also has a greatly overrepresented Early sClass. In this region, approximately 171,000 acres of the Early sClass are eligible for restoration treatment (Table 3; Figure 11). The "Succession" only treatment is necessary for the reestablishment of the Late Closed which has the greatest deficit at approximately 116,000 acres, and the Mid Closed sClass which has a deficit of approximately 73,000 acres. Smaller portions of the landscape should also receive the "Succession and thinning" treatment to promote a shift into the Late Open sClass which currently has a deficit of approximately 16,000. Portions of the Mid Open sClass, which is within HRV, should receive the "Succession" only treatment to allow for transition to the Mid Closed sClass.

Table 3.3. Northern California Interior Coastal Ranges Restoration Needs Assessment.

Class	Current Acres	Current Prop	Min Acreage w/i Min HRV Prop	Acre Deficit/Surplus	Restoration Treatment
Early	226508	0.49	55,210	+/171,298	Succession
Mid Open	125,607	0.27	119,622	w/i	Succession
Late Open	94534	21	110,420	-/15,886	Succession + Thinning
Mid Closed	208	0.0004	73,614	-/73,406	Succession
Late Closed	13,228	0.03	129,694	-/116,466	Succession

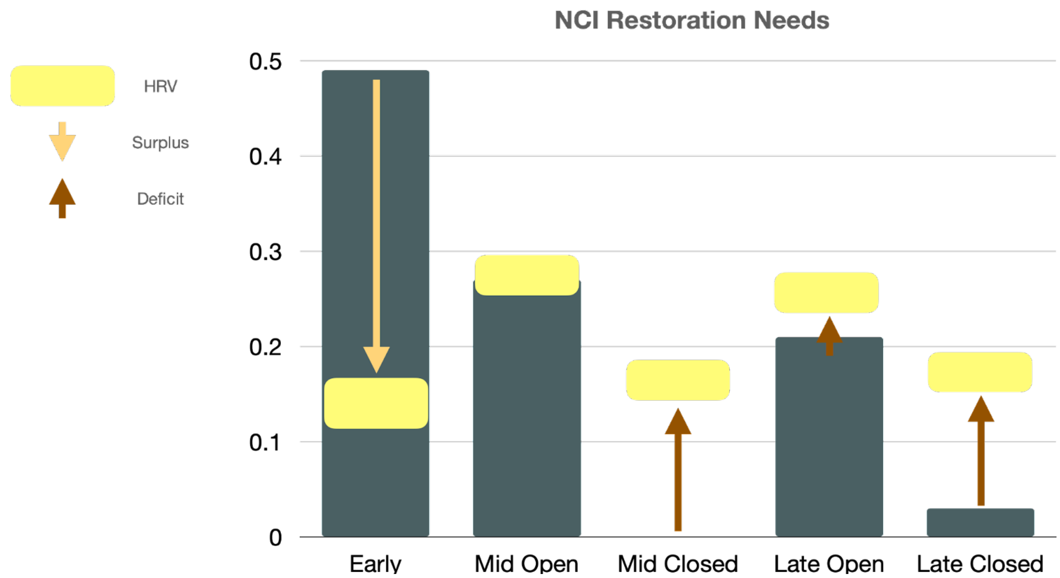


Fig. 3.11. The Northern California Interior Coast Ranges (NCI) Blue oak woodlands and savannas restoration needs assessment. Within NCI Blue oak woodlands, the Early successional class is overrepresented by approximately 171,298 acres thus requiring the "succession" treatment to return it to within HRV proportions. The Mid Open successional class is within HRV requiring "Surface Fire/Thinning" treatment to maintain it within HRV. All other successional classes are below HRV. The Late Open successional class has a deficit of approximately 15,886 acres requiring a restoration treatment of "Succession + thinning" to return it to within HRV. Similar to CCR both closed canopy states are underrepresented with Mid Closed 73,406 acres below and Late Closed 116,466 acres below HRV, thus, the "Succession" treatment should be implemented.

The SNF ecoregion requires a slightly different approach. While the Early sClass is overrepresented with a surplus of approximately 119,000 acres (Table 4; Figure 12), it is closer to HRV proportions than the other ecoregions. In SNF, the Late and Mid Open sClass are also overrepresented – the Late Open more so – with a surplus of approximately 148,000 acres. The Mid Open SClass has a surplus of approximately 84,000 acres. Both Closed sClasses are underrepresented, the Mid Close sClass with an approximate deficit of 178,000 acres and the Late Closed with one of approximately 84,000 acres, thus suggesting the need to primarily apply the “Succession” only treatment and reduce disturbances to the open sClasses so that they may transition to more closed states.

Table 3.4. Sierra Nevada Foothills Restoration Needs Assessment.

Class	Current Acres	Current Prop	Min Acreage w/i Min HRV Prop	Acre Deficit/Surplus	Restoration Treatment
Early	271,766	0.21	152,561	+/119,205	Succession
Mid Open	415,027	0.32	330,548	+/84,479	Succession
Late Open	453,084	0.36	305,121	+/147,963	Succession
Mid Closed	12,488	0.01	190,701	-/178,213	Succession
Late Closed	118,972	0.09	203,414	-/84,442	Succession

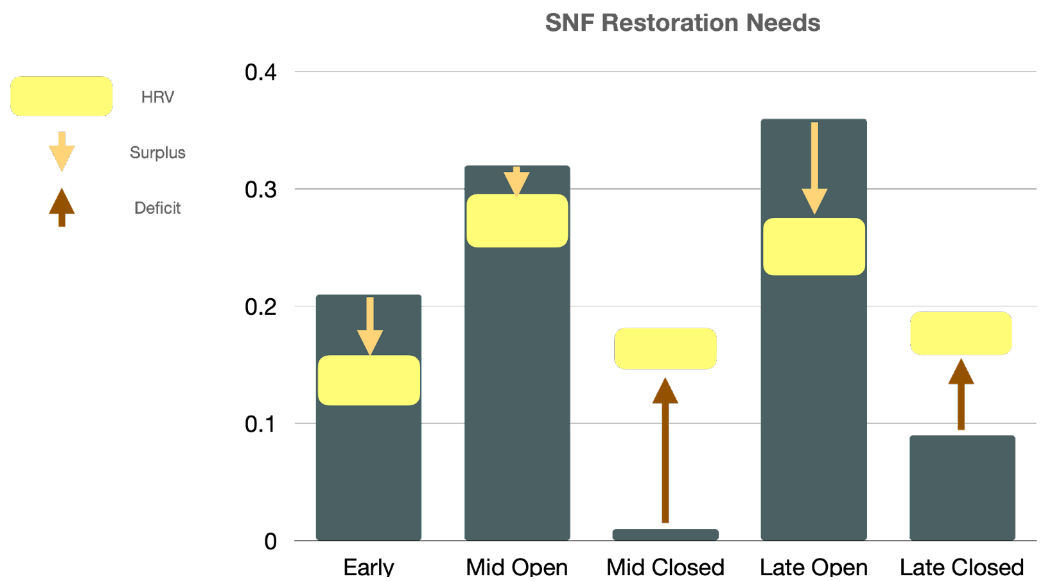


Fig.3.12. The Sierra Nevada Foothills Blue oak woodlands and savannas restoration needs assessment. For the Sierra Nevada Foothills Blue oak woodlands, the Early successional class is overrepresented by approximately 119,205 acres thus requiring the “Succession” treatment to return it to within HRV proportions. The Late and Mid Open successional classes are overrepresented with surpluses of 147,963 and 84,479 acres respectively, again necessitating the “Succession” treatment to return them to within HRV. Both closed canopy states are underrepresented with Mid Closed 178,213 acres below and Late Closed 84,442 acres below also requiring the “Succession” treatment.

STSM Restoration Treatment Assessment Case Study

The use of STSM to determine the best management approach and effort necessary to restore the structural diversity of the Blue oak woodlands and savannas of the NCI ecoregion demonstrated that multiple treatments were necessary. First, to allow for the reestablishment of later successional stages, the "succession" treatment needed to be implemented and disturbances needed to be reduced. In Blue oak woodlands and savannas, fire return intervals are key to successional progression in the STSM, with longer intervals allowing for early stages to transition and mature into later stages. Using current fire return intervals determined through prior analysis (Ch.2), we adjusted fire return intervals to half of their current levels.

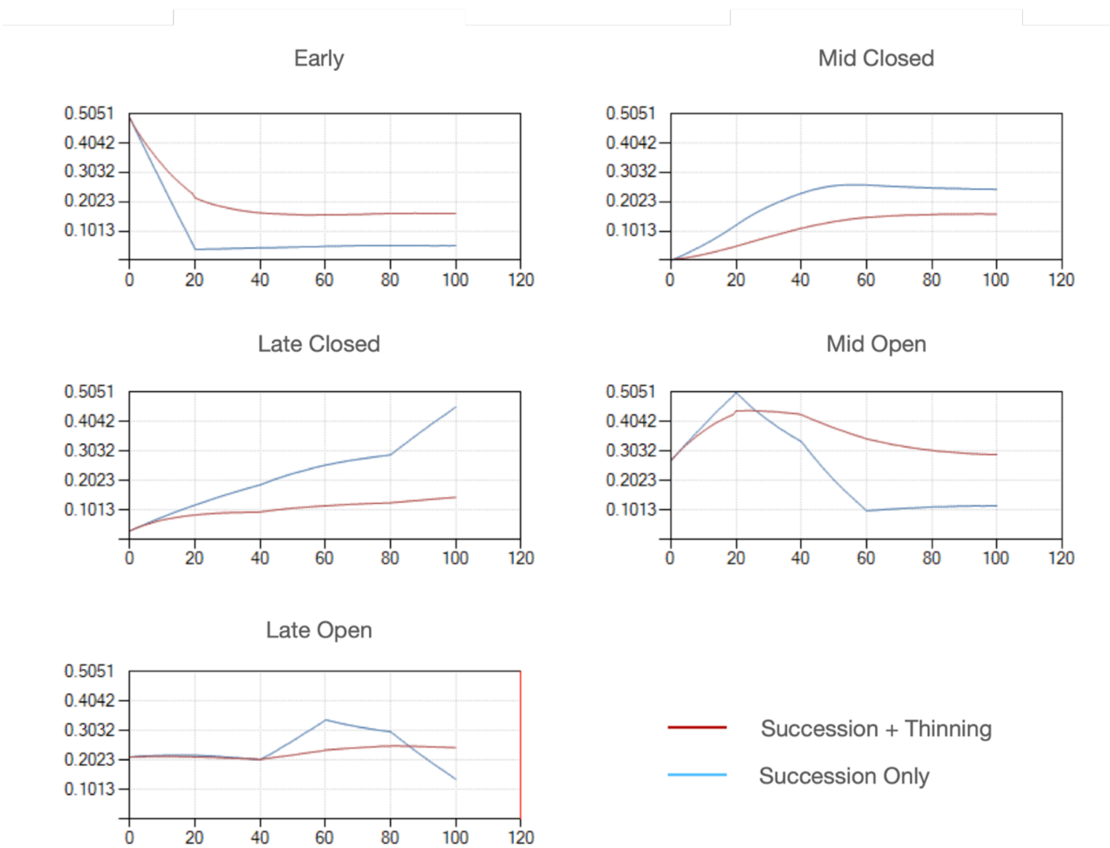


Fig. 3.13. State and Transition Simulation Modeling Restoration Treatment Assessment for the Northern California Interior Coastal Ranges Blue oak woodlands and savannas ecoregion. Through modeling, we determined that just implementing the Succession only treatment (Blue line) resulted in an underrepresentation of the Early, Mid Open, and Late Open successional states and overrepresentation of Late and Mid Closed successional states. Incorporating strategic thinning of 1,000 acres to both the Mid and Late Open states, and of 500 acres to both the Late and Mid closed states of 100 years or older resulted in the return of NCI's Blue oak woodlands and savannas to within historical ranges of structural diversity.

But if allowing for succession was the only treatment applied, then the Late and Mid Closed sClasses came to dominate, becoming overrepresented with regard to HRV (Figure 13, Blue line). Additionally, the Early sClass decreased dramatically and while the Mid and Late Open sClasses saw initial gains, over time they started to drop below HRV. So, following our restoration needs assessment, we introduced targeted thinning treatments. We found that two additional treatments were necessary: 1) targeted low-intensity thinning treatments of 1,000 acres per year for Mid Open areas and Late Open of over 30 years (a relatively small proportion of current extent – only 2/10ths of a percent), and 2) targeted Replacement/mechanical thinning treatments of areas of 100 years or older in the late sClasses of 500 acres each. These three treatments together returned the Blue oak woodlands and savannas to within structural HRV (Figure 13, Red line).

Discussion

In all three ecoregions, Blue oak woodlands and savannas have experienced higher rates of conversion to exotic species than their respective regions as a whole, confirming part 1 of our first hypothesis. And while Blue oak woodlands and savannas do account for large portions of each region, that disproportionately more of these landscapes have experienced conversion of land cover to exotic species than other vegetation types reconfirms findings that oak woodlands are an especially threatened landscape (Wilson, Sleeter, and Davis 2015). These rates may also be underreported as the methods used to classify the imagery used for our analysis do not reliably detect exotic species below canopy covers (Rollins 2009).

Part 2 of our first hypothesis is only partially supported with the SNF and NCI ecoregions both having higher rates of land use conversion to agriculture than overall regional rates in accordance with our hypothesis. But in the CCR ecoregion, Blue oak woodland savannas have actually seen lower conversion to agriculture rates than the region as a whole. The CCR ecoregion has an extensive history of agriculture, a slightly milder climate influenced by maritime conditions (Thorne et al. 2002), and also has the lowest proportion of Blue oak woodlands and savannas (22 percent) of all the ecoregions. Perhaps much of the other vegetation communities of this region have also been converted to agricultural land use. A targeted study of risk in this ecoregion could look at conversion rates for each of the different BpS classifications of the region to determine which systems have been most affected by conversion to other land uses and which are most at risk.

The CCR region experienced the highest conversion to exotic species (44%), while SNF had the greatest conversion to urban development (6 percent). These regional differences in land cover/land use conversion, although not great, are interesting and likely explained by the geography of the regions. For example, the Sierra Nevada Foothills are adjacent to, and within commuting distance of, several large urban centers located in the central valley and are adjacent to three of the state's most revered national parks: Yosemite, Kings Canyon, and Sequoia. This has likely contributed to the growing exurban development in the foothills (McBride, Russell, and Kloss 1996; Orlando 2008).

Future research could explore these differences more thoroughly, perhaps looking at the post-European settlement history of the regions, exurban development, and land use planning and policies.

Support for our second hypothesis varied by region. All ecoregions do indeed have a high proportion of the landscape in the Early sClass – CCR with the highest at 62 percent. This is considerably higher than the SNF (21 percent) and NCI (49 percent) ecoregions. This may be related to the region's less extensive Blue oak woodland and savanna cover coupled with its agricultural and ranching history. Blue oak woodland and savanna landscapes may have been more attractive for clearing than nearby shrublands and forests due precisely to its characteristic lower tree densities and more open, "park-like" settings (Alagona 2008).

SNF has the highest rates of private land ownership of all the ecoregions – approximately 80 percent (Thorne et al. 2002; (Huntsinger et al. 2004). The fact that much of the Blue oak woodlands in SNF are privately held ranchlands might explain why it varies so much in Departure from HRV than the other two regions. Both the more open woodland states are slightly overrepresented proportionately in this region (although not by acreage), while the more closed states are underrepresented. Privately owned ranchlands represent extensive tracts of this landscape that have remained relatively intact with the presence of grazing as the predominant disturbance. Grazing by cattle may be having an effect analogous to low-intensity surface fires, keeping the landscape free from shrubs, young trees, herbaceous growth, and other dead fuels such as leaf litter (Bond and Keeley 2005; McCreary and George 2005; Sulak and Huntsinger 2007). This form of disturbance may be maintaining the characteristic open “park-like settings” of Blue oak woodlands (Jepson 1923; Muir 1979; Lewis, Bean, and Lawton 1973; Anderson 2006; Sulak and Huntsinger 2007). It may also be preventing succession to the closed canopy sClasses that are underrepresented in this region. While research indicates grazing has negative impacts on oak regeneration (Tyler, Kuhn, and Davis 2006), research is conflicting about the effect of grazing specifically on blue oaks. Some findings suggest it may be partially responsible for declines in recruitment (Bartolome, McClaran, and Allen-Diaz 2002), while other results indicate that grazing at low to moderate densities has little effect (Allen-Diaz and Bartolome 1992; Lillian M. Hall et al. 1992; Tecklin, Connor, and Mccreary n.d.; McCreary and George 2005; Koenig and Knops 2007), and perhaps in some cases, a positive effect (Lillian M. Hall et al. 1992). In addition to stocking densities, seasonality of grazing appears to be an important factor with summer grazing being the most detrimental (Lillian M. Hall et al. 1992; McCreary and George 2005). Future research could more thoroughly compare and contrast the grazing practices and histories among the regions and compare grazing effects, seasonality, and small mammal herbivory with low-intensity surface fires.

We have demonstrated the use of HRV, current Departure (both of vegetation structure and fire severity distributions), and STSM modeling in a restoration needs assessment to inform restoration treatments. We determined in our case study that current fire return intervals needed to be reduced by approximately half in the Northern California interior coast ranges region. This differs from some findings in western forested regions where there are fire deficits (Mallek et al. 2013; R. D. Haugo et al.

2019). However, we found that reducing disturbance was insufficient and resulted in the predominance of later, more closed canopies well outside of HRV (Figure 13). We found that when we coupled fire suppression with targeted low-intensity "thinning" of 1,000 acres each of the Open-canopy states and high-intensity "replacement" treatments of 500 acres of each of the Closed-Canopy states per year, we were able to maintain the heterogeneous mosaic of sClass distributions within their historical distribution. Due to the extensive conversion and changes these landscapes have experienced, restoring historic fire regimes may not be possible or even beneficial (Dudley et al. 2020). And, while we have estimates of historic fire return intervals, many of the low-intensity, surface fires – the largest component of historical fire regimes in these landscapes (Van de Water and Safford 2011) – were likely applied locally and strategically, by indigenous peoples (Anderson 2006; Klimaszewski-Patterson et al. 2018). Our modeling approach allows for the exploration of how a more strategic application of disturbance can help influence Blue oak woodlands and savannas structural diversity.

Our findings do indeed suggest the need for some limited reforestation of Blue oak woodlands and savannas – with a few caveats. Restoration needs for each of these regions primarily requires a reduction of disturbance so that later successional stages can reestablish (reforestation) – this is true for all ecoregions. However, the reintroduction of low-severity disturbances, such as surface fires or targeted grazing and/or thinning activities are also necessary to maintain open sClasses in CCR and NCI and will likely continue to be necessary to maintain these states in SNF. This will allow for succession but with reduced tree density and competition resulting in more resilient landscapes (Fry

2008). And while our results indicate that a regional approach is likely necessary for the restoration of Blue oak woodland savannas, overwhelmingly, the greatest restoration need across regions is allowing for succession (exclude fire, grazing, and other disturbances). As seen in our previous study, fire has been increasing in extent and intensity in these regions (Syphard, Keeley, and Abatzoglou 2017). Reducing disturbance in these regions will likely require not only more ecologically informed grazing practices, but also more extensive fire suppression actions – beneficial both ecologically and sociologically. Targeted local prescribed burning or thinning practices might be beneficial in more ways than one, allowing for transitions to open canopy states and also reducing the risk of high-intensity replacement fires which return both open and closed states to the currently overrepresented Early sClass (McCreary and George 2005; Keane et al. 2019; Syphard, Keeley, and Abatzoglou 2017).

Our study was broad in scale and likely missed finer-scale impacts on Blue oak woodland and savanna structure, conversion, and restoration needs. For example, while the "urban" category in the land use data from the Monitoring Trends in Burn Severity Project (Eidenshink et al. 2007) does detect small towns, it does not account for the subdivision of ranches into smaller residential properties. Additional spatially heterogeneous environmental characteristics within regions, such as slope, aspect, elevation, soils, etc., are crucial to consider prior to the application of restoration treatments. While our suggestions are meant to inform restoration efforts broadly, site-specific applications will have to take into consideration these finer-scale factors. Further research should seek to incorporate these factors both in estimates of departure and in assessing where restoration treatments will be most effective.

Our findings confirm that Blue oak woodlands and savannas have experienced high rates of conversion to other land uses and vegetation - close to or greater than 50 percent in some regions. In what remains of Blue oak woodlands and savannas, much has departed from historical ranges of within-system structural variation, with a large over representation of Early successional classes and no region completely within HRV. Thus, these landscapes require active restoration efforts, including primarily the reduction of disturbances in order to allow woodland succession to proceed, and the spatially limited reintroduction of low-severity thinning to expand Mid and Late Open classes – whether that be through prescribed fire, strategic grazing, and/or mechanical thinning of vegetation.

Despite limitations, our study provides broadscale insight into the conversion, departure, and restoration needs of a sociologically, economically, culturally, and ecologically important landscape. The history and ecology of Blue oak woodlands and savannas are complex, involving multiple drivers of change and extensive human use and management, both pre- and post-European settlement. Although our work could not, and was not intended to, conclusively demonstrate that these landscapes are an anthropogenic landscape, we do demonstrate that they require active ecological management if we are to return them to the open "park-like settings" that Indigenous peoples and early European settlers enjoyed and depended upon.

Works Cited

Ackerly, Kozanitas, Oldfather, and Papper. 2019. "Mortality and Resprouting in California Oak Woodlands Following Mixed-Severity Wildfire." *International Business Review (Oxford, England)*.

Alagona, Peter S. 2008. "Homes on the Range: Cooperative Conservation and Environmental Change on California's Privately Owned Hardwood Rangelands." *Environmental History* 13 (2): 325–49.

Allen-Diaz, Barbara H., and James W. Bartolome. 1992. "SURVIVAL OF QUERCUS DOUGLASII (FAGACEAE) SEEDLINGS UNDER THE INFLUENCE OF FIRE AND GRAZING." *Madroño* 39 (1): 47–53.

Anderson, M. K. 2006. "The Use of Fire by Native Americans in California." *Fire in California's Ecosystems*, 417–30.

Bailey, Robert G. 1998. *Ecoregions of North America*. U.S. Department of Agriculture, Forest Service.

Bartolome, McClaran, and Allen-Diaz. 2002. "Effects of Fire and Browsing on Regeneration of Blue Oak." *Symposium on Oak ….* https://www.fs.usda.gov/treesearch/pubs/26130.

Beckmann, Jill J. 2019. "Competition, Climate, and Drought Effects on Tree Growth in an Encroached Oak Woodland in Northern California." Cal Poly Humboldt.

Bestelmeyer, Ash, and Brown. 2017. "State and Transition Models: Theory, Applications, and Challenges." *Rangeland Ecology & Management*.

Blankenship, Frid, and Smith. 2015. "A State-and-Transition Simulation Modeling Approach for Estimating the Historical Range of Variability." *AIMS Environmental Science*.

Blankenship, Kori, Randy Swaty, Kimberly R. Hall, Sarah Hagen, Kelly Pohl, Ayn Shlisky Hunt, Jeannie Patton, Leonardo Frid, and Jim Smith. 2021. "Vegetation Dynamics Models: A Comprehensive Set for Natural Resource Assessment and Planning in the United States." *Ecosphere* 12 (4). https://doi.org/10.1002/ecs2.3484.

Bond, William J., and Jon E. Keeley. 2005. "Fire as a Global 'Herbivore': The Ecology and Evolution of Flammable Ecosystems." *Trends in Ecology & Evolution* 20 (7): 387–94.

Cooke, Steven J., Joseph R. Bennett, and Holly P. Jones. 2019. "We Have a Long Way to Go If We Want to Realize the Promise of the 'Decade on Ecosystem Restoration.'" *Conservation Science and Practice* 1 (12). https://doi.org/10.1111/csp2.129.

Daniel, Colin J., Leonardo Frid, Benjamin M. Sleeter, and Marie-Josée Fortin. 2016. "State-and-transition Simulation Models: A Framework for Forecasting Landscape Change." *Methods in Ecology and Evolution / British Ecological Society* 7 (11): 1413–23.

Daniel, and Frid. 2012. "Predicting Landscape Vegetation Dynamics Using State-and-Transition Simulation Models." *The First Landscape State-and-Transition Simulation …*. https://www.fs.usda.gov/treesearch/pubs/42566.

Das, Adrian J., Nicholas J. Ampersee, Anne H. Pfaff, Nathan L. Stephenson, Tedmund J. Swiecki, Elizabeth A. Bernhardt, Patricia K. Haggerty, and Koren R. Nydick. 2020. "TREE MORTALITY IN BLUE OAK WOODLAND DURING EXTREME DROUGHT IN SEQUOIA NATIONAL PARK, CALIFORNIA." *Madrono; a West American Journal of Botany* 66 (4): 164–75.

Dudley, Nigel, Luca Eufemia, Martina Fleckenstein, Maria E. Periago, Ilka Petersen, and Jean F. Timmers. 2020. "Grasslands and Savannahs in the UN Decade on Ecosystem Restoration." *Restoration Ecology* 28 (6): 1313–17.

Dwomoh, Francis K., Jesslyn F. Brown, Heather J. Tollerud, and Roger F. Auch. 2021. "Hotter Drought Escalates Tree Cover Declines in Blue Oak Woodlands of California." *Frontiers in Climate* 3. https://doi.org/10.3389/fclim.2021.689945.

Eidenshink, Jeff, Brian Schwind, Ken Brewer, Zhi-Liang Zhu, Brad Quayle, and Stephen Howard. 2007. "A Project for Monitoring Trends in Burn Severity." *Fire Ecology* 3 (1): 3–21.

Elena-Rosselló, Ramón, Maggi Kelly, Sergio González-Avila, Alexandra Martín, David Sánchez de Ron, and José M. García del Barrio. 2013. "Recent Oak Woodland Dynamics: A Comparative Ecological Study at the Landscape Scale." In *Mediterranean Oak Woodland Working Landscapes: Dehesas of Spain and Ranchlands of California*, edited by Pablo Campos, Lynn Huntsinger, Jose Luis Oviedo Pro, Paul F. Starrs, Mario Diaz, Richard B. Standiford, and Gregorio Montero, 427–59. Dordrecht: Springer Netherlands.

Fry, Danny L. 2008. "Prescribed Fire Effects on Deciduous Oak Woodland Stand Structure, Northern Diablo Range, California." *Rangeland Ecology & Management* 61 (3): 294–301.

Hankins. 2015. "Restoring Indigenous Prescribed Fires to California Oak Woodlands." *Gen. Tech. Rep. PSW-GTR-251. Berkeley, CA: US.*

Haugo, Ryan D., Bryce S. Kellogg, C. Alina Cansler, Crystal A. Kolden, Kerry B. Kemp, James C. Robertson, Kerry L. Metlen, Nicole M. Vaillant, and Christina M. Restaino. 2019. "The Missing Fire: Quantifying Human Exclusion of Wildfire in Pacific Northwest Forests, USA." *Ecosphere* 10 (4): e02702.

Haugo, Ryan, Chris Zanger, Tom DeMeo, Chris Ringo, Ayn Shlisky, Kori Blankenship, Mike Simpson, Kim Mellen-McLean, Jane Kertis, and Mark Stern. 2015. "A New Approach to Evaluate Forest Structure Restoration Needs across Oregon and Washington, USA." *Forest Ecology and Management* 335 (January): 37–50.

Huntsinger, Lynn, Adriana Sulak, Lauren Gwin, and Tobias Plieninger. 2004. "Oak Woodland Ranchers in California and Spain: Conservation and Diversification." *Advances in Geoecology* 37: 309–26.

Jepson, Willis Linn. 1923. *The Trees of California*. Sather gate Bookshop.

Keane, Robert E., Kathy Gray, Brett Davis, Lisa M. Holsinger, and Rachel Loehman. 2019. "Evaluating Ecological Resilience across Wildfire Suppression Levels under Climate and Fuel Treatment Scenarios Using Landscape Simulation Modelling." *International Journal of Wildland Fire* 28 (7): 533–49.

Keane, Robert E., and Rachel Loehman. 2019. "Historical Range and Variation (HRV)." In *Encyclopedia of Wildfires and Wildland-Urban Interface (WUI) Fires*, edited by Samuel L. Manzello, 1–12. Cham: Springer International Publishing.

Klimaszewski-Patterson, Anna, and Scott Mensing. 2020. "Paleoecological and Paleolandscape Modeling Support for Pre-Columbian Burning by Native Americans in the Golden Trout Wilderness Area, California, USA." *Landscape Ecology* 35 (12): 2659–78.

Klimaszewski-Patterson, Anna, Christopher T. Morgan, and Scott Mensing. 2021. "Identifying a Pre-Columbian Anthropocene in California." *Annals of the Association of American Geographers. Association of American Geographers* 111 (3): 784–94.

Klimaszewski-Patterson, Anna, Peter J. Weisberg, Scott A. Mensing, and Robert M. Scheller. 2018. "Using Paleolandscape Modeling to Investigate the Impact of Native American–Set Fires on Pre-Columbian Forests in the Southern Sierra Nevada, California, USA." *Annals of the Association of American Geographers. Association of American Geographers* 108 (2): 1–20.

Koenig, Walter D., and Johannes M. H. Knops. 2007. "Long-Term Growth and Persistence of Blue Oak (Quercus Douglasii) Seedlings in a California Oak Savanna." *Madrono; a West American Journal of Botany* 54 (4): 269–74.

Lewis, H. T., L. J. Bean, and H. W. Lawton. 1973. "Patterns of Indian Burning in California: Ecology and Ethnohistory."

Lillian M. Hall, Melvin R. George, Douglas D. McCreary, and Theodore E. Adams. 1992. "Effects of Cattle Grazing on Blue Oak Seedling Damage and Survival." *Journal of Range Management* 45 (5): 503–6.

Mallek, Chris, Hugh Safford, Joshua Viers, and Jay Miller. 2013. "Modern Departures in Fire Severity and Area Vary by Forest Type, Sierra Nevada and Southern Cascades, California, USA." *Ecosphere* 4 (12): art153.

Mansourian, Stephanie, Nigel Dudley, and Daniel Vallauri. 2017. "Forest Landscape Restoration: Progress in the Last Decade and Remaining Challenges." *Ecological Restoration* 35 (4): 281–88.

McBride, Russell, and Kloss. 1996. "Impact of Human Settlement on Forest Composition and Structure." *Sierra Nevada Ecosystem Project, Final Report to*.

McCreary, and George. 2005. "Managed Grazing and Seedling Shelters Enhance Oak Regeneration on Rangelands." *California Agriculture*.

Mensing, Scott A. 1992. "THE IMPACT OF EUROPEAN SETTLEMENT ON BLUE OAK (QUERCUS DOUGLASII) REGENERATION AND RECRUITMENT IN THE TEHACHAPI MOUNTAINS, CALIFORNIA." *Madroño* 39 (1): 36–46.

Muir, John. 1979. *John of the Mountains: The Unpublished Journals of John Muir*. Univ of Wisconsin Press.

Orlando, Anne M. 2008. "Impacts of Rural Development on Puma Ecology in California's Sierra Nevada." Ann Arbor, United States: University of California, Davis.

Pavlik, Bruce M., Pamela Muick, and Sharon Johnson. 1993. *Oaks of California*. 2nd ed. Cachuma Pr.

Picotte, Long, Peterson, and Nelson. n.d. "LANDFIRE 2015 Remap – Utilization of Remotely Sensed Data to Classify Existing Vegetation Type and Structure to Support Strategic Planning and Tactical Response." *Earthzine*.

Rollins, Matthew G. 2009. “LANDFIRE: A Nationally Consistent Vegetation, Wildland Fire, and Fuel Assessment.” *International Journal of Wildland Fire* 18 (3): 235–49.

Sulak, Adriana, and Lynn Huntsinger. 2007. “Public Land Grazing in California: Untapped Conservation Potential for Private Lands?” *Rangelands* 29 (3): 9–12.

Swaty, Randy, Kori Blankenship, Kimberly R. Hall, Jim Smith, Megan Dettenmaier, and Sarah Hagen. 2021. “Assessing Ecosystem Condition: Use and Customization of the Vegetation Departure Metric.” *Land* 11 (1): 28.

Swiecki, Bernhardt, and Drake. 1997. “Stand-Level Status of Blue Oak Sapling Recruitment and Regeneration.” *Of Symposium Oak*

Swiecki, Tedmund J., Elizabeth A. Bernhardt, and Christiana Drake. n.d. “Factors Affecting Blue Oak Sapling Recruitment 1.” Accessed July 27, 2022.

Syphard, Alexandra D., Jon E. Keeley, and John T. Abatzoglou. 2017. “Trends and Drivers of Fire Activity Vary across California Aridland Ecosystems.” *Journal of Arid Environments* 144 (September): 110–22.

Tecklin, Jerry, J. Michael Connor, and Douglas D. Mccreary. n.d. “Rehabilitation of an Oak Planting Project on Cleared Rangeland Using Treeshelters and Grazing: A Ten-Year Saga 1.” Accessed August 10, 2022.

Temperton, Vicky M., Nina Buchmann, Elise Buisson, Giselda Durigan, Lukasz Kazmierczak, Michael P. Perring, Michele Sá Dechoum, Joseph W. Veldman, and Gerhard E. Overbeck. 2019. “Step Back from the Forest and Step up to the Bonn Challenge: How a Broad Ecological Perspective Can Promote Successful Landscape Restoration.” *Restoration Ecology*, no. rec.12989 (May). https://doi.org/10.1111/rec.12989.

Thorne, J., Cameron, D., & Jigour, V. (2002). A guide to Wildlands Conservation in the Central Coast Region of California.

Tyler, Claudia M., Bill Kuhn, and Frank W. Davis. 2006. “Demography and Recruitment Limitations of Three Oak Species in California.” *The Quarterly Review of Biology* 81 (2): 127–52.

Van de Water, Kip M., and Hugh D. Safford. 2011. “A Summary of Fire Frequency Estimates for California Vegetation before Euro-American Settlement.” *Fire Ecology* 7 (3): 26–58.

Westoby, and Walker. 1989. “Opportunistic Management for Rangelands Not at Equilibrium.” *Rangeland Ecology & Management*.

Wilson, Tamara Sue, Benjamin Michael Sleeter, and Adam Wilkinson Davis. 2015. “Potential Future Land Use Threats to California’s Protected Areas.” *Regional Environmental Change* 15 (6): 1051–64.

Zavaleta, Erika S., Kristin B. Hulvey, and Brian Fulfrost. 2007. “Regional Patterns of Recruitment Success and Failure in Two Endemic California Oaks.” *Diversity & Distributions* 13 (6): 735–45.

Discussion

Anthropogenic global change has resulted in multiple, large-scale ecological crises, including unprecedented rates of extinction, a changing climate, and lost or degraded landscapes, natural resources, and ecosystem services (Temperton et al. 2019). To step up and meet these challenges, ecological restoration and conservation needs to be implemented at landscape to regional scales and we must be explicit about incorporating ecological complexity into assessment, implementation and management (Temperton et al. 2019; Bullock et al. 2022). Incorporating complexity requires improving our understanding of within-system vegetation structural diversity and the ability to restore or mitigate the heterogeneous processes that drive this heterogeneity. It is also imperative that we incorporate multiple ecosystem types, and recognize the importance of restoring non-forest systems, in addition to those forested systems that have to date been the focus of large-scale restoration, in order to fully meet the challenges of preserving biodiversity and mitigating the effects of climate and other drivers of anthropogenic global change.

My research addresses these challenges by taking a landscape-to-regional approach to the assessment, monitoring, and management of ecological restoration. It also incorporates ecosystem heterogeneity and complexity by using state and transition models, iterative and adaptive approaches, and recognizing the heterogeneous nature of disturbance regimes. And I demonstrate how these approaches can be applied in non-forested systems, specifically woodlands and savannas, and alluvial shrubland systems.

First, I demonstrated how state and transition models and the L-TEAM framework can be used to move toward a more systematic and scientifically informed approach to long-term management and monitoring of restoration and conservation at the landscape-level. L-TEAM uses an adaptive approach and combines STMs, objective oriented goals, and decision support tools into a framework that aids scientists and managers by 1) clearly communicating understanding of ecosystem function and drivers through the use of STMs, 2) assisting stakeholders, through the use of OOGs, in establishing clear, actionable goals that can be assessed through the development of rigorous experiments, the results of which can then be used to 3) develop DSTs to support management decisions and actions, and ensure continuity. I used two case studies, one an alluvial shrubland and another in oak woodlands and savannas, to show that L-TEAM is especially well suited to capture the complexity inherent in landscape-level projects, which include habitat heterogeneity, multiple land uses and land use histories, and multiple management goals.

To further assess the applicability of L-TEAM, it should be applied to different systems and at different scales. Through its emphasis on stakeholder engagement, L-TEAM also has the potential to include a diversity of ecological knowledge and management practices, including traditional ecological knowledge (TEK) (Anderson and Barbour 2003; Reyes-García et al. 2019). And while some challenges remain to the wide-scale application of L-TEAM, such as uncertainty with regard to the impacts of climate change, the incorporation of new technologies and data management, L-TEAM's adaptive and flexible approach make it a promising step toward a more lucent, scientifically informed, and strategic framework for the long-term monitoring and management of restoration projects.

Second, my research examined how changes in disturbance heterogeneity from historical ranges – specifically fire severity – varies regionally in two of California's foothill ecoregions that are dominated by oak woodland and savanna. I also assessed how regional vegetation structure has departed from historical distributions. While my findings do suggest some regional differences, fire severity has increased across both regions. Also, in both the Sierra Nevada foothills and the Northern Coast Interior ranges, more acreage is burning than would typically burn prior to European settlement. It is important to note that this contrasts with findings in California's forested regions where researchers have found deficits in area burned (Mallek et al. 2013; McGarigal et al. 2018; Haugo et al. 2019). Large fire complexes are becoming more frequent in these regions – with severe social, economic, and ecological implications. Moreover, these shifts in the extent and distribution of fire severities are compounded by land use change, introduced species and disturbances, and a changing climate. This is demonstrated by my findings that much of the vegetation in these regions has experienced high rates of conversion and shifts in native vegetation cover structure.

While this study was broad in scale and was limited temporally by the modern fire severity records available (35 years), it found clear trends in vegetation structure and fire severity distributions. To manage these landscapes for resilience under future uncertainties it is necessary to consider actions that help return historic ranges of variation (HRV) in vegetation structure and fire severities. Moreover, these findings emphasize that it is important to consider not only the restoration of a particular vegetation, but of restoring or maintaining both a heterogeneous mosaic of vegetation state classes and a heterogeneous disturbance regime (i.e. fire with varying severity classes).

Finally, to further emphasize the need to focus on the restoration of complexity (Bullock et al. 2022) – in this case specifically heterogeneous vegetation structure – I evaluated the current conditions and restoration needs of California's Blue oak woodlands and savannas. As one of California's most extensive and biodiverse, but also one of its most degraded, vegetation communities (Bernhardt and Swiecki 2001; Thorne et al. 2018; Bernhardt and Swiecki 2001), the restoration of California's Blue oak woodlands and savannas remains both crucial and challenging (Bernhardt and Swiecki 2001; Whipple, Grossinger, and Davis 2011). As even the name implies (i.e. woodlands *and* savannas), maintaining a diverse mosaic of vegetation in various successional stages is key to the identity of this system (George and Alonso 2008; Bullock et al. 2022). My research confirms that Blue oak woodlands and savannas have experienced high rates of conversion to other land uses and vegetation – with most regions experiencing rates of conversion of close to or greater than 50 percent. In what remains of Blue oak woodlands and savannas, departure from historical ranges of within-system structural variation is common, with over-representation of Early successional classes ubiquitous and no region completely within HRV. My restoration assessment indicates that these landscapes require broad-scale restoration efforts, including reducing disturbances in order to allow woodland succession to proceed, and spatially targeted reintroduction of low-severity thinning to expand Mid and Late Open classes – whether that be through prescribed fire, strategic grazing, and/or mechanical thinning of vegetation.

This study was also broad in scale, covering the three California ecoregions where Blue oak woodlands and savannas are most extensive. This may limit how specific restoration treatment recommendations can be at the site scale, however, it reveals broad trends in the conversion and departure of Blue oak woodlands and savannas across the landscape. It also identifies general restoration approaches necessary to shift the system back within historical structural diversity. Moreover, multiple drivers of change and a history of extensive human use, both pre- and post-European settlement, have and will continue to influence the extent and structure of woodlands. Future research, modeling, and planning should seek to more thoroughly explore how these factors interact and affect the heterogeneous mosaics of these landscapes. This future work can build on our findings which demonstrate that active ecological management is required to return Blue oak woodlands and savannas back to within their historical range of structural diversity.

Ecological restoration is one of our most promising tools to mitigate species loss, landscape degradation, and the loss of natural resources and ecosystem services. As we scale these efforts up to face the challenges and uncertainties associated with climate change, a growing human population, and the 6th largest extinction event in history, it is necessary to shift our focus to the restoration of complexity and heterogeneity (Bullock et al. 2022). My research has demonstrated that an adaptive landscape to regional approach aided by tools such as the L-TEAM framework and state and transition simulation models, has the potential to help assess, monitor, and manage ecological restoration projects in a way that incorporates ecosystem complexity and heterogeneity in a strategic and scientifically informed manner.

Works Cited

Anderson, M. Kat, and Michael G. Barbour. 2003. "Simulated Indigenous Management: A New Model for Ecological Restoration in National Parks." *Ecological Restoration* 21 (4): 269–77.

Bernhardt, and Swiecki. 2001. "Ecological Importance of California Oak Woodlands." *Restoring Oak Woodlands in California: Theory*.

Bullock, James M., Elisa Fuentes-Montemayor, Ben McCarthy, Kirsty Park, Rosie S. Hails, Ben A. Woodcock, Kevin Watts, Ron Corstanje, and Jim Harris. 2022. "Future Restoration Should Enhance Ecological Complexity and Emergent Properties at Multiple Scales." *Ecography* 2022 (4). https://doi.org/10.1111/ecog.05780.

George, Melvin R., and Maximo F. Alonso. 2008. "Oak Woodland Vegetation Dynamics: A State and Transition Approach." In *In: Merenlender, Adina; McCreary, Douglas; Purcell, Kathryn L., Tech. Eds. 2008. Proceedings of the Sixth California Oak Symposium: Today's Challenges, Tomorrow's Opportunities. Gen. Tech. Rep. PSW-GTR-217. Albany, CA: US Department of Agriculture, Forest Service, Pacific Southwest Research Station: Pp. 93-104*, 217:93–104. fs.usda.gov.

Haugo, Ryan D., Bryce S. Kellogg, C. Alina Cansler, Crystal A. Kolden, Kerry B. Kemp, James C. Robertson, Kerry L. Metlen, Nicole M. Vaillant, and Christina M. Restaino. 2019. "The Missing Fire: Quantifying Human Exclusion of Wildfire in Pacific Northwest Forests, USA." *Ecosphere* 10 (4): e02702.

Mallek, Chris, Hugh Safford, Joshua Viers, and Jay Miller. 2013. "Modern Departures in Fire Severity and Area Vary by Forest Type, Sierra Nevada and Southern Cascades, California, USA." *Ecosphere* 4 (12): art153.

McGarigal, Mallek, Estes, and Tierney. 2018. "Modeling Historical Range of Variability and Alternative Management Scenarios in the Upper Yuba River Watershed, Tahoe National Forest, California." *Gen. Tech. Rep. RMRS*.

Reyes-García, Victoria, Álvaro Fernández-Llamazares, Pamela McElwee, Zsolt Molnár, Kinga Öllerer, Sarah J. Wilson, and Eduardo S. Brondizio. 2019. "The Contributions of Indigenous Peoples and Local Communities to Ecological Restoration." *Restoration Ecology* 27 (1): 3–8.

Temperton, Vicky M., Nina Buchmann, Elise Buisson, Giselda Durigan, Lukasz Kazmierczak, Michael P. Perring, Michele Sá Dechoum, Joseph W. Veldman, and Gerhard E. Overbeck. 2019. "Step Back from the Forest and Step up to the Bonn Challenge: How a Broad Ecological Perspective Can Promote Successful Landscape

Restoration." *Restoration Ecology*, no. rec.12989 (May). https://doi.org/10.1111/rec.12989.

Thorne, James H., Hyeyeong Choe, Peter A. Stine, Jeanne C. Chambers, Andrew Holguin, Amber C. Kerr, and Mark W. Schwartz. 2018. "Climate Change Vulnerability Assessment of Forests in the Southwest USA." *Climatic Change* 148 (3): 387–402.

Whipple, A. A., R. M. Grossinger, and F. W. Davis. 2011. "Shifting Baselines in a California Oak Savanna: Nineteenth Century Data to Inform Restoration Scenarios." *Restoration Ecology*. https://doi.org/10.1111/j.1526-100X.2009.00633.x.

Appendix 1. Model Descriptions

Ch.2 Fire and vegetation structure HRV models

Purpose

The purpose of these models was to determine the Historical Range of Variation (HRV) for fire and vegetation structure of California's foothill woodlands. Specifically, it will be used to determine how far current conditions have departed from historical ranges in fire severity and vegetation successional stages.

Each BpS model has its own specific documentation accessed through the LANDFIRE Program website (https://landfire.gov/bps-models.php). What follows is a general description/example of these models.

States, Transitions, and Scale

States.

- Early
- Mid Open
- Mid Closed
- Late Open
- Late Closed

Transitions:

- Surface Fire
- Mixed Fire
- Replacement Fire
- Succession
- Insects or Disease
- Windthrow or weather
- Native Grazing

All Transition probabilities were based on LANDFIRE documentation and source models (www.landfire.gov). However, following Blankenship et al. (2015) we used LANDFIRE and ST-Sim's beta distribution and probability multipliers to account for uncertainty and variability in fire return intervals. The beta distribution is used to represent variability within a fixed range (i.e. the minimum and maximum fire return interval). Simulations were then run using ST-Sim's transition multiplier function to sample from the beta distributions defined for each model and each fire severity type.

Scale:

Models were set to each ecoregion's acreage with each cell representing 10 acres. Each time step was equal to one year. Initially, each state represented equal proportions of the landscape but stabilized within 200 to 250 years (similar to findings by others; Haugo et al. 2019).

Example Source Model Parameters from BpS 10300

Table A.1: Mediterranean California Lower Montane Black Oak-Conifer Forest and Woodland LANDFIRE model parameters

Deterministic Transitions

From Class	Begins at (yr)	Succeeds to	After (years)
Early1:ALL	0	Mid1:OPN	24
Mid1:OPN	25	Mid1:OPN	999
Mid1:CLS	25	Mid1:CLS	999

Probabilistic Transitions

Disturbance Type	Disturbance occurs In	Moves vegetation to	Disturbance Probability	Return Interval (yrs)	Reset Age to New Class Start Age After Disturbance	Years Since Last Disturbance
Mixed Fire	Early1:ALL	Early1:ALL	0.003	333	No	0
Replacement Fire	Early1:ALL	Early1:ALL	0.0057	175	Yes	0
Surface Fire	Early1:ALL	Early1:ALL	0.0667	15	No	0
Alternative Succession	Mid1:OPN	Mid1:CLS	1	1	Yes	20
Replacement Fire	Mid1:OPN	Early1:ALL	0.0057	175	Yes	0
Mixed Fire	Mid1:OPN	Mid1:OPN	0.03	33	No	0
Surface Fire	Mid1:OPN	Mid1:OPN	0.0667	15	No	0
Replacement Fire	Mid1:CLS	Early1:ALL	0.0057	175	Yes	0
Mixed Fire	Mid1:CLS	Mid1:OPN	0.03	33	Yes	0
Surface Fire	Mid1:CLS	Mid1:CLS	0.0667	15	No	0

Table. A.2 Model Project and Scenario Settings for A regional assessment of California woodlands historical and modern fire severity and vegetation trends

Regional HRV Model Project Settings		
Strata: Vegetation Type		
	Name	**SNF:**10270;10280;10290;10300;10310;10970;10980;11050;11140;11520;11540 **NCI:**10270;10280;10290;10970;10980;11050;11130;11140;11380;11510;11520
	Description	BpS Name
States		Selected from State list in each BpS source model
Transitions	Transition type	Deterministic: Succession/age
		Probabilistic: Selected from BpS source model
Model Scenario Settings		
Run Control		
	Start timestep	0
	End timestep	1,000
	Total iterations	100
Transition Pathways: States		
	Class	Selected from BpS source model state list
	To class	Following rules defined in BpS source model
	Age min	Defined in each BpS source model for each State; youngest 0
	Age max	Defined in each BpS source model for each State; oldest 999

Transition Pathways: Transitions		Only none/one transition per cell per time-step possible
	Class	Selected from among states defined in BpS source models (see example Table >>>)
	To class	Selected from among states defined in BpS source models
	Transition type	Selected from Probabilistic transitions (from BpS source models)
	Probability	Attributed as the annual probability of a transition occurring (from BpS source models and modified with transition multipliers (Blankenship et al 2015); the reciprocal of the probability is the return interval)
	Age reset	Attributed in accordance with the rules defined in BpS source models
	Time-Since-Transition (TST) min	Attributed based upon BpS source model rules; default is 0
Initial conditions	Non-spatial	
	Total Acres	SNF: 5,587,353 acres; NCI: 1,707,873 acres
	Acres per Cell	10
	Distribution	Equal Proportions among states
Model Output options		
	State classes	Every timestep with “include zero values” option
	Transitions	Every time step

Ch.3 Model 1: Vegetation Departure

Purpose

This model's purpose is to determine the Historical Range of Variation (HRV) of vegetation structure for California's Blue oak woodlands and savannas. Specifically, it was used to determine how far current conditions have departed from historical ranges in vegetation successional stages for each ecoregion.

States, Transitions, and Scale

States.

- Early
- Mid Open
- Mid Closed
- Late Open
- Late Closed

Transitions:

- Surface Fire
- Mixed Fire
- Replacement Fire
- Succession
- Alternative succession
- Native Grazing

Transition probabilities were based on LANDFIRE documentation and source models (www.landfire.gov).

Scale:

Models were set to each ecoregion's acreage with each cell representing 10 acres. Each time step was equal to one year. Initially, each state represented equal proportions of the landscape but stabilized within 200 to 250 years (similar to findings by others; Haugo et al. 2019).

Table A.3. Model Parameters for Blue oak woodlands and Savannas (11140) from LANDFIRE documentation

Deterministic Transitions

From Class	Begins at (yr)	Succeeds to	After (years)
Early1:OPN	0	Mid1:OPN	19
Mid1:OPN	20	Late1:OPN	59
Late1:OPN	60	Late1:CLS	99
Mid1:CLS	60	Late2:CLS	999
Late1:CLS	100	Late1:CLS	999

Probabilistic Transitions

Disturbance Type	Disturbance occurs In	Moves vegetation to	Disturbance Probability	Return Interval (yrs)	Reset Age to New Class Start Age After Disturbance	Years Since Last Disturbance
Replacement Fire	Early1:OPN	Early1:OPN	0.005	200	Yes	0
Native Grazing	Early1:OPN	Early1:OPN	0.02	50	No	0
Surface Fire	Early1:OPN	Early1:OPN	0.1	10	No	0
Replacement Fire	Mid1:OPN	Early1:OPN	0.005	200	Yes	0
Alternative Succession	Mid1:OPN	Mid1:CLS	0.017	59	Yes	0
Surface Fire	Mid1:OPN	Mid1:OPN	0.1	10	No	0
Replacement Fire	Late1:OPN	Early1:OPN	0.01	100	Yes	0
Surface Fire	Late1:OPN	Late1:OPN	0.12	8	No	0
Replacement Fire	Late1:CLS	Early1:OPN	0.01	100	Yes	0
Mixed Fire	Late1:CLS	Late1:OPN	0.02	50	Yes	0
Replacement Fire	Mid1:CLS	Early1:OPN	0.01	100	Yes	0
Mixed Fire	Mid1:CLS	Mid1:OPN	0.02	50	Yes	0

Table A.4. Model Project and scenario settings for Ch.3 Assessing California Blue oak woodland and savanna land use conversion, structural diversity departure, and restoration needs Departure assessment.

Model Project Settings		
Strata: Vegetation Type		
	Name	11140
	Description	California Lower Montane Blue Oak-Foothill Pine Woodland and Savanna
States		Early, Mid Open, Mid Closed, Late Open, Late closed
Transitions	Transition type	Deterministic: Succession/age
		Probabilistic: Surface Fire, Mixed Fire, Replacement Fire
Model Scenario Settings		
Run Control		
	Start timestep	0
	End timestep	1,000
	Total iterations	100
Transition Pathways: States		
	Class	Selected from BpS source model state list
	To class	Following rules defined in BpS source model (see table 3)
	Age min	Defined in BpS source model for each State; youngest 0
	Age max	Defined in BpS source model for each State; oldest 999
Transition Pathways: Transitions		Only none/one transition per cell per time-step possible

	Class	Selected from among states defined in BpS source model (see example Table 3)
	To class	Selected from among states defined in BpS source model
	Transition type	Selected from Probabilistic transitions (from BpS source model)
	Probability	Attributed as the annual probability of a transition occurring (from BpS source model; the reciprocal of the probability is the return interval)
	Age reset	Attributed in accordance with the rules defined in BpS source model
	Time-Since-Transition (TST) min	Attributed based upon BpS source model rules; default is 0
Initial conditions	Non-spatial	
	Total Acres	SNF: 5,587,353 acres; NCI: 1,707,873 acres; CCR 842,226 acres
	Acres per Cell	10
	Distribution	Based on Current distributions for each ecoregion
Output options		
	State classes	Every timestep with "include zero values" option
	Transitions	Every time step

Ch.3. Model 2: NCI Restoration Treatments.

Purpose:

This model's purpose was to help determine the effect of implementing restoration treatments to return Blue oak woodlands and savannas of the Northern California Interior Coast Ranges ecoregion to within structural HRV. Specifically, this model was used to determine the level/amount of treatment necessary.

States, Transitions, and Scale

States.

- Early
- Mid Open
- Mid Closed
- Late Open
- Late Closed

Transitions:

- Surface Fire
- Mixed Fire
- Replacement Fire
- Succession
- Alternative succession
- Native Grazing

Scale:

Models were set to NCI's remaining Blue oak woodlands and savannas' acreage with each cell representing 10 acres. Each time step was equal to one year. Each state was set to NCI's current proportions.

Table A.5. Model project and scenario settings for Ch.3 Assessing California Blue oak woodland and savanna restoration treatment assessment.

Model Project Settings		
Strata: Vegetation Type		
	Name	11140
	Description	California Lower Montane Blue Oak-Foothill Pine Woodland and Savanna
States		Early, Mid Open, Mid Closed, Late Open, Late Closed
Transitions	Transition type	Deterministic: Succession/age
		Probabilistic: Surface Fire, Mixed Fire, Replacement Fire
		Targeted: "surface thinning/fire", "replacement thinning/fire"
Model Scenario Settings		
Run Control		
	Start timestep	0
	End timestep	100
	Total iterations	100
Transition Pathways: States		
	Class	Selected from BpS source model state list
	To class	Following rules defined in BpS source model
	Age min	Defined in BpS source model for each State; youngest 0
	Age max	Defined in BpS source model for each State; oldest 999
Transition Pathways: Transitions		Only none/one transition per cell per time-step possible

	Class	Selected from among states defined in BpS source model (see Table 3)
	To class	Selected from among states defined in BpS source model
	Transition type	Selected from Probabilistic transitions (from BpS source model)
	Probability	Attributed as the annual probability of a transition occurring; fire probability determined from current fire trends (ch.2) then reduced by half
	Age reset	Attributed in accordance with the rules defined in BpS source model
	Time-Since-Transition (TST) min	Attributed based upon BpS source model rules; default is 0
	Transition Targets	Each time step: 1,000 acres surface thinning/fire Mid open, Late Open (=/> 30 years); 500 Acres replacement thinning/fire Mid Closed, Late Closed (=/>100 years)
Initial conditions	Non-spatial	
	Total Acres	NCI: 1,707,873 acres
	Acres per Cell	10
	Distribution	Current sClass distribution for the NCI region
Output options		
	State classes	Every timestep with “include zero values” option
	Transitions	Every time step