DISSERTATION

TRANSFORMATIONAL CHANGE IN CONSERVATION

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Theodore Patrick Toombs

Department of Human Dimensions of Natural Resources

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Doctoral Committee:

Advisor: Richard L. Knight
Co-Advisor: Jennifer E. Cross

Tara L. Teel
Rebecca Niemiec

ABSTRACT

TRANSFORMATIONAL CHANGE IN CONSERVATION

This dissertation explores a fundamental question for the conservation profession and society at large: *How can we more effectively create the transformational change necessary to solve complex conservation problems?* To do so, it's important to understand processes of transformational change and how they can be strategically utilized to address conservation problems. The lack of inclusion of social and systemic sciences into conservation science and practice hinders the profession's understanding of transformational change. Socio-ecological systems theory and social science have many insights to offer, but these insights have not been systematically incorporated into science and practice or coalesced into an integrated theory, despite repeated appeals from social scientists. Each chapter of this dissertation takes a unique perspective on change. Chapter 2 explores the value orientations of Illinois farmers as important knowledge in the process of creating changes in individual behavior. Chapter 3 is a case study of conservation program that failed to materialize in part due to lack of attention to broader social issues. Chapter 4 is a synthesis of critiques of the current conservation paradigm that illustrate its bias toward individualistic, agentic theories of change that result from mainstream adoption of individual, neoliberal ideology. Many conservation problems are social and systemic in nature, yet the professions dominant theory of change is based on a theoretical perspective of these problems as individualistic, behavior problems. To address this, a more integrative set of theoretical perspectives is needed. Chapter 5 articulates a new, integrative theory of change (TTC) composed of four interdependent sets of mechanisms that can be enacted through

strategic, conservation action in collaborative, place-based settings: (a) building communities of practice; (b) empowering individual catalysts; (c) reconfiguring the system; and (d) connecting across dimensions. I propose a set of testable propositions related to each of these components. The aim of the TTC is to integrate existing social and systems science insights into conservation science and practice, expand the set of potential interventions available, and improve the profession's ability to create the change necessary to address the world's most pressing conservation issues.

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Aldo Leopold, beliefs, biodiversity offsetting, collective action, conservation, conservation banking, distributed action, farmers, habitat credit trading, individualism, Land Ethic, land ownership, market-based instruments, mitigation, neoliberalism, no-net-loss conservation, private lands conservation, social change, social science integration, social sciences, socio-ecological systems, stewardship, systems thinking, transformational change, transitions, value orientations

CHAPTER ONE: INTRODUCTION

1. INTRODUCTION TO THIS DISSERTATION

This dissertation explores a fundamental question for the conservation profession and society at large: *How can we more effectively create the transformational change necessary to solve complex conservation problems*? Conservation practice is an applied discipline that uses science to design interventions intended to reduce environmental impact. Because environmental degradation is increasing rapidly in most places, conservation's most fundamental goal must be to change this trajectory. The conservation profession and the organizations and individuals in it therefore are change agents. In this dissertation, I assume that the more effective the profession is at generating change, the more likely it is that urgent conservation issues can be solved. If this assumption is true, then a fundamental question for the profession is, how does it currently conceptualize change and strategically approach transformational change, and how can it do so more effectively? This dissertation is focused on understanding transformational change and how it can be statically generated to address conservation problems.

Research in the natural sciences tells us we need dramatic change (in terms of magnitude and speed) to address issues such as climate change and biodiversity loss. Environmental impacts are worsening on a global scale, despite substantial conservation effort (Butchart et al. 2010; Diaz & Rosenburg, 2008; Diaz et al., 2019; Jones et al., 2018; Keppel et al., 2012; Theobald et al., 2020). Public statements from scientists are increasingly dire. For example, the most recent United Nations report on climate change describes the situation as "code red for humanity" and warns of a bleak, irreversible future if dramatic, immediate, large-scale reductions in emissions do not occur within very short time frames (Meredith, 2021). Social scientists and the conservation profession increasingly understand that solving large-scale, complex conservation problems requires change at multiple levels of society (Amel et al., 2017; DeFries & Nagendra, 2017; Steffens et al., 2017). The combined need for dramatic environmental change and the need for multiscale social change that incorporates multiple influences over behavior suggest that conservation's overarching theory of change must include these qualities.

To create change, we must first understand it. In this dissertation, I define transformational change in conservation as the act and result of strategically and significantly reconfiguring or designing new socio-ecological systems to serve the social and ecological common good. Existing theories in social science focus on agentic, social, and structural influences over behavior but seldom integrate the three influences into one theory (Ritzer, 1996). Any theory of transformational change, by definition, must describe how these three influences work across multiple dimensions in complex systems. Simply put, transformational change is systems change.

The lack of inclusion of social and systemic sciences into conservation science and practice hinders the profession's understanding of transformational change. Socio-ecological systems theory and social science have many insights to offer, but these insights have not been systematically incorporated into science and practice or coalesced into an integrated theory despite repeated appeals from social scientists (Bennett et al. 2017; Mascia et al., 2003). Nearly twenty years after Mascia et al., (2003) made this argument, the situation is not much better (Bennett & Roth, 2019). The same or similar argument has been made about systems thinking and science (Knight et al., 2019). As a result, many insights and theoretical perspectives from the social sciences and socio-ecological systems research remain underutilized, misunderstood, and overlooked (Bennett & Roth, 2019; Bennett et al., 2017; Manfredo et al., 2019; Shove, 2010).

Major, global research investments (e.g., climate change research) continues to prioritize understanding the causes and consequences of environmental degradation over how to generate systemic social change to address it (Overland & Sovacool, 2020). These insights represent a large source of untapped potential for creating change that has so far been overlooked by the conservation profession.

2. RESEARCH EPISTEMOLOGIES

I realized the untapped potential of the social and systemic sciences about ten years ago when I began my research. As a conservation professional with almost 20 years of experience at that time, I had become frustrated with what I viewed as a lack of success and progress in my profession, combined with a lack of openness to new ideas and approaches, particularly with respect to social science perspectives. My own personal failures to create change were salient and combined with an eagerness for change, weighed heavily on me. I felt my profession was stuck, and I believed that many of my colleagues agreed. Now, as I complete this dissertation with nearly 30 years of experience, my hope is that my research, along with that of others, can help push the profession toward a new paradigm that frees it to make more effective and rapid change in the world. It is within this personal context that I want to make my research epistemologies explicit.

2.1. Holism

Holism is the idea that the whole is greater than the sum of its parts and the theory that the parts of a whole are inextricably interconnected. My prior training in ecology makes this perspective familiar and natural to me. Systems thinking is consistent with holism in that it focuses on the interrelationships between all the parts of a system, rather than the individual parts or relationships between specific parts (Meadows, 2008). Systems are characterized by mutualism, embeddedness, reciprocal determinism, feedbacks, etc. (Table 1). A holistic epistemology focuses on these processes and how they explain emergent outcomes.

In the first two chapters of this dissertation, I present applied research and a case study of conservation practice that are not holistic in their perspective and therefore lack the ability to generate transformational change on their own. In Chapter 5, I propose an integrative theory of change that exemplifies a more holistic approach by integrating theoretical perspectives from many disciplines at multiple scales. Chapter 5 reflects an epistemology that suggests the power of transformational change lies not within the specific theories of change (i.e., the parts), but from their integration (i.e., the whole).

Another aspect of *holism* is diversity and inclusion to achieve recognitional, procedural, and distributional justice. Diversity is the representation of many perspectives, whereas inclusion is the process of incorporating those perspectives so that they can be manifested in conservation science and practice (Mohai et al., 2009) The conservation profession has long been criticized for excluding the perspectives of local and indigenous communities and inadvertently causing harm to these communities through its conservation approaches (Mohai et al., 2009). In Chapter 4, I explore the lack of inclusion of the social and systemic sciences into conservation practice, and I find that not all social sciences are equally excluded. Specifically, my research suggests that there is a strong bias toward what kinds of sciences are integrated toward agentic, individualistic theories and against social and systemic perspectives. My review of the literature suggests that the root cause of the lack of inclusiveness is cultural bias that excludes, more broadly, many forms of social perspective in society. It suggests that this cultural bias is a root cause of both the lack of attention to environmental justice and its failure to create sufficient transformational change, and those problems both can be addressed by including more diverse perspectives and thinking about conservation issues from a holistic, systemic perspective.

2.2. Applied Relevance

The second epistemology is *applied relevance*. Fostering change is an applied goal. In Chapter 4, I explore the cultural bias that prevents the profession from adopting more holistic theoretical perspectives into its core theory of change. This bias causes reliance on narrow, micro-scale theories of change that can't explain transformational change on their own, because transformational change is a multiscale process. To create change, conservation will need to shift to a multiscale theory of change that describes how change is transmitted across social, spatial, and temporal dimensions. Chapter 5 proposes an integrative theory of change meant to provide a theoretical framework for the profession in dealing with particularly challenging systemic conservation problems. The Theory of Transformational Change (TTC) is proposed as an alternative to the current dominant approach in conservation today. The aim of this research is to help shift the profession to applied approaches that are more likely to generate change to address the world's most urgent conservation issues.

2.3. Learning from Failure

Finally, my experience as a conservation professional has taught me the importance of *learning from failure*. Here, I define failure simply as not achieving a stated objective or as a "deviation from expected and desired results" (Cannon & Edmondson, 2001). If the goal of the conservation profession is to reverse declines of environmental goods, and those goods are in rapid decline despite considerable effort, then one might conclude that the conservation profession itself has failed, but I believe this is unrealistic for a couple of reasons. First, conservation is an obligation of society; it is not the responsibility solely of the conservation profession to encourage conservation behavior. Secondly, the conservation profession has had many successes worth celebrating (Bolam et al., 2020; Brooks et al., 2009). The problem is more nuanced.

Learning from failure as an epistemology is a mindset that suggests being better at generating change requires that we first change ourselves. As Catalano et al. (2018) describe it, the conservation profession has adopted a "culture of success" that encourages replicating past wins and avoids deep examination of ineffective efforts. Others suggest this leads the conservation profession to jump from one fad to another in search of solutions (Redford et al., 2013). A culture of success inhibits learning from failure, a more powerful pathway to long-term success than focusing on wins (Catalano et al., 2018). This epistemology encourages examination of the relationship between the conservation profession's internal processes (e.g., strategy development, planning, diversity, and inclusion) and its external effectiveness.

3. OVERVIEW OF THE FOUR RESEARCH CHAPTERS

Each chapter of this dissertation takes a unique perspective on change. Chapter 2 explores the value orientations of Illinois farmers. The goal of this study was to use methodologies previously used to explore value orientations in citizens of the western United States to understand their perspectives on wildlife issues and apply them to understand farmers' value orientations toward land stewardship (Teel et al., 2007). The framework for the research was the cognitive hierarchy, which is the idea that values, value orientations, beliefs, attitudes, and intention exist in a hierarchy wherein the concepts closer in proximity to behavior (i.e., intention) are more predictive of behavior, and the concepts further away are more fundamental in organizing consistency among the concepts (Whittacker et al., 2006). This study was intended to provide insights on fundamental farmer beliefs to help explain their conservation behavior.

Farmers showed strong orientation toward land values (i.e., mutualism and responsibility) that are consistent with conservation behavior and conservation practice adoption, yet the adoption of conservation practices in Illinois is low for most practices, and environmental outcomes (e.g., water quality) suffer as a result. While these studies are valuable in understanding behavior, they show that additional theoretical perspectives beyond the individual and cognition are necessary to fully understand the behavior of farmers, and those perspectives are not accessible through individual-level theories.

In Chapter 3, I conduct a case study of a conservation program that I helped develop called “Habitat Exchanges.” The goal of this program was to provide a tool that could be used to voluntarily measure and coordinate the activities of multiple organizations to offset the impacts of energy developments (e.g., building a well pad to drill for natural gas) on Greater Sage-Grouse habitat (i.e., sagebrush steppe) in Western Colorado. Although the stakeholder group successfully completed the tool, the program never came to fruition because it was not adopted by the State of Colorado or any of the companies as the standard mitigation approach for this species, although that was the original intent. As a measure of the technical quality of this tool, I compare the program’s structure and rules to the highest standards of the biodiversity offset profession and find that the program meets or exceeds all criteria and standards. Despite its high quality, the tool was not adopted, suggesting that quality was not why it was rejected. Reasons for the lack of adoption included the lack of policy requiring the offsets and the high cost of the offsets to energy companies. The tool, developed through a consensus approach of multiple scientists, showed that the impacts of energy development were higher than previously expected and that the cost of offsetting those impacts was potentially not cost effective for energy companies. The lack of a federal or state requirement to offset meant that companies could avoid using the tool and thereby avoid the high cost of offsets. The fact that this tool did not materialize shows the limits of market-based, voluntary approaches to conservation. When costs are high, voluntary approaches are unlikely to be sufficient. The lesson here is that while neoliberal approaches such as market-based, voluntary mitigation may fit the conservation paradigm by aligning with neoliberal values, they are not necessarily effective at changing behavior without careful consideration of other socio-political aspects.

In Chapter 4, I conduct a literature review of critical perspectives of the conservation field. This review reveals that individual, neoliberal ideology is the fundamental model of thought underlying the dominant conservation paradigm, and this bias helps formulate what types of conservation structures (e.g., tools, approaches, processes, policies, etc.) the profession deems legitimate. Understanding of this ideology and its influence on the profession provides perspective for understanding the results and approach of the previous two chapters and helps shed light on why the conservation profession has failed to integrate multiple theoretical perspectives. Considering this review, the research on Illinois farmers is part of a broader perspective on the adoption of conservation practices that focuses primarily on cognitive theories at the individual scale (e.g., theory of reason action, cognitive hierarchy, rational choice, etc.). Over 35 years of conservation practice adoption research, this literature has concluded that individual-level factors fail to have significant or consistent predictive power on the adoption of conservation practices (Prokopy et al., 2019). The review of critical perspectives sheds light on why research has focused so much on the individual perspective while neglecting social and systemic perspectives. The habitat exchanges program also represents a program consistent with neoliberal bias. The lesson here is that while neoliberal conservation approaches are a cultural fit with U. S. society, there are limits to these approaches because their core models of thought conflict with the goals of conservation. Conservation requires collective action toward the common good, while neoliberal approaches seek to privatize and commodify public goods into private transactions. Their goal is to find win-wins between development and nature, but when these win-wins are not possible, the approaches become inoperable.

Chapter 4 adds to the essential literature of the current conservation program by providing a possible explanation for the lack of inclusion (i.e., individual, neoliberal bias), whereas previous studies assumed the lack of inclusion was relatively equal across the social sciences (Bennett et al., 2019). Instead, I show that individualistic social sciences such as microeconomics and micro psychology are well-integrated, while theoretical perspectives at higher social levels (e.g., organizational change, community-level collaboration, cultural perspectives, etc.) are not well integrated. This research shows that conservation, like many other professions, has been tangibly influenced by individual neoliberalism (e.g., law enforcement and the correctional system, public health, education, etc.), and that these trends and affects are best understood as cultural, ideological phenomena.

Finally, Chapter 4 is the culmination of my learning from the previous chapters and my experience as a professional. I came to understand that addressing complex conservation issues requires understanding transformational change, that transformational change relies on multiple theoretical perspectives that transcend social, spatial, and temporal scale, and that integrative theories of transformational change in conservation are rare. In this chapter, I provide an integrative theory of change that can address a specific set of systemic conservation problems that I call “distributed action problems” to fill this gap. Because the most challenging problems can be characterized as systemic distributed action problems, my hope is that the proposed theory provides the basis to help shift the conservation profession toward an underlying theory of change that is better equipped to address transformational change for these types of problems.

CHAPTER TWO: OPERATIONALIZING LEOPOLD'S LAND ETHIC

1. SYNOPSIS

Understanding the decision-making process of farmers is a key to the conservation of private lands. Aldo Leopold's Land Ethic essay espouses the idea that land stewardship should be motivated by a moral obligation to the land (i.e., a value orientation). The cognitive hierarchy examines relationships between general value orientations and specific attitudes/norms to predict how cognitions influence behavior. This article examines Illinois farmers' value orientations (domination-mutualism) toward land, specific beliefs (rights-responsibilities) toward land ownership, and relationships among these cognitive concepts. The goal of this research was to empirically operationalize Leopold's Land Ethic. Results highlighted the relationships among the concepts advanced in contemporary cognitive hierarchy logic and Leopold's Land Ethic. The general domination-mutualism constructs predicted specific rights and responsibility beliefs, which in turn predicted the Land Ethic. Farmers were more oriented toward mutualism than domination. Most respondents slightly agreed with Leopold's Land Ethic ideology, but they viewed conservation as an individual, private, voluntary decision in which the public should have little role. Based on our research, this article suggests potential conservation strategies that encourage the socially responsible use of private land with emphasis on mutualism and responsibility.

2. INTRODUCTION

Understanding the decision-making process of farmers is a key to successful private lands conservation. Farmers have significant influence over how land is managed, particularly in the Midwest. In Illinois, for example, more than 97% of the land is privately owned (Natural

Resources Council of Maine, 2015), and 70% of the State's land is cropland (Laingen, 2014). Farmers' land management decisions affect themselves, their families, neighbors, and the public at large. Solving large-scale, conservation problems (e.g., reducing eutrophication caused by use of fertilizers) will require dramatic shifts in behavior among thousands of individual farmers (David et al., 2014).

Aldo Leopold believed that the primary pathway to conservation on private farmland is to foster a moral obligation in farmers to take better care of their land. His essay "*The Land Ethic*" in "*A Sand County Almanac and Sketches Here and There*," advanced the concept that farmers should take on this responsibility because it was expedient (i.e., practical, cost effective, profitable), and the right thing to do (Leopold, 1949). The message was one of social responsibility, and conservationists were tasked with instilling this sense of responsibility in farmers (Freyfogle, 2007).

A Sand County Almanac and Sketches Here and There has inspired generations of conservationists, and some considered it to be one of the most influential conservation books ever written (Meine & Knight, 1999). Over 2 million copies have been printed in 12 languages (https://www.aldoleopold.org/about/aldo-leopold/sand-county-almanac/). Since its publication 55 years ago, the frequency with which the book is cited continues to increase (Leopold, 2004). Some conservation organizations use Leopold's writings to teach about and encourage land stewardship by private landowners (e.g., Point Blue Conservation Science, Wendell Gilgert, personal communication, http://www.pointblue.org/our-science-and-services/conservation-science/working-lands/).

Researchers have suggested that "activating" farmers' stewardship beliefs can encourage participation in conservation practices and programs (Thompson et al., 2015). In the United

States, farmers often are encouraged to conserve land by participating in financial incentive programs such as the U. S. Department of Agriculture Conservation Reserve Program (CRP), which pays farmers to retire highly erodible cropland. Barriers to the adoption of conservation practices by farmers, however, often are a complex mix of social, personal, and economic factors, and there is no evidence that financial factors play a large role (Prager & Posthumus, 2010). Farmers adopt practices based on their own sense of religious, spiritual or moral obligation, not for financial gain (Osmond et al., 2015). Riemer and Prokopy (2014), for example, found that farmers were more motivated to engage in conservation because they believe it is the right thing to do for the environment, and they concluded that financial incentives were not the primary motivation for participation in programs.

This article seeks to operationalize Leopold’s Land Ethic by examining the value orientations of Illinois farmers toward land and their beliefs about the rights and responsibilities of land ownership. The value orientations of mutualism and domination were adopted from contemporary theories in social psychology. Their concepts of rights and responsibilities reflect how some farmers believe they should use their land. I measured the accuracy of Leopold’s Land Ethic by examining farmers’ responses to Leopold’s own statements.

3. BACKGROUND

3.1. The Cognitive Hierarchy

Cognitions refer to the mental states (e.g., values, beliefs, attitudes) people use in thinking about situations (Homer & Kahle, 1988; Vaske & Manfredo, 2012). Such cognitions are part of a “hierarchy” that ranges from general to specific (Figure 1).

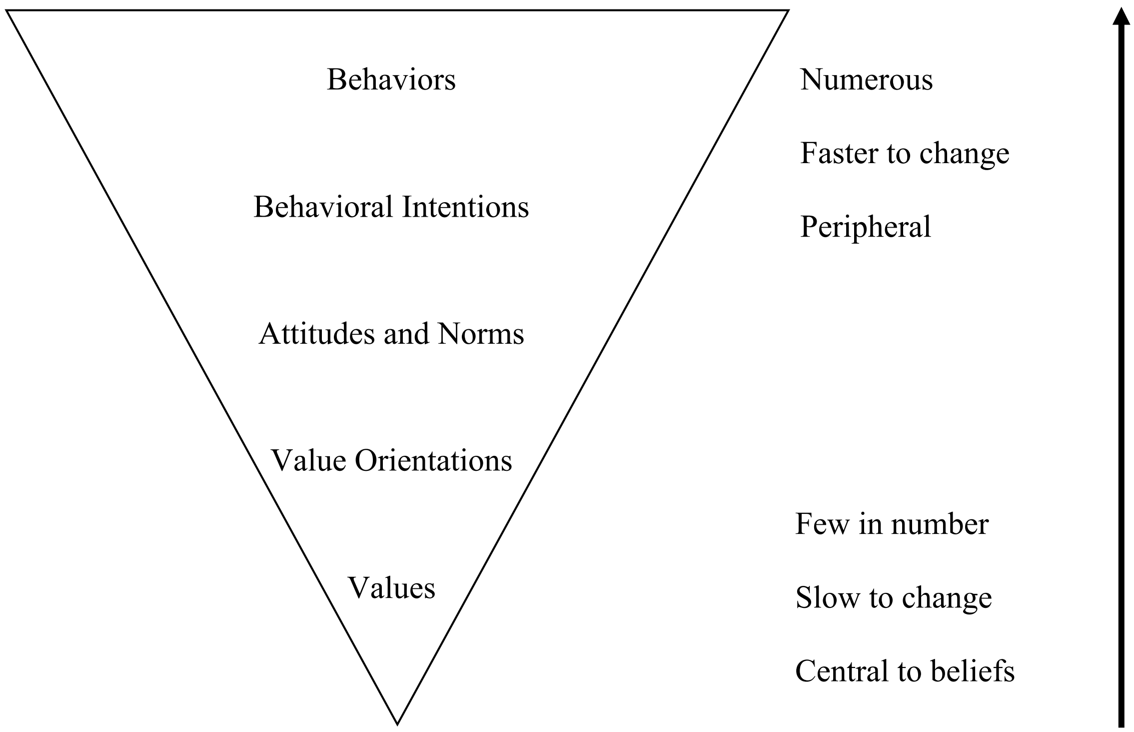

Figure 1. The cognitive hierarchy (adapted from Fulton et al., 1996)

The cognitive hierarchy examines the relationships between general values/value orientations and specific attitudes/norms to predict how cognitions influence behavior. This theoretical approach has been applied to environmental issues (Stern et al., 1999; Van Liere & Dunlap, 1980), forests (Bengston, 1994; Vaske & Donnelly, 1999; Vaske et al., 2001), wildlife (Fulton et al., 1996; Manfredo et al., 2009; Teel and Manfredo, 2009; Whittaker et al., 2006), and nature in general (Buijs, 2009).

The cognitive hierarchy distinguishes values from value orientations. *Values* are desired end states or modes of conduct that people hold dear (e.g., freedom, equality, honesty) (Rokeach, 1973). Values are general and are not linked to specific situations or objects.[1] A person who values honesty is likely to be honest when conducting business deals, completing tax forms, or interacting with friends. Values represent basic desires and goals and define what is important to us (e.g., fairness). Because values are established early in life, are derived from culture, and are linked to person's identity, they are difficult to modify (Manfredo et al., 2016).

Given that values often are shared by all individuals in a culture, they are not likely to explain much of the variance in specific behaviors. *Basic beliefs*, on the other hand, represent our perceptions of general classes of situations (e.g., land use) or issues (e.g., climate change) and offer meaning to the broad cognitions reflected in values. *Value orientations* are networks of basic beliefs that provide contextual meaning to values relative to a specific domain such as the environment (Manfredo et al., 2009; Vaske & Manfredo, 2012). Value orientations reflect our ideology in the cognitive hierarchy (Schwartz, 2006). At the group level, ideology refers to consensually held beliefs that enable the people who share them to define themselves, to understand meaning, and to relate to one another (Pratto, 1999). The strength of a given ideology, and hence its associated value orientations, varies among individuals, and differences in attitudes and behaviors stem from this variation.

3.2. Mutualism and Domination

Basic beliefs include environmental values, world views, or images of nature. All of these concepts contain patterns of basic beliefs that give meaning to values in a specific domain. These domains reflect a *mutualism–domination* value orientation (Manfredo et al., 2004). Individuals with a domination value orientation believe the environment should be managed for human

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[1] In social psychology, an *object* can be any entity that is being evaluated (e.g., a person, situation, wildlife, management action or policy) (Eagly & Chaiken, 1993).

benefit and tend to prioritize human well-being over the environment in their attitudes and behaviors. Such individuals also tend to treat land in utilitarian terms and to rate actions that harm the environment as acceptable. A mutualism value orientation reflects an egalitarian ideology that fosters perceptions of social inclusion and equality that extend to human–land relationships (Wildavsky, 1991). People with a mutualism orientation view the environment as an extended family, deserving of rights and care. These individuals are less likely to support actions resulting in harm to the environment and are more likely to engage in actions that protect the land.

Leopold presaged mutualism and domination views toward land in his discussion of an "A-B cleavage" between different types of farmers (Leopold, 1949). Group A focused on the commodities that land can produce for humans, a domination-oriented relation to land. Leopold believed that the domination-oriented view that humans were separate from nature and capable of molding it to their benefit was a key conservation challenge to overcome (Freyfogle, 2012). Group B regarded the land as a community that humans were merely a part of, a mutualism-oriented relation (Leopold, 1949). Leopold (1949) believed that Group B would be more likely to develop the "love, respect, and admiration for land" necessary to develop a sense of obligation toward it.

3.3. Rights and Responsibility

Whereas mutualism and domination can be value orientations about land in general, rights and responsibility relate to more specific beliefs about land ownership. Due to the prominence of property rights as a social construct and legal mechanism and the large portion of land in private ownership, farmers have great influence over land use decisions in the United States. (Freyfogle, 2007). Property rights represent a fundamental balance between the rights of the individual to use the land as they wish versus the rights of society to reap benefits of responsible land stewardship (Singer, 2000). Many rights regarding the use of private land are not encoded in law, but instead are presumed (Bromley & Hodge, 1990). For example, if there is no law against plowing to the edge of a stream, it often is presumed that a landowner has the individual right to do so, even though it may harm water quality for downstream users. Property rights are a key to understanding policy debates about topics that involve farmers but also affect society, such as water quality, growth management, wildlife management, and endangered species.

Despite public debates over property rights, little empirical research has explored property rights from the perspectives of farmers. The conservation community sometimes assumes that farmers emphasize their right to choose over their responsibility to society (Jackson-Smith et al., 2005). Paying farmers for engaging in conservation practices through government or other incentive programs represents a way to compensate landowners to forgo their presumed rights. Previous research has found that ranchers believe that their rights to use their land come with corresponding responsibilities to use the land in a way that does not harm it or other people (Jackson-Smith et al., 2005; Stroman et al., 2016). These researchers called for more work demonstrating the empirical utility of property rights orientations in private landowners which suggests that farmers might also be motivated to use land responsibly.

4. HYPOTHESES

The following eight hypotheses were examined to determine the relationships between the value orientations: mutualism, domination, rights, responsibility, and the Land Ethic (Figure 2).

H_1 Responsibility will be positively related to mutualism.

H_2 Responsibility will be negatively related to domination.

H_3 Rights will be negatively related to the mutualism.

H_4 Rights will be positively related to domination.

H_5 Land Ethic will be positively related to mutualism.

H_6 Land Ethic will be negatively related to domination.

H_7 Land Ethic will be negatively related to rights.

H_8 Land Ethic will be positively related to responsibility.

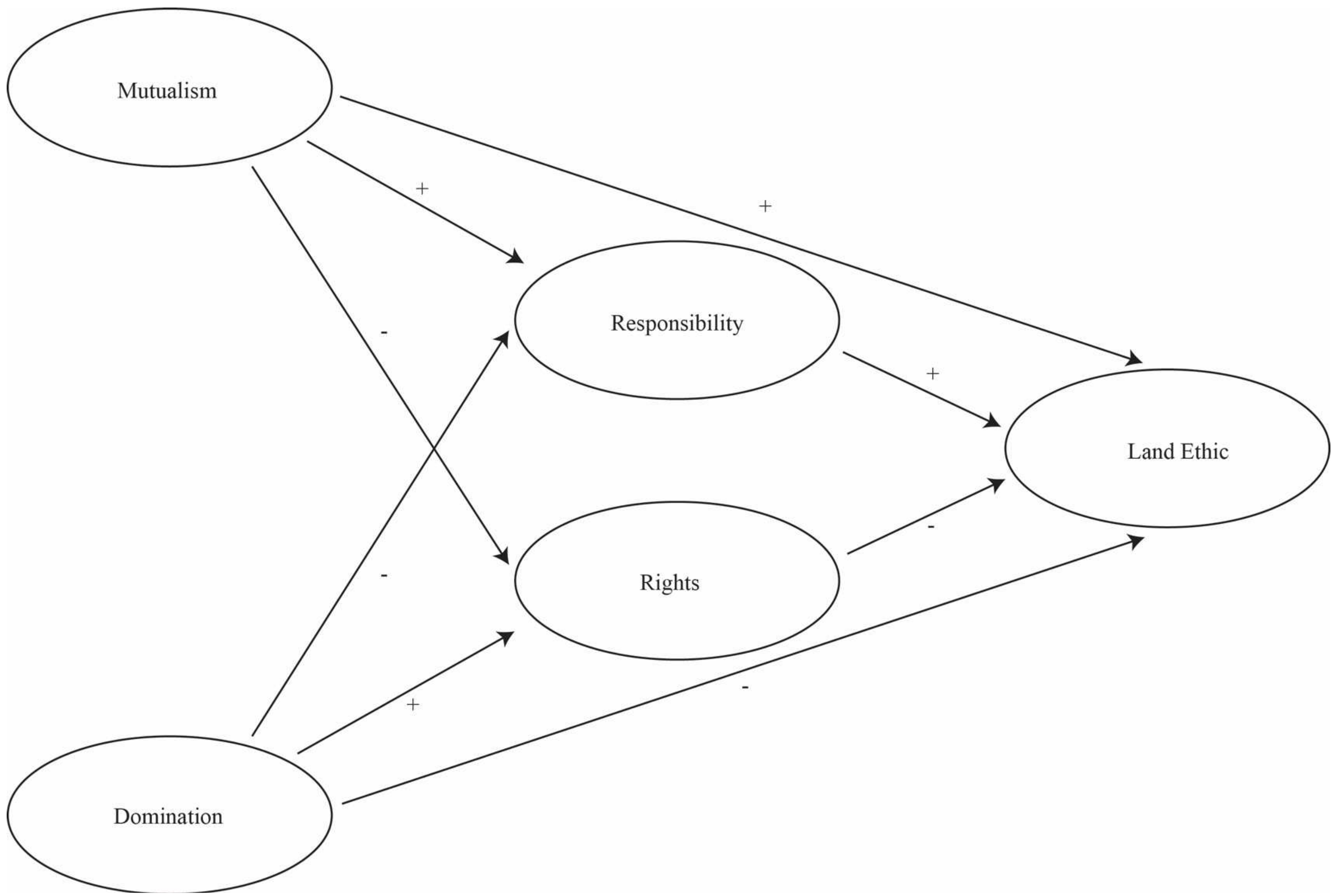

Figure 2. Hypothesized relationships among belief constructs

5. METHODS

I conducted a repeat-mail survey using a random sample of 3,000 Illinois agriculture producers stratified by enrollment in U.S.D.A. conservation programs (e.g., Conservation Reserve Program). One-third of this sample was enrolled in conservation programs, mirroring the proportion enrolled in the population as a whole. Names and addresses of participants were selected by Survey Sampling, International. Each selected participant was mailed the questionnaire along with a cover letter explaining the study and a stamped return envelope (hereafter termed "questionnaire packet"). Non-respondents were mailed a reminder/thank you postcard approximately 14 days after mailing the initial questionnaire packet, followed by a second packet 14 days later. A second postcard reminder/thank you was mailed 16 days later, followed by a third questionnaire packet. Of the initial 3,000 producers on the mailing list, 2,808 surveys were deliverable. Of these, 910 usable surveys were returned (response rate = 32%).

6. VARIABLES/LATENT CONCEPTS

Domination and mutualism indices were derived from previous research (Teel & Manfredo, 2010). Domination was constructed from four variables. Respondents provided their level of agreement with the following statements: (a) "Lands should be managed to benefit people." (b) "Needs of people should take priority over land protection." (c) "Land is primarily for people to use." (d) "Primary value of land is to provide products useful to people." Mutualism was also constructed from four variables: (a) "Land has value whether people use it or not." (b) "Land should be managed so that the environment benefits." (c) "I feel an emotional bond with the land." (d) "Conserving land is important for future generations." All variables were coded on 7-point scales ranging from "strongly disagree" (-3) to "strongly agree" (+3) with 0 as a middle point.

I developed two scales to reflect rights and responsibility beliefs. The four rights variables were: (a) “Landowners have a right to use their land as they see fit” (b) “Other people have no right to tell private landowners how to manage their land” (c) “Private landowner rights outweigh any responsibilities the landowner has to manage land for public benefit” (d) “Conservation is a voluntary choice of the landowner.” The three responsibility variables were: (a) “Landowners have an obligation to consider how their management affects other people” (b) “Landowners have an obligation to manage the land for future generations” (c) “Conservation is one of the responsibilities of private landownership.” These variables were coded on the same 7-point scales as domination and mutualism.

The land ethic was measured using variables derived from direct quotes from Leopold’s writings. Quotes were selected based on their relation to responsibility for land stewardship of private lands (Meine & Knight, 1999). The eight variables were: (a) “Conservation is a state of harmony between people and land” (b) “When people see land as a community to which they belong, they may begin to use it with love and respect” (c) “Land management is right when it tends to preserve the integrity of the land” (d) “People abuse land because they regard it as a commodity belonging to them” (f) “Landowners have an obligation to manage the land in the interest of the community” (g) “A landowner is a custodian of the land” (h) “Landowners have a responsibility to manage land for private and public benefit” (k) “The private landowner is the custodian of wildlife.” Land Ethic variables were coded on the same 7-point scales as all other variables.

7. ANALYSES

The internal consistency of belief constructs was examined using Cronbach’s alpha. A confirmatory factor analysis examined whether the belief constructs provided a good fit to the data. LISREL 9.2 (Joreskog & Sorbom, 2015) was used for these analyses based on the maximum likelihood estimation procedure and the variance covariance matrix. A structural equation path analysis was used to test the predictive validity of the model, as well as assess the mediation role of the rights and responsibilities beliefs. A given variable functions as a mediator to the extent that it accounts for the relation between the predictor and the criterion (Baron & Kenny, 1986). Structural equation models were fitted for direct, partial and full mediation models. In structural equation modeling, three separate models are required to demonstrate mediation (Hayduk, 1987). In the full mediation model, the predictors (domination and mutualism) only influence the criterion (land ethic) indirectly through their effect on the mediators (rights and responsibilities). In the partial mediation model, the predictors influence the criterion variable directly and indirectly through their effect on the mediators. In the third model, direct effects, the predictors directly affect both the criterion and the mediators, but the mediators are constrained to not affect the criterion.

Mediation occurs under the following conditions. First, the predictors must significantly influence the mediators, and the predictors must significantly affect the criterion variable (direct effects model). Second, the paths between the predictors and the mediators and between the mediators and the criterion must be significant in both the full and partial mediation models. Full mediation occurs when the direct path from the predictors to the criterion are not significant in the partial mediation model. Third, a comparison of the nested models using the change in chi-square statistics indicates that the full mediation model fits better than the direct effects model, and the partial mediation model fits no better than the full mediation model (Baron & Kenny, 1986; Hayduk, 1987).

8. RESULTS

Confirmatory factor analysis (CFA) demonstrated that the data provided an acceptable fit to the domination and mutualism belief constructs (Table 1). Standardized factor loadings ranged from .48 to .75 for domination, and .51 to .82 for mutualism. Additional support for combining the belief statements into their associated constructs was evident from the reliability analysis (Table 1). Reliability coefficients for domination and mutualism were .71 and .76, respectively. All item total correlations were > .40. Deleting any item from these basic belief dimensions did not improve reliability. Farmers responded positively to all domination and mutualism belief statements with one exception (Needs of people should take priority over land protection = -.43, Table 1). On average, farmers were more mutualism (*M* = 2.01) than domination (*M* = .62) oriented.

Table 1. Means, confirmatory factor and reliability analyses for domination and mutualism

		Confirmatory Factor Analysis		Reliability Analysis		
Basic belief construct/survey item	Mean	Standardized factor loading	*t*	Item-Total Correlation	Alpha if Item Deleted	Cronbach Alpha
Domination	.62					.71
Lands should be managed to benefit people	.99	.61	17.17	.49	.65	
Needs of people should take priority over land protection	-.43	.65	18.45	.54	.62	
Land is primarily for people to use	.36	.75	20.63	.57	.59	
Primary value of land is to provide products for people	1.54	.48	13.23	.38	.71	
Mutualism	2.01					.76
Land has value whether people use it or not	1.91	.51	15.00	.44	.77	
Land should be managed so that the environment benefits	1.65	.48	27.20	.63	.66	
I feel an emotional bond with the land	2.08	.59	17.80	.55	.71	
Conserving land is important for future generations	2.40	.71	22.44	.65	.69	

The data also provided an acceptable fit to the rights and responsibility belief constructs (Table 2). The CFA standardized factor loadings were .49 to .66 for rights, and .55 to .64 for responsibility. Additional support for combining the belief statements into their associated constructs was evident from the reliability analyses (Table 2). Reliability coefficients for rights and responsibility were .67 and .63, respectively. All item total correlations were > .40. One item was deleted (the public has a role in deciding how private land is used) from the responsibility construct to bring reliability to an acceptable level. This finding will be discussed more in the Discussion. Deleting any additional items from their basic belief dimension did not improve reliability. Farmers responded positively to all rights and responsibility statements (Table 2). On average, farmers were more responsibility (M = 1.99) than rights (M = 1.16) oriented.

Table 2. Means, confirmatory factor and reliability analyses for rights and responsibilities

		Confirmatory Factor Analysis		Reliability Analysis		
Basic belief construct/survey item	Mean	Standardized factor loading	*t*	Item-Total Correlation	Alpha if Item Deleted	Cronbach Alpha
Rights	1.16					.67
Landowners have a right to use their land as they see fit	1.44	.69	13.03	.50	.57	
Other people have no right to tell private landowners how to manage their land	.94	.49	18.59	.41	.64	
Private landowner rights outweigh any responsibilities a landowner has	.78	.66	14.17	.52	.56	
Conservation is a voluntary choice of the landowner	1.46	.50	18.50	.40	.64	
Responsibilities	1.99					.63
Landowners have an obligation to consider how their management affects others	1.77	.64	16.30	.45	.50	
Landowners have an obligation to main maintain land for future generations	2.30	.55	18.48	.43	.53	
Conservation is one of the responsibilities of private landownership	1.90	.61	17.26	.42	.55	

Finally, the data supported the Land Ethic construct (Table 3). Standardized factor loadings were .51 to .78 (Table 3). The reliability coefficient was .84; all item total correlations were > .40. Deleting any additional items from their basic belief dimension did not improve reliability. Overall, farmers slightly agreed with the Land Ethic construct (*M* = 1.19).

Table 3. Means, confirmatory factor and reliability analyses for land ethic

		Confirmatory Factor Analysis		Reliability Analysis		
Basic belief construct/survey item	Mean	Standardized factor loading	*t*	Item-Total Correlation	Alpha if Item Deleted	Cronbach Alpha
Land Ethic	1.19					.84
Conservation is a state of harmony between men and land	1.39	.72	17.51	.65	.81	
When people see land as a community, they begin to use it with love and respect	1.05	.73	24.99	.69	.81	
Land management is right when it tends to preserve the integrity of the land	1.75	.78	21.12	.65	.82	
People abuse land because they regard it as a commodity belonging to them	.63	.51	14.14	.48	.84	
Landowners have an obligation to manage land in the interest of the community	.67	.62	17.03	.66	.81	
A landowner is a custodian of the land	2.17	.58	16.16	.49	.84	
Landowners have a responsibility to manage land for private and public benefit	.52	.53	14.66	.56	.83	
The private landowner is a custodian of wildlife	1.32	.56	15.64	.52	.83	

Full and partial mediation models were examined using the chi-squared test for the respective models. The partial mediation model ($\chi^2 = 1080.32$, $df = 218$, $p < .001$) had a significantly better fit than the full mediation ($\chi^2 = 1097.89$, $df = 220$, $p < .001$) model ($\Delta\chi^2 = 17.57$, $df = 2$, $p < .001$). In structural equation modeling, the model with the smaller chi-square fits the data better. The partial mediation model was used to describe the data.

Overall fit of the partial mediation model was assessed using six indicators (χ^2/df, GFI, NFI, CFI, RMR). Marsh and Hocevar (1985) suggest that a χ^2/df ratio between 2:1 and 5:1 indicates an acceptable fit. The model was in this range ($\chi^2/df = 1080.32/218 = 4.96$). Values for goodness of fit index (GFI), the normed fit index (NFI) and the comparative fix index (CFI) were .91, .94, and .95, respectively, indicating an acceptable fit for the model (Bollen 1989). Finally, the root-mean-square residual (RMSR), which measures the average discrepancies between the observed and the model-generated covariances was .16, suggesting a close fit of the data (Church and Burke, 1994).

Having accepted the partial mediation model, the hypotheses in the model were examined (Figure 3). Hypotheses 1 and 2 predicted that responsibilities would be positively related to mutualism and negatively related to domination. The standardized regression coefficient for mutualism ($\beta = .77$, $p < .05$) was significant and in the predicted direction (Hypothesis 1) (Figure 2). The relationship between responsibilities and domination was not significant at the $p < .05$ level; Hypothesis 2 was rejected. Mutualism and domination explained 61% of the variance in responsibilities.

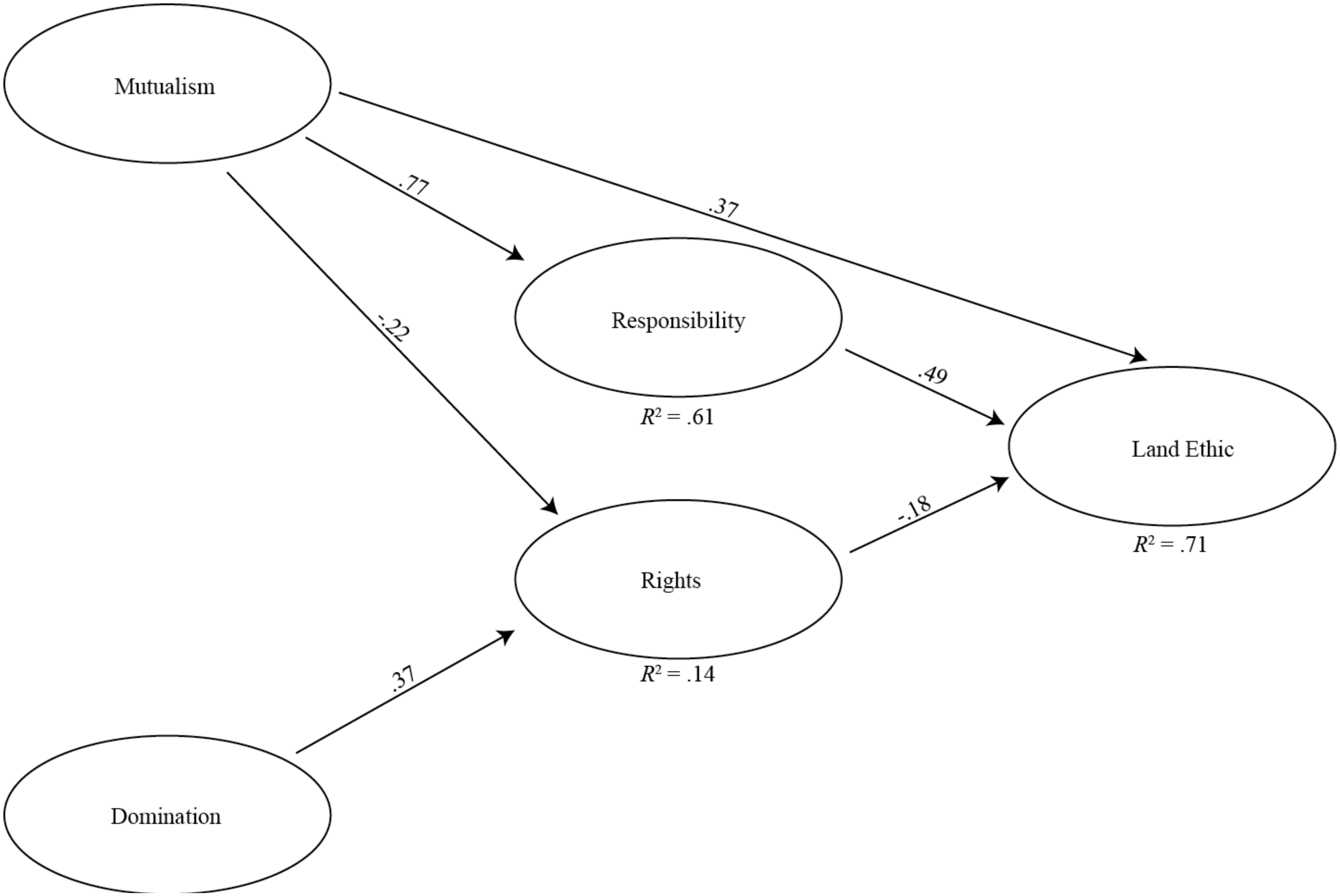

Figure 3. Empirical relationships among belief constructs (Only significant paths are displayed.)

Hypotheses 3 and 4 predicted rights would be negatively related to mutualism and positively related to domination. Standardized coefficients for mutualism ($\beta = -.22$, $p < .05$) and domination ($\beta = .37$, $p < .05$) were significant and in the predicted direction; Hypotheses 3 and 4 were accepted (Figure 2). Mutualism and domination explained 14% of the variance in rights.

Hypotheses 5 to 8 examined the relationships between the belief constructs and the Land Ethic. The standardized coefficients for mutualism ($\beta = .37$, $p < .05$) and rights ($\beta = -.18$, $p < .05$) and responsibilities ($\beta = .49$, $p < .05$) were significant and in the predicted direction; Hypotheses 5, 7, and 8 were accepted (Figure 2). There was no significant relationship between domination and the Land Ethic; Hypothesis 6 was rejected. Together, the four constructs explained 71% of the variance in the Land Ethic.

9. DISCUSSION

The general domination-mutualism constructs predicted the more specific rights and responsibility concepts, which in turn predicted the Land Ethic. Although this theoretical approach has been applied to a range of topics, this is the first application to farmer beliefs about land and land ownership. A strong mutualism relationship with land was observed. By comparison, farmers' domination orientation was more neutral. The findings suggested that mutualism is partially derived from an emotional bond with the land (e.g., "I feel an emotional bond with the land"; $M = 2.08$) and the importance of conserving land for future generations (e.g., "Conserving land is important for future generations"; $M = 2.40$). The mutualism orientation was somewhat unexpected given the age ($M = 62$ years), sex (90% male), and rural location of our sample population. Prior research has found that mutualism is more common in younger, female, and urban samples (Lloyd & Miller, 2010; Martinez-Espineira, 2006; Vaske et al., 2001).

The mutualism findings were consistent with previous research that showed farmers generally self-identify as stewards of the land; 80% or more of farmers claimed to be land stewards (Ahnstrom et al., 2009). In a review of agricultural practice adoption, Prokopy et al. (2008) found that positive environmental attitudes and awareness were always positively associated with practice adoption. Beedell and Rehman (2000) showed that some farmers' decisions were influenced by their own ethical beliefs and social pressure from important referents. Thompson et al. (2015) found that Indiana farmers held strong profit and stewardship beliefs but never held negative stewardship beliefs.

Whereas farmers' strong stewardship beliefs were expected, I also predicted them to hold a domination value orientation. Farming is a pursuit that involves altering the land for human benefit. Other research shows hunters to hold stronger domination- than mutualism-oriented beliefs about wildlife than non-hunters (Teel et al., 2005). Hunters may believe that the primary purpose of wildlife is human use yet may still hold strong ethical beliefs toward wildlife. Domination was also the primary value orientation of most Americans and related to Judeo-Christian religion (Kluckholn & Strodtbeck, 1961; Schwartz, 2006; Manfredo et al., 2009). Burton (2004) sheds light on this nuance in his discussion of the symbolic role of utilitarian-oriented 'productivist' behavior in which farmers maximize production to obtain financial rewards. The symbolic value of production becomes part of a farmer's self-identity and is supported through community norms and behavior (Burton, 2004). The symbolic value of behavior emerges from farmers' strong connection with the land and their role in taking care of the land (Burton, 2004). The nurturing role embodied in mutualism is not necessarily contradictory to 'productivist' behaviors in which farmers maintain tidy, weed-free farms or maximize their economic utility. This may explain why farmers responded to individual domination and mutualism items positively in all but one case.

The rights and responsibility constructs examined more specific farmer beliefs about land ownership. These constructs related to whether farmers emphasize their own right to use the land as they wish, even if it harms the land (e.g., threatens wildlife, reduces water quality) or other people (e.g., through pollution of the water supply of neighbors or nearby communities) versus their responsibility to make socially responsible land use decisions. In other words, farmers might refrain from an agricultural practice if they believe it could harm others or the land. Leopold recognized this fundamental balance of responsibilities that comes with land ownership when he noted that "the crux of the problem is that every landowner is the custodian of two interests, not always identical, the public interest and his own" (Freyfogle, 1999).

In this research, farmers agreed with both rights and responsibilities items suggesting that they understood the balance between the rights of individuals and the rights of society to benefit from responsible ownership. The finding that farmers more strongly agreed with responsibilities than rights items is consistent with the mutualism orientation. Finding that farmers emphasize responsibilities over rights is counter to conservation's assumptions and to public discourse about property rights (Jackson-Smith et al., 2005). The results, however, were consistent with research on ranchers in Utah and Texas, where 38% of the respondents believed the interests of society must be considered when making land-use decisions (Jackson-Smith et al., 2005; Stroman et al., 2016).

To enhance the reliability of the responsibility construct, one of the items was dropped. Overall, 68% of farmers disagreed with the item, "the public has a role in deciding how private land is used." indicated that 75% of farmers agreed that "Conservation is a voluntary choice of the landowner." The farmers believed they are responsibility for the stewardship of land, but on a voluntary basis that respects their perceived rights of ownership.

Farmers consistently slightly agreed with all Land Ethic items. This suggested that many Illinois farmers fall into Leopold's group B, which he contended would be more likely to develop an ethical relation to land (Leopold 1949). When analyzed together with mutualism-domination and rights-responsibilities questions, a more complete picture emerges. Farmer responses were strong to questions related to the whether or not farmers are responsible for taking care of the land. Consistent with many other studies, farmers clearly believed they are obligated to conserve the land and wildlife (Ahnstrom et al., 2009).

Findings here also were consistent with "dual-interest theory," which suggests that egoistic-hedonistic based self-interest and empathy-sympathy based other interests (e.g., moral obligation) are internalized within an integrated self-interest (Czap et al., 2012; Lynne, 1999). According to this theory, empathy and sympathy for others are expected to "temper self-interest on the way to one's own self-interest" (Czap et al., 2012). Sheeder and Lynne (2009) found that Nebraska farmers who empathized with downstream water users were more likely to practice conservation tillage. These farmers sometimes also held seemingly contrary beliefs (domination and mutualism, and rights and responsibility) simultaneously suggesting support for dual interest theory.

Overall, farmer responses showed an orientation toward mutualism and responsibility. Combined with the positive response to the Land Ethic items, it might seem that farmers were in line with Leopold's vision of the Land Ethic. Responses to some individual items, however, suggested that farmers may hold a somewhat different view. Whereas many farmers believed they have a responsibility to take care of the land, most see conservation as a voluntary choice in which the public should have little role. In general farmers agreed with the Land Ethic but felt that the public has no role in their decision making. This suggests that Illinois farmer's motivation for stewardship might be more inward facing (e.g., toward future generations of farmers that might use their land) than what Leopold espoused (i.e., obligation to the broader land and human community). I suggest that future researchers explore the relationships between the belief constructs measured in this article and motivation for land stewardship.

Leopold made a connection between mutualism, moral obligation (responsibility) and the Land Ethic in his writings. He believed that farmers were more likely to use their land responsibly if they felt that they were a part of the land community and had an emotional appreciation for it (mutualism) (Freyfogle, 2012). The results of the structural equation model demonstrated this connection empirically by showing the close relationship between social psychological constructs of mutualism and responsibility, and the land ethic. The model shows that mutualism influenced the land ethic, and that rights and responsibilities mediated this relationship. Thus, in the case of Illinois farmers, our research confirms Leopold's views. Farmers with a mutualistic orientation were more likely to believe they should use their land in a responsible manner.

Our research also suggests a close connection between the way social psychologists operationalize these concepts and Leopold's Land Ethic. Comparing responses to Leopold's own statements to concepts from the cognitive hierarchy reveals the core components of the Land Ethic. Whereas mutualism on its own is predictive of the Land Ethic, rights and responsibilities are important additional components that I suggest being included in future studies of farmers and other private landowners. Domination did positively influence the rights construct as predicted but was not statistically associated with either responsibility or the Land Ethic as hypothesized. The potential influence of domination in the model was likely overshadowed by mutualism and responsibility, both of which are more closely aligned the Land Ethic ideology.

10. CONCLUSION

Social psychologists have used the theoretical framework of the cognitive hierarchy and value orientations to understand environmental behavior in a variety of contexts (Stern et al., 1999; Bengston, 1994; Vaske and Donnelly, 1999; Fulton et al., 1996; Manfredo et al., 2009; Buijs, 2009). This article extended the research to understand farmers' value orientations about the land and beliefs about the rights and responsibility that comes with land ownership. I demonstrated that domination, mutualism, rights and responsibility within the hierarchy provided a useful framework for predicting the nature and orientation of a Land Ethic in farmers. The more specific responsible land ownership belief was more predictive of the Land Ethic than the more general mutualism value orientation. Future research should explore the degree to which these cognitions including the Land Ethic influence other elements of the hierarchy and ultimately farmer behavior. Although this article focuses on Illinois farmers, the framework can be applied to range of conservation topics.

Researchers have suggested that activating farmer stewardship values is important for engaging farmers in conservation (Thompson et al., 2015). This article suggests that conservation program design and communications that convey mutualism and responsibility could be effective. Solving large-scale, conservation problems will require change in the behavior of thousands of farmers. Conservation solutions that tap into deeply held value orientations have the potential to result in longer-lasting changes in farmer behavior (Jones et. al., 2016). Illinois farmers agree with many of the components of Leopold's message highlighting the potential for conservation strategies that encourage socially responsible private land use.

CHAPTER THREE: HABITAT EXCHANGES FOR GREATER SAGE-GROUSE – A CASE STUDY OF BIODIVERSITY OFFSET DESIGN IN THE UNITED STATES

1. SYNOPSIS

Mitigation of development impacts through biodiversity offsetting often is used to conserve biodiversity while allowing development to continue. Biodiversity offsetting (BO) programs are expanding globally, yet the number of transactions remains relatively low. A new approach to biodiversity offsetting, called Habitat Exchange (HE), was introduced in the United States to conserve habitat for the greater sage-grouse in two states. These HEs are reviewed in terms of the extent to which their design features address common theoretical and practice challenges facing biodiversity offsetting, including currency, no net loss, equivalence, longevity, time lag, uncertainty, reversibility, and thresholds. I find that HE design rules and processes thoroughly address each of these challenges. But despite an apparently sound design, HEs have garnered few transactions overall and have not been broadly applied in the United States. While overcoming the theoretical and practical challenges is important, I conclude from this case study that the primary barriers to widespread use of HEs are not practical, theoretical, scientific or technical, but sociopolitical in nature. The HE experience serves as a reminder of the importance of considering and integrating multidisciplinary social science, concepts, and approaches as part of design to improve chances of success. Use of existing socio-ecological frameworks as part of a thorough, science-based situational analysis prior to design, is recommended to more comprehensively incorporate solutions into the design phase and position BO to generate greater conservation benefit.

2. INTRODUCTION

As global development pressures increase, biodiversity continues to decline. These losses continue despite mounting policy responses and localized successes (Butchart et al., 2010). Mitigation of development impacts through biodiversity offsetting is often is used as a means of conserving biodiversity while allowing development to continue. Biodiversity offsets are defined using three criteria: (1) "They provide additional substitution or replacement for unavoidable negative impacts of human activity on biodiversity, (2) they involve measurable, comparable biodiversity losses and gains, and (3) they demonstrably achieve, as a minimum, no net loss of biodiversity" (Bull et al., 2013).

Biodiversity offsetting is a multi-faceted conservation approach. It began with wetlands mitigation in the mid-1970s in the United States and originated as a policy tool to permit development while recognizing and offsetting the losses of biodiversity incurred in the process. From this beginning, BO has evolved into an increasingly science-based conservation approach (Devictor, 2015; Gordon et al., 2015). BO is an approach that is often viewed as a means of satisfying multiple disparate stakeholder interests and thereby reducing conflict between stakeholders. For example, from the developer point of view, BO can be a means to secure permits to operate or build (Brownlie & Botha, 2009). From the investor standpoint, BO is intended to minimize risks associated with impacts on biodiversity (Burgin, 2008). BO may be legally required (e.g. Australia; Dupont, 2017; Hillman & Instone, 2010), or voluntary (e.g. the Rio Tinto mine in Madagascar; Bidaud et al., 2015; Doswald et al., 2012).

Biodiversity offsetting has rapidly expanded to more than 50 state-based programs in 45 countries including Australia, Canada, the European Union, Brazil and the United States, and privately-run programs in South Africa, New Zealand and Madagascar (Bennett et al., 2017; Bull et al., 2015; Madsen et al., 2011; Maron et al., 2015). Despite widespread diffusion, the effectiveness of BO in conserving biodiversity remains unclear, and the number of transactions completed through these programs remains relatively low (Bennett et al., 2017). Scientists generally conclude that the approach is sound in theory, and there is common agreement about the conceptual and practical challenges of implementation (Bull et al., 2013; Gibbons & Lindenmayer, 2007; Maron et al., 2012; Moilanen et al., 2009). However, due to a lack of empirical studies using well-established and monitored baselines or counterfactuals, the actual conservation benefit generated by BO (i.e. no net loss or net gain) has not been demonstrated (Zu Ermgassen et al., 2019; Curran et al., 2014).

Limited scientific evaluation of ecological outcomes is a problem throughout conservation programs and is not unique to BO (Ferraro & Kiss, 2002; Neugarten et al., 2011). Rigorous evaluation is needed to assess how BO design choices influence biodiversity outcomes. However, many BO programs, including Habitat Exchanges (HEs), have not yet generated a sufficient number of offsetting transactions through which to evaluate outcomes (Bennett et al., 2017). HEs are one type of BO design being advanced in the United States for the greater sage-grouse (GSG; *Centrocercus urophasianus*) and other species. In lieu of outcomes evaluation, case studies provide a contextual resource to contribute to the refinement of design, standards and policy by illuminating design choices and intentions (Norton, 2009).

Our motivation for conducting this case study comes from the low number of transactions obtained through two existing HEs designed for GSG in Colorado (0 transactions) and Nevada (3 transactions). This experience is consistent with the global performance record of BO. In 2013, Bull et al. introduced an overarching conceptual framework and set of considerations to address common theoretical and practical design challenges. Since then, the framework has become a common reference in the literature on BO (Google Scholar counts 322 citations as of January 30, 2020). Our primary goal is to compare the design features of the HEs to the challenges outlined by Bull et al. (2013) as a way of assessing the extent to which HE designs are sufficient to encourage adoption. Our basic question regarding technical aspects of the program was: Are there considerations that should or could have been made that would have increased the chances of uptake? Secondarily, I also reflected upon other considerations that could have influenced acceptance and uptake of the program.

This analysis is informed by the experiences of the authors affiliated with Environmental Defense Fund (EDF) and Environmental Incentives, LLC (EI) as progenitors and stakeholders in the two HE-launch initiatives that are the subject of this article. It also is informed by the Cornell University-affiliated author's experience as an engaged researcher studying the HE launch initiatives. By drawing on our group's experience to describe how design choices were incorporated as potential solutions to key BO challenges—and, critically, with what consequences—I aim for this analysis to provide a resource for conservation practitioners faced with addressing similar challenges with BO.

3. MATERIALS AND METHODS

The design features of the two HEs under investigation are documented in their respective operations manuals (Colorado Habitat Exchange, 2015b; State of Nevada Sagebrush Ecosystem Program, 2017b). I reviewed these documents looking for similarities and differences between the two programs. Because I identified few substantive differences, in this paper I present one generalized model of HE design. Using this model, I evaluated the extent to which the design features of the HEs addressed the challenges with BO identified by Bull et al. (2013). In addition, I address BO governance as it forms the context for dealing with the practical and technical challenges that may arise. Governance itself can also entail practical and technical challenges.

Given the experiences of the two HE initiatives under study, I view this analysis as an opportunity to openly learn from our experience, a vital yet often overlooked aspect of conservation and adaptive management (Catalano et al., 2018). HEs have not been as widely adopted nor have they generated offsets as rapidly as was expected at the outset. This article provides us, as lead designers and facilitators involved in these first HEs, with an opportunity to reflect on lessons I learned and disseminate these lessons to the conservation community. I emphasize that description of the HE design features and our discussion are based on the thinking and circumstances that guided efforts to develop the CHE and NCCS during a specific time period (2010-2017). At the time of this writing, both the CHE and NCCS continue to be actively under development with a focus on scaling these programs up.

4. THE ORIGIN OF GREATER SAGE-GROUSE HABITAT EXCHANGES

U.S. conservationists have called for new, innovative, proactive approaches to conserve species that are listed or proposed for listing under the Endangered Species Act (ESA; Bean, 2015; Donlan & Rothacker, 2015). Over the past decade, HEs emerged in response to this call as on-the-ground pilot initiatives to expand upon policy and practice associated with species conservation banking (a long-tenured approach to conserving listed species in the United States), recovery credits trading, and other market-based conservation approaches (Kreuter et al., 2017).

The program designers set out to advance the conservation banking model in three ways. First, I aimed to expand mitigation from a piecemeal, parcel-by-parcel approach to a landscape-scale approach more typical of global BO approaches. Second, I aimed to reduce conflict between energy production, wildlife conservation, and agricultural production. Third, I emphasized the need to bring multiple stakeholders together in the design of the solution (e.g., agriculture, government, development, and non-governmental organizations). During this period, draft mitigation policy in the United States seemed to be heading conservation banking toward a no-net-loss standard (USFWS, 2016b). Given that policy development is underway, HEs served as models for demonstrating how the shift towards no net loss could be made in practice (Carreras Gamarra & Toombs, 2017). Development of the Colorado Habitat Exchange (CHE) and the Nevada Conservation Credit System (NCCS) was initiated in 2010 and 2012, respectively, to address mitigation for the declining GSG.

The HEs were promoted as an innovative approach to address specific conservation needs that EDF recognized as strategically important to respond to and to address. First, energy and mineral development in GSG habitat was contributing to habitat losses and the decline of GSG populations, despite some voluntary mitigation offered by developers. Second, stakeholders agreed that a proactive approach was needed to stimulate mitigation to help prevent the need to list. Third, current mitigation efforts lacked sufficient provisions to measure habitat impacts and corresponding offsets to establish equivalency. Finally, the landowner community, which was impacted by both energy and mineral development and potential GSG listing, was willing to support mitigation, but not yet extensively involved in mitigation. EDF, EI, and the Colorado Cattlemen's Association (CCA) partnered to address these problems by stimulating a mitigation offset market through which private landowners could offer habitat improvements for GSG on their land as offsets.

In Nevada, the Department of Conservation and Natural Resources (DCNR) recognized a similar set of conservation needs as were identified in Colorado. DCNR worked with EI to develop the NCCS by adapting the design of the CHE to conditions relevant for Nevada. At the time of this writing, the CHE is still under design and negotiation, while the NCCS has been in operation since its approval by the State of Nevada in December 2014. To date, the NCCS has enrolled approximately 50,000 acres in offsets and facilitated three transactions (State of Nevada Sagebrush Ecosystem Program, 2018).

The GSG HEs in both states were designed with the input of stakeholder committees comprised of conservation, industry, agriculture, and federal and state representatives. Both design processes were led jointly by EDF and EI following principles consistent with the Business and Biodiversity Offsets Programme (BBOP) in the resulting programs, including: (1) net gain, (2) mitigation hierarchy, (3) equivalency, and (4) adaptive management (BBOP, 2012; Norton, 2009). With these principles as baselines, the stakeholders were led through the design process.

5. HABITAT EXCHANGE GOVERNANCE AND REGULATORY CONTEXT

5.1. Governance

Central to HE design is its programmatic structure (Figure 4). This structure was intended to address lack of consistency and transparency associated with current mitigation approaches by (1) clearly defining roles and responsibilities, (2) making reports publicly available, (3) bringing stakeholders into the decision-making process, (4) using standardized mitigation rules and tools to promote consistency, (5) streamlining the mitigation process for mitigation buyers and sellers to encourage more efficiency and participation, (6) ensuring compliance and accountability to regulators, and (7) protecting confidentiality for buyers and sellers.

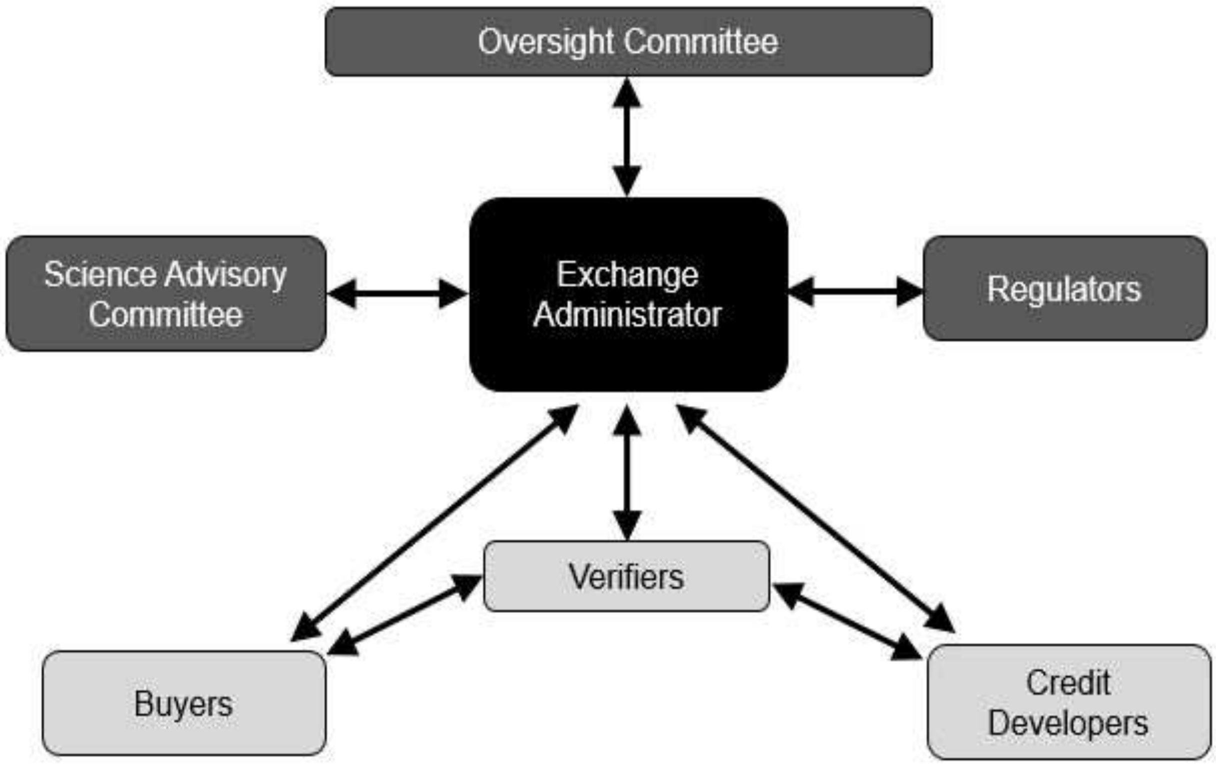

Figure 4. Programmable habitat exchange governance structure

As shown in Figure 4, the *Administrator* fulfills a central role by managing day-to-day operations in accordance with rules outlined the HE manual. Primary duties of the Administrator include facilitating and overseeing all credit generation and transaction activities, releasing credits and reporting results, and managing a registry. The registry is used for tracking credits, debits and transactions. The *Oversight Committee* is a formal stakeholder group that includes representatives from conservation interests, industry, agriculture, and government, and which is responsible for overseeing the operations of the HE. *Credit Developers* typically are private landowners who produce and sell credits. Credit developers may also be bank facilitators, such as conservation bank companies, or other types of aggregators who work with multiple landowners to implement conservation projects, secure financial assurances, and sell credits.

Buyers are entities that purchase credits for mitigation or to meet other conservation objectives. *Regulators* are the agencies that authorize the use of credits for compensatory mitigation. *Verifiers* are third-party contractors that are certified by the Administrator to assess the accuracy of credit and debit calculations. The *Science Advisory Committee* is composed of scientific experts on the target species and its habitat. This committee develops and manages biological standards for the target species and its habitat and makes technical recommendations to the Oversight Committee through the Administrator. The Oversight Committee is responsible for ensuring that the Administrator conducts all HE operations in accordance with the HE manual. Additional responsibilities of the Oversight Committee include approving Science Advisory Committee members, reviewing and approving reports, and reviewing and adopting any changes resulting from adaptive management.

The programmatic structure of HEs and policy and procedures for HE administration are described in three key documents (Figure 5). The HE manual details the roles and responsibilities of the governing entities and associated committees, describes the process for buyer and credit developer participation, includes or references standardized rules and tools for participation, defines how confidentiality is to be addressed, and provides details on the contents of reports and the timing of their submittal. The habitat quantification tool (HQT) methods document outlines the methods, metrics, processes, and scientific justification for the currency of the HE, which quantifies the biodiversity impact and offset (Colorado Habitat Exchange, 2015a; State of Nevada Sagebrush Ecosystem Program, 2017a).

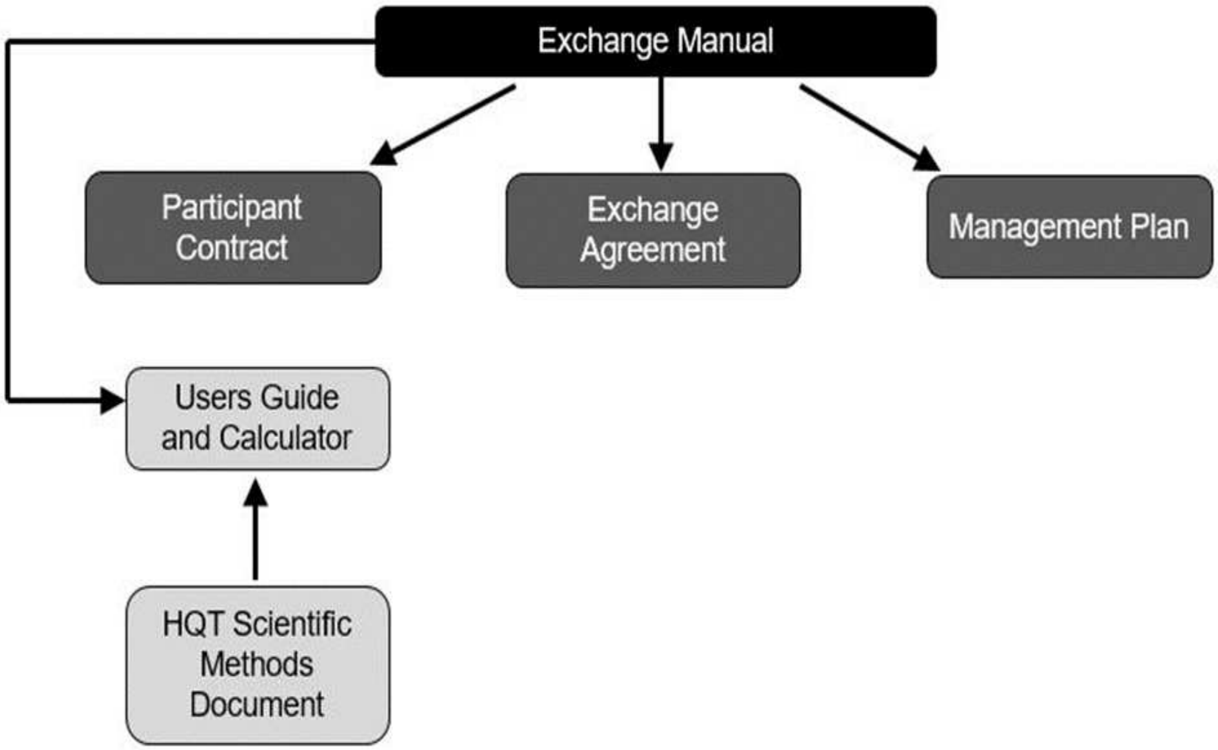

Figure 5. Documents and tools that facilitate habitat exchange operations

The third key document is the Exchange Agreement, which played a role in the development of the CHE but not the NCCS. This agreement serves as a contractually binding document that references the laws, policies and regulations relevant to a particular exchange, provides details on how credits are determined and released, how credits are to be used to offset debits, how landowner confidentiality is to be addressed, and how regulatory predictability may be provided. Exchange Agreements provide templates for the documents needed by credit developers and buyers to participate in the HE (e.g., participant contracts and management plans). Signatories to the Exchange Agreement typically include the Administrator and relevant Regulators.

Exchange Agreements are an evolution of U.S. conservation banking agreements. By establishing one legally binding reference point for stakeholders, these agreements addressed a number of commonly identified biodiversity offset challenges. These include lack of specificity on the extent to which adverse impacts should be offset (McKenney & Kiesecker, 2010), absence of robust landowner confidentiality provisions (Kreuter et al., 2017), and lack of a clear mechanism for landowners and buyers to receive regulatory assurances. By specifying these details, Exchange Agreements also provide a direct connection between biodiversity offsets (credits) and adverse impacts (debits) that has been lacking in U.S. approaches to species mitigation.

At best, the latter approaches require that the spatial extent of the adverse impact be measured, then a subjective mitigation ratio be applied to determine the amount (i.e. the spatial extent) of biodiversity offset required (USFWS Director, 2003). Exchange agreements also address landowner confidentiality and a lack of transparency between stakeholders and between stakeholders and the general public, both of which have been identified as challenges to generating voluntary participation (Kreuter et al., 2017).

5.2. Regulatory Context

The HEs were developed amid a complex, uncertain, and shifting regulatory context. This volatile regulatory landscape posed a significant challenge in the design process. When the HEs initially were being developed in 2010 and 2012, the U.S. Fish and Wildlife Service (USFWS) issued a finding that listing the GSG as endangered was "warranted but precluded by higher priority listing actions" (USFWS, 2010). The GSG remained under active endangered species listing consideration until October 2015 when USFWS issued a finding that conservation plans developed since 2010 were adequate to deem the species not warranted for listing (USFWS, 2015). This finding restored certainty that authority and responsibility for GSG management would be remain with the state governments.

At the time of the 2010 USFWS finding, Colorado and Nevada had differing rules and procedures for mitigation for non-listed species. In addition, none of the relevant federal agencies involved in conservation or management of lands co-extensive to GSG habitat had guidance or policy for pre-listing conservation (USFWS, Bureau of Land Management, and U.S. Forest Service). Under these circumstances, the CHE and the NCCS worked with their respective state agencies and the state and regional offices of federal agencies to connect the HEs to legal authorities and establish agreements recognizing the HEs. Where possible, HEs were intended to fit within existing legal standards and procedures for review and approval of industry projects, not to change these processes.

As an emerging option for compensatory mitigation, HEs were intended to provide consistent, high-quality BO standards for developers that were required to use the HE to meet their mitigation obligations imposed by state or federal agencies. In Colorado and Nevada, developers also had the option to conduct permittee-responsible mitigation or purchase from an approved conservation bank. For those developers who wished to use the HEs but were not so required, the CHE and NCCS were developed with a view to providing a voluntary option to offset impacts. Because regulatory assurances from USFWS can encourage credit developers and buyers to invest in the conservation, the design of HEs contemplated integration with regulatory mechanisms such as Candidate Conservation Agreements with Assurances (CCAAs) to provide developers and buyers with incidental take-protection assurances.

6. HABITAT EXCHANGE DESIGN FEATURES

As discussed, HEs were developed to address perceived shortfalls in the prevailing forms of mitigation being practiced in the United States. Next, I compare specific design features to theoretical and practical challenges identified by Bull et al. (2013) (Table 4).

Table 4. Summary of practical and technical biodiversity offset challenges and design recommendations (adapted from Bull et al., 2013) with corresponding habitat exchange design features

Challenge	Design Recommendation	Habitat Exchange Design Feature
Currency	Use multiple or compound metrics, incorporate measure of ecological function as well as biodiversity	• Habitat Quantification Tool (HQT) is a compound metric used to measure functional acres for both impacts and offsets • Currency is functional acres (area x quality) • Credits and debits calculated for impacts and offsets
No Net Loss	Measure no net loss against dynamic baseline, incorporating trends; state whether no net loss is at project or landscape level; consider discounting rate	• Mitigation Standard • Mitigation Hierarchy • Programs Scope • Landscape Scale Approach • Valid Compensatory Mitigation Measures • Mitigation Ratio (in NV, 'Landscape Importance Factor') • Credit Baseline
Equivalence	Do not allow 'out of kind' trading unless 'trading up' from losses that have little or no conservation value	• Reasonable Relation • Strategic Investment (Trading Up) • Service Area • Proximity Factor • Credit Site Eligibility
Longevity	Offsets should last as long as impacts of development; offsets should be adaptively managed for change	• Matching the Duration of Credits & Debits • Minimum Credit Term • Credit Durability • Credit Project Documentation • Credit Site Protection Instrument • Performance Standards • Site Assessment, Verification & Monitoring • Endowment Funds
Time Lag	Require offsets to be delivered through biodiversity banking mechanisms	• Timeliness (except Advanced Credit Release) • Temporal Loss • Debit Project Duration (reflects impact to species, not strictly duration) • Debit Project Rehabilitation (required)
Uncertainty	Development of a framework for	• Periodic spot checks (Admin) • Credit Project Selection & Design

	uncertainty in offsets is a research requirement	• Adaptive Management (project scale) • Credit Release Schedule & Advanced Credit Release • Best available science
Reversibility	Define reversibility; require all losses to be reversible	• Credit Reversals (Intentional/Unintentional) • Financial Assurances • Reserve Account • Reserve Account Contribution • No Imminent Threat
Thresholds	Define explicit thresholds for impacts that cannot be offset	• Un-mitigatable impacts

6.1. Currency

Currency refers to the metric used to quantify and compare impacts to offsets. Metrics are proxies for biodiversity (Bull et al., 2013). Many types of metrics are used globally, and policies often lack details specifying what kind of metrics should be used, leading to considerable variability and a lack of comparability across projects (Carreras Gamarra et al., 2018; Gardner et al., 2013). Simple metrics such as habitat area alone are insufficient to capture biodiversity and to serve as a BO currency or to measure cumulative impacts and offsets across landscapes (Carreras Gamarra & Toombs, 2017). For these reasons, Bull et al. (2013) recommend using multiple or compound metrics and incorporating measures of ecological function as well as biodiversity.

Habitat Exchanges use a standardized scoring method with compound metrics to define the currency used for comparing impacts to offsets. The currency is "functional acres," which is a measure of habitat function (i.e., quality or condition) multiplied by habitat area. Including function or quality as a measure accounts for variation in quality over space and time. Habitat function is quantified on three spatial scales: the landscape, the surrounding area, and the mitigation site. Included are direct impacts (i.e., surface area disturbance) and indirect impacts (i.e., habitat avoidance) on quality and habitat requirements throughout the complete life cycle of the GSG. Project developers know that their impacts will be measured using the HQT before the project occurs and therefore can use the HQT to predict and reduce their potential impacts prior to project initiation via changes in project siting. Consistent use of the same method of assessing impacts and offsets also allows for comparability between projects, which makes it possible to assess cumulative impacts and benefits over time (Quétier & Lavorel, 2011). Finally, use of a standardized quantification method combined with the unique programmatic design of the HE enables the Administrator to aggregate credits from multiple offset projects into bundles for sale, potentially increasing the efficiency of the program.

6.2. No net loss

Biodiversity offset programs often fail to meet no-net-loss or net-gain standards because baseline rarely is specified or considered as dynamic but instead is assumed to be fixed at the point of project initiation (Bull et al., 2013; Maron et al., 2018). Thus, there is no comparison with which to judge progress, making it impossible to determine whether the standard has been met.

Habitat Exchanges were designed with the goal of meeting a net-gain standard at the project level. The mitigation hierarchy is used first to avoid and then to minimize before compensating for unavoidable impacts. Additionality is defined as an intervention or outcome that is above and beyond what otherwise would have happened (i.e., counterfactual) and therefore is an important consideration for meeting no-net-loss/net-benefit goals. HEs use a different credit calculation method than species conservation banks, helping HEs meet a higher additionality standard. Conservation banks generally calculate credits by applying a multiplier of one for any area of the bank that includes habitat, regardless of whether credits are gained from protection, restoration or both. A lesser multiplier is applied for less important areas that are included in the bank (e.g., buffer areas), resulting in a one-to-one credit ratio (one credit per acre) (Carreras Gamarra & Toombs, 2017). In contrast, HEs calculate credits by subtracting from the assessed habitat quality (expressed as a percentage from 0 to 100) an amount that represents the expected long-term average habitat quality in the geographic region in which the site is located. This amount is referred to as the "credit baseline." For example, a 100-acre site with an assessed habitat quality of 80% and a credit baseline of 30% would result in 50 credits (100 acres * (80% - 30%)).

The concept of credit baseline offers a number of advantages. First, it provides a reasonable measure of the uplift on which to base credit calculations. Second, it allows for the identification and protection of above-average-quality habitats in a region. Third, it avoids penalizing landowners who have done a good job at managing their land in the past, by allowing them to earn credit for excellent existing conditions (i.e., protection without restoration). Finally, it avoids creating a perverse incentive to degrade habitat in order to generate credits. Setting the credit baseline at the condition of the vegetation at the initiation of the offset would make protection of existing habitat alone as a lone strategy obsolete and would make restoration the only way to earn credits.

Still, a complete accounting of baseline provides an understanding of regional trends and specific limiting factors across the landscape (Bull et al., 2014). In many regions, GSG habitat loss is inevitable without intervention. For example, invasive species and climate change can reduce the functionality of existing habitat even when it is not influenced by development. Since the HQT is not used to assess regional habitat trends across the landscape, it is not possible to establish and monitor this dynamic baseline and use it as a comparison for landscape scale attainment of no net loss/net gain. The standard at the landscape scale is only addressed indirectly through comparing cumulative HQT scores over time to trends in GSG populations. The specific or modeled relationship between habitat and population is not made explicit through the HQT or any other means.

6.3. Equivalence

A key challenge for BO is to establish sufficient ecological similarity between the biodiversity impacts and offsets. Technically, no BO achieves true equivalence because not all components of biodiversity are measured or fungible (Salzman & Ruhl, 2000). Habitat Exchanges establish equivalency through use of the same metric for assessing impacts and offsets, enabling an apples-to-apples comparison. In addition, HE rules allow trading up (i.e., out of kind) under special circumstances at the discretion of the regulators (e.g. when the impacts are small or minimal, or when the offsets are known to meet a specific higher need). For example, in some locations within the Nevada GSG distribution, dryland sagebrush habitat is abundant, but utilization of that habitat may be limited by the lack of late brood-rearing habitat, which is dependent on wet meadow vegetation (Casazza et al., 2011; Schreiber et al., 2015). Even if development impacts sagebrush, enhancing wet meadows and protecting the existing surrounding sagebrush habitat can represent trading up.

Other rules that contribute to establishing equivalence include service areas and eligibility criteria. Service areas, defined as geographic boundaries drawn around subpopulations, are used to ensure that the offset benefits the same subpopulation that was harmed by the impact. Eligibility criteria include being located in GSG habitat, meeting a minimum standard for habitat quality as measured by the HQT, demonstrating no imminent threat of development, and agreeing to maintain the site quality throughout the duration of the project. In addition, the Administrator may require some management actions for all offsets that reflect best management practices for the species (e.g., fence flagging, escape ramps in livestock watering facilities, etc.).

6.4. Longevity

To fully compensate for impacts, BO must supply benefits commensurate with the impacts for a duration that is equivalent to the impact, including considerations for lags between the beginning of the offset and the provisioning of habitat benefits. In addition, Bull et al. (2013) recommend that offsets also be managed adaptively for changing conditions. Unlike conservation banking, HEs enable credits of variable term lengths, allowing for these considerations to be taken into account.

HEs have several mechanisms to ensure longevity. Offsets must be contemporaneous with impacts and be maintained until the impact is rehabilitated. To ensure this, site assessment, verification and monitoring rules require third-party verification that the impacts have returned to baseline conditions before offsets can expire. Annual monitoring generally is conducted by the owner of the credits (i.e., landowner) and verification is completed by verifiers certified by the program at the beginning of the project and every few years thereafter.

During this time, a site protection instrument (i.e., agreement) is required in association with the participant contract and the site management plans. The instrument outlines all pertinent legal arrangements, management and enforcement of any restrictions that will ensure protection of the credits on the site, and any other information required by applicable laws. A minimum term duration is established to ensure offsets exist long enough to provide meaningful benefit for the species (e.g., 30 years is the minimum credit term in the NCCS).

The following types of credit projects are allowed: 1) habitat conservation – maintenance of existing high-quality habitat, 2) habitat restoration – creation of new habitats where habitat has been lost, and/or 3) habitat enhancement – improvements to existing low-quality habitat. Permanent credit projects must secure a covenant, conservation easement, deed restriction or similar device to demonstrate durability. Term credit projects must have in place a *Participant Contract and Management Plan* (Figure 5) that includes appropriate language to ensure durability for the duration of the project. Public lands credit sites also are required to ensure sufficient site protection by entering into an agreement with the applicable public lands agency. Management plans require the use of adaptive management to adjust to changing conditions and to ensure the success of restoration projects. Performance standards are described in the management plan which the credit project must achieve and maintain in the specified time period.

To help ensure longevity of projects, endowment funds are established for all offsets to fund the long-term management and monitoring of the offset. Endowment fund levels are determined by the Administrator and set to reflect a full cost accounting of the project over its lifetime.

6.5. Time Lags

There is a risk that offsets will not be realized prior to the occurrence of the impact (Quétier & Lavorel, 2011). Offsets must be contemporaneous with impacts and account for temporal loss. To address this issue, HEs require that most of the offset credits be in place and verified by a third party before they can be released for sale and used as offsets.

The overall intent of these credit-release rules is to eliminate the risk that benefits will not accrue due to failed restoration or time lags in restoration, and to encourage private investment in advanced mitigation prior to the impact. If buyers can anticipate their future mitigation needs, then they can purchase restoration credits that are likely to be in place prior to their need to develop. The goal of the HEs is to encourage this type of pro-active mitigation marketplace.

Debit project durations are based on permit durations but also include considerations for rehabilitation and the return of the species to the impacted site (in Nevada, debit terms are extended by 10 years to allow for GSG repopulation of the site**).** Debit project rehabilitation is required for term debit projects but is only enforceable by the permitting agency.

6.6. Uncertainty

The outcomes of offsets are uncertain, and this often is addressed in conservation banking through the use of mitigation ratios (Carreras Gamarra & Toombs, 2017). The HEs seek to use more accurate accounting through the HQT to minimize uncertainty that the offset is equivalent to the impact. Credit project selection and design rules reduce uncertainty that the offset project will provide the benefits expected. The credit release schedule is defined in the project's management plan and is tied to achievement of performance standards as verified by a third party. Except in cases of advanced credit release, credits are release only after achievement of habitat quality and functional acres corresponding to the amount of credits released.

HEs institute adaptive management at the site and programmatic level, which is a strategy to address uncertainty related to lack of knowledge at the outset and to account for changing conditions and new scientific information over time. The HE adaptive management process has been adapted from the Conservation Measures Partnership's *Open Standards for the Practice of Conservation* (2013).

Site eligibility criteria can also increase confidence that the project will benefit the GSG. To be eligible, projects must be designed to meet minimum habitat function by the end of the project. Sites must be located within the same service area as the impact and must not have an elevated risk of development within the time frame of the contract.

6.7. Reversibility

Reversibility can refer to the potential for restoration through offsets to achieve conditions similar to those impacted (Bull et al., 2013), or it can refer to a situation in which the credits secured on the offset site are lost for some reason. Since I address the former definition above, this section is focused on the latter.

Credit reversals occur when credits are lost to impact, degradation, force majeure, neglect or other causes, and may be intentional or unintentional. According to the HE manual, when unintentional reversals (e.g., because of force majeure) occur, the Administrator withdraws credits from the reserve account (explained below) to cover the invalidated credits at no cost to the Credit Developer. The Administrator then uses the remaining funds in the project site's financial assurances to remediate the credit project or replace credits off-site to the degree possible. When intentional reversals occur, the Administrator withdraws credits from the reserve account to cover invalidated credits. The Credit Developer is responsible for fully replacing all invalidated credits using the project site's financial assurances and also must pay a 10% administrative fee.

The programmatic structure of the HE makes it possible for the reserve account to address these reversals, should they occur. The reserve account is an account of excess credits, not dollars. In the CHE, each Credit Developer contributes a total of 11% of their credits to the reserve account, which includes a 4% base contribution and a 7% split-estate risk contribution. If a reserve account is not utilized (no reversals), then these credits are counted toward net gain.

The HEs require that Credit Developers establish appropriate financial assurances for each credit project site in order to sell credits. Financial assurances are fiscal mechanisms that are used to ensure the durability of credits generated throughout the full duration of a credit project. Financial assurances are defined in each Credit Developer's Participant Contract and documented in an accompanying Management Plan, and can consist of contract terms, such as financial penalties for intentional reversals and specific payment terms, and financial instruments, such as long-term stewardship funds and contract surety bonds.

Financial assurances must ensure that funds are available: 1) for the implementation and long-term management of each credit project, including remedial actions in the event of unintentional reversals, and 2) to promptly replace credits that have been transferred but become invalidated due to intentional reversals. Credit site eligibility requirements, including requirement of no imminent threat, also reduce uncertainty.

6.8. Thresholds

In order for offsets to compensate for impacts, it must be practically and ecologically possible to restore those impacts (Godden & Vernon, 2003). If not, those impacts should technically be considered off-limits to developers with these restrictive thresholds explicitly defined in policy (Quétier & Lavorel, 2011). What is off-limits to offsetting (i.e., unmitigable impacts) is defined in the HE manual or in accompanying policy. For example, in Colorado, GSG breeding areas or leks are off-limits to development, and therefore not eligible for offsetting.

7. DISCUSSION

Our case study shows that the specific design features incorporated into HEs address all of the theoretical and practical challenges previously identified by Bull et al. (2013) (Table 4) as part of the overall intent of the program. Despite systematically addressing these challenges, the HEs have generated few transactions and therefore very little direct conservation benefit to date. This brings us back to our original research question: are there considerations that should or could have been made that would have increased the chances of uptake? In this vein, I suggest the primary barriers to widespread use of HE may not be practical or theoretical, or even scientific or technical, but social and political in nature. Therefore, I turn my attention toward sociopolitical factors to consider during design and that influenced HEs and may be relevant for BO globally.

By sociopolitical factors, I refer to the broad social and political contexts in which initiatives to roll-out HEs in Colorado and Nevada were situated. I characterize these contexts in terms of what incumbent norms and institutions were relevant to introducing HEs to the conservation policy field of these states and to the broader region and community of practice as a new model of BO, and the institutional legitimacy (Suchman, 1995) with which the HE-launch initiatives were received.

Norms refer to relatively informal practices undertaken by a group or groups based on general understandings of what constitutes “normal” courses of action and objectives. Institutions in turn refer to norms that have been formalized as guidance, policy, rules, and laws. By extension, institutions also include the organizations that create and administer guidance, policy, rules, and laws. Institutional legitimacy refers to the “generalized perception or assumption that the actions of an entity are desirable, proper, or appropriate within some socially constructed system of norms, values, beliefs, and definitions” (Suchman, 1995).

At the most fundamental level, I suggest that legitimacy was a barrier to the acceptance and adoption of HEs (Dowling & Pfeffer, 1975). Building collaborative governance arrangements such as HEs involves bringing together multiple institutions and stakeholders that may hold disparate goals, values, and approaches which can be difficult to reconcile, and lead to failure (Kraft & Wolf, 2016). Fundamental lack of legitimacy can be a barrier to success in environmental governance that transcends technical or practical challenges (Kraft & Wolf, 2016).

Conflicting stakeholder beliefs and norms regarding private property rights and mitigation were observed during the design of the HE, yet some of these were either not overcome or not explicitly acknowledged or discussed in the design process. Fundamentally, the HE is a rights-based conservation approach in that it is relies on the presumption of secure and somewhat absolute property rights (i.e., the right to develop oil and gas or the right of a rancher to destroy sage-grouse habitat), yet these rights in practice are not absolute in the sense that they are sometimes associated with legal or social constraints. It is likely that stakeholders differed in their perception of how absolute property rights were in this context, which could influence stakeholder willingness to accept various levels of constraint (i.e., regulations requiring HE use, or the amount of mitigation required for an oil and gas project).

Norms surrounding mitigation likely were different among stakeholders, as well. Expectations and behavior, including those formalized in law and those not formalized, for what constitutes an acceptable level of mitigation were not evenly agreed upon by stakeholders. From our perspective, the HE program design ushered in a dramatic change to mitigation norms. As Galik et al. (2017) mention, competition with cheaper low-quality credits is a function of the lack of regulatory standard for purchasing high quality credits. Prior to the initiation of the HE design process in CO, the state lacked a robust and consistent mitigation framework for sage-grouse, which influenced stakeholder expectations for the HE. Switching from the old process to the new represented a big leap for some stakeholders. While norms are powerful predictors of behavior, they are not easy to change (Keizer & Schultz, 2018), and are unlikely to change over short time periods. For HEs to be viewed as a legitimate approach by all stakeholders, policy and social norms would be mutually reinforcing over time, but during the course of our project, the legal structures (i.e., current requirements and legal drivers of mitigation) and the norms that HE conveyed were in conflict (i.e., not suggesting the same level of mitigation as appropriate).

I argue that the HE development process and related discussions shifted the norms (i.e., what to expect from mitigation projects and programs) surrounding mitigation for the GSG. Although it was later rescinded by the Trump administration, the inclusion of HEs in the U.S. Fish and Wildlife Service final compensatory mitigation policy is a perfect example (finalized and published) of changing norms at the level of the broad policy field (USFWS, 2016a). Another piece of evidence of a norms shift, is that, more locally and recently, the states have been pursuing legal strategies that will expand government authority to require offsetting through the HEs (Colorado Senate Bill 19-181, 2019; State of Nevada Sagebrush Ecosystem Council, 2019).

With regard to guidance, policy, rules, and laws, the novel status of the HEs as a new model for BO complicated recognition by state and federal agencies of the HE as an approach to compensatory mitigation. Federally, policy for compensatory mitigation had been premised on a 1981 policy statement under which USFWS had recognized conservation banks, but nothing like the HEs. Without an established policy precedent, the process of designing a HE that met federal standards and projected regulatory certainty to the regulated community took place amid a dynamic unfolding internal conversation in which it was unclear whether USFWS would ever recognize the HEs. Although USFWS later acknowledged the HEs in a policy update (USFWS, 2016a), the agency's early reluctance to assign its imprimatur was a major barrier to traction.

The Bureau of Land Management (BLM) also was a primary federal agency relevant to policy affecting the HE design process. HE program designers sought for BLM to recognize HEs for compensatory mitigation within agency rules and processes. Under the Federal Land Policy and Management Act, BLM had long worked with oil and gas, mining, and other developers on federal land on compensatory mitigation projects using mechanisms distinct from the approach embodied in the HE (e.g., permittee-responsible mitigation, fixed offsetting ratios). The HE approach to quantifying offsetting obligations, the involvement of new actors, and other distinguishing features hampered recognition of the HE by BLM as a legitimate, normative approach to compensatory mitigation. For example, after a Memorandum of Understanding (MOU) was established between the State of Nevada and the BLM, HE inclusion in the state BLM permitting process remained inconsistent.

At the state level in Colorado and Nevada, both states had regulatory frameworks to provide a legal basis for the HEs to operate when the HE-launch initiatives began. However, as with the USFWS and the BLM, the HE-launch initiatives encountered difficulties related to norms and legitimacy at the state level. In Colorado, oil and gas industry disagreed with the high offset obligations produced by the HE HQT. At the same time, the industry took advantage of a loophole to BO requirements. These circumstances produced quasi-voluntary aspect to the HE program that dampened demand for credits (Large & Wolf, manuscript in preparation). In Nevada, the NCCS was premised on a statute which called for the NCCS' creation but did not explicitly specify the state's authority to require developers to purchase NCCS offsets. Again, the lack of clear authority created uncertainty in the market of potential credit suppliers and buyers, limiting traction.

Conflicts between stakeholders were also present with respect to how habitat was quantified. Galik et al. (2017) suggest that, for habitat quantification, tradeoffs between accuracy, complexity and practicality of use, and that repeatability and consistency are important. While I generally agree, from my perspective the complexities in habitat quantification in HEs were not primarily technical or scientific. HQTs were designed using panels of highly qualified and respected scientists and credit determination processes were rapid (one hour to complete initial assessment), repeatable, and consistent. Instead, I suggest that trust between stakeholders, including differences in how willing those stakeholders were to follow the science, posed primary barriers to achieving mutually agreeable solutions.

The science also revealed impacts that resulted in fundamental economic challenges for oil and gas developers that could not be overcome in a voluntary framework. Oil and gas development is a diffuse but sprawling impact (Trainor et al., 2016) and high levels of impact on GSG were reflected in the CO HQT. Thus, the CHE would have required much higher levels of offset and hence cost to developers than prior mitigation schemes in that state. Without strong regulatory drivers to require purchase of credits, the purchase of HE credits was left to voluntary participation. In our experience, voluntary purchase is unlikely when credit cost is high. These barriers were lower in NV where the primary impacts are mining, which has a more concentrated footprint resulting in different impacts to GSG than oil and gas.

This experience suggests that the process of stakeholder engagement deserves greater consideration in BO design. Participatory approaches to design hold the promise of achieving greater trust and learning among stakeholders, leading to more acceptable and durable conservation programs (Reed, 2008). HEs were developed through a collaborative stakeholder development process which was largely successful in generating collaborative design decisions but failed to help the group overcome difficult tradeoffs between disparate interests, such as the tradeoff between a scientifically robust offset and the financial cost of the offset to project developers. The process was not inclusive of all stakeholders, for example, I did not include non-governmental organizations who opposed market-based solutions or preferred different solutions. I did not fully consider alternative approaches, but instead built partnerships with organization who generally agreed with my predetermined approach. Galik et al. (2017) suggest that the lack of adequate process standards was a barrier to HE success. Reed (2008) and Luyet et al. (2012) offer frameworks for determining the most appropriate process for stakeholder participation, and neither of these frameworks were used to inform HE design processes. The lessons here are to give serious, careful, and systematic consideration to sociopolitical relations among stakeholders, inclusion, and to consult experts to determine the most appropriate stakeholder process, and to utilize experts in facilitation and conflict resolution, in order to increase the chances for success.

7. CONCLUSIONS

Through this case study, I have demonstrated how HE design features address the practical and theoretical challenges outlined by Bull et al. (2013). Yet, despite an apparently sound design, HEs have garnered few transactions overall and have not been broadly applied in the U.S. for GSG or other species, hindering the ability of their innovative design to provide conservation benefit. The lack of uptake in this case is consistent with many other global instances of BO, which, despite the proliferation of programs, lack widespread uptake (Bennett et al., 2017). While I agree with Bull et al. (2013) that overcoming the theoretical and practical challenges is a key to achieving BO’s promise, I conclude from this case study that the primary barriers to widespread use of HE are not practical or theoretical, or even scientific or technical, but sociopolitical in nature.

The HE experience illustrates the limits of market-based approaches, and serves as a reminder of the importance of considering and integrating multidisciplinary social science, concepts, and approaches as part of design to improve chances of success (Bennett, 2016). Long-term conservation success hinges on its full integration with the values and goals of society, which must be understood by program designers (Mascia et al., 2003). Use of existing socio-ecological systems frameworks (e.g., Biermann et al., 2010; Polski & Ostrom, 1999) in the planning phase could help highlight potential sociopolitical barriers and assist with incorporating more comprehensive design solutions, and position BO to generate greater conservation benefit.

CHAPTER FOUR: CRITICAL PERSPECTIVES OF THE MAINSTREAM CONSERVATION PARADIGM – INDIVIDUAL NEOLIBERAL IDEOLOGY AND THE CROWDING OUT OF SOCIAL SCIENCE INSIGHTS

1. SYNOPSIS

This article critically examines the systemic root causes of the inadequate efforts of the conservation profession to incorporate the social sciences into its theories of change. I review existing criticisms that the mainstream conservation has adopted neoliberalism as its core ideology. When combined with individualism, this ideology serves as the basis for the paradigm and has deep influence over conservation science and practice and culminates in the over-reliance on theories of change focused on individual mental states and practices focused on altering the decisions of individuals using financial incentives. The result of individual, neoliberal influence is a bottom-up theory of change that lacks attention to social and systemic process which are necessary to transfer behavior change across individuals, higher social scales, geographic space, and time. Shifting to a new paradigm that identifies and addresses distributed action problems as social and systemic may be necessary to address critically urgent conservation issues.

2. INTRODUCTION

Meeting urgent and globally significant conservation goals is not possible without transformational change across many levels of society (Amel, Manning, Scott, & Koger, 2017; Steffen et al., 2015; Masson-Delmotte et al., 2018). Despite some success (Bolam et al. 2020; Hayward, 2011; Brooks, Wright & Sheil, 2009), conservation has failed to reverse widespread environmental decline (Diaz et al., 2019; Keppel et al., 2012; Diaz & Rosenburg, 2008). The magnitude and pervasiveness of global human impact is progressing (Theobald et al., 2020; Tierney et al., 2020; Diaz et al., 2019; Butchart et al., 2010). At the same time, an increasing willingness to confront failures and explore new alternatives in the profession may indicate that a paradigm shift is underway (Kuhn, 1970).

A prominent criticism is that conservation has failed to incorporate insights from the social sciences into policy and practice (Manfredo et al., 2019; Amel et al., 2017; DeFries & Nagendra, 2017; Redford, 2011; Schulz, 2011; Balmford & Cowling, 2006; Cowling, 2005). Despite this decades-old critique, major global research investments continue to prioritize understanding the consequences of environmental decline rather than strategies to produce systemic change (Overland & Sovacool, 2020), and many insights from the social sciences remain underutilized, misunderstood, and overlooked by practitioners across the field (Bennett & Roth, 2019; Bennett et al., 2016; Fox et al., 2006; Mascia, 2003). Related critiques include the lack of a proper process to diagnose problems (Knight et al., 2013, Game et al., 2014), poor collaboration between conservation organizations toward common goals (Freyfogle, 2006), inadequate empirical evaluation of outcomes (Baylis et al., 2016), and a "culture of success" that inhibits learning from failure (Catalano, 2017), leading conservationists to jump from one fad to another in search of solutions (Redford et al., 2013). The articles cited above express a growing recognition that conservation science and practice are not currently equipped to address urgent, large-scale conservation goals (Orbasli, 2017), and that the full integration of the social sciences is a key to restructuring to achieve greater success (Bennett et al., 2016).

3. METHODS

I start with identifying the core issues preventing incorporation of social science as integral to conservation and any transformational change effort. Potential barriers include: 1) the lack of common vocabulary and shared mental models between social and natural scientists, 2) academic reward systems that discourage interdisciplinary collaboration, 3) lack of funding for collaboration and applied social science in conservation, and 4) limited opportunity (Fox et al., 2006). Interrelated problems include: biocentrism in the profession (Redford, 2011; Freyfogle, 2006; Mascia, 2003), a lack of understanding of the complexities of behavior change (Schulz, 2011), naïve understandings of conservation by social scientists (Redford, 2011), the limited number of social scientists employed by conservation organizations (Mascia, 2003), and the lack of understanding of the distinct and varied contributions that many fields of social science can provide to conservationists (Bennett et al., 2016).

Articulation of how the current, mainstream conservation paradigm is structured could further illuminate root causes for the lack of integration of the social sciences and provide insight on how to improve conservation success (Knight et al., 2013). The goal of this article is to review and examine critical perspectives on mainstream conservation to conceptualize its basic structure. I then ask how this structure relates to the integration of the full range of insights that the social sciences can provide. I focus on how science and practice are applied to distributed action problems (DAPs), which have been particularly challenging for conservationists to address (Diaz et al., 2008). Distributed action problems occur when the patterned actions of many individuals, distributed widely across space and time, harm the common good and thriving ecological systems. The archetype DAP is non-point-source pollution in agricultural landscapes that create hypoxic zones in waterbodies (e.g., Gulf Hypoxia). Other examples of DAPs include other kinds of agricultural conservation on private lands (e.g., wildlife habitat), land-based climate change action, water conservation in urban areas, and household energy conservation. Distributed action problems commonly are misdiagnosed as behavioral problems that can be addressed through individualistic behavior-change methods and frameworks, when, in fact, they are multifaceted, multi-scaler, systemic issues often called “wicked conservation problems” that require systems change to solve (DeFries & Nagendra, 2017). To contextualize my research, I first provide a brief case study of how the State of Illinois addresses stream pollution that results in Gulf Hypoxia. Then, I review critical perspectives of the current mainstream conservation paradigm. In particular, I explore how its ideological basis shapes conservation science and practice and hinders a useful theory of change.

4. GULF HYPOXIA: A DISTRIBUTED ACTION PROBLEM

The Mississippi River Basin is the largest drainage in North America, covering 41% of the contiguous United States. In the 1970s, a “dead zone” devoid of marine life was discovered where the river enters the Gulf of Mexico (Figure 6). Gulf Hypoxia (GH) is a state of deprived oxygen that kills marine organisms and is caused by an over production of algae which are stimulated by excess nutrients from agricultural fertilizers, wastewater, nitrogen fixation by leguminous crops, and fossil fuel combustion (Bianchi, 2010; Rabalais et al., 2002). The size of the “dead zone” has been generally stable or increased over time since 1985, according to the Mississippi River Gulf of Mexico Watershed Nutrient Task Force (EPA, 2021; Figure 7). Hypoxic zones occur worldwide and are increasing in number, magnitude, and extent (Rabotyagov et al., 2014; Diaz et al., 2008).

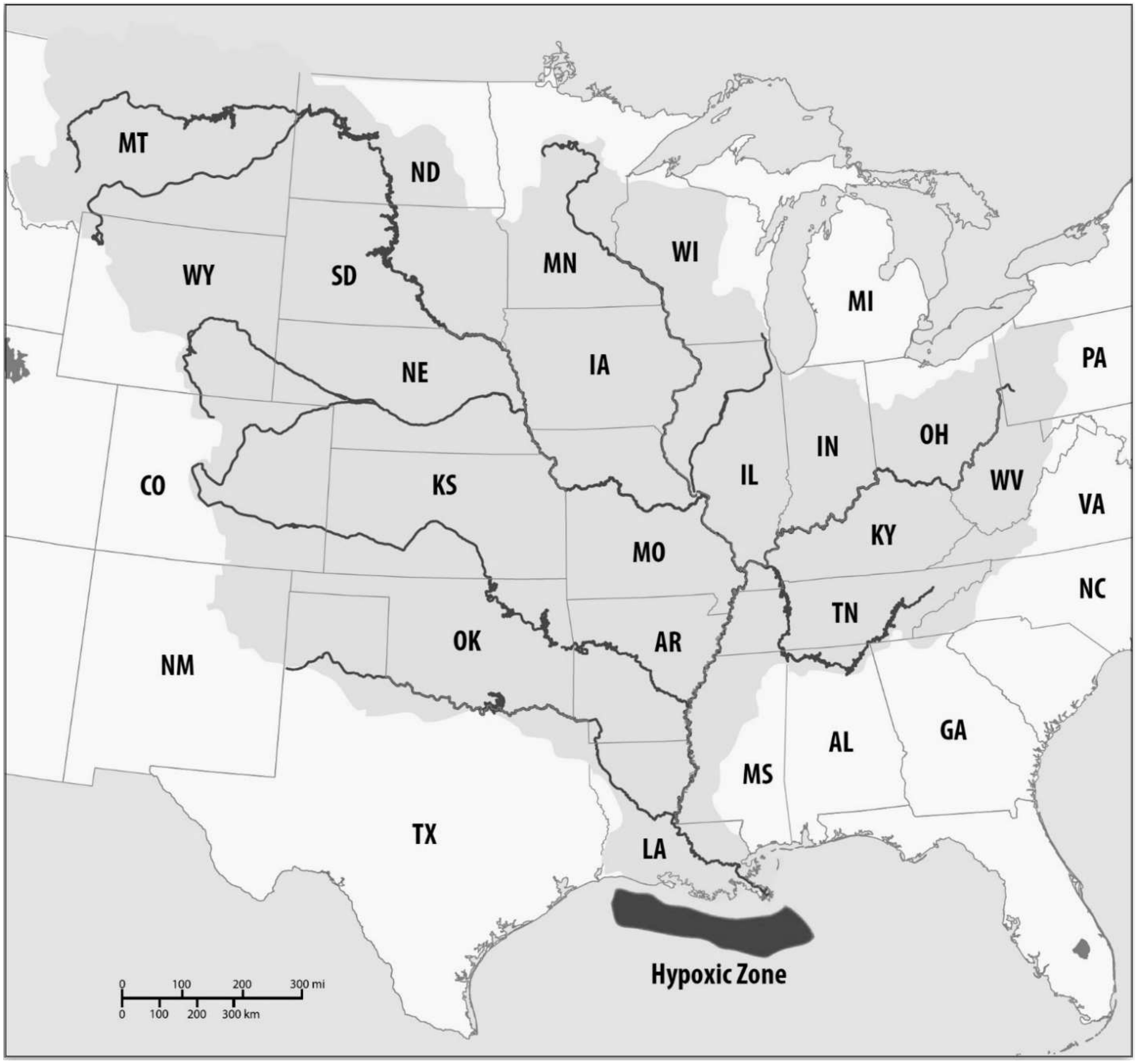

Figure 6. Map of the Mississippi River watershed showing the location of the "dead zone" caused by Gulf Hypoxia

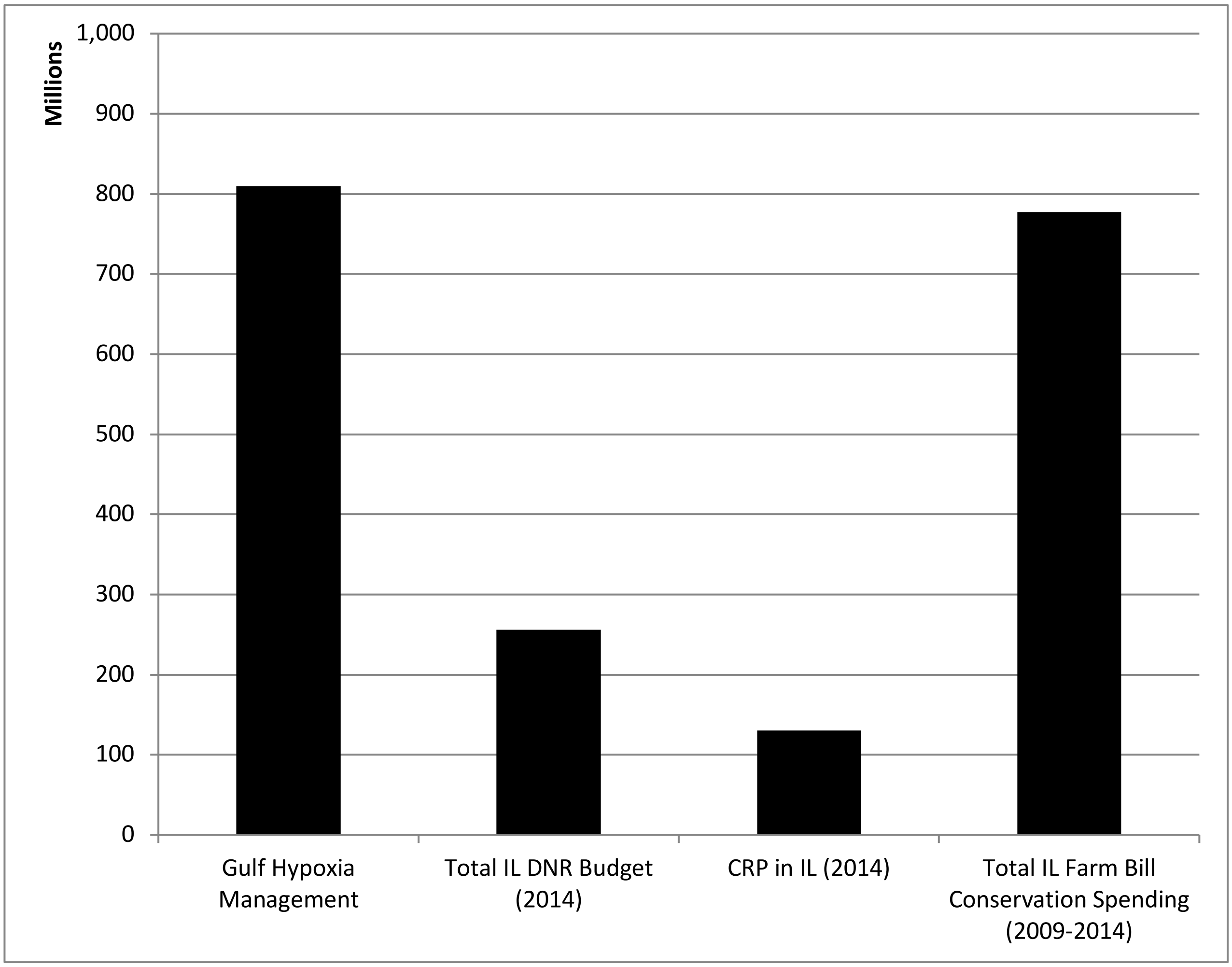

Figure 7. Comparison of Illinois conservation expenditures to projected annual cost of Gulf Hypoxia management. *Note: The projected annual cost of direct payments to reduce nitrogen and phosphorus stream loads by 45% in Illinois in 2014 exceeded the total annual Illinois Department of Natural Resources budget, annual Conservation Reserve Program expenditures in the state, and total Farm Bill conservation program spending over the previous 5 years.*

Increases in nitrogen load from the Mississippi River are the primary factor in the worsening of GH; most of this this nitrogen load (N; 74%) originates from agricultural non-point sources (i.e., farm fields) (Rabalais et al., 2002). Load increases result from three forms of human activity: 1) flood control and navigational channelization of the Mississippi, 2) loss of forests, wetlands, riparian vegetation, and the expansion of agricultural drainage, and 3) increases in nitrogen and phosphorus (P) inputs into the Mississippi River drainage primarily from agricultural fertilizer application (Rabalais et al., 2002).

Gulf Hypoxia reduction is a high conservation priority for federal and state natural resource agencies and cooperatives. In 1997, the U.S. Environmental Protection Agency (EPA) established the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, made up of 12 states, 5 federal agencies and tribes, to understand the causes and effects of GH and to coordinate activities to reduce the size, severity and duration of GH, and to ameliorate the effects of hypoxia (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2008). The task force developed an action plan in 2001 that coordinates the various state-based efforts and began implementing that plan in 2008 (https://www.epa.gov/ms-htf/hypoxia-task-force-nutrient-reduction-strategies). Its original goal was to reduce the size of the GH zone from the current five-year running average of 14,000 square kilometers to 5,000 square kilometers by 2015. To meet this goal, the plan recommended a 45% reduction in 1980 through 1996 stream loads of N and P. The primary activities under the plan are monitoring, decision-support tools, modeling, permitting and regulatory program support, outreach, education, partnerships, and financial and technical assistance (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2016). A reassessment of the plan in 2013 showed that both N and P inputs had increased since 2008, and the 45% reduction goal was not achieved (Mississippi River/Gulf of Mexico Watershed Nutrient

Task Force, 2013). In fact, the area of the Hypoxic zone has not significantly decreased since monitoring began in 1985 (Figure 7).

All state plans emphasize a voluntary, incentive-based approach focused on funding through USDA conservation programs to provide direct payments to landowners to implement nutrient-reduction practices (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2016). Illinois accounts for the largest percentage of N (17%) and P (13%) to stream loads in the Mississippi watershed (Willhite, 2014). In May 2014, Illinois conducted a science assessment (Assessment) to support its Nutrient Loss Reduction Strategy (David et al., 2014). Its purpose was to determine the effectiveness of management practices that could reduce nutrient input to surface waters and estimate the cost of direct payments to farmers for implementing the practices. The Assessment modeled various combinations of practices and other land use changes that needed to reduce N and P by 45%. Three scenario models achieved the goal and ranged in cost from $791 million to $810 million per year (David et al., 2014). For example, one of the scenarios would require:

- 10% of all farmers to reduce N applications to manufacturer recommended rates;
- Change timing of N applications on all farms;
- Installation of bioreactors on 15% of tile-drained fields;
- Eliminating all P applications on 12.5 million acres for six years;
- Reducing tillage on 1.8 million acres;
- Planting cover crops on 87% of all corn and soybean acres;
- Planting riparian buffers on all cropland currently without them; and
- Converting 2.5 million acres of cropland to perennial crop (hay, alfalfa, etc.).

This example illustrates the dramatic change necessary to meet nutrient-reduction goals. Some changes are impossible without altering the basic structure of agriculture in the state; for example, increases in hay supply would require more livestock than currently exist to fill demand (David et al., 2014).

This example also illustrates the inadequacies of addressing this problem primarily through financial incentives. Most obvious is the difficulty of overcoming two challenges. The first is to provide payments large enough to motivate farmers to switch to conservation practices. The other challenge is to allocate and sustain enough funding to enroll a sufficient number of landowners and keep them enrolled over time. Maintaining public support of the U.S. Farm Bill is critical to addressing this challenge. To put these costs in perspective, the estimated cost ($791 million to $810 million annually) of N and P reduction to 45% below current loads was six times larger than current annual Conservation Reserve Program expenditures in Illinois, and three times larger than the total annual budget of the Illinois Department of Natural Resources (Figure 8; USDA, 2014; State of Illinois, 2014). Given federal and state budget constraints and competing needs for conservation funding, it will be an enormous political and financial challenge to allocate and sustain the additional funds necessary for an approach based on financial incentives to create large-scale change.

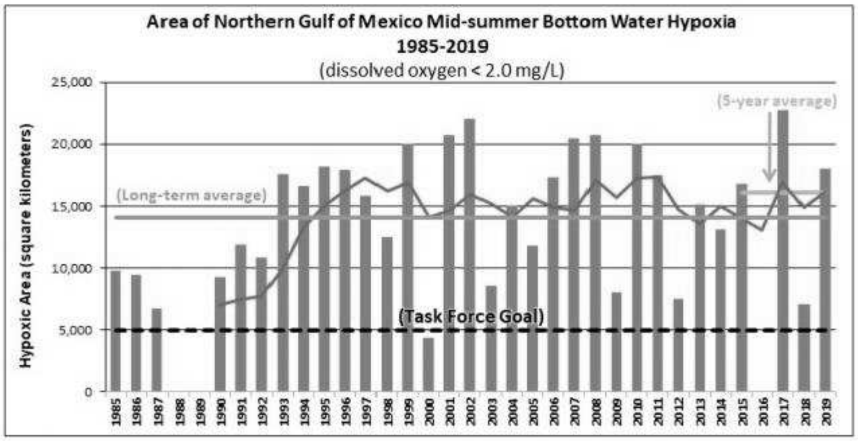

Figure 8. Area of Northern Gulf of Mexico mid-summer bottom water hypoxia 1985-2019
Note: The hypoxic area in the Gulf of Mexico has not significantly decreased since 1985 (EPA, 2021).

The magnitude of change necessary to address Gulf Hypoxia points to the need for more dramatic, fundamental change (Steffen et al., 2015). Insights from the social sciences could help address the need to increase the pace, scale, and durability of change, but the lack of incorporation of the social sciences limits the conservation profession's access to these insights (Bennett et al., 2016). In the next section, I review critical perspectives on the structure of the current conservation paradigm to determine how it encourages or inhibits the inclusion of insights that could address DAPs like Gulf Hypoxia.

5. STRUCTURE OF THE CURRENT CONSERVATION PARADIGM

Paradigms are structured by mental models, which are the most fundamental components of socio-ecological systems (Meadows, 2008). These underlying cultural ideologies, assumptions, and beliefs guide scientists' and practitioners' views of what is important, possible, or legitimate, and are manifested in the types and varieties of theoretical lenses utilized, and ultimately in the policies and approaches pursued, preferred, and practiced (Moon et al., 2018; Jones et al., 2011; Kuhn, 1970). Congruence between the major components of systems characterizes a paradigm (Meadows, 2008; Kuhn, 1970; Figure 8). Mutual reinforcement between the various elements of systems and rationalizations and framings that defend and justify the status quo help to maintain social systems (Jost et al., 2019; Stroh, 2015). This review focuses on criticisms of the mainstream conservation paradigm, which includes the ideologies, science, and practice of dominant government and non-governmental organizations.

6. IDEOLOGICAL BASIS

Ideology is cultural, and. like other cultural phenomenon, it has hidden, unconscious influence over behavior (Huaco, 1986). As such, all members of society swim in this cultural soup. Throughout history, conservation has aligned with dominant cultural ideology to guide its own paradigm (Orbasli, 2017). Individualism is a dominant ideology in the United States, the most individualistic society on earth (Hofstede, 2001). Americans are more likely to value freedom, independence and self-reliance, to believe they have control over their lives, and to attribute outcomes to their own individual traits than in collectivist societies (Fischer, 2008). Paradoxically, it is a myth that colonial America favored individualistic organization (Shain, 1994), and Americans today are more likely to favor the collective over the individual in many ways, i.e. belonging to churches and local organizations, or respecting authority (Fischer, 2008). This suggests that American values may better reflect voluntarism, a particular form of individualism (Fischer, 2008; Proctor, 1980). Voluntarism's core principles are reliance on voluntary action, which includes both individual and community, and free choice, but with commitment to those choices, e.g., it's ok to quit a job, but not to be subordinate (Fischer, 2008).

Neoliberalism as a dominant global ideology over the past 40 years has "pervasive effects on ways of thought to the point where it has become incorporated into the common-sense way that many of us interpret, live in, and understand the world" (Harvey, 2005). Neoliberalism is more than just a form of economic production or a political project of de-regulation; it is a widely influential cultural paradigm that touches on all aspects of life (Brown, 2018; Harvey, 2005; McCarthy & Prudham, 2004; Larner, 2003; Peck & Tickell, 2002; Bowles, 1998). Through adopting and prioritizing its core assumptions, systems structures, and framing, it has concretely shaped the science and practices of many fields, such as crime and incarceration, education, and health care (Schrecker, 2016; Koechlin, 2013; Dhont, 2012; Rowden, 2009; Fischer, 2008; LeBonte & Stucker, 2008; Herbert & Brown, 2006; Fitzsimmons, 2002; Coburn, 2000). Mainstream conservation has been similarly influenced (Fletcher, 2020; Adams, 2017; Orbasli, 2017; Buscher & Fletcher, 2015; Potter & Wolf, 2014; Arsel & Buscher, 2012; Buscher et al., 2012; Sullivan, 2012; Igoe et al., 2010; Burns et al., 2002; Ackerman & Gallagher, 2000; Fitchen, 1987).

Neoliberalism structures society by emphasizing individual autonomy, private rights, rationality, self-reliance and independence, freedom of choice, marketization, commodification, financialization, de-regulation, de-centralization, and competition (Bromley, 2019; Holmes & Cavanagh, 2016; Harvey, 2005; McCarthy & Prudham, 2004; McGregor, 2001). In doing so, neoliberal ideology mutually reinforces the individual side of voluntarism, rather than the collective. The net effect is a de-emphasis or crowding out of the social or relational aspects of human life, e.g., relationships, community orientation, collaboration, morality, and the collective good (Brown, 2018; McGregor, 2001). Next, I discuss how these ideologies manifest in mainstream conservation in the United States.

7. CRITICISMS

7.1. Criticisms of Conservation Science

Twelve meta-analyses represent the current state of knowledge of agricultural conservation practice adoption (Yoder et al., 2019; Prokopy et al., 2019; Ranjan et al., 2019; Lui et al., 2018; Carlisle et al., 2016; Wauters & Mathijs, 2014; Lesch & Wachenheim, 2014; Baumgart-Getz et. al., 2012; Prager & Posthumus, 2010; Ahnstrom et al., 2009; Knowler & Bradshaw, 2007). A key finding from these studies is that while some individual/farm scale factors that are positively associated with adoption, overall, these factors are not "powerful or consistent predictors" of practice adoption (Prokopy et al., 2019). Researchers report that "the literature is full of contradictions and paradoxes and generalizations are hard to make" (Ahnstrom et al., 2009), that "most factors have inconsistent and in fact mostly insignificant impact" (Wauters & Mathijs, 2014), and many factors have "small influence on adoption when examined individually" (Baumgart-Getz et al., 2012).

A second key finding is that across all studies "there is no evidence that economic factors are more predictive of adoption than non-economic ones" (Prager & Posthumus, 2010). Many studies show that the perceived high cost and actual cost of establishment and maintenance of non-productive lands is a barrier to adoption, and the availability of incentives, the potential for economic benefits, and the potential cost savings from removing non-productive lands from production are motivations for farmers to engage in conservation practices. However, across all studies these factors have small influence when analyzed individually (Baumgart-Getz et al., 2012). Instead, "the literature reveals that not all influencing factors have been financial and some factors considered financial do not affect decision-making as one would expect under a model of maximizing net present value." (Lesch & Wachenheim, 2014).

Finally, contextual variables (e.g., supply chains, social networks) are sometimes recognized as vitally important to understanding and changing behavior, but practice adoption research overlooks them and focuses primarily on individual or farm-level factors. The lack of consistent theoretical frameworks and the overuse of individual theoretical perspectives contribute to exclusion and limited focus on contextual factors (Prokopy et al., 2019). The three most common theoretical models used in practice adoption research are: 1) economic models, 2) innovation-diffusion models, and 3) perception-based models, e.g., the *Theory of Reasoned Action* by Prager and Posthumus (2010). Each of these perspectives omits or de-emphasizes social and ecological context as a key determinant of behavior. Only half of practice adoption studies on farmer conservation adoption consider context as a predictor of conservation behavior (Yoder et al., 2019). Several researchers have highlighted this problem as a critical shortfall of the research and recommend that practice adoption researchers "deploy more comprehensive theoretical lenses and examine both individual-level and structural factors" (Prokopy et al., 2019).

Overall, the practice adoption research reveals that behavioral insights were drawn primarily from individual, agent-based, theoretical perspectives. A similar analysis of 134 research articles on land use found that in 75% of the studies, the subject of change was the individual rather than social groups, e.g., communities, institutions (Groeneveld et al., 2017). My review of articles directed toward conservation professionals as guidance on theories applicable to solve various kinds of conservation problems shows a similar emphasis on individualistic perspectives (Masuda et al., 2021; Battista et al., 2018; World Bank Group 2015; Ardoin et al., 2013; Steg & Vlek, 2009; Heimlich & Ardoin, 2008; Monroe, 2003; Vining & Ebreo, 2002; Cook & Berrenberg, 1981).

Individualistic theoretical perspectives are derived primarily from microeconomics and psychology. The most prominent and frequently used in studies of land use is Expected Utility Theory (Homans, 1984), a form of rational choice theory wherein individuals make decisions that maximize utility or profit (Groeneveld et al., 2017). Rooted in neoclassical economics (Ritzer, 1996), Homans (1967) saw his rational choice theory as an applied form of behaviorism where behavior results from informed choices modified by a conditioned response to external punishments or rewards (Delprato & Midgley, 1992). Psychological perspectives include the Theory of Reasoned Action, (Fishbein & Azjen, 2010), Value Belief Norm Theory (Stern, 2000), Norm Activation Theory (Schwartz, 1992) and Diffusion of innovations (Rogers, 2003). In these theories, behavior results primarily from choices driven by mental states (e.g., personal norms, beliefs, attitudes, perceptions) and other personal characteristics (e.g., demographics).

Several assumptions are common to these microeconomic and psychological theories. The first is that the scale of focus is the individual, and individuals (or individual actors) are assumed to possess the capacity to make free choices, i.e., agency (Friedman & Hechter, 1988), but models that assume agency will have little explanatory power in cases where individuals do not possess it (Satz & Ferejohn, 1994). Second, the choices that individuals make and the behaviors they engage in as a result are assumed to be conscious and intentional (Burke et al., 2009; Sarver, 1983). Context may influence behavior unconsciously, for example through norms (Griskevicius et al., 2008) or habits (Wood & Neal, 2007), leading researchers and practitioners to underappreciate its power to influence on behavior (Johns, 2006). Third, these theories omit social and ecological context, fail to explain a mechanism whereby it influences behavior directly, or assume it only influences behavior indirectly through choice (Sarver, 1983). Thus, the key determinants of behavior are assumed to be internal cognitive states that influence choices (e.g., attitudes, beliefs, preferences, perceptions of risk, etc.), or the perceived consequences of those choices (e.g., costs and benefits). Perhaps the most problematic aspect of over-reliance on these theories is methodological individualism, the idea that the most fundamental element of social life is individual action (Elster, 1989), and simple aggregation of individual thoughts, choices, and actions can explain larger-scale social phenomena (Hechter, 1983; Heath, 2005). The result is a bottom-up theory of change described below. Shove (2010) critically labelled these perspectives the "paradigm of attitude-behavior-choice."

7.2. Criticisms of Conservation Practice

Consistent with individualistic theoretical perspectives, mainstream conservation approaches currently emphasize three general types of conservation interventions: 1) financial incentives, 2) influencing individual choice through information and awareness, and 3) regulations, or some combination thereof (Prager & Posthumus, 2010; Steg & Vlek, 2009; Echiaveria, 2005). Typical conservation activities that support these interventions include developing decision support tools (Schwartz et al., 2017), collecting, analyzing and disseminating information (Pullin & Knight, 2003), increasing awareness of environmental problems (Lemke et al., 2010), and widespread use of individually targeted incentives (Echiaverria, 2005). Regulations have a potential advantage of addressing Gulf Hypoxia more efficiently, effectively, and fairly (Echeverria). However, regulation is sometimes unpopular and difficult to enact, and its critics suggest that its costs (e.g., monitoring and enforcement) outweigh the benefits (Stern, 2006). Non-point, agricultural sources of nutrient pollution currently are unregulated by the federal Environmental Protection Agency; regulation is under state jurisdiction, and few states have adopted it (Kling, 2013). Thus, for DAPs there are two general approaches, both of which emphasize individual choice.

Consistent with their underlying theoretical basis, incentive-based interventions frame the problem as economic, despite research to the contrary (Prager & Posthumus, 2010). The goal is to seek to make conservation behaviors more financially rational so that individuals voluntarily choose to engage in them (Reganold et al., 2011). The financial cost of conservation is assumed to be the primary barrier to conservation; therefore, conservationists must make conservation pay by exploring how conservation practices save money, enhance financial productivity, or overcome costs (Fletcher 2020). According to this rationale, fundraising is the key to enrolling more landowners in programs, so proposing and defending public incentive policy becomes vital work (Barbier et al., 2018). It also is essential to monetize the costs and benefits of conservation action or inaction to inform decision-making or establish markets (Keeler et al., 2019; Costanza et al., 2014; Adams, 2014). This perspective has the effect of over-simplifying the conservation challenge. In extreme cases, this narrative is expressed simply as, "Cut farmers a check," (Leonard & Russel, 2019) or "You get what you pay for" (Ferraro & Kiss, 2002).

The goal of psychological, behavioral interventions similarly aligns with the theoretical determinants of choice, which are cognitive mental states that influence individual decisions. Thus, the conservation goal under behavioral approaches is similarly to understand and influence these mental states to make conservation choices more likely (e.g., in line with attitudes, more convenient, less risky).

The role of the state under neoliberalism is to support its core tenants of individual rights, freedoms, and free markets (Harvey, 2005). This is evident in that the structure of the two most prominent examples of the direct payment approach are the conservation programs administered by the U.S. Department of Agriculture (U. S. Department of Agriculture, 2021) and the emergence of markets for ecosystems services (Forest Trends, 2021; https://www.forest-trends.org/who-we-are/initiatives/who-we-areinitiativesecosystem-marketplace/). In each case, private landowners receive direct payments that compensate for the cost of adopting specific conservation practices on their land. Participation in government programs is voluntary, and technical and financial assistance is delivered individually, farm by farm, through private contracts administered through a vast, countrywide, county-based delivery system (U. S. Department of Agriculture, 2021). Payments for ecosystem services is similarly designed as a private, contractual, transactional approach to conservation where individual landowners voluntarily participate in markets to sell services derived from their land (Salzman et al., 2018).

The most important criticism of direct government payments and market-based instruments is the persistent lack of results despite the high cost (Fletcher, 2020). For example, from 1995-2015, $2.7 billion was spent on water quality in Iowa through USDA programs in part to support GH nutrient reduction strategy, yet loads increased, in part due to tile draining that reversed positive impacts of conservation practice adoption (Environmental Working Group, 2015; Jones et al., 2018). Other non-target conservation goals in this landscape also worsened, for example, monarch butterfly populations declined by more than 80% over the same time period (Thogmartin et al., 2017). In addition, current expenditures are small compared to projected costs of fully implementing state nutrient reduction plans through direct payments, e.g. Illinois $800 million (Davis et al., 2014). Despite a proliferation of programs globally, growth of market-based instruments has stagnated in many places and not lived up to their promise of producing significant benefits for biodiversity, ecotourism, climate mitigation and other issues up to their promise to ameliorate conservation problems (Fletcher, 2020; Sunderlin et al., 2015).

A criticism of direct payments as a policy approach is that they are highly economically inefficient compared to alternative approaches (Bromley & Hodge, 1990). For example, rather than paying farmers to transition from harmful farming practices to beneficial ones, the less costly, more efficient incentives-based approach would be for farmers to pay society when it is necessary for them to divert from collective goals, e.g., protection of highly erodible soils (Bromley & Hodge). Under this kind of a system, farmers would pay society for the right to plow highly erodible land, rather than being paid by society NOT to plow such land, as in the case of current U. S. Department of Agriculture Conservation Reserve Program. Thus, direct payments through this critical lens are viewed more as a politically expedient conservation approach (i.e., do not challenge farmers presumed right to plow) than an economically efficient one.

Incentives have potential perverse social effects that are largely ignored under direct payment approaches, such as the "crowding out" of intrinsic motivation as a result of financial "extrinsic" rewards and market discourse (Dayer et al., 2017; Neuteleers & Engelen, 2015; Rode et al., 2014; Muradian et al., 2013). If crowding effects are occurring, and financial incentives sources decline in the future, it could make the job of sustaining changes to private landowner behavior over the long term even more difficult. Theoretically, the overall cost of direct payments over time would be expected to increase as a result of payment norms, i.e., conservation practices should only be engaged in if costs are paid (Neuteleers & Engelen, 2015). For marginal costs to decrease over time, direct payment approaches would have to *crowd-in* or produce more behavior than their incremental cost over time, and there is no evidence of this effect or of a strategy for doing so. Direct payment approaches also have been criticized for ignoring non-financial barriers (Prokopy et al. 2019), lacking a strategy of engagement (Wright, 2008), and being unethical (Stuart & Gunderson, 2020; Luck et al., 2012). These issues remain mostly unaddressed in agricultural conservation policy.

Direct payments also may have the additional perverse effect of maintaining the current neoliberal systems. Incentives sometimes are considered examples of fundamental structural change (Roesch-McNally et al., 2018). However, through a systems lens, incentives are at the very best a low-level, incremental structural change, and at worst pushing the system in the wrong direction (Carlisle, 2016; Meadows, 2008). An alternative view is that conservation payments are public subsidies to the current system that enable it to sustain itself by reducing its negative impacts, legitimizing its ideology, and distracting attention away from changing the system in a more fundamental way (Buscher et al., 2012; Igoe et al., 2010; Vatn & Bromley, 1997). Environmental markets are similarly viewed as enabling continued capital accumulation (Buscher & Fletcher, 2015; Lohmann, 2012). In other words, neoliberal conservation approaches are not transformational change strategies because they attempt to make minor, incremental adjustments to the current systems' problematic outcomes as a means of sustaining it, rather than fundamentally and proactively reconfiguring it (Carlisle, 2016). Thus, neoliberal conservation may include some useful tools, but is not up to the task of meeting society's need for more fundamental, transformational change (Ackerman & Gallagher, 2000).

There are, of course, exceptions to mainstream practice. For at least 20 years, an emerging movement toward collaborative conservation has been underway (Lubell, 2002). Collaborative conservation includes the recognition that distributed action problems require collective, not just individual, action, and that supportive social environments must exist for collaboration to flourish (Prokopy, 2014). USDA is experimenting with collaborative approaches through its programs, e.g., RCPP, CARM; Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2016). These approaches represent variation in the system, pre-conditions and opportunities catalyze change, yet collaboration requires effort and entails costs (Lubell, 2002).

There are many kinds of collaboration (e.g., planning, policy, partnership, conflict resolution) that may not be oriented around action (Margerum, 2008). These collaborative efforts deserve further attention as bright spots, but they remain too fragmented and experimental to have had serious impact yet.

8. CONSERVATION'S DOMINANT THEORY OF CHANGE

While conservationists increasingly understand that conservation is not primarily about biology, the perception that conservation behavior stems primarily from individual choice is still prevalent (Bennett & Roth, 2019; Balmford & Cowling, 2006). This persistent individual choice perspective evident in the critical perspectives described here culminates in a dominating individualistic, bottom-up theory of change. Under this theory, the goal of the conservation is to influence individual mental states and financial conditions to alter decisions related to the adoption of specific actions, then aggregate or replicate those individual actions across space and time. The underlying assumption is that environmental and social benefits arise from the cumulative effects of individual actions (i.e., methodological individualism).

The term "scaling up" is an often-used metaphor for a theory of transformative change in conservation and is formally defined as the process of "expanding, adapting, and sustaining successful policies, programs or projects in different places and over time to reach a greater number of people" (Holcombe, 2012). Scaling up is understood by some as multi-dimensional process of change and adaptation (Nguyen et al., 2019; Wyborn & Bixler, 2013; Hartmann & Linn, 2008). Yet, many conservation projects lack understanding of the mechanisms that foster transformational change and lack attention to specific strategies for scaling up that are truly multi-dimensional (Battista et al., 2017; Harmann & Linn, 2008). Rather, they typically focus on understanding and applying tools to facilitate and enhance the spread of behaviors or innovations from one individual to another (Battista et al., 2017). The term "scaling up" falsely implies that approaches implemented effectively in one area can be replicated with the same effectiveness in another, which denies variations in communities and geographic regions that would likely result in corresponding variation in conservation solutions across social and geographic scale (Prokopy & Genskow 2015).

From a practical perspective, individualistic conservation approaches are difficult to implement because multiple or contradictory factors in different individuals may motivate the same conservation action (Knowler & Bradshaw, 2007). This leaves conservationists with two impractical options: 1) develop a separate behavioral theory of change for each individual, or 2) ignore individual perceptions and use incentives to override them. The high degree of variation between individual personal factors related to the same behavior could partially explain why individualistic theoretical perspectives have failed to find consistent trends in factors the drive adoption in agriculture (Yoder et al., 2019; Knowler & Bradshaw, 2007).

By prioritizing individual mental states, personal factors, and financial choices, this theory of change excludes the very mechanisms that could provide scaling effects – systemic and social processes. As a result, mainstream conservation lacks a coherent strategy for: 1) how one change adopter will maintain a new behavior over time without ongoing external inputs; 2) how adoption will spread from one person to others and to larger social scales (i.e., communities, institutions, etc.); 3) how adoption of one practice will spread to other kinds of practices; or 4) how a new practice will become widespread and normative across space and time.

9. CONCLUSIONS

This analysis describes conservation as a system of science and practice and highlights its embeddedness within dominant cultural ideologies. The critical perspectives reviewed here suggest that individual, neoliberal ideology forms the basis of the mainstream conservation paradigm, and this cultural bias strongly influences and structures science and practice by legitimizing economic problematizing and framing and normalizing individualistic delivery and implementation approaches (Figure 9). In prioritizing the individual, mainstream conservation undermines social, moral, and systemic understandings, discourse, theory, and approaches (Brown, 2019; Wood, 2018; Sandel, 2010; Lockie, 2009; Higgins & Lockie, 2002; Stenson & Watt, 1999). That a similar effect is occurring in many fields across the United States, is additional evidence that its roots are cultural (Davis et al., 2015).

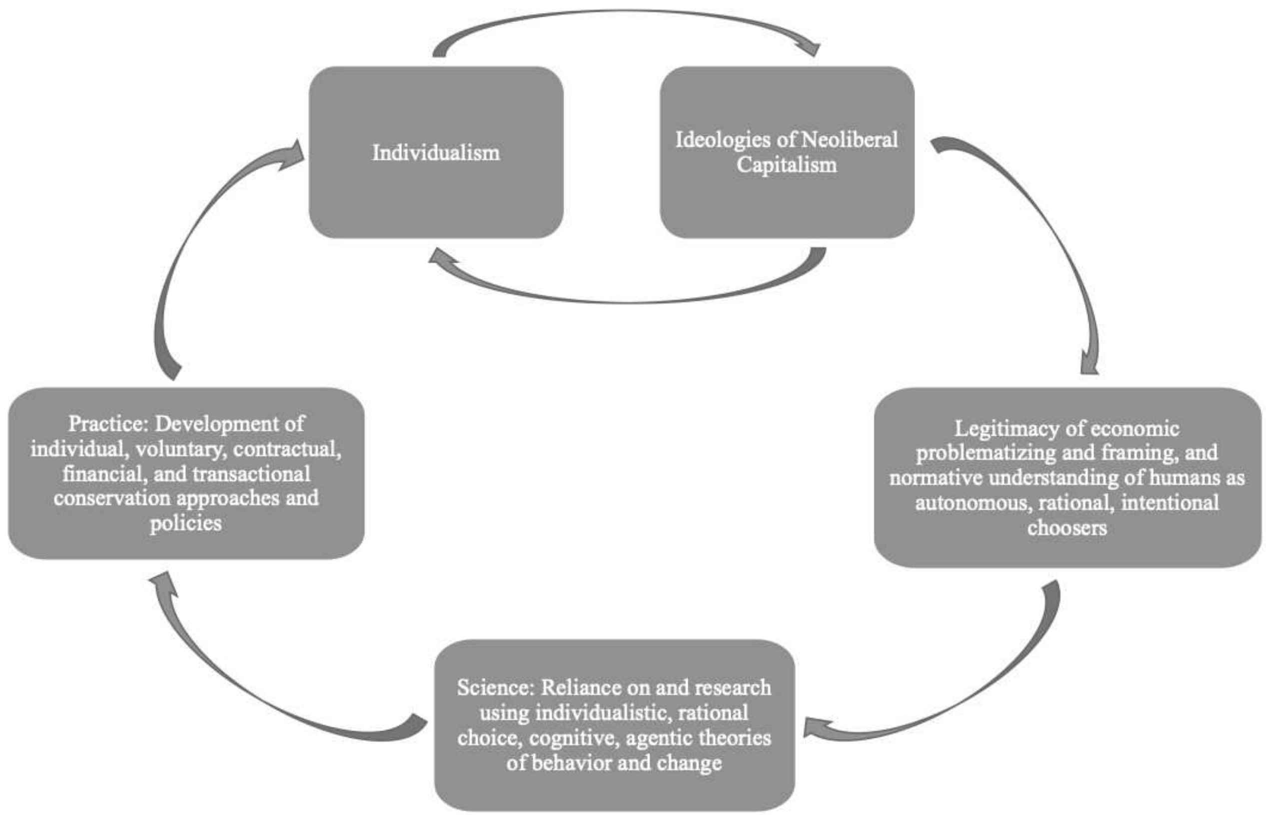

Figure 9. Causal loop summary of the current mainstream conservation paradigm

Many scientists have previously highlighted the individual bias present in conservation science and practices and encouraged research and application to take a broader theoretical perspective (Prokopy et al., 2019; Yoder et al., 2019; Shove, 2010). The individual, neoliberal bias described here creates a gap between existing social and systemic science insights and those that are applied to understand and address conservation issues. While many theoretical perspectives in the social sciences are relevant to conservation problems (Bennett et al., 2016), this bias narrows the number of existing perspectives that are drawn from and applied to conservation science and intervention. Contrary to other researchers (e.g., Bennett et al., 2016), I find that social science perspectives that emphasize individual decision-making are well integrated (i.e., economics and, to a lesser extent, psychology). My analysis suggests that the problem is not the lack of applicable theories or integration of social sciences per se, but the crowding effect of the individual, neoliberal bias on the inclusion of a broader range of theoretical perspectives.

The result of this bias is that the profession's dominant approaches stem from an overarching theory of change designed for temporary, small-scale, behavior change, when what is often needed is multi-scale systems change (Amel et al., 2017; Steffen et al., 2015). The current theory of change lacks attention to the social and ecological contextual factors that can generate transformational change (Shove, 2010). It also may exacerbate conservation problems by overlooking potential perverse social effects (e.g., motivational crowding). The problem is not with applicability or relevance the validity of micro-scale individualistic theories, but the misapplication of them to complex, multi-scaler social problems like DAPs. The result is a profession guided by limited theoretical perspectives and most importantly lacking a coherent theory of change capable of producing the necessary widespread, transformative change. Gulf

Hypoxia serves as an example of how this theory of change informed by limited perspectives is failing to generate the degree of change necessary to address distributed action problems in the United States.

CHAPTER FIVE: AN INTEGRATIVE THEORY OF TRANSFORMATIONAL CHANGE FOR CONSERVATION

1. SYNOPSIS

Solving large-scale, complex conservation problems requires transformational change at multiple levels of society. Distributed action problems (DAPs) have been particularly challenging because they result from the cumulative negative consequences of the patterned actions of hundreds, thousands, or millions of individual and group actors distributed across large spatial and temporal scales. Unfortunately, DAPs often are misdiagnosed and addressed by scientists and practitioners as individual, behavioral problems, when they are systemic problems. I reviewed the literature on collaborative, place-based social change and combined it with the literature on socio-ecological systems and socio-technical transitions to outline a new, integrative theory of transformational change (TTC) for addressing DAPs. The TTC is composed of four interdependent sets of mechanisms that can be enacted through strategic, conservation action in collaborative, place-based settings: (a) building communities of practice; (b) empowering individual catalysts; (c) reconfiguring the system; and (d) connecting across dimensions. I propose a set of testable propositions related to each of these components. The aim of the TTC is to integrate existing social and systems science insights into conservation science and practice, expand the set of potential interventions available, and improve the profession's ability to create the change necessary to address the world's most pressing conservation issues. The theory also provides a vision for a new conservation paradigm by re-orienting conservation away from the current individualistic, neoliberal, behavioral perspective toward a more inclusive, comprehensive, socio-systemic perspective.

2. INTRODUCTION

Recent studies show that environmental impacts are worsening at a rapid pace on a global scale, despite substantial conservation efforts (Butchart et al., 2010; Diaz et al., 2019; Jones et al., 2018; Keppel et al., 2012; Theobald et al., 2020; Tierney et al., 2020). Some researchers recognize that solving large-scale, complex conservation problems will require transformational change at multiple levels of society (Amel et al., 2017; DeFries & Nagendra, 2017; Steffens et al., 2017). Many disciplines and theoretical perspectives can inform transformational change (Bennett et al., 2017; Mascia et al., 2003), but so far, the conservation profession has not combined these perspectives into a coherent theory capable of addressing global challenges (Guerrero et al., 2018; Mahajan et al., 2021; Muhar et al., 2018). Existing insights from social and systems science are essential but remain underutilized, misunderstood, and overlooked (Bennett & Roth, 2019; Bennett et al., 2017; Manfredo et al., 2019; Shove, 2010; Virapongse et al., 2016). Research investments in conservation continue to prioritize understanding the ecological causes and consequences of environmental degradation over investments in how to generate transformational change (Overland & Sovacool, 2020).

Many conservation problems are difficult to address because they are distributed action problems (DAPs); they result from the cumulative consequences of the patterned actions of hundreds, thousands, or millions of individual and group actors that occur at multiple levels of society and across large spatial and temporal scales (Barry & Bateman, 1996; Biggs et al., 2010). Patterns in the actions of multiple actors across many dimensions suggest that they result from distributed, networked systems (Biggs et al., 2010). DAPs are different from collective action problems primarily because they are theorized to result from different root causes and therefore rely on different theories of change. In collective action, public goods are underprovided, and common-pool resources are degraded because there is a disincentive for self-interested individuals to act, even though it would be in the group's best interest to do so. The solution is to limit free-riding, allocate exclusive-use rights, impose regulation, or foster collaboration (Hussain, 2018; Olsen, 1965; Ostrom, 2000). Collective action problems often are conceived as decision problems resulting from conflicts between individual and collective interests (Barry & Bateman; 1996; Hussein, 2018).

While collective action is an essential part of addressing DAPs, the root cause of DAP's is not only the need for coordinated action, but also the more fundamental need to reconfigure the socio-ecological systems that are shaping harmful, patterned actions (i.e., DAPs). These systems may be intentionally configured to benefit the self-interests of a limited set of actors (e.g., corporate supply chains), designed to foster purposes that are incompatible with conservation (e.g., profit, efficiency), or may simply be disorganized or inefficient for supporting conservation goals. For example, agricultural production systems in the United States are configured to benefit corporate actors such that farmers become "locked-in" to actions that are harmful for the environment (Hendrickson et al., 2017; Roesch-McNally et al., 2018). For cities working towards aggressive greenhouse gas reduction, a common good but not a common pool resource, the challenge is a disorganized system that does not support individual actions across the city that collectively would reduce greenhouse gas production. Table 8 describes one such case, where the City of Fort Collins reconfigured the system in several substantial ways to increase the number of homeowners participating a home energy retrofit program as well as the comprehensiveness of the changes they adopted (Tools of Change, 2021). This case illustrates how systems reconfiguration were substantially more impactful than the previous behavioral interventions.

Other examples of DAPs include land-based, climate-change mitigation (Rose et al., 2012), non-point-source pollution from farm fertilizer that causes hypoxic zones in waterbodies (Diaz & Rosenburg, 2008), private lands biodiversity conservation (Burger et al., 2019), marine plastics pollution (Casoli & Ramkumar, 2020), and water and home energy use (Biggs et al., 2009; Guerrero et al., 2020). The solution to DAPs is to reconfigure systems to encourage individual and collective conservation action across social, spatial, and temporal dimensions.

Unfortunately, conservation scientists and practitioners misdiagnose and treat DAPs as individual behavioral problems that can be solved by altering individual choices and actions, an approach that has been critically labeled the “paradigm of attitude-behavior-choice” (Shove, 2010). For example, in 75% of land use studies, the subject of change is the individual rather than social groups (e.g., organizations, communities, institutions; Groeneveld et al., 2017), and only half of studies on farmer conservation adoption consider context as a predictor of conservation behavior (Yoder et al., 2019). Applied scientific guidance to practitioners similarly emphasizes individualistic theoretical perspectives and interventions over collective or systemic perspectives (Ardoin et al., 2013; Cook & Berrenberg, 1981; Mahajan et al., 2020; Monroe, 2003; Rare, 2019; Steg & Vlek, 2009; Vining & Ebreo 2002). The assumption that DAPs are behavioral problems results in over-reliance on individualistic, agentic, theoretical perspectives as the basis for strategic theories of change (e.g., Theory of Reasoned Action; Fishbein & Azjen 2010; Expected Utility Theory; Homans, 1967; see also Groenveld et al., 2017; Prager & Posthumus, 2010; Yoder et al., 2019). These perspectives, primarily from microeconomics and psychology, assume that the primary determinants of behavior are internal mental states (e.g., attitudes, beliefs, preferences, perceptions etc.) that influence choices, or the perceived consequences of choices (e.g., costs and benefits; Barnes & Sheppard, 1992; Burke et al. 2009;

Friedman & Hechter, 1988; Sarver 1983). As a result, the most common conservation interventions applied to DAPs – incentives, regulations, providing information, and increasing awareness – focus on influencing individual choice (Echiaveria, 2005; Prager & Posthumus, 2010; Steg & Vlek 2009). Thus, the dominant theory of change is an individualistic one in which the primary objective is to influence specific behaviors by altering the mental states and choices of individuals, then to aggregate or replicate those individual actions or programs across space and time (e.g., U.S. Farm Bill Natural Resources Conservation Service approach to addressing Gulf Hypoxia). This approach relies on the false assumptions of methodological individualism, that the most fundamental element of social life is individual action (Elster, 1989), and aggregation of individual thoughts, choices, and actions can explain larger-scale social phenomena (Heath, 2005; Hodgson, 2000). Simply put, the conservation profession is misdiagnosing complex, multi-scalar, socio-systemic issues as behavioral issues and addressing them with primarily individualistic theories and strategies. The problem is not the availability of social and systemic theories of change. It is the misapplication of individualistic theories to systemic problems.

The lack of breadth in the selection of social science in conservation science and practice is a direct cause of this limited theory of change, but its deeper origins are cultural. People in individualistic cultures, such as the United States, tend attribute behavior to dispositional (e.g., intentions, personal characteristics, etc.) rather than contextual or situational factors (Miller, 1990). Laypersons and scientists alike underestimate the power of situational context and overestimate the role of disposition in influencing behavior (Johns, 2006; Sabini et al., 2001). In addition, over the past 40 years, neoliberal ideology has "become incorporated into the common-sense way that many of us interpret, live in, and understand the world" (Harvey, 2005). The combined effect of individual neoliberalism is to crowd out the systemic and social aspects of human life (e.g., holistic, systemic thinking, collaboration, interdependence, collective action, solidarity, moral, affective, and ethical framings, etc.) in favor of individual rights, freedom of choice, rationality, competition, and free-market economic thinking and framing (Barker & Carman, 2000; Brown, 2018; Giddens, 1984; Higgins & Locke, 2002; Lockie, 2009; Lynch & Kalaitzake, 2020).

Individual neoliberalism has concretely shaped conservation science and practice (Adams, 2017; Bromley, 2019; Burns et al., 2002; Buscher & Fletcher, 2015; Buscher et al., 2012; Fitchen, 1987; Fletcher, 2012; Fletcher, 2020; Igoe et al., 2010; McCarthy & Prudham, 2004; McGregor, 2001; Sullivan 2012) and many other fields (Coburn, 2000; Dhont, 2012; Fischer, 2008; Fitzsimmons, 2002; Herbert & Brown 2006; Koechlin, 2013; Kopnina, 2015; LeBonte & Stucker, 2008; Rowden, 2009; Schrecker, 2016) around these principles. This evidence suggests that incorporating systems and social sciences into conservation practice is impeded not only by ideological differences between ecologists and social scientists within the profession (Bennett et al., 2017), but also by the broader societal influence of individual, neoliberal ideology on the profession (Fletcher, 2020; Orbasli, 2017).

Many scientists have previously highlighted the individual bias present in conservation science and practices and encouraged research and application to take a broader theoretical perspective (Prokopy et al., 2019; Yoder et al., 2019; Shove, 2010). By decontextualizing individual action from its social and systemic influences, these cultural ideologies have deprived the profession of many existing theoretical perspectives which contain the mechanisms of change it must harness to solve DAPs. The current dominant theory, drawn from a narrow range of perspectives, fails to explain how individual behavior change spreads across social, spatial, and temporal dimensions, and therefore how it becomes normative and transformational. The profession does not need new theories, per se (Sovacool & Hess, 2017). Rather, it needs to coalesce existing theories that encompass multiple sources of influence and dimensions of change into a comprehensive, multi-scalar, integrated theory that can be applied to a DAPs in a variety of settings (Yoder et al., 2019).

To address this need, I reviewed and synthesized literature from socio-ecological systems, socio-technical and sustainability transitions, collective action, social practices theory, and other social and behavioral theories into an integrative theory that explains how social, agentic, and structural influences combine to produce change that spreads across social, spatial, and temporal dimensions. I focus on theories applicable to DAPs because they underly the world's most pressing conservation issues and because of their challenging nature. Our practical aim is to provide a theoretical foundation to aid practitioners in enacting strategic change, to expand the range of potential interventions deployed to address DAPs, and to provide a vision for a new conservation paradigm.

3. DEFINING TRANSFORMATIONAL CHANGE

I define transformational change in conservation as the act of and result of strategically reconfiguring or designing new socio-ecological systems to serve the social and ecological common good. Transformational change is "dramatic" or "radical" change that transforms a socio-ecological system from one relatively stable system state to another through the punctuated disruption and subsequent re-ordering of a social system's "deep structure" (Gersick, 1991; Kuhn, 1970; Scrase et al., 2009; Wollin, 1999). Change is transformational only if the systems' core structural elements and interactions are altered by disruption or significantly reconfigured after disruption (Gersick, 1991; Moore et al., 2014) such that it produces dramatically different, more beneficial social, and ecological outcomes than the status quo. In contrast, incremental change is the optimization of the existing system through change to less fundamental aspects. (Moore et al., 2014; Scrase et al., 2009). Incremental change is incapable of addressing distributed action problems because it does not result in change to the deeper-level components (e.g., core worldviews and socio-cultural and situational factors) around which the system is configured and that ultimately influence human action (Muhar et al., 2017). Incremental adjustments do not have the power to overcome the more fundamental aspects of the system that remain intact and encourage the opposite kinds of behavior (e.g., maximizing yield and reducing ecological diversity). Incremental change strategies also can easily backfire by inadvertently by reinforcing deeper elements of the current system (i.e., change strategies reinforce harmful cultural ideologies), which results in "pushing the system in the wrong direction" (Meadows & Wright, 2008). Change cannot be transformational if dominant, core structural elements and interactions remain after disruption (Moore et al., 2014).

4. HOW SYSTEMS SHAPE HUMAN ACTION

The Theory of Transformational Change (TTC) depends on a conceptualization of how systems shape human action (Figure 10) that contrasts with the attitude-behavior-choice paradigm. Building upon Meadow's (2008) iceberg metaphor that suggests systems contain four basic components (mental models, system structures, behavioral patterns, and outcomes), the TTC adds two components (action situations and actors). Mental models, or cultural-level (not individual level) values, worldviews, ideologies, and assumptions are the most fundamental components of systems that unconsciously and consciously influence actors' views of what is important, possible, or legitimate and affect the types and varieties of human system structures pursued and preferred (Jones et al., 2011; Kuhn, 1970; Moon et al., 2018). Human system structures (e.g., rules, norms, policies, procedures, operations, resources, etc.) are designed to align with cultural mental models (Foster-Fisherman, 2007). Systems structures also include the biophysical conditions and the built environment (e.g., roads, buildings, farms, fields, etc.; Virapongse et al., 2016). Together, mental models and system structures form the socio-ecological context in which human action is embedded.

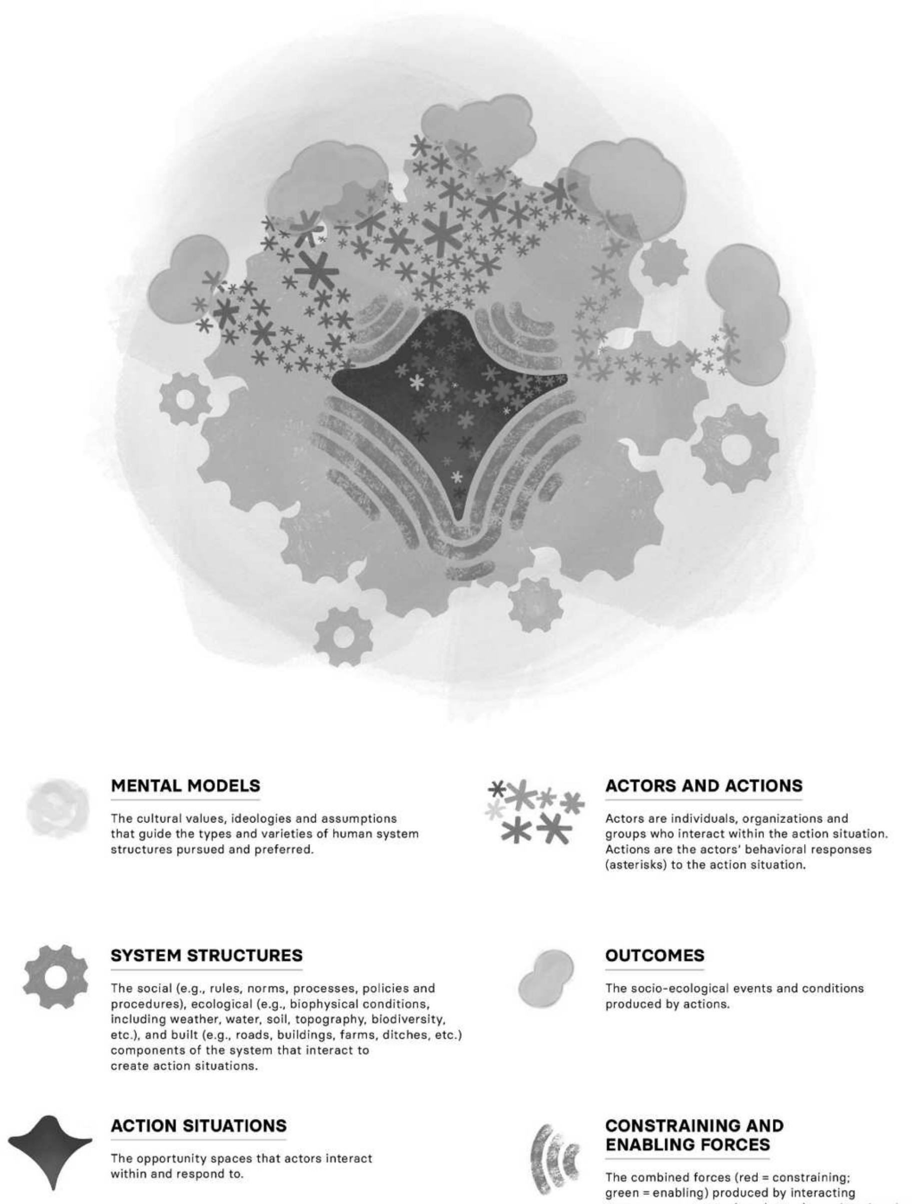

Figure 10. A model of how systems shape action

The interactive effects of mental models and systems structures produce action situations, which are multi-dimensional “opportunity spaces” that groups and individuals respond to with action (Fligstein & McAdam, 2012; McGinnis, 2011). Action situations are the proximate manifestations of conditions produced by systems at given points in time or space. Action situations are dynamic, and they constrain and/or enable behavior in patterned ways over time (McGinnis, 2011). The patterned action situations that are created by systems produce patterned behavior. Thus, action situations are the proximate determinants of action, and they mediate the relationship between systems and action.

Action situations have characteristics of strength, direction, and time that interact to make certain behaviors more or less likely to emerge from them. Strength is the net cumulative force of multiple interacting and counteracting forces and conditions produced by the system toward a type of behavior (as in Lewin’s Force Field Theory; Burnes & Cooke, 2012). Direction is the general action pathway that, over time, multiple action situations push actors toward. The direction of an action situation defines the opportunity to act in particular ways for specific actors at specific times and for multiple actors over time. Action situations define the boundaries of choice (i.e., the potential decision space); however, in contrast with Ostrom (2011), our view is that choice is only one of many factors influencing action within situations. In other words, the action situation always is what actors are responding to, but it may not be a conscious, intentional response to known conditions. In many cases, actors may be unaware of the conditions the action situation presents to them, and they may respond unconsciously to those conditions, as in habits, (Wood, 2018), or norms (Farrow et al., 2017).

Actors are individual persons or groups (i.e., organizations, governing bodies, coalitions of multiple actors, etc.) that engage in actions within action situations. Actors respond to action situations, but they also can configure systems and thereby shape action situations depending on their individual characteristics, power, position, and agency. Social processes are centripetal forces that tend to push multiple actors to converge toward common actions. Agency produces divergent responses from individuals, but usually within action situation boundaries. The agency of actors always is constrained to some degree by the situation. Actors, embedded within systems and subjected to social process, generally are assumed to be subordinate to these forces, but this is not structural determinism. Agents can modify the effect of systems on behavior by altering the action situation or by changing the system itself (e.g., conservationists, as agents, can strategically reconfigure the system).

Actions are the responses of actors to action situations and are the primary mediator of socio-ecological outcomes. Actions are specific behaviors at points in time, sets of linked of behaviors (i.e., a sequence of related behaviors), and the patterned, complex responses (e.g., multiple behavior sets) of multiple actors to dynamic action situations over time. This implies that patterned actions across multiple actors are more important focal points for intervention in DAPs than specific actions at specific points in time. Accordingly, the TTC focuses on action patterns as the primary mediators of socio-ecological outcomes, not individual behavior, or cognitive processes. Cognitive processes often are assumed to be rationalizations of actions already taken (i.e., self-perception theory), whereas other theories emphasize the opposite causal pathway (i.e., Theory of Reasoned Action). Thus, behavior is an actor's response to the system, and attitude is a cognitive response to action, rather than a cause of behavior. When attitude aligns with action, it reinforces and encourages repeated action in a mutually reinforcing relationship. This two-way interactive view of the attitude-behavior relationship shifts the applied emphasis of change toward proactively reconfiguring systemic conditions to favor action (focus on systems), rather than retroactively changing individual attitudes to influence decisions (focus on cognition).

The basic components of systems described above interact through systemic processes to shape human action (Hallett & Hobbs, 2020; Senge, 1990; Table 5). First, interacting elements and systems processes in socio-ecological systems create action situations. Individuals and groups respond to action situations with actions. Behavioral patterns resulting from patterned action situations over time generate environmental outcomes, which in turn influence the structure of the system in a circular, recursive fashion. Change of all kinds (e.g., behavior change, attitude change, policy change) flows through the structures and actors in systems which are networked. This conceptual model of how systems shape human action provides the framework for integrating a variety of theoretical perspectives into TTC.

Table 5. Key properties of complex socio-ecological systems and implications for conservation intervention

Property	How it works	Implication	References
Mutualism	Social and ecological aspects of systems are interdependent.	Common systemic root causes of undesirable social and environmental outcomes are key opportunities for motivating collective action	Jellinek, et al. 2018; Ban, et al., 2013; Foster-Fishman, et al., 2007
Embeddedness	Systems are multi-scalar; smaller-scale systems are influenced by larger, more dominant systems in which they are situated.	Connect place-based reconfigurations with larger scale societal transitions to create change opportunities.	Meadows & Wright, 2008; Foster-Fisherman, et al., 2007; Kim, 1999
Mutual constitution	Systems and sub-systems co-produce each other (e.g., culture produces the self and vice versa) and are self-organizing	While individual behavior is shaped by systems, individual actors also have power to change systems	Kirkwood et al. 2013; Markus & Kitayama, 2010; Burns et al., 2002; Giddens, 1984
Reciprocal determinism	System factors interrelate in non-linear relationships through feedbacks (e.g., behavior determines attitude and vice versa)	Building feedbacks between components is key to understanding, strengthening, and sustaining change over time	Meadows & Wright, 2008; Gielen & Green, 2015; Bandura, 1978
Joint influence, interacting effects, and pivotal influence	Multiple interacting factors have mediating and moderating effects on action; some elements have more influence than others	Apply interventions to multiple leverage points simultaneously, target structures and actors with the most leverage or power; create beneficial feedbacks (e.g., between policy and norms)	Fischer & Reichers, 2018; Abson et al., 2017; Hobbs et al., 2011; Foster-Fishman, et al., 2007
Constraining and Enabling conditions	Socio-ecological factors combine to form constraining or enabling conditions. These opposing forces and feedbacks shape action situations by defining opportunity and decision space.	Reduce systemic constraints and increase enabling conditions	Hendrickson et al., 2017; Burnes & Cooke, 2012; Lewin, 1951
Networked systems; ripple and cascading effects, emergence	System content flows through networked relationships between and among actors and structures. Change ripples and cascades through network linkages.	Systems may react to conservation interventions in unpredictable ways and new emergent states are opportunities for change	Gray et al., 2019; Lichtenstein & Plowman, 2009; Wheatley & Frieze, 2006; Hodgson, 2000

5. AN INTEGRATIVE THEORY OF TRANSFORMATIONAL CHANGE

Integrative theories are theories that combine the concepts and central propositions from two or more prior existing theories into a new single set of integrated concepts and propositions. In this section, I outline an integrative theory of transformational change for addressing DAPs. This theory is derived from the review and synthesis of three sets of existing theoretical perspectives from multiple literature sets representing the basic influences over change in socio-ecological systems: agentic, social, and structural. Agentic theories focus on how individual actors can take intentional, strategic action to influence themselves, others, and their surroundings, to improve their circumstances (Bandura, 2007). Social influence theories address the processes and relations between individuals and between groups and individuals that effect individual and collective behavior (Ritzer, 1996). Structural influences include the human factors such as technology, rules, regulations, and procedures (e.g., political, and legal procedures, supply chains, decision-making, etc.), as well as the built and natural environment that formulate context of behavior and action (Virapongse et al., 2016). Systemic processes integrate these influences by determining how they interact (Table 5). The defining feature of transformational change is that it must be transmitted across three dimensions: social, spatial, and temporal (Figure 11). In the social dimension, change is transmitted from individual to individual and up and down social levels (e.g., from society to communities to individuals, or the reverse). In the spatial dimension, change moves across ecosystems (e.g., species to communities), or physical space (e.g., parcel, farm, landscape). Change also progresses over time as actions become durable or fleeting.

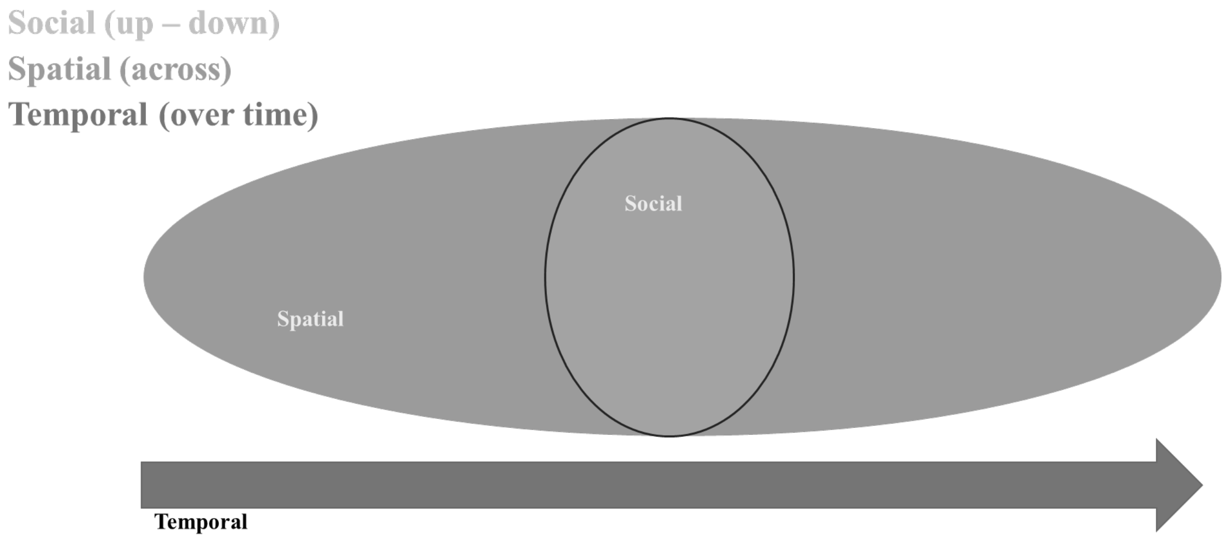

Figure 11. Three dimensions of transformational change

The TTC suggests that enduring and transformational change results from strategically activating the sources of influence across multiple dimensions. Three interrelated strategic approaches activate the social, agentic, and structural influences within reconfigurations: (a) building communities of practice (social process); (b) empowering individual catalysts (agentic process); and (c) reconfiguring the system (structure and systemic process, Table 6). The final strategic approach, connecting across dimensions, integrates the changes made in reconfiguration across social, spatial, and temporal scales (structure and systemic process). Below, I present twelve testable propositions for the TTC and describe how each component contributes to change.

Table 6. Strategic actions for reconfiguration

Change Component	Goal	Theories and Concepts	How It Works	Ideal Outcomes	Intervention Guidelines
Build Communities of Practice	Prioritize the common good	collaborative visioning, planning, and goal setting	Actors integrate social and ecological goals, and prioritize common goals over individual self-interest	Actors share common purpose, responsibility, risk, uncertainty, costs, and benefits	Provide human or financial capital to build collaborative capacity
	Foster social cohesion	social capital; social norms; group and place identity	Actors build trust which establishes group identity and norms, increases commitment and reciprocity	Increased capacity to collectively act in complex ways	Design and facilitate collaborative processes; celebrate group success together
	Collectively Learn and Act	Social practices; social learning; modelling; behavioral norms; self-perception; group and self-efficacy	Actors engage regularly in common endeavors and new actions are learned, socialized, and legitimized; actors gain experience and confidence with new actions	New actions normalize, become routine, and are internalized by individuals; cognitive states re-align with new behavioral norms	Participate in communities of practice as a co-equal learner
	Practice Inclusive decision-making making	Environmental justice theory; inclusion; procedural equity	Influence over outcomes is equitably distributed; diverse perspectives recognized, valued, and incorporated	Solutions that work for the majority; recognitional, procedural, and distributional justice	Diversify stakeholders and involve them in all stages of planning and implementation
	Institute Iterative Learning	Adaptive management; agile design; individual and collective efficacy; learning from failure	Communities pilot and refine strategies as they learn, adapt, and innovate through a continuous learning cycle;	Increased capacity to adapt to change; strategies are more effective and mutual goals achieved	Participatory monitoring and experimentation locally to test stakeholder hypotheses
Empower Individual Catalysts	Identify, empower, and connect catalysts to others	agency-systems dynamics; inspiration; contagion; diffusion of innovation	Catalysts inspire, disrupt, organize, facilitate, innovate, model, legitimize, and teach	Others are recruited to engage in new actions; innovative solutions are discovered	Identify positive deviants and connect them to others through social networks

Reconfigure the System	Remove structural constraints and create enabling conditions	Multiple properties and processes of complex systems; constrained choice; systems analysis, thinking, planning, and strategy development; force-field analysis	New system structures formalize and institutionalize the rules, resources, and processes that shape action toward mutualistic goals	System is optimized for social and ecological common good; structural conditions support beneficial patterns of behavior across multiple actors	Use systems thinking to identify leverage points,; design multi-pronged strategies to re-align deep structures with desired outcomes
	Weave social networks	Social network theory; diffusion; social contagion	New niches are created, new connections made, and missing roles are filled; network content modified	Attitudes and actions spread through networked relationships	Establish new organizations to fill necessary or missing roles
	Change the Nature of System Content	Communication; social networks; diffusion; social norms	Content flows influence actor decisions and help them overcome barriers	Improved ability for individuals to adopt new practices	Use normative messaging in communications
Connect Across Dimensions	Connect local scale reconfigurations across geographies	networked systems; community level diffusion, social contagion	Collaboratives learn from other successful reconfigurations and adapt them to their local conditions	Successful reconfigurations spread through social networks and are adopted in other geographies	Provide opportunities for local reconfigurations to learn from each other and coordinate their efforts
	Enact multi-level policy to support local scale reconfigurations	Multi-scale governance; embeddedness; polycentric governance; disruption; transitions; social networks	Coherent, integrated policy across scale supports common elements of multiple reconfigurations and reinforces normative, action	Consistency and feedback between policy and norms across scale creates stability and durability over time	Identify common elements of place-based reconfigurations and integrated those into overarching policy
	Capitalize on opportunities created by disruptions	Societal transitions; disruption; sorting	Disruptions motivate actors to restructure locally to avoid negative consequences; differential survival under disruption provides ideas for restructuring locally	Local efforts are inspired to restructure systems to be more resilient to disruption	Monitor status of societal transitions and disruptions; understand how other reconfigurations survive or fail under disruption

5.1. Situated Reconfiguration

Proposition #1: Transformational change in socio-ecological systems follows a process of transition in which local-scale reconfigurations sort, compete, converge, and emerge over space and time into new system states.

Societal transitions theories, such as punctuated equilibrium, socio-technical, socio-ecological and sustainability transitions explain how society changes and evolves over long time periods at the macro-scale (40-50 years; Table 7). These theories have been applied to explain de-carbonization of energy and transport systems (Geels, 2002; Schot et al., 1994; Verbong & Geels, 2007), ecosystem management (Olsson et al., 2004), decline in tobacco use and improvements in motor vehicle safety in the United States (Gielen & Green, 2015). These theories describe transformational change as a multi-scalar process in which innovations arise and develop at the meso-scale in reconfigurations that are socially, spatially, and temporally nested within society (Geels, 2002).

Table 7. Stages of societal transition

Stage	Theories and Concepts	How it works	Time
Pre-existing equilibrium	Cultural alignment, embeddedness, networked systems	Mental models, systems structures and behaviors of sub-systems align with those of dominant system; some local departure and variation exists, but is suppressed by dominant system	
Disruption	Triggers, catalyst events	Exogenous or endogenous events, movements, communications, technologies, and observations challenge or alter one or more levels of pre-existing, deep structure, and serve as triggers catalyzing change	
Activation	Place-based inspiration; leadership; social and resource capacity	In places with sufficient capacity, individual leaders inspire others and establish groups to respond collectively to disruption	
Meso Scale Reconfiguration	Build communities of practice, empower individual catalysts, restructure systems, connect across dimensions	Collaborative groups find common purpose and establish shared, local, socio-ecological goals; reconfigure place-based, bounded systems to fit local conditions and achieve mutual goals; develop, test, and socialize new practices that over time normalize; variation across geographies increases as different responses to disruption emerge across regions	
Sorting	Differential survival and competition, networked systems, social evolution	Some place-based re-configurations are successful; others fail and dissipate; successes diffuse through socio-structural networks to groups in other locations that adopt modified versions that fit local circumstances	
Retention	Embeddedness, networked systems, convergence	As more and more local groups adopt elements of successful reconfigurations, new structures and practices normalize and institutionalize across geographies; deep structure of dominant system converges around new structures as it adopts elements and norms of successful reconfigurations into policy	
Transformation	Deep systemic socio-ecological change, cultural change, societal change	A new dominant system emerges as sub-systems align toward mutual socio-ecological goals; individual actors and groups align their actions with the new, dominant system; society meets mutual social and ecological goals across wide geographic regions and the new, regenerative system sustains new actions and outcomes over time.	

Transitions theories describe societal change in similar ways. In relatively stable systems, mutually reinforcing feedback among cultural, social, political, and economic structural conditions, actions, and outcomes maintain equilibrium states (Burns & Dietz, 1992; Geels, 2011; Gelfand et al., 2011; Jones et al., 2011; Jost et al., 2004; Markus & Kitayama, 2010; Meadows & Wright, 2008; Moon et al. 2018). Transitions begin when exogenous or endogenous forces or events disrupt deep structures of the broader systems and trigger local reconfigurations. Over time, successful reconfigurations are retained, and the system as a whole is transformed into a new equilibrium state (Baumgartner et al., 2009; De Haan & Rotmans, 2011; Fischer-Kowalski & Rotmans, 2009; Geels, 2002; Geels, 2011; Gersick, 1991; Kemp et al., 2007; Markard & Truffer, 2012; Moore et al., 2014; Prokopy et al., 2014; Wollin, 1999). Disruptions occur at a variety of scales and trigger purposeful strategic reconfiguration for a variety of motives (i.e., activation stage) when they are fast-paced and disrupt deep components of the system (e.g., dominant mental models; de la Sablonniere & Taylor, 2020; Moore et al., 2014; Sweetman et al., 2013; Taijfel & Turner, 1986; Usborn & de la Sablonniere, 2014), and when place-based groups can overcome barriers to activation, such as lack of capacity, collective trauma (e.g., indigenous peoples), confusion, lack of collective identity, or the inability to identify mutual goals (Prokopy et al., 2014; de la Sablonniere & Taylor, 2020).

The TTC draws from transition theory by accepting the general theorized pattern of transition and by adopting the central importance of the meso-scale reconfiguration, which is situated within broader, societal transition. This means that the processes that occur within reconfiguration are of central importance, but it does not imply that change always occurs in a linear fashion along this pathway. The networked, embedded nature of systems means that change is multidirectional, circular, and recursive, not adhering to singular pathways (Sallis et al., 2008; Stokols, 1996). While transitions theory is useful in these ways, it cannot suffice for transformational change in conservation on its own because: (a) it neglects place and other spatial dimensions; (b) it underemphasizes the role of agency and social process; (c) it lacks application to conservation problems; and (d) it focuses mostly on de-centralized pathways (Coenen et al. 2012; Geels 2011). The TTC fills those gaps and provides more socio-ecologically oriented description of the applicable social and agentic theory at play within the reconfiguration.

5.2. Place-based Collaborative Settings: The Context for Reconfiguration

Proposition #2: Collaboration in a bounded place (e.g., a watershed) helps groups communicate about and formulate reconfigurations around mutual social and ecological goals and shared interests and experiences.

Research suggests that nested, place-based collaborative settings are an essential precondition for the reconfiguration of socio-ecological systems (Druschke et al., 2013; Mahajan et al., 2021). These settings are mutually constituted socio-ecological spaces that comprise the local context that shapes individual and collective action and, therefore, change (Cheng et al., 2003; Hansen & Coenen, 2015; Low & Altman, 1992; Seamon, 2020). These studies suggest that socio-ecological systems change is more feasible when there is a geographic boundary of focus, and place provides that boundary.

Place is an integrative concept that links the social and ecological aspects of systems (Cheng et al., 2003), but it has not been widely incorporated into socio-ecological models or transition theory (Hansen & Coenen, 2015; Masterson et al., 2017). Place is important because individuals are more likely to interact with each other and share history, material dependence, experiences, impacts, and the consequences of actions (Cross, 2015; Manzo, 2005; Seamon, 2020; Yoder & Chowdhury, 2018). Cultural, social, and psychological processes (e.g., material dependence, ideology, spirituality, and sensory experience) interact to create individual and collective place attachment and identity, which can foster social cohesion (Cross, 2015; Manzo, 2015).

Collaboration involves regular, transparent communication and engagement around that which is shared (e.g., goals, responsibility, risk, knowledge, consequences), and allows actors to prioritize the common good over individual self-interest (Dillenbourge, 1999; Kliskey et al., 2021; Stern, 2011). Collaborative settings are essential for DAPs to enable activation of social processes that are an essential prerequisite for establishment of new social and behavioral norms and to diffuse behaviors from the group to individual actors and across actors within the group (Ehrlich & Levin, 2005; Minkler et al., 2008). Place-based collaboration is more likely to be activated in places with stocks of human, social and financial capital from internal and external sources, which are necessary to overcome transaction costs (Jablonski et al., 2020; Lubell et al., 2002), and when the environmental issue (a potential disruption) is more severe (Prokopy et al., 2014). A community's readiness to engage in collaborative action as a response to disruption can be assessed by measuring its collective attitude, knowledge, ongoing efforts, available resources and leadership, and represents opportunity for conservation action (Dannenberg & Barrett, 2018; Knight & Cowling, 2006; Oetting et al., 2014). Collaboration can improve the chances of meeting socio-ecological goals, but most collaboratives don't measure outcomes (Wilkins et al., 2021).

5.3. Building Communities of Practice

Proposition #3: Communities of practice activate social processes that enable collaborative groups to learn about and socialize complex, new action sets and strengthen their capacity to collectively act in increasingly complex ways.

Conservation practices are multi-step, complex behaviors that are often poorly understood and not well integrated within everyday routine (e.g., planting, tillage, harvest procedures in farming; Baumgart-Getz et al., 2012), therefore they may need to be socialized in groups before most individuals will adopt them. Conservation practices are not only specific technical prescriptions, but "social practices" or mutually constituted, routinized sets of behaviors, along with associated mental activities, material resources, knowledge, tools, language, processes, and structures (Reckwitz, 2002; Shove, 2010; Shove et al., 2015; Wenger, 2000). For new practices to remain durable over time, they must become normal, routine, and integrated into normal ways of doing things (e.g., conservation practices integrate into farm production systems; Church et al., 2020). This implies that conservation practices may not be widely adopted unless they are socialized in group settings where social processes that support their adoption can be activated.

Communities of practice are groups of people who engage in ways of doing things together (e.g., cooking, practicing agriculture, shopping, etc.) that help activate social processes helpful in diffusing action across individuals in the group (Eckert, 2006; Wenger, 2000). They are the social settings, processes, and actions in which the feasibility, effectiveness and legitimacy of new practices are demonstrated; group and self-efficacy is built; social norms are activated; and new actions are internalized by individuals in the group (e.g., attitudes change to support new actions; Pickering et al., 2018; Van Bavel et al., 2020; Wenger, 2000; Yoder & Chowdbury, 2018). Communities of practice also are social networks, and the individuals within them can be viewed as carriers of action (e.g., early adopters) who help diffuse actions across individuals in the group (Reckwitz, 2002). Social norms are more influential when they are salient (Cialdinia & Goldstein, 2004) and behavioral norms are more influential when visible

(Fielding et al., 2005). Communities of practice help to activate normative influences that can be powerful influences over change (Cialdini et al., 1990; Farrow et al., 2017).

Communities of practice typically include collaborative visioning, planning, and goal setting, which enables groups to prioritize the common good and share responsibility for it (Minkler et al., 2008). Most conservation targets (e.g., biodiversity, air quality, water quality, etc.) are common goods, a special set of public goods that may not benefit each individual member equally (Hassain, 2018). Common goods are inherently social, because they represent compromise and reconciliation between individual and collective interests (Hassain, 2018). Linking social goals and ecological goals during the goal-setting stage can help groups find common ground, build support for action, and find leverage for strategic reconfiguration by asking what systemic elements are common to both environmental and social problems. The common vision and goals are vitally important to success because they are the foundations of the new system to be configured (Scrase et al., 2009).

Communities of practice are collaborations that can foster social cohesion through familiarity, frequent interaction, shared identity, trust, and reciprocity (Mahajan et al., 2020; Olson, 1965; Wenger 2000). Social cohesion is a centripetal force that facilitates convergence toward common goals (Ehrlich & Levin 2005; Fehr & Schurtenberger, 2018; Heinrich & Boyd, 2005). It enables groups to prioritize the "we" over the self-interests of the members, and it helps build social capital, which can lead to higher levels of collective action (Auer et al., 2020). A recent study found that countries in which people were most in favor of precautionary measures for COVID were those that fostered a sense of national attitude of "we're all in this together" (Van Bavel et al., 2020). A variety of community engagement theories help explain how collaborative action groups can be maintained, facilitated, and sustained (Kliskey et al., 2021;

Margerum, 2008; Sarkissian, 2009; Wilkins et al., 2021). Social cohesion is a source of solidarity that can keep groups aligned and together through difficult times and ultimately enable them to act in more complex ways and to act more effectively together (Auer et al., 2020; Bruggerman & Corten, 2021).

Proposition #4: Democratic, inclusive decision-making processes produce reconfigurations that are fair and acceptable to more people and therefore are more effective and durable over time.

The common good can only be identified by consulting a diversity of perspectives, particularly of those who are underrepresented but affected (Pellow, 2019; Salomon et al., 2018; Tallis & Lubchenco, 2014). Diversity is dependent on inclusive decision-making because diverse perspectives are manifested through inclusion (Sabharwal, 2014). Inclusive process can improve the chances that ecological outcomes are met by resolving dissenting views, reducing uncertainty, dismantling inequalities, and increasing cooperation with communities most immediately affected (Curseu et al., 2017). Research suggests that, when most individuals affected by a conservation effort can participate in making and modifying the rules, the effort is more likely to be acceptable to local communities, successful, and durable over time (Ostrom, 2000; Salomon et al., 2018). Transformational change that benefits a wider swath of society requires that recognitional, procedural, and distributional justice are pursued as goals and achieved as social outcomes conservation (Gooden & Sas-Rolfes, 2020; Pellow, 2019).

Essential tools to achieve environmental justice outcomes include stakeholder analysis, planning, and diverse participation (Vogler et al., 2017), but designing and executing processes in which actors share equal power and influence over decisions and outcomes also is essential to enable diverse perspectives to be manifested (Calfucura, 2018). An inclusive process must strike a balance between encouraging expression of individual uniqueness and diverse perspectives with a sense of belonging, collective identity, and social cohesion toward its common goals and group processes (Shore et al., 2018). This balance can produce innovation, legitimacy of conservation approaches, greater commitment, and more effective and durable collective action (Shore et al., 2011).

Proposition #5: An iterative learning process fosters shared truth, increases the effectiveness of reconfigurations, and increases the group's ability to adapt to exogenous forces.

Collaborative monitoring processes are essential elements of transformational change as a means of group learning and adaptation within communities of practice, but they also are an important social process that improves a group's capacity to act collectively. A shared source of information arising from the integration of science and practical knowledge can generate a "shared truth" that supports the common good because it because it engenders trust and provides a means of overcoming disagreements and reducing polarization (Radzvlavicius, 2021; Wilmer et al., 2018). Collaborative monitoring structures should: (a) recognize complexity, non-linearity, and multiple scales; (b) integrate social and ecological variables; (c) be predictive; and (d) monitor both outcomes and processes of implementation (Cundill & Fabricus, 2009). Monitoring also should bolster the social processes of collaboration by: (a) encouraging ongoing reflection; (b) involving decision-makers directly and collaboratively; (c)) feeding directly into decision-making; and (d) encouraging working toward a system's potential through collective sense-making (Cundill & Fabricus, 2009; Sanford, 2019). For example, participants in reconfigurations might develop the hypotheses that are tested and evaluated in the field as conservation practices are being implemented, and this can demonstrate the effectiveness of a practice and increase its legitimacy (Pickering et al., 2018; Wilmer et al., 2018; Yoder et al, 2019). Collaborative monitoring at the watershed scale can dramatically increase cooperation and be a cost-effective means of generating useful conservation interventions (Nichols & Williams, 2006; Ostrom, 1993; Radzvilavicius et al., 2019; Yoder et al., 2019).

Adaptive management is a valuable process of continual learning to solve problems and adjust actions and goals over time, but it is insufficient to address key issues such as connecting to broader scales, conflicting stakeholder perspectives, and institutional limitations (Virapongse et al., 2016). The TTC posits that iterative learning is broader and more consistent with social practice theory wherein conservation practices are envisaged as "activity systems" or relational bundles of knowledge, action, and socio-ecological context (West et al., 2018). Under this view, practices inherently are relational constructs that are learned, practiced, and evaluated in groups that include scientific and practical expertise. Collaborative monitoring processes are structures that facilitate these social activities across communities and landscapes (Calfucura, 2018; West et al., 2018). This is a departure from common, dominant approaches in which conservation practices are informed by external scientific information and viewed as specific, independent technical applications and financial transactions that are engaged in by individuals at the parcel or farm scale (e.g., U.S. Department of Agriculture, Natural Resources Conservation Service). Thus, iterative learning through collaborative monitoring is not a standalone activity but an integral structure for reconfiguration as a mechanism for learning, adapting, and socializing new practices that are a part of the reconfiguration (Nichols & Williams, 2006).

5.4. Empowering Catalysts

Proposition #6: Empowering individual change agents helps recruit actors to engage in new actions and produces more innovative and practical solutions.

The TTC contends that social change processes interact with agentic processes to advance transformational change in unique but complimentary ways. Social processes are a convergent influence that helps groups unify and act collectively around common goals, whereas agentic processes contribute to change in the opposite way by providing a system with a source of diversity, innovation, or variation that can be accessed or will emerges when the system is disrupted (Geels, 2011). For example, innovators are essential for change because they are already engaged in the behaviors necessary for change, but they represent a small percentage of individuals who are acting outside of the current norm (Rogers, 2003). The variation provided by diverse actors in the system provides new ideas, inspiration, etc. that are potential sources of change. Individuals in positions of power in an organization may be resistant to change because they are acting out, promoting, and benefitting from the status quo (Llopis, 2015). Under the TTC, leaders are not necessarily in positions of power but are connected and respected agents of change throughout the process of reconfiguration. They provide entrepreneurship, facilitation, advice, coordination, modelling, networking, inspiration, networking, and trust-building in ways that contribute to change (Hussein, 2018; Westley et al., 2013). Leaders may promote collaboration by highlighting common ground and shared identity, connecting collaborative efforts to policymakers, and modeling their actions to others (Van Bavel et al., 2020). Thus, social processes and agentic processes are distinct-but-complimentary influences on reconfiguration. Connecting change agents to others can be an effective intervention that diffuses new actions across social networks (Neimiec et al., 2021; Neimiec et al., 2019)

This concept is consistent with actor-systems dynamics theories in which actors and systems are recursive, mutually constitutive, and reciprocally deterministic (Bandura, 1977; Burns et al., 2002; Dietz & Burns, 1992; Giddens, 1984; Markus & Kitayama, 2010; Oppong, 2014). In stable systems, actors may possess little agency, especially when actions are unconscious, ineffective at producing change, highly constrained, or have invisible consequences (Burns et al., 2002; Dietz & Burns, 1992). System disruptions may remove constraints, creating opportunities for individual actors to act independently in ways formerly suppressed by the system (Jost et al., 2004). An actor's degree of agency to create change varies continuously as a reverse function of constraint and is relative to actor characteristics, power relations, and other factors (e.g., timing; Bandura, 2018; Dietz & Burns, 1992; Elder, 1994).

In rational choice theory and some collective action frameworks (e.g., Ostrom's Institutional Analysis and Design Framework), agents are conscious, self-interested actors acting on intention (Bandura, 2018). In norms-oriented theories, agency is highly constrained by or sometimes unconsciously absent from elements of the system such as norms (Bandura, 2018). The TTC can accommodate both conscious and unconscious actions, and social practices are carried out consistently according to the rules and norms of the system but with some variation among actors related to individual characteristics. That diversity is expressed and capitalized upon during reconfiguration. Individual actors are neither fully autonomous nor conformist "dopes" (Bandura, 2018). This view of agency is consistent with a reciprocally deterministic view of the attitude-behavior relationship (Gelfand et al., 2017). In other words, self-perception theories emphasize cognitive responses (e.g., attitudes) to behavior (cognitive dissonance; Cooper, 2007), (self-perception; Bem, 1972), whereas attitude-behavior theories emphasize the other causal direction (e.g., Theory of Reasoned Action; Fishbein & Azjen, 2010), and the TTC recognizes both directions.

5.5. Reconfiguring Systems

The configuration of system structures creates the conditions that support or constrain coordinated conservation action across multiple actors and at multiple levels in the system. As communities of practice harness social process, they build the capacity to act together in complex ways. The degree to which collaborative groups use this capacity to restructure their local system to support and encourage the actions necessary to achieve their mutual goals is critical to achieving transformational change. There are three basic ways to reconfigure a system: (a) remove structural constraints and create enabling conditions; (b) reconfigure social networks; and (c) change the nature of the content exchanged between actors. Restructuring contributes to transformational change by creating action situations that support new behavioral patterns. Restructuring helps establish mutually reinforcing feedbacks among new behavioral patterns and the processes, rules, material, and physical conditions necessary to practice them, thereby sustaining changes in behavioral patterns over space and time (Stokols, 1996). Reconfiguring system structures is akin to institutional change because institutions are embedded systems (Acheson, 2006).

Systems thinking, planning, and analysis are subjective, dynamic processes essential for strategic restructuring (Foster-Fishman et al., 2007; Knight et al., 2019). Systems planning involves: (a) defining the system and diagnosing the problem; (b) identifying fundamental elements and patterns of interaction; (c) identifying leverage and power in the system; (d) visioning and goalsetting; and (e) developing change strategies that alter deep structure to accomplish systemic change goals (Foster-Fishman et al., 2007; Hallet & Hobbs, 2020; Kennedy et al., 2017). Many tools and approaches are available to implement systems thinking, planning, and analysis (Abercrombie et al., 2018; Abson et al., 2017; Gray et al., 2013; Kennedy et al., 2017; Meadows & Wright, 2008; Omidyar Group, 2021; Stroh, 2015); however, systems thinking has not been well integrated into conservation (Knight et al., 2019).

Proposition #7: Removing structural constraints and creating enabling conditions fosters change in behavioral patterns across multiple actors at multiple levels in the system.

The COVID pandemic in the United States is an example of a disruption that has temporarily altered behavioral patterns, but those patterns are more likely to remain if they are accompanied by structural change. The pandemic resulted in a dramatic increase in online shopping (e.g., Amazon sales increased 200%) which caused retailers to close more than 8,000 stores in 2020, with another 10,000 expected to close in 2021 (Takefman, B., 2021; Thomas L., 2021). In this case, the pandemic is a disruption that altered a long-standing behavioral pattern (shopping in stores) and subsequently caused a potentially longer-lasting structural change in the system (i.e., the absence of physical buildings). In a strategic response to the opportunity created by the pandemic disruption, Amazon expanded its infrastructure and refined its order-fulfillment process to better enable online shopping (a structural change; Takefman, 2021). The combined effect of new constraints (absence of stores) and improved enabling conditions (better online shopping process) is an example of reconfiguration, in this case to reshape shopping conditions. The new configuration creates action situations that are favor a shift to online shopping.

This is an example of how restructuring can reconfigure systems in ways that influence widespread, long-term behavioral patterns. In conservation, opportunistic, strategic restructuring is similarly important to reshape action situations to support the targeted behavioral patterns. Reconfiguration may involve re-aligning systems with new mental models (e.g., collaboration), strengthening structures and interactions that are already aligned, or creating new structures that currently are missing (e.g., a new monitoring system; Foster-Fisherman et al., 2007; Scrase et al., 2009). It also may involve creating constraints in the system to discourage negative action (e.g., publicly reporting point-source pollutants).

Leveraging change in one part of the system will lead to desired outcomes only if concurrent shifts happen in the relational and compositional elements of the system, suggesting that multiple interventions working to reconfigure the system in the same direction will be necessary for transformational change (Foster-Fisherman, 2007; Higgins, 2015; Sallis et al., 2008; Stokols, 1996). For example, in the farmland surrounding the Florida Everglades, multiple actors in response to a lawsuit and invasive species research (disruptions that triggered collective action), shifted from individual, voluntary, incentive-based conservation approach to a collaborative, jointly compliant, regulatory approach (Yoder, et al., 2019; Table 8). These shifts involved multiple structural changes adopted in a collaborative setting, including new requirements to adopt conservation practices, group compliance standards, monitoring systems, social learning through interaction and demonstration, and multi-level governance. This resulted in dramatic declines in phosphorus pollution, exceeding initial goals (Yoder et al., 2019).

Table 8. Case studies of two successful U.S. reconfigurations

Features	**Florida Everglades**	**City of Fort Collins**
Target	314 farmers, businesses, public sector, residential	70,500 residential and business customers
Place	Everglades Agricultural Area (approx. 202,343 ha), FL USA	City of Fort Collins, CO, USA (approx. 14,245 ha)
Goal	Reduce Phosphorus pollution from farm fertilizer runoff	Improve energy efficiency of buildings
Common goods	Water quality, biodiversity, treasured public place through	Clean air, energy conservation, greenhouse gas reduction, climate change reduction
Outcomes	Reduced Phosphorus loads to 55% below baseline exceeding goals Realization of personal benefits (e.g., cost savings) Self-efficacy increased	Tripled enrollment rates in program resulting in 60% greater carbon savings and increased Therm savings per home by 70% and kWh/home by 50% over the previous program (9,543 kWh/year/household) Converted 44% of assessments into comprehensive home upgrades and 52% of projects chose the most comprehensive of the 3 packages offered
Dimensions		
Build Communities of Practice (social processes)	Targeted watershed rather than individual farmers Collective goals linked farmer livelihood/reputational risk (individual), protection of treasured place (social) with reduced P loading and invasive species impacts (ecological) Trainings facilitate peer-to-peer and expert to farmer learning Bridging capital and regular formal and informal communication facilitated shared understanding and activated normative motivations Monitoring generates regular feedback on the effectiveness of BMPs and increases legitimacy of practices, collective efficacy, and innovation Regular coordinated action normalizes practices	Targeted neighborhoods rather than individual homeowners Utilized neighbor to neighbor open houses and social network recruitment into the program The City worked with local contractors and energy efficiency providers to collectively redesign the program to meet the needs of homeowners, the City's GHG goals, and local business partners Social norms "join your neighbor" (yard signs, brochure boxes) All contractors working on these projects increased adoption of a new Quality Assurance process using photo documentation Currently working with the local MLS to provide home energy information to realtors, buyers, and appraisers to incorporate into home pricing and appraisals

Empower Catalysts (agentic process)	Farmer success awards: recognition and success created pride in achievement, and activated personal stewardship norms and sense of responsibility toward community	Tried enlisting neighborhood association leaders, but didn't work Tailored messages to different demographic groups
Reconfigure the System (structural influences)	Shift from individualized, voluntary incentives to joint compliance regulation establishes water quality standard Tax imposed to collectively pay for storm water treatment areas Locally based, transparent monitoring system established Permitting program requires each farm to adopt BMPs, but only offers socially acceptable practices, and offers flexibility	On-bill financing New coordination between City departments (billing, vendors, etc.) Repositioned customers in the supply chain to remove barriers related to time, complexity, and trust of vendors. New intermediary with contractors Simplification of choices and selection of packages. New financing capital acquired New roles for Energy Advisors Standardized pricing for certain components, and contractors who agreed to the pricing were placed on a rotation for projects Free home energy efficiency assessments Bundled packages from unbiased city specialist
Connect Across Dimensions	Coordinated multi-level governance between Sugar Cane Growers Coop (local), South Florida Water Management District and University of Florida (state) and Everglades Forever Act (federal)	Designated as a Landmark case study and written up as a case study in ToolsofChange.org based on impact, innovation, replicability, and adaptability.

Restructuring in collaborative settings requires that groups come to a collective understanding of which structures are most pivotal in the system, how they interact with others, and which are most practical to address with limited resources (Foster-Fisherman et al., 2007; Higgins, 2015; Reichers et al., 2021). It also involves understanding what differences in the system could serve as leverage for change (Abson et al., 2017; Foster-Fisherman et al., 2007; Sanford, 2019;). For example, the diverse social and ecological needs of people in the community (e.g., employment opportunities, public health, a clean environment) may be at odds with narrow goals of the system (e.g., efficiency, profit). How can the system be reconfigured to produce more diverse outcomes? Identifying bright spots, or changes that have already been made or capacities for change that exist (e.g., a community collaborative with a paid coordinator), and capitalizing on them, is also critical (Omidyar Group, 2021).

Proposition #8: Social networks structure the relationships between actors resulting in more effective transmission of change through the system.

Social networks are the structures of relationships through which the material, informational, technological, and social content of a system flows (Summers et al., 2013). Network weaving involves understanding and changing the configuration of a social network to optimize the system toward common goals (Krebs & Holley, 2006). Transformational change depends on an understanding of the dynamic relationships between actors in a system, because the way in which these relationships are configured has powerful influence over the spread of new behaviors (Kerr & Coviello, 2020; Law, 1992; Lockton, 2012; Summers et al., 2013; Zaheer et al., 2010). The networked nature of systems means that flows may not adhere to discrete hierarchies (e.g., individual, interpersonal, community, etc.) but instead interact through social network linkages. Therefore, understanding the configuration of networks (e.g., density, specificity, cohesiveness, interconnectivity, centralization, etc.) is key to fostering change (Bodin et al., 2009; Hearney & Israel, 2008; Latkin & Knowlton, 2016). The structure of social networks determines how content flows through the system, which may not adhere to arbitrary hierarchies (Henry & Vollan, 2014).

Meadows (2021) suggests that the structure of information flows is a relatively high leverage point for change in systems, but the power of social networks to foster change goes well beyond information to any kind of system flow (Zaheer et al., 2010). Attending to and intentionally building dense, interconnected social networks can help build and maintain social cohesion, build social capital, community support, facilitate learning and innovation through social influence, and help groups initiate and coordinate action (Auer et al., 2020; Hearney & Israel, 2008). Potential strategic interventions to weave social networks include: (a) changing the governance of the network; (b) changing the overall configuration of the network; (c) connecting existing actors to each other; (d) adding new actors to fill new roles; and (e) filling structural holes (Dietz, 2019; Hoang & Antoncic, 2003; Hoang & Yi, 2015; Kerr & Coviello, 2020; Latkin & Knowlton, 2015; Zaheer et al., 2010).

Networks are the structures through which social content flows, therefore social networks represent the combined effect of structural and social influence on behavior. Social networks are created, maintained, and changed through social processes (Latkin & Knowlton, 2015). Social networks interventions therefore must simultaneously consider the structure of the network, the social processes that maintain it, the content that flows through it, and how these elements influence behavior (Latkin & Knowlton, 2015). As system structures, social networks also can be understood in terms of how they constrain or enabling behavioral patterns, for example, corporate agricultural supply chains can create dependency in farmers that constrains choice in ways that make conservation action less possible (Hendrickson et al., 2017).

Proposition #9: The nature of the content exchanged between actors in a social network influences their ability to adopt new conservation actions.

Content is the material, informational, technological, and social substance (e.g., information, money, advice, communications, norms, attitudes, resources, capabilities, trust, power, etc.) that flow through social networks and other system structures. (Summers et al., 2013). A common conservation intervention is to influence decision-making by providing information or financial incentives (Echiaveria, 2005; Prager & Posthumus, 2010; Steg & Vlek, 2009). The emphasis in the TTC is to expand the scope of what is considered content to include social aspects, such as personal support and interpersonal normative communication, and an awareness that content flows can influence behavior directly and unconsciously, not just through decisions. Information alone is notoriously ineffective at changing behavior, but orienting information such that it provides feedback on the effectiveness of action has a stronger effect (Fischer, 2008; Lockton, 2012).

5.6. Connecting Across Dimensions

Because local-scale reconfigurations are nested socially, spatially, and temporally within broader-scale societal transitions, transformational change depends on connecting the changes made within local-scale reconfigurations across these dimensions (Geels et al., 2002; Kark et al., 2015). Connecting across dimensions is a late stage of reconfiguration that can move societal transitions along as reconfigurations begin to move into sorting and retention stages (see Table 6). Connecting across dimensions differs from the common conception of “scaling up” frequently used in conservation. “Scaling up” refers to increasing the impact of more local conservation efforts, but this idea falsely implies expanding or replicating a program, innovation, skill, or policy that has been proven in one place to another place or across individuals (Battista et al., 2017; Holcombe, 2012; Rao & Power, 2019). Scaling up ignores the fact that biophysical and social aspects of place vary and interact differently across geographies and influence action in different ways (Cheng & Daniels, 2003; Cheng et al., 2003; Prokopy & Genskow, 2015; Seamon, 2020). A variety of scientists have suggested that social change strategies developed in one place cannot be effectively replicated across geographies without attention to their specific attributes (Cheng & Daniels, 2003; Chess & Gibson, 2007; Coenen et al., 2012; Hansen & Coenen, 2015; Jablonski et al., 2020; Masterson et al., 2017). Thus, the TTC asserts that reconfiguration must take place in local, place-based settings before they are connected across socio-political scales and geographies and can become durable over time.

Proposition #10: Multi-level coordinated governance institutionalizes the common elements of successful local reconfigurations, creates consistency in action across social scale and actors, and establishes positive feedback between new local norms and high-level policy that sustains normative action over time.

Multi-level governance and polycentric institutional theory explains how transformational change may require a mix of diverse, decentralized solutions manifested in local reconfigurations and centralized policy that supports and institutionalizes the common elements of these reconfigurations (Biggs et al., 2010; McGinnis, 2011; Ostrom, 2005; Rijke et al., 2013). These arrangements can address problems in which the scale of management fails to match the scale of social and ecological processes, which is common to DAPs (Folke et al., 2007). To accomplish this, policy leaders must identify the common, successful elements of local reconfigurations and incorporate those into overarching policy (e.g., federal) that aligns with and supports policy at lower levels. Policies can support common structural needs of local reconfigurations, for example, by providing human and social capital to enable collaboratives to succeed, or financial resources for them to collectively implement their plans (Rao & Power, 2019). Leaders of reconfigurations can connect to higher level political leaders and support these kinds of policies. For example, George Floyd's public murder at the hands of Minneapolis police in 2020, led to a bill in the House of Representatives that, if passed, would require several types of nationwide policing reforms, including: providing mental health training and assistance for officers; collecting use-of-force data; providing de-escalation training; and certifying officers and training courses at the federal level (Ray, 2021). These examples of structural reconfiguration have already been tested in more than 30 states (Ray, 2021).

To create durable change over time requires the formation of mutually reinforcing feedbacks among groups (social and normative processes) and individuals (individual processes) and systemic reconfigurations (policies, processes, procedures, patterns). Policies are ways of institutionalizing change that has already occurred in local reconfigurations. Under the TTC, policies are viewed as methods of establishing mutually reinforcing feedbacks between new social and behavioral norms and formal institutions, as opposed to a "driver" of change which suggests a unidirectional, top-down process of change (Green, 2016). The TTC posits policy is a means of supporting local change that has already occurred, and in the process, influencing more change in places that have not yet adopted the new norms, a multi-directional process (Green, 2016). Decentralized change also is a more a democratic way of implementing policy than conservationists typically deploy and one in which local communities have more control over their resources (Salomon et al., 2018; Calfucura, 2018; Horning et al., 2016). It also can be more effective at accomplishing environmental objectives (Wright et al., 2016). Multi-level governance therefore helps stabilize new system states over time by fostering these mutually reinforcing feedbacks between policies and norms across socio-political scale. In stable systems, these norms and policy feedbacks must be continually nurtured and maintained over time across multiple societal levels. For example, the Water Sustainability Act of British Columbia encourages local place-based governance that is connected through province-wide overarching policy (Horning et al., 2016).

Proposition #11: Connecting local reconfigurations across geographies enables the spread of successful reconfigurations through social networks to other geographies

Connecting across geography implies a form of community-scale diffusion across leaders in different reconfigurations (Davidson & Loe, 2014; Mahajan et al., 2020; Mascia & Mills, 2018), but few community-based conservation efforts utilize this strategy (Cheng et al., 2019). Connecting across geography implies forming knowledge networks of reconfiguration leaders to facilitate cross-boundary communication, information sharing, capacity sharing, and coordinated action (Wheatley & Frieze, 2006; Wyborn, 2015). De-centralized approaches that succeed at making these connections can help stimulate novel solutions (Salomon et al., 2018). Bridging organizations may serve this connecting role (Rathwell & Peterson, 2012); for example, the Monarch Joint Venture coordinates the actions of many other organizations and enables communication among them through a website and regular meetings. (Monarch, 2021).

Proposition #12: Connecting local reconfigurations to the broader, ongoing societal transitions accelerates the pace and scale of change by integrating ideas, restructuring configurations, and behavioral norms into larger-scale movements.

Connecting the experiences of place-based collaboratives with the broader, ongoing changes in society (e.g., sustainability, agroecology, regenerative agriculture, etc.) can help increase the pace and scale of change by stimulating new collective action in other places. This requires identifying the societal transitions underway and tracking endogenous and exogenous disruptions that can stimulate additional collective action (Westley et al., 2013). It also involves using societal transitions and disruptions as opportunities to communicate the inadequacies of the current system to support local values and goals and motivate additional local action in response. Creating awareness of disruptions and their potential to impact local communities can motivate collaborative groups to act in order to avoid internal or external threats to local values (Calfucura et al., 2018).

6. SUMMARY

The integration of existing social, agentic, and systemic theoretical perspectives suggests that strategic transformational change occurs in place through collaborative reconfiguration, which then is connected across multiple dimensions (Figure 12). Change begins when an endogenous or exogenous disruption alters or challenges the deep structure of the dominant or local system. In response, leaders in local communities inspire others to form collaboratives that coalesce around common, mutualistic socio-ecological goals. These groups regularly engage in common endeavors; new actions are socialized, and actors learn from each other and establish new social and behavioral norms. Through strategic, inclusive collaboration, groups reconfigure local systems and formalize and institutionalize new behavior norms into policy, process, and practice. Through community-scale diffusion, local reconfigurations are communicated with other collaborative groups in other geographies and with actors at higher socio-political scales. Through time, the different types of reconfigurations that local collaboratives make are tested and adapted as they respond to disruption, and some fail while others succeed. The behavioral norms and structural elements of successful reconfigurations are incorporated into multi-level coordinated policy that supports and reinforces them, and, over time, these converge. Divergent elements remain locally and represent diversity in the system, which helps maintain resilience to future disruption. Through this process, a new, dynamic equilibrium state emerges, and the old system is transformed.

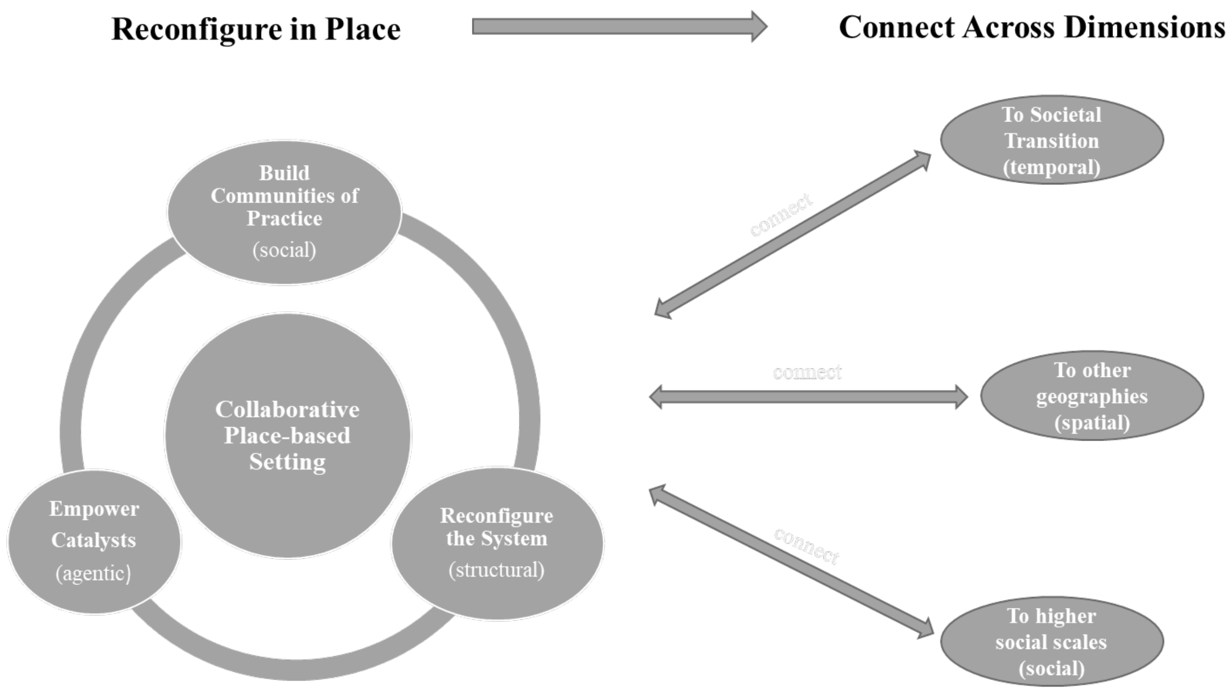

Figure 12. Theory of transformational change

7. DISCUSSION

While socio-ecological systems theorists have advanced a conceptual understanding of the interconnections between social and ecological systems, the integration of agentic, social, and structural influences and the incorporation of social, spatial, and temporal dimensions into these models remain incomplete (Manfredo et al., 2014; Muhar et al., 2017; Redman et al., 2004; Restall & Conrad, 2015). The TTC endeavors to integrate insights from various theories into a cohesive theory with testable hypotheses useful for explaining past efforts and leading new efforts seeking to create lasting change. Here I review the key points of divergence from existing theories.

The first difference between the TTC and existing theories is the shift from choice to action as the key mediator of conservation actions. Much applied research has focused on refining Ostrom's SES Framework and Institutional Design and Analysis Framework, which originates from institutional and economic theory (Cole et al., 2019; Collins et al., 2010; Ostrom, 2009; Polski & Ostrom, 1999). Like Ostrom's framework, the TTC focuses on how systems influence action situations that shape behavior and outcomes. Under Ostrom's theory, however, action situations are viewed as settings in which actors consciously evaluate, deliberate, and consider options to make operational, collective, and constitutional choices, often in policy settings (Cole et al., 2019; McGinnis, 2011). Ostrom's theory reinforces the idea that conscious evaluations made within the rules and norms structures of the action situation are the primary mediators of action (McGinnis, 2011), but this perspective does not account for social and cultural theory that suggests systems through action situations also shape behavior directly and unconsciously, for example, through cultural ideology, social norms, or ecological factors and the built environment that could eliminate conscious choices and opportunities. Action rather than choice, therefore, is central in to TTC because action is a more comprehensive mediator than choice.

The TTC builds upon other valuable integrative theories that do not prioritize choice as the primary mediator of behavior, but the TTC more comprehensively integrates agentic, structural, and systemic perspectives. For example, the TTC incorporates some of the same elements (e.g., developing social networks and vision building) as Westley et al.'s (2013) theory of transformative agency. Their theory emphasizes the point that agents must act in concert with the evolving context of the systems at the macro level, which creates opportunity for change (Westley et al., 2013). This point is critical for transformational change, but their theory is less cognizant of the social and systemic processes that constrain agency, and that must be activated in these times of opportunity. Additionally, the TTC component "reconfigure the system" aligns with the view of Foster-Fishman et al. (2007) that reconfiguring systems is essential to transformational change. Their theory, however, is not specific to conservation problems in which place requires more emphasis and is focused on the community scale without describing the processes of how change moves across social, spatial, and temporal dimensions. Virapongse et al. (2016) present a framework for translating socio-ecological systems theory into environmental management, and they make some similar conclusions, such as the need for monitoring, adaptive learning, collaborative engagement, and learning, but the framework lacks attention to process, a key to strategically transformational change. Similarly, Muhar et al.'s (2017) model of socio-ecological interaction is an important contribution in that it helps conceptualize the components that need integration and how those impact behavioral patterns, but it does not explain how change moves across dimensions. Our TTC draws from these efforts to produce a more process-oriented theory of change and integrates across dimensions.

The TTC perhaps is most like Mahajan et al.'s (2020) theory-based framework, which is a process-oriented theory that explains how community-based conservation establishes, persists, and diffuses across scale. Like the TTC, it incorporates agentic, structural, and systems perspectives into a socio-ecological framework; however, it de-emphasizes structural reconfiguration by viewing it narrowly and generically as "governance." Furthermore, it does not explicitly integrate across social scales by explaining how mesoscale, place-based collaboratives related to societal change or microlevel individual behavior change. Integration across social scale is important to understand and create durable, transformational change.

8. SUGGESTIONS FOR FUTURE RESEARCH AND APPLICATION

The components of the TTC and its propositions provide a framework for future research. There is a need to validate and refine this theory based on case studies of existing successful and unsuccessful collaboratives and other conservation efforts. The TTC provides a framework for assessing what elements of reconfiguration are currently taking place, and which of those contribute most to success. Each proposition is grounded in past research and provides a testable hypothesis, that if examined in comparison to other approaches, can help provides the opportunity to advance conservation science and application in more systematic ways. Case studies could determine gaps in the structure of collaboratives that could improve the theory and provide insights for intervention (Wilkins et al., 2021). I have conducted two such case studies on what I deem as successful reconfigurations, in that they increased adoption and demonstrated environmental results (Table 8). In each case, I see many of the elements of the TTC at play, but there is no clear evidence that either of them is sufficiently addressing all aspects. Additional case studies are needed to validate the theory, help fill gaps, and evaluate the relative importance of elements to change in specific contexts. Transformational change is a long-term process which calls for the need for longitudinal studies (10 years or more in length) at the scale of the place-based collaborative (e.g., watershed) that compare the interventions and achievement of successful outcomes of reconfiguration in one place to another (Wilkins et al., 2021). Studies that take a multi-scale perspective of these collaboratives as nested in broader societal shifts also are important. The propositions I suggest should be tested within this research framework.

Addressing DAPs will require better diagnostic tools that are able to identify systems nodes ripe for intervention and diagnose key structures shaping action situations. This will require holistic systems analysis to understand the net effect of interactive system components on action situations and behavioral patterns, but current reductionistic scientific methods tend to understand system relationships by isolating system components from their broader context, which could be a barrier to testing and implementing this theory (Sallis et al., 2008). Implementing systems thinking within stakeholder settings will be essential to developing collective understandings of how systems shape behavior in specific contexts (Nguyen et al., 2011), as will mainstreaming innovative techniques of systems analysis (Walters et al., 2016). System archetypes have been advanced as short cuts for diagnosing typical systemic problems (Stroh, 2015), and there may be ways to simplify and normalize these ways of thinking to diagnose systemic conservation issues more clearly, by asking simple questions. For example, does the system constrain behavior by limiting choice (e.g., path dependency in agriculture); does it provide too many choices or choices that are too complex (e.g., municipal energy programs); or are the actions that need to occur misplaced in time or space (e.g., drinking and driving)? I suggest that development of diagnostics tools that stem from the TTC could be a valuable applied research contribution for conservationists.

Overall, the TTC suggests what the primary components of large-scale profession wide conservation efforts should entail to be comprehensively contributing to transformational change. As such, it could inform the roles of specific organizations and conservation efforts within the broader context of the conservation profession.

9. CONCLUSIONS

The power of transformational change lies not in specific theories, but in their integration. DAPs are the world's most pressing conservation issues, and solving these systemic problems requires an overarching, testable theory of change that integrates agentic, social, and systemic perspectives across multiple dimensions. The TTC is intended to advance the conservation fields' power to strategically produce change by providing a broad, multiscale framework for understanding and developing specific strategic change interventions. Future research should explore the validity of its propositions and consider the integration of additional theories (e.g., political, macro-economic) while maintaining parsimony.

The TTC represents a significant shift in the conservation profession away from the ideology of individual choice to a science-based, multidimensional view of socio-systemic change (Virapongse et al., 2016; Table 9). Its adoption would significantly alter what it means to practice conservation and would greatly expand the available conservation toolbox. Rather than asking how individual behaviors can be changed then scaled up, conservationists would ask how systems can be reconfigured in place and diffused across dimensions. This orientation focuses conservation on changing the conditions in which behavioral patterns develop, rather than changing specific behaviors. Its focus on social and systemic theories of change represents a shift toward a more proactive, collaborative, democratic approach that has potential to foster more resilient and durable solutions. It is more proactive because it focuses on interventions that address root causes that influence conditions that exist prior to action, not correcting (e.g., through punishments or rewards at the individual level) behavior that already has been encouraged by the system. It is more democratic because it relies on meaningful, collaborative, local involvement of a wide range of stakeholders, which is likely to improve local acceptance, legitimacy, and durability (Horning et al., 2016). Finally, decentralized, locally diverse, and networked solutions are likely to be more adaptive and resilient to disruption (Biggs et al. 2010).

Table 9. A comparison between the current mainstream conservation and transformational change paradigms

Element	**Current**	**Transformational Change**
Mental Models	Domination, competition, hierarchy, individual, efficiency, centralization	Mutualism, collaboration, democracy, multi-dimensional, resilience, localization
Core Question	How do we alter decisions so that many individuals choose to conserve?	How do we reconfigure socio-ecological systems to support individual and collective conservation action?
Pathway	Bottom up; starts with individual	Middle out; starts with community
Theory of Change	Change individual decisions, remove barriers, and provide benefits; aggregate choices and actions; replicate programs across scale	Reconfigure systems in place around mutualistic community-based, socio-ecological goals; diffuse change through social networks and system structures across dimensions
Key processes	Cognitive states and preferences; individual decision-making, costs, risks, and incentives; barriers and benefits of specific behaviors	Processes of complex, distributed, networked systems; social; agentic; multi-level governance; disruption, transition, and reconfiguration
Primary Methods	Provide incentives, information, and education; enact financial policy; change attitudes; increase awareness	Activate social processes; build communities of practice; restructure systems; empower catalysts; connect across dimensions
Durability	Requires sustained external financial inputs across space and time; permanent attitude change	Requires building and sustaining mutually reinforcing feedbacks within and across dimensions

Acting collectively for the common good is a key to the long-term evolutionary success of humans in the past and today (Amel et al., 2017; Boyd & Richerson, 2009; Henrich, 2016; Ostrom, 2000). Systems that encourage rather than discourage collective action must be a priority for society. The expansion of collaborative conservation over the past 20 years (Koontz et al., 2020; Lubell et al., 2002; Wilkins et al., 2021) represents a bright spot for change in conservation, but the success of these efforts likely is undermined by the dominant cultural paradigm that imagines a private society in which individuals are independent, competitive choosers who seek to maximize utility and resources for themselves and their families but not for others or broader society (Hussain, 2018; Rawls, 1999). This implies that significant change to dominant social systems and the mental models that guide them may be necessary for an integrative socio-systemic theory to flourish. Some signs of societal shifts are underway, such as efforts to democratize the economy (Bell, 2015; Hewlett Foundation, 2020; Johanisova & Wolf, 2012; Kelly & Howard, 2019). Rather than just adapting to these societal shifts, conservationists could use disruptions as opportunities to stimulate the bigger, more transformational changes needed and lead society by providing vision, substance, and tools to make the shift as rapid as possible.

CHAPTER SIX: CONCLUSION

Transformational change is dramatic change that requires the integration of existing theories that incorporate agentic, social, and structural influences across social, spatial, and temporal dimensions. Micro-scale theories like those explored in Chapter 2 are useful to understand the relationships between cognitive processes and behavior, but they do not address social or structural influences. The lack of integration of multiple theoretical perspectives into the conservation profession is related to the pervasive adoption of an individual, neoliberal cultural bias that influences the theories deemed legitimate. This bias crowds out social and systemic perspectives. Chapter 3 is a case study of a neoliberal approach to conservation that failed because of its lack of attention to broader socio-political concerns. Chapter 4 describes how these broader concerns have been crowded out through a bias toward individual, agentic perspectives in conservation science and practice.

To remedy this problem, the conservation profession will need to better integrate multiple theories across multiple dimensions. The TTC proposed in Chapter 5 is intended to provide a framework for doing so, and for understanding and developing specific strategic change interventions. The proposed TTC represents a significant shift in the conservation profession away from the ideology of attitude-behavior-choice to a science-based, multidimensional view of socio-systemic change (Virapongse et al., 2016; Figure 12). Its adoption would significantly alter what it means to practice conservation and would greatly expand the available conservation toolbox. Rather than asking how individual behaviors can be changed and then scaled up, conservationists would ask how systems can be reconfigured in place and diffused across dimensions. This orientation focuses conservation on changing the conditions in which behavioral patterns develop rather than changing specific behaviors. Its focus on social and systemic theories of change represents a shift toward a more proactive, collaborative, democratic, approach that has potential to foster more resilient and durable solutions. It is more proactive because it focuses on interventions that address root causes that influence conditions that exist prior to action, not correcting (e.g., through punishments or rewards at the individual level) behavior that is already been encouraged by the system. It is more democratic because it relies on meaningful collaborative, local involvement of a wide range of stakeholders which is likely to improve local acceptance, legitimacy, and durability (Horning et al., 2016). Finally, decentralized, locally diverse, and networked solutions are likely to be more adaptive and resilient to disruption (Biggs et al., 2010)

Acting collectively for the common good is a key to the long-term evolutionary success of humans in the past and today (Amel et al., 2017; Henrich, 2016; Boyd & Richerson, 2009; Ostrom, 2000). Systems that encourage rather than discourage collective action must be a priority for society. The expansion of collaborative conservation over the past 20 years (Koontz et al., 2020; Lubell et al., 2002) represents a bright spot for change in conservation, but the success of these efforts is likely undermined by the dominant cultural paradigm that imagines a private society where individuals are viewed as independent, competitive choosers who seek to maximize utility and resources for themselves and their families, but not of others or broader society (Hussain, 2018; Rawls, 1999). The emphasis placed on individualistic approaches undermines collective action. If we define conservation as collective human action for the common good, then conservation is a counterculture endeavor in many places. This implies that significant change to dominant social systems and the mental models that guide them may be necessary for an integrative socio-systemic theory to flourish. Some signs of these societal shifts are underway (Hewlett Foundation, 2020; Zabel, 2020; Kelly & Howard, 2019; Bell, 2015; Johanisova & Wolf, 2012). Rather than just adapting to these societal shifts, conservationists could use disruptions to it as opportunities to stimulate the bigger, more transformational changes needed, and lead society by providing vision, substance, and tools to make the shift as rapid as possible.

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