This is the accepted manuscript version of the contribution published as:

Anderies, J.M., Cumming, G.S., Clements, H.S., Lade, S.J., **Seppelt, R.**, Chawla, S., **Müller, B.** (2022):
A framework for conceptualizing and modeling social-ecological systems for conservation research
Biol. Conserv. **275** , art. 1097693

The publisher's version is available at:

http://dx.doi.org/10.1016/j.biocon.2022.109769

A framework for conceptualizing and modeling social-ecological systems for conservation research

John M. Anderies[*] Graeme S. Cumming[†] Hayley S. Clements[‡] Steven J. Lade[§]
Ralf Seppelt[¶] Sivee Chawla[‖] Birgit Müller[**]

July 18, 2022

Acknowledgments. This work was supported by the National Socio-Environmental Synthesis Center (SESYNC) of the U.S.A., under funding received from the National Science Foundation DBI-1639145. HSC acknowledges funding from a Jennifer Ward Oppenheimer Research Grant and Kone Foundation.

[*]School of Human Evolution and Social Change and School of Sustainability, Arizona State University, Tempe AZ, USA

[†]ARC Centre of Excellence for Coral Reef Studies, James Cook University, Townsville 4811, Australia

[‡]Centre for Sustainability Transitions, Stellenbosch University, South Africa; Helsinki Lab of Interdisciplinary Conservation Science, Department of Geosciences and Geography, University of Helsinki, Finland

[§]Stockholm Resilience Centre, Stockholm University, Stockholm, Sweden and Fenner School of Environment & Society, Australian National University, Canberra, Australia

[¶]Helmholtz Centre for Environmental Research - UFZ, Department of Computational Landscape Ecology, Leipzig, Germany and Institute of Geoscience & Geography, Martin-Luther-University Halle-Wittenberg, Halle (Saale), Germany

[‖]ARC Centre of Excellence for Coral Reef Studies, James Cook University, Townsville 4811, Australia

[**]Helmholtz Centre for Environmental Research –UFZ, Department of Ecological Modelling, Leipzig, Germany

Abstract

As conservation biology has matured, its scope has expanded from a primarily ecological focus to recognition that nearly all conservation problems involve people. At the same time, conservation actions have been increasingly informed by ever more sophisticated quantitative models. These models have focused primarily on ecological and geographic elements of conservation problems, such as mark-recapture methods, predicting species occurrences, and optimizing the placement of protected areas. There are many off-the-shelf ecological models for conservation managers to draw upon, but very few that describe human-nature interactions in a generalizable manner. We address this gap by proposing a minimalistic modeling framework for human-nature interactions, combining well-established ideas in economics and social sciences (grounded in Ostrom's social-ecological systems framework) and accepted ecological models. Our approach begins with a systems breakdown consisting of an ecosystem, resource users, public infrastructure, and infrastructure providers; and interactions between these system elements, which bring together the biophysical context, the relevant attributes of the human society, and the rules (institutions, such as protected areas) currently in use. We briefly review the different disciplinary building blocks that the framework could incorporate and then illustrate our approach with two examples: a detailed analysis of the social-ecological dynamics involved in managing South African protected areas and a more theoretical analysis of a general system. We conclude with further discussion of the urgent need in conservation biology for models that are genuinely designed to capture the complexities of human socioeconomic behaviour, rather than the more typical approach of trying to adapt an ecological model or a stochastic process to simulate human agency and decision-making. Our framework offers a relatively simple but highly versatile way of specifying social-ecological models that will help conservation biologists better represent critical linkages between social and ecological processes when modeling social-ecological dynamics.

Introduction

Humans are now the primary driver of global environmental change (Vitousek et al., 1997; Rockström et al., 2009). In recent years, the several-fold increase in production of renewable resources and associated land use change have become the primary drivers for biodiversity decline (Foley et al., 2005; Green et al., 2005; Seppelt et al., 2014). As a result, applied ecology and conservation biology have become increasingly aware of the importance of understanding the role of people in conservation and resource management problems. The conceptual growth trajectory of conservation biology, from pure ecology to a more inclusive and interdisciplinary perspective (Mace, 2014), can be tracked through the kinds of quantitative analyses undertaken in the name of conservation. Early models focused primarily on ecological dynamics, addressing such questions as acceptable off-take rates and carrying capacities for harvested wild populations; home range sizes and habitat requirements of animal species; and vegetation successional dynamics in relation to the management of fire, floods, and other perturbations (Starfield and Bleloch, 1986). In many of these models, people were seen as external to the problem, often viewed in a god-like role as 'the manager' supposedly responsible for undertaking actions (e.g., culling, burning, land clearing, restocking) recommended by the model. Later attempts to develop more nuanced approaches (e.g., 'management strategy evaluation', or MSE, as proposed by Bunnefeld et al. (2011)) have often retained a focus on the ecosystem as the primary component of conservation models. MSE, for instance, recognizes that 'harvesters' may make independent decisions but still treats managers as external agents who dictate rules to harvesters. Others (e.g. Daw et al., 2015) have recognized the complexity of social-ecological dynamics, but adopted weaker quantitative approaches to model development.

A series of conservation failures, or partial successes, alerted conservation biologists to the need for a more holistic world view in analyses of management problems (e.g. Larkin, 1977). The late 1990s and early 2000s saw a backlash against simple 'command and control' perspectives and a greater focus on complexity and system dynamics (Holling and Meffe, 1996; Ludwig, 2001; Holling, 2001). These developments were reflected in a series of more sophisticated models that tried to include both social and ecological dynamics (Carpenter et al., 1999). However, most conservation models used simple rules for social system elements and decisions and minor modifications of standard ecological models (e.g., logistic growth curves, metapopulation models, and Lotka-Volterra predator-prey models) to reflect human impacts on ecosystems.

Increasing awareness of the need for better ways to include people and institutions in models of ecological management led to the entry into conservation biology of some parallel developments from the social sciences. Early development of modeling in the social sciences was largely centered in economics around general equilibrium models, repeated games, and dynamic optimization. Although relevant to conservation, e.g. understanding cooperative behavior and the optimal inter-temporal management of living populations, these models relied on techniques that made them difficult to generalize and communicate across different research communities. The development of multi-agent modeling based on repeated games to understand the evolution off cooperation (Axelrod, 1984) and the notion of complex adaptive systems (Holland, 1992, 1995) lead to a rapid expansion of modeling to other social science disciplines, e.g. evolutionary anthropology in particular. Prior to 2007, mathematical modeling in the social sciences with relevance to conservation primarily occurred via multi-agent models and social network analysis to explore the ways in which fundamental assumptions about how people interact to collaborate or compete for resources influenced conservation outcomes (e.g. Barreteau et al., 2001; Bousquet and Le Page, 2004). Social network analysis was also used to shed light on the relationships between social structure and conservation-relevant processes, such as the creation of institutions (Schneider et al., 2003) and the success of adaptive management (Bodin and Norberg, 2005; Crona and Bodin, 2006). However, these approaches often did not interface readily with standard conservation models, and challenges (many of which are still unresolved) arose in capturing individual processes of cognition and decision-making in more realistic ways (Schlüter et al., 2019a,b).

The publication of Ostrom's Diagnostic Framework (2007) highlighted the importance of institutions (broadly defined as rules, laws, customs, and traditions) as an important missing element in social-ecological models. Ostrom's framework provided a much stronger focus on the dynamic feedbacks between ecosystems, human actions, regulatory frameworks, and politics. However, challenges in translating from Ostrom's lists of relevant variables to dynamic processes have resulted in relatively poor uptake of Ostrom's social-ecological systems (SES) framework into the models used in conservation biology. From a conservation biology perspective, the language of existing frameworks for modeling social-ecological systems is rooted in political science and economics. It is based around such examples as Ostrom and Kiser's Institutional Analysis and Development Framework (Kiser and Ostrom, 1982); Ostrom's 'SES' Framework (Ostrom, 2009)); game theory such as Maskin's notion of (policy) mechanism design (Maskin, 2008); Long's (1958) Ecology Of Games Framework and Lubell's (2013) extension of it; and resource economics, such as standard dynamic optimization models that underlie notions of maximum sustainable or economic yield (Clark, 2010; Gordon, 1954). Deploying these frameworks requires significant investment and worked examples relevant to applied conservation problems are few. This limits their usage and appeal in conservation science. At present, in conservation biology, the study of social-ecological systems is characterized by a plethora of frameworks and relatively abstract models on the one hand; and localized, highly detailed models of individual case studies on the other. Unsurprisingly, this situation is not conducive to theoretical development and general advances in the field.

We regard this trend as 'disturbing' because purely ecological models are fundamentally incapable of capturing the dynamics of human social processes and how they interact with ecological systems. People and human societies are different from other organisms and ecological communities in many ways. For example, people act both cooperatively and competitively at many different scales and levels of social organization; they show intentionality, bias, and reflexivity (i.e., they act to achieve broader goals, based on their world views and values, and respond to predictions); and they build infrastructure and use technologies that alter their relationships to each other and to the natural world. These differences mean that the basic assumptions of many ecological models are untenable when applied to people. In our view, truly social-ecological models should include key elements of human uniqueness, not ignore it.

At the same time, from an ecological perspective, models of human behavior in economics and sociology often ignore critical ecological dynamics. Ostrom's categorization of ecosystems as 'resource providers' downplays the importance of ecosystems as life support systems (Epstein et al., 2013; Seppelt et al., 2020); and successful resolution of many conservation problems hinges on a deep understanding of complex ecological dynamics, such as processes of colonization, succession, and species turnover (Pereira and Daily, 2006). Thus, social-ecological models, whether conceptual or quantitative, should capture the fact that 'management' or more appropriately 'governance' is not something we 'do' but, rather, refers to social structures in the form of norms and formal rules that *emerge* from the interactions among people and between people and the environment (Schill et al., 2019). A wider, more established between-ground that bridges different disciplines is thus needed in order for social-ecological models to fulfill their true potential.

Social-ecological systems research in general and conservation research in particular currently lacks a widely accepted general, unified conceptual and formal modeling framework that encompasses the social, biophysical, and technological dimensions of SES dynamics and that transcends specific interest areas (particular species, ecosystem types, etc.), modeling approaches (e.g. ordinary differential or difference equations, individual/agent-based, stochastic processes, age/stage structured models), and analytical techniques (e.g. simulation, stability, viability, and bifurcation analyses, dynamic programming, optimization, and optimal control). We propose such a modeling framework derived from a combination of ideas from ecology (Mangel and Clark, 1988; Clark et al., 2000; DeAngelis, 2018), economics (Beltratti, 1997; Stiglitz, 1974; Grossman and Krueger, 1995; Anderies, 2003), political science (Ostrom, 2009; Anderies et al., 2004; Hinkel et al., 2014; Anderies et al., 2019a), bioeconomics (Clark, 2010; Gordon, 1954, 1953; Brander and Taylor, 1998; Sanchirico and Wilen, 2001; White and Wadsworth, 1994) and multi-agent systems (Lynam et al., 2002; Parker et al., 2003; Janssen, 2002). The framework can be combined with ideas from other relevant disciplines (geography, planning, architecture, engineering, policy, law) as part of a 'toolkit' to serve as a guide to develop conceptual models and translate them into formal dynamic models for SESs. The framework provides a "simple but not too simple" starting point for modeling any social-ecological system and offers social-ecological researchers an entry point for an 'off the shelf' approach that can be used and adapted to fit a given context or question. The remainder of this paper first lays the theoretical foundations for the framework including situating it in the constellation of existing frameworks. It then presents the framework itself, disciplinary building blocks for operationalizing the framework, and two use cases for conceptual and formal model development, respectively.

Theoretical foundations for SES model development

Concerns about over-exploitation of natural resources go back at least to Malthus (1798) and Ricardo (1817) with their work on agricultural production and human population dynamics. More recent work focused on theories of common-pool resource management (Gordon, 1954, 1953; Schaefer, 1957) while raising interest in sustainability which were made more dynamic and mathematically rigorous by Clark's (1976; 2010) work on Bioeconomics. It is no coincidence that concerns about species loss and the rising impacts of human population growth on ecosystems were formalized in the field of conservation biology at around the same time, being described as a 'new synthetic discipline' by Soulé (1985). While conservation biology saw itself initially as the mission-oriented application of ecological theory to 'saving biodiversity,' parallel research by mathematicians and ecologists in the 1970s and 80s (e.g. Holling, 1973; Westoby et al., 1989; Ludwig et al., 1997) on system-level concepts such as resilience led to deeper engagement in natural resource management by social scientists (e.g. Berkes et al., 2003) who were interested in the dynamic interactions of social and ecological processes. Subsequent research in a range of disciplines connected ideas from political science, policy studies, and ecology (e.g. Ostrom, 2007; Anderies et al., 2004) leading to a general framework for analyzing social-ecological systems (Ostrom, 2009) which is now strongly associated with the terms 'SES Framework' and 'SES Theory'. Although conservation biology has undertaken a 'social turn' in recent years, with some notable exceptions (e.g. Cinner et al., 2022; Büscher and Fletcher, 2019) many analyses in conservation biology remain naïve about social and economic theory and associated best practices (Bennett et al., 2017).

There are several 'frameworks' for studying SESs, and the terms 'framework', 'theory' and 'model' are often conflated. Frameworks refer to core sets of conceptual elements and the general relationships between them required to frame problems in particular research domain. Theories add layers of specificity and assumptions about the nature of the core elements and relationships; while models specify formal representations of these elements and relationships. For problems that involve groups of people interacting with natural resources (i.e., the majority of problems encountered by conservation biology), there are at least six directly relevant frameworks: the Institutional Analysis and Development (IAD) Framework (Kiser and Ostrom, 1982), the SES Framework (Ostrom, 2009), the Robustness of SES Framework (Anderies et al., 2004), the Advocacy Coalition Framework (Sabatier, 1988), the Ecology of Games Framework (Long, 1958; Lubell, 2013), and the Human Ecology Framework (Dyball and Newell, 2014). All of these frameworks are well-suited to addressing specific types of questions and can be used, at various levels of generality and depending on the research question, to guide formal model development. As such, modeling practitioners in the conservation space may benefit from an awareness of these frameworks. Interested readers may refer to (Anderies et al., 2019a) for a more detailed comparison of the various frameworks.

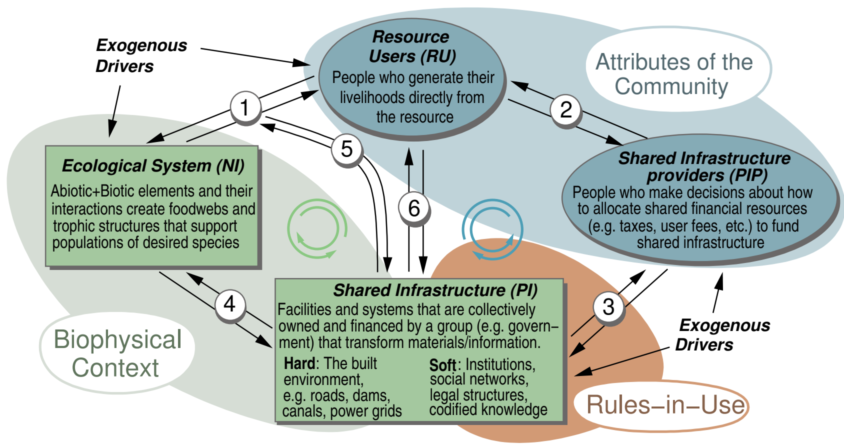
A SES modeling framework should transcend particular questions and application contexts and provide for the representation of key features of any SES. The essential element in any SES with persistent structure (e.g., an ecological community and its human dependents) is feedback (Csete and Doyle, 2002; Carlson and Doyle, 2002). In SESs, these feedbacks take the form of information-action loops wherein human individuals or groups extract information about the state of a system (e.g., an ecosystem), decide how to act on the system (e.g., which species to protect and which to harvest), and undertake the action, generating a response from the ecosystem (e.g., changing population size or distribution), that over time triggers system change and restarts the cycle (loop) (Anderies et al., 2007, 2019b).

Because of the critical importance of social-ecological feedbacks in natural resource management and conservation, we focus on the Robustness of SES framework. This framework has grown over the years to incorporate the built environment consisting of several human-made infrastructures present in most, if not all, SESs such as canals, roads, fences, communication systems, etc. SESs are thus seen as a subset of a broader class of systems known as coupled infrastructure systems (CIS) as shown in Figure 1 and referred to as the CIS Framework. The IAD framework also captures the notion of feedbacks but it is not rich enough to act as a framework for a minimal model. The SES framework formalizes the interactions between key elements in SESs, but potential pathways through which feedbacks operate are not explicit. As a result, the importance of feedbacks is lost; while the SES Framework provides an ontology for categorizing SESs, it is not useful for building a dynamic model.

The CIS Framework, shown in Figure 1 describes SESs/CISs using four key elements:

- **Resource users**. The actors who derive benefits (livelihoods) directly from the natural infrastructure by extracting resources (e.g. land system). Many commonly studied resource users derive material flows (e.g., fish, food, fiber). Others may derive benefits less directly, such as ecotourism operators manager whose livelihood depends on visitors. Note that actors who consume products extracted from a resource system are not necessarily treated as resource users in the framework. For example, consumers of salmon purchased in a market represent an exogenous driver on resource users through market demand. Abbreviated as "RU" below.

- **Ecological System**. The landscapes and ecosystems (biotic and abiotic entities and their interactions, such as animals, water, soils, and fish production) that support life and generate benefits to people.



Element	Relevant Disciplinary Knoweldge
RU	Psychology, Sociology, Economics
PIP	Psychology, Sociology, Economics
PI	Engineering, Public Policy, Historical Analysis
NI	Ecology, Hydrology, Climate Science, Conservation Biology
Link 1	Bioeconomics, Natural Resource Economics, Human Geography
Link 2	Political Science, Political Ecology
Link 3	Public Policy, Planning
Link 4	Conservation, Civil Engineering, Environmental Sociology, Human Geography
Link 5	Human Geography, Resource Management
Link 6	Behavioral Economics, Public Policy, Human Geography
Exogenous Drivers	
NI	Climate Science
RU/PIP/PI	Disaster and Risk Management

Figure 1: The CIS Framework. The framework depicts the fundamental elements and interactions (arrows 1-6) that comprise any 'Coupled Infrastructure System' of which SESs and by definition, all conservation problems, are special cases. The relationship to the elements of the IAD Framework - the biophysical context (living and built environments) and the attributes of the community (our social worlds) encompass elements of the CIS-Framework as shown. The rules-in-use (institutions) connect the social and biophysical domains. Finally, the embedded table shows various academic traditions that may inform the conceptual and formal modeling of elements and links in the CIS Framework. Note that "conservation" appears in two places; near the Ecological System element and along link 4. The former reflects more traditional approaches to biodiversity and ecosystem function. The latter refers to specific species conservation infrastructure such as game preserves, and wildlife corridors. The CIS Framework emphasizes that the whole system must be grappled with if one wants to stop biodiversity loss and restore degraded ecosystems. See Table 1 for further details on relevant literature.

This element also incorporates structures and processes that support ecosystem functions and human wellbeing, such as nitrogen cycling, predation, storm or flood protection, or religious and spiritual values. In various iterations of the CIS Framework this system element is also termed 'natural infrastructure'; and in Ostrom's writings, the 'resource system'. Note, the observation above that natural systems are used in various ways other than for 'resources' is the basis for referring to the 'resource system' as 'natural infrastructure' as the later allows for generalizing the way we 'use' the natural world. In the same way a fisher uses a hook (hard infrastructure) to catch a fish, ecotourism operators use ecosystems to 'catch' tourists. This is the motivation behind shifting from thinking about social-ecological systems to coupled infrastructure systems (CIS). Abbreviated as "NI" below.

- **Public (shared) infrastructure**. Facilities and systems accessible for public use which affect resource users (link 6), resource systems (link 4) or interactions between them (link 5) and are owned and operated by groups. *Hard infrastructure* are tangible objects such as roads, dams, and machinery. It may also include 'green infrastructure' that has been deliberately cultivated by people within built environments, such as shade trees or picnic lawns. *Soft infrastructure* includes non-tangibles that serve to structure social and economic systems, such as institutions, legal systems, culture, and social networks. Abbreviated as "PI" below.

- **Public (shared) infrastructure providers**. The agents who participate in governance, such as governments, NGOs, or community organizations, and who make collective decisions about resource allocation to public infrastructure. Providers may be individuals or entire organizations. Abbreviated as "PIP" below.

These four key elements of a generic SES/CIS are connected in the framework by six different links that represent material, energy, and information transfers between elements. For example, through link one, fishers (resource users) might exert effort (energy) and extract biomass (material) from the Ecological System. Along link 4, a wildlife conservation agency might exert effort to monitor a hunted species (part of the Ecological System), extracting information that it uses to restrict hunting activity through link 5 or hunting season length in link 6.

Consideration of the links between different elements of the framework provides a basis for thinking about dynamic feedbacks within an SES/CIS and their relevance for effective and sustainable management. If the different links between system elements are broken or dysfunctional, or if critical processes are missing within a system element (e.g., keystone species are lost from the Ecological System or decision-making processes are stalled by political rivalries) the functioning of the entire system will be affected. Similarly, errors and uncertainty can easily propagate through the system if information is incorrect or only partially correct. There are 4 prominent types of feedbacks shown in Figure 1 that are essential to understanding the governance of shared resources in general and solving conservation problems in particular. The green loops on the left represent the 'management' or 'operational' feedbacks. The counter clockwise loop represents traditional, formal environmental policy and resource management processes. The clockwise loop represents 'co-management', where information may flow in the other direction. On the right, the blue loops represent the political economy, i.e. collective action and joint decision-making. The outer, clockwise loop represents standard political and public investment processes while the inner, again, represents co-management. The extent to which the 4 types of feedbacks are operating and the extent to which they are coupled, i.e. form a *feedback network*, varies widely with context. The question of 'effective conservation' or 'good governance' boils down to the function of these feedback processes.

Disciplinary building blocks for operationalizing the CIS Framework

The CIS Framework draws attention to the need to consider all elements and links in model development. In this process, careful arguments for what details of links and elements are included and excluded are made. For example, conservation models frequently model the ecosystem in great detail, may include rudimentary descriptions of the resource user, but typically omit public infrastructure, especially soft infrastructure such as institutions, and any providers of that public infrastructure. We propose that the development of any social-ecological model should at least consider the inclusion of all elements listed by the CIS/Robustness framework and justify any omissions. We are not claiming that social-ecological models must necessarily include more complicated descriptions of the social components of the SES/CIS than the ecological components. The key is that while this choice ultimately depends on the research questions being studied, the CIS Framework can be used to systematically provide a rationale for such choices. The modular structure of the CIS Frameworks allows system elements to be readily included or omitted as appropriate.

There are several published examples that illustrate how the CIS Framework can be used to analyze a particular case (Cifdaloz et al., 2010; del Mar Mancha-Cisneros et al., 2018; Tellman et al., 2018), conduct a comparative case-study analysis (Therville et al., 2019), help develop a game for stakeholder engagement (Bonté et al., 2019), or develop mathematical models (Muneepeerakul and Anderies, 2017, 2020). Although these studies provide examples of the CIS Framework in action, they are not necessarily a useful guide to applying the framework without some additional conceptual background. Thus, before presenting worked examples, we will map disciplinary building blocks onto the components of the CIS Framework.

When developing a social-ecological model, it is important to leverage established knowledge for each component (Figure 1 and Table 1). For example, we see far too often a supposedly social-ecological model being constructed by simply renaming the 'predator' in an ecological model as a human actor. Different disciplines have different traditions and degrees of acceptance of dynamical modeling. In ecology, mathematical modeling is commonplace and there are well-accepted mathematical building blocks, such as the Lotka-Volterra model and its many variations for predator-prey interactions or the logistic growth model for population dynamics. Mathematical modeling in economics is also extremely well-developed, both in general (e.g. consumer and firm behavior, general equilibrium) and in the specific case of natural resources (e.g. Gordon, 1954; Clark, 2010, 1976). Mathematical bioeconomic treatments of resource management also frequently have limitations, most notably around assumptions of perfect rationality required in mathematical representations of economic decision making. Overcoming these limitations is a key motivation for recent developments in agent-based modeling.

In other academic disciplines relevant to building models of SES such as anthropology, human geography, psychology, sociology, political ecology, and history, formal mathematical modeling is less prominent and relevant knowledge exists in the form of a diversity of theories and basic principles which guide quantitative analyses. It is nonetheless important to incorporate key insights from these fields into formally modeled relationships whenever possible. In data-rich settings, relationships required to build the model could also be estimated systematically. However, the theoretical building blocks often form the basis for empirically-estimated or participant-constructed model components and their role should not be underestimated.

The academic traditions shown in Figure 1 inform our understanding of the elements and links in the CIS and provide its backbone for modeling and analysis. Considering each of the different links, bioeconomic models (e.g. Gordon, 1953; Clark, 1976, 2010) have been used extensively to model effort allocation decisions of resource users along link 1 counter clockwise (CCW); how much biomass extraction will result along link 1 clockwise (CW); and whether this level of extraction results in biological or economic over-exploitation based on endogenous ecological dynamics. Such simple models of decision making and ecological dynamics (Table 1) should be understood as starting points for building richer models for specific contexts; for instance, humans do not behave as *Homo-economicus* as most bioeconomic models assume, particularly when there is uncertainty (e.g. Kahneman and Tversky, 1979), or in social contexts (e.g. Gintis et al., 2005, 2008; Lieberman, 2013; Ostrom, 1998) that are easily influenced by framing effects or a lack self-control (e.g. Thaler, 1980; Thaler and Sunstein, 2009). Social structure and socioeconomic context (e.g. Barnes et al., 2016; Clausen and York, 2008; Cinner et al., 2009) also affect how individuals interact with the resource (link 6 $\rightarrow$ RU $\rightarrow$ link 1). Thus, psychology, behavioral economics, sociology, and political science all provide valuable insights for models of decision making (effort allocation - link 1) and how decision-making may respond to policy (link 6 CCW; see Schill et al. (2019) for further details). Similarly, human geography provides knowledge of how resource appropriation relates to shared infrastructure along links 5 and 6 CCW and indirect effects of infrastructure on the resource (e.g. distance to markets) along link 4 (e.g. Cinner et al., 2022; Epstein et al., 2021).

Just as simple human decision-making models do not consider social contexts and the complexity of behavior, the logistic growth models used to represent the 'ecology' in bioeconomic models typically do not consider space or community-level interspecific interactions. Concepts from ecology, hydrology, and earth system science provide guidance for increasing the representativeness of ecological models. Link 4 is seldom considered in policy models. This link represents how shared infrastructure (e.g. conservation/resource management agencies, transportation departments, etc.) measures/monitors (e.g. population surveys) and modifies landscapes (builds dams and canals, installs power lines, etc.). These activities are expensive and have important implications for ecological dynamics (e.g., species movement, hydrological processes) and, as a result, the long-term trajectories of landscapes (Cumming and Epstein, 2020). Modeling this link can be informed by basic principles in conservation (i.e. meta-population models, species area relationship models) and engineering. Not only must agencies monitor natural infrastructure states, they must monitor resource users' activities and sanction appropriately. Monitoring and sanctioning, i.e. (non-)compliance, is a central issue in resource governance and is particularly important in conservation contexts (Epstein et al., 2021; Arias, 2015; Solomon et al., 2015; Keane et al., 2008; Epstein et al., 2021). Compliance issues are captured in link 5 where the PI monitors RU interactions with the resource and enforces sanctions (e.g. fines) through link 6. Compliance can also be affected through link 4 wherein PI modifies flows in the NI directly through, for example, fencing or security cameras. Link 3, which captures the negotiation between those who allocate shared financial resources (taxes, user fees, etc.) and actors who build (private contractors), operate (government and private sector workers), and maintain shared infrastructure is also seldom captured in 'social-ecological systems' models. Engineering, planning, economics, and public policy provide guidance for modeling this interaction.

Link 2 has probably received the most attention in the literature on SESs. This link represents collective action (CW) and power relations (CCW) among communities in relation to the group that in some way represents their interest through producing shared infrastructure. The infrastructure provider group can be equal to, a subset of, or distinct from the resource user community. The literature on collective action and the evolution of cooperation in the context of shared resources is vast. Ostrom's Nobel Prize winning work *Governing the Commons* (1990) is the most well-known work in this area. Similarly, power relations have received a lot of attention in the literature. Power relations appear within the RU, PIP, and PI elements and across links 2, 3, and 6. The framework can help think about critical power relations in a given system in broad terms but mathematical modeling of endogenous power relations is a significant research challenge; it is difficult to avoid building power asymmetries into the model. While the details of the elements and links are important, there are two general features of the CIS Framework. First, note that the climate drives (exogenous driver to NI) all finer-scale systems, such as a conservation area, fishery, forest, or watershed, both in terms of its endogenous dynamics and the risk portfolio it faces (exogenous drivers to RU, PIP, and PI). Second, the evolution of large shared infrastructure systems is slow. It can take generations for decisions in a given CIS, e.g. the ecological impacts of dam construction (Kingsford, 2000) or the recovery of over-exploited fisheries (Phillips et al., 2022; Marschoff et al., 2012), to play out. Thus, historical examples are critical to consider when developing models.

Note that we are not suggesting any particular balance of levels of complexity in the social or ecological components in models of SES. Rather we here show that for each of the elements and feedbacks in the CIS Framework, various disciplines provide building blocks for quantitative analysis and modeling and consequently provide a multidisciplinary and modular approach. Neither is Table 1 intended as a systematic review, it is merely illustrative of key literatures where useful modeling building blocks have been developed. Finally, we do not argue that all elements need to be fully developed to include dynamic interactions. But given the research question at hand, our approach allows researchers to follow a roadmap to decide and sufficiently discuss which potential feedbacks in the CIS Framework should be developed and which can be safely neglected for a particular problem. We provide an illustration of implementing this roadmap in the following worked examples.

Example applications of the CIS Framework

Is discussed throughout this paper, the CIS Framework has been used in a number of modalities ranging from comparative analysis to conceptualizing to building formal mathematical models of SES. Here we illustrate its use in two modalities: conceptualization and formal model building. Of course, the first is a necessary step for the second and the CIS helps guide both steps. Analysis is the final step in any modeling exercise and, as the name suggests, the framework was originally developed to guide the analysis of robustness in SES. The framework does not dictate the type of analysis conducted and details of model analysis is beyond the scope of this paper (but see, e.g. Anderies, 2006; Cifdaloz et al., 2010; Homayounfar et al., 2018; Muneepeerakul and Anderies, 2020, for examples of robustness analysis). Rather, the examples show how the Framework helps uncover key feedbacks at the conceptual level, how the Framework elements and linkages translate to formal model structures, and how insights from various fields are used to enrich models. The first case illustrates the CIS in the conceptualization modality to dissect a real-world conservation example in southern Africa to identify and characterize critical feedback processes that would be essential

CIS Element: Natural Infrastructure - Land, Marine, Climate Systems

Ecology (Original work: Verhulst 1845, 1847, Lotka 1925, Volterra 1926. Modern guide: Jörgensen et al. 2001)

Building Block	**Description**	**Relevance**
Logistic growth: $\frac{dp}{dt} = rp\left(1 - \frac{p}{K}\right)$	Density dependent population ($p(t)$) growth with limiting resources (K).	Minimal biologically reasonable model for harvestable natural populations.
Lotka-Volterra - (linear version): $\dot{x} = rx - \alpha yx$ $\dot{y} = \beta yx - dy$	Predator-Prey/trophic interactions. Linear model can be generalized to any species interactions and any level of ecosystem complexity if x and y are vectors, i.e. $x \in \Re^m$, $y \in \Re^n$.	Standard model for multi-trophic species interactions. Generalizable as $\dot{x} = f(x, y)$, $x \in \Re^m$ $\dot{y} = g(x, y)$, $y \in \Re^n$

Conservation (Original work: Arrhenius 1921. Recent work: Pereira & Daily 2006)

Species-area curves $S = c\left(\sum_j h_j A_j\right)^z$	Estimates number of species S per patch, considers habitat/patch h_j quality	Indicator of ecological stability and functioning, used to set conservation goals.

CIS Element: Public Infrastructure, Links 1-4-5/6 Management Feedback Structures (green loops, Figure 1)

Bioeconomics (Gordon 1953; 1954, Schaefer 1957, Clark 1976; 1990)

Building Block	**Description**	**Relevance**
Gordon-Schaefer $\dot{x} = f(x) - h(x, e)$	Generic model of renewable resource biomass ($x(t)$) growth ($f(x)$) and harvesting ($h(x, e)$). Lotka-Volterra with human 'predators'.	Fundamental model of common-pool resource management problems. Effort, $e(t)$, often treated as a lumped stock variable.

Geography, Planning, Civil Engineering (Tobler 1970, Von Thünen, 1966)

$\gamma(h) = \frac{\sum_{i,j}(x_i - x_j)^2}{2N(h)}$	Tobler's 1st law: everything is related to everything else, but near things more so.	Tobler's 1st law is mathematically translated into geo-spatial correlation measures.
$R = R_0 - \gamma\|z\|$	Thünen's location theory of spatial organization of land use surrounding a central market.	Land rent R is a function of the distance $\|z\|$ to this central market R_0.

CIS Element: Public Infrastructure, Links 2-3-6 Social and Political Feedback Structures (blue loops, Figure 1)

Political Science (Black 1948, Liggett 1985, Ostrom 1990)

Building Block	**Description**	**Relevance**
Median Voter Model Voter models, Social Choice Theory	Generic models of opinion dynamics and social choice based on probabilistic, and game-theoretic models.	Provides a foundation for modeling behavior of voters and politicians in collective choice situations.

CIS Element(s): Resource Users and Public Infrastructure Providers

Behavioural Economics (Schiller 2005, Brown 2010, Schlüter et al. 2017, Schill et al. 2019)

Building Block	**Description**	**Relevance**
Variations on e $e_i = g(\hat{x}_i, \hat{e}_o, c)$ $\dot{s}_i = s_i(\pi(s_i) - \phi)$	Models of 'action' based on bounded rationality ($\hat{x}_i$), other-regarding preferences ($\hat{e}_o$), cultural context (c). Strongly game-theoretic with frequent use of replicator dynamics.	Allows for the modeling of decisions where agents rely on estimates/perceptions of state variables and incorporate others actions and social context, norms, etc.

Psychology, Cognitive and Decision Sciences (Raiffa 1968, Stillings et al. 1995, Thagard 2005, Kolkman et al. 2005)

Planned behavior, mental models $e_i = g(\hat{x}_i, \hat{e}_o, c, b)$	Similar to behavioral economics but more focused on information processing and beliefs (b) and less on strategic behavior.	Foundation for understanding how knowledge, data, learning, and cognitive processes impact individual decision making.

Table 1: Examples of relevant models or conceptual underpinnings from relevant disciplines that may serve as building-blocks for more elaborate models of social-ecological systems.

to include in a mathematical model. The next step of translating the resulting conceptual model into a formal mathematical model is detailed in the Supplementary Materials. The second example illustrates the Framework used in a formal model building mode and focuses on the classic natural resource management problem and illustrates how to move from the naive Gordon-Schaefer model to models with richer behavioral, market, and political contexts. Again, additional details and analysis is provided in the Supplementary Materials.

Multi-tenure conservation areas in southern Africa

It is increasingly recognized that protected areas (PAs) are social-ecological systems. They are created by people, for people, and interact with biological, social, economic, cultural and political contexts from local to global scales (Pollnac et al., 2010; Palomo et al., 2014; Cumming et al., 2015; Cumming and Allen, 2017). Understanding the capacity of PAs to conserve biodiversity into the future, and meet the growing expectation that they contribute to local livelihoods and local-to-global ecosystem services thus requires an understanding of their social-ecological dynamics and key feedbacks.

The case presented here is a case study of a multi-tenure PA system (a collective of private landowners as well as a state PA) in southern Africa, to demonstrate that the CIS Framework can be applied to conservation landscapes with a range of resource rights, including private, communal and/or state ownership. It speaks to several commonly researched conservation questions: (1) how to ensure PAs are effective and resilient in conserving biodiversity; (2) how to incentivise non-state actors to adopt pro-conservation land uses and behaviours; and answering questions one and two in an African context often links to (3) how to facilitate successful and sustainable wildlife-based tourism enterprises. Figure 2 summarizes the key features of the case study using the CIS Framework diagram.

A shift from livestock to wildlife ranching occurred across southern Africa in the 20th century due to the introduction of legislation (public infrastructure) that enabled landowners (resource users) to own, manage and benefit commercially from wildlife (ecological system/natural infrastructure) on their properties (e.g., through hunting, ecotourism or live trade) (Child et al., 2012). This legislation significantly increased the abundance and distribution of wildlife across private and communal land in southern Africa (Child et al., 2012; Taylor et al., 2020). The conservation value of wildlife ranching has been questioned, however, due to the general lack of monitoring of the ecological impacts of the industry and the temptation for some landowners to prioritize shorter-term profits over longer-term conservation. Such temptation can lead to management practices that ultimately erode ecosystem resilience, including enhancing the densities of large charismatic mammals (through artificial waterholes, vegetation cutting), unsustainable hunting rates or predator persecution (Cousins et al., 2010; Child et al., 2013; Clements and Cumming, 2017). There is thus the risk of a positive feedback through link 1, mediated by the exogenous driver of tourist demands, whereby the actions of resource users increase the densities of large charismatic mammals in the resource system and thus visitor revenues (Clements and Cumming, 2017, 2018). This encourages resource users to continue managing their wildlife at profitable yet ultimately unsustainable levels. This risk depends on the extent to which landowners are making decisions through link 1 according to cashflow versus ecological monitoring (Clements and Cumming, 2017, 2018).

Conservancies are promoted as a means of aligning wildlife ranching more closely with conservation objectives, and entail neighbouring landowners removing internal fencing to create larger, cooperatively managed wildlife areas (Lindsey et al., 2009; Leménager et al., 2014; Chidakel et al., 2020). In addition to increasing the area protected and thus the sustainability of the resource system, resource users can in theory benefit from economies of scale associated with tourism and wildlife, by sharing management and infrastructure costs over a larger area (Lindsey et al., 2009; Child et al., 2012; Chidakel et al., 2020). Wildlife management within conservancies is guided by cooperative agreements among landowners, creating shared public infrastructure. These rules are intended to reduce the frequency of undesirable management practices

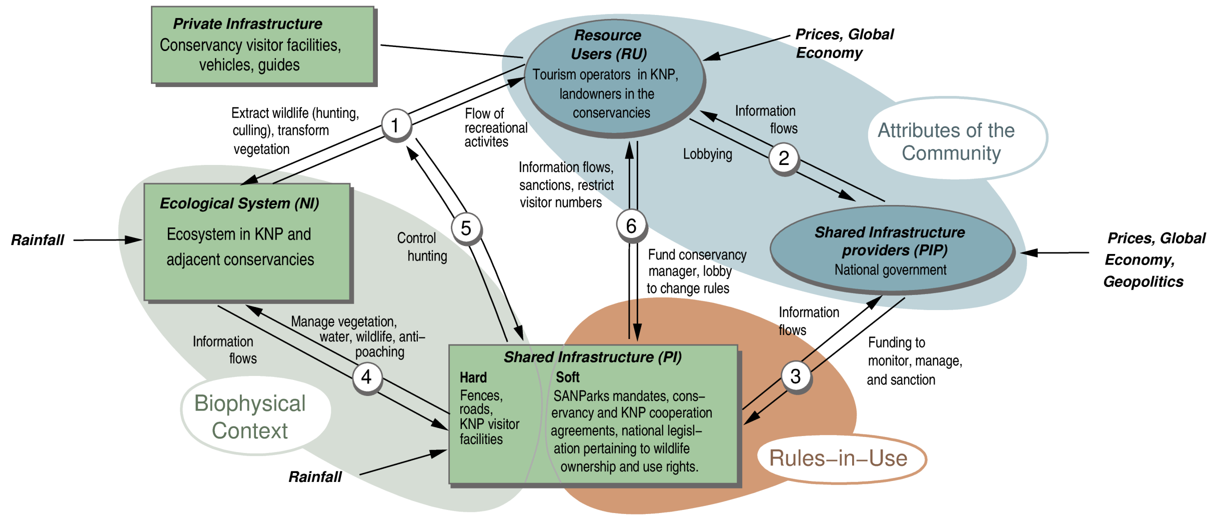


Figure 2: Visualization of the components and links of the CIS framework, as applied to a multi-tenure conservation system including Kruger National Park (KNP) and the surrounding private reserves.

by individual resource users through link 6 (e.g. hunting quotas) and link 5 (e.g. hunting season, wildlife stocking limits, limited waterhole numbers, limited tourist numbers). Undesirable practices would impact all resource users through their shared resource system (Lindsey et al., 2009; Child et al., 2013). Typically, conservancy members will co-fund a conservancy manager (who is thus part of the shared public infrastructure) to ensure the co-created rules are enforced. In theory this creates a feedback counter clockwise through links 1-4-5/6 whereby the impacts of resource users on the resource system are monitored by the manager and used to adjust or better enforce the conservancy rules (public infrastructure) and thus the actions of landowners and their visitors. Landowners themselves can also collectively decide on, or individually lobby for, changes to the public infrastructure based on what they observe happening in their connected resource systems, creating a clockwise feedback through these links 1-4-5/6.

Of course, it is not usually so straightforward in practice. Landowners may disagree on the collective rules and may have different capacities and willingness to abide by, and contribute to, this shared public infrastructure. If some resource users allow more wildlife hunting, build more roads, have higher volumes of tourists or attract more wildlife to their property through provision of extra waterholes or vegetation management, these resource users may benefit through increased revenues, but negatively impact the shared resource system (Child et al., 2013). Unequal investment in shared public infrastructure by resource users can also present a challenge. For example, in the Save Conservancy in Zimbabwe, the landowner investing the least in anti-poaching spent 43 times less than the landowner spending the most (Lindsey et al., 2009). The public infrastructure provider element of the CIS Framework is typically not relevant in this type of example, since the landowners (resource users) also act as the public infrastructure provider.

There are many important unanswered questions regarding how to promote sustainable wildlife conservancies (Lindsey et al., 2009; Child et al., 2013) which a dynamical model could address. For example, exploring options for balancing the costs and benefits of maintaining shared public infrastructure and understanding resource user incentives to cooperate (under what conditions is worth their while?). These options could include national legislation such as tax breaks and sustainability certifications. It would be interesting to explore which local agreements best balance individual freedoms with sustainable wildlife management practices (Lindsey et al., 2009). Incorporating feedbacks between links 1-4-5/6 is critical to understanding these dynamics. Such questions and dynamics also have relevance for community based natural resource management and conservation, which is also common in southern Africa (Child and Barnes, 2010). Models and theory from many disciplines are relevant to these problems (Figure S.1 and Table S.1, Supplementary Materials) including the social sciences (psychology, behavioural economics), bioeconomics, ecology and conservation biology.

In some instances, these wildlife conservancies have developed agreements with adjacent state-run PAs, and fences have been removed between the state PA and the private or communal conservancies (Child et al., 2012; Leménager et al., 2014). A well-known example is the partnership between Kruger National Park (KNP) and conservancies adjoining the park to the west in South Africa (Chidakel et al., 2020). These conservancies comprise several hundred landowners and cover an area of 2,400 square kilometers; 12.5% the size of KNP.

KNP is one of South Africa's 20 national parks and is managed by South African National Parks (SANParks), a national government-funded organization, in alignment with national legislation including the National Environmental Management: Biodiversity Act, and National Environmental Management: Protected Areas Act. Therefore, in the case of KNP, SANParks is the public infrastructure, partially funded via link 3 by the Department of Environmental Affairs, who are the public infrastructure providers. Employees in KNP (e.g. game rangers, gate, lodge, anti-poaching, and maintenance staff) are thus employed by SANParks as part of the public infrastructure. The only resource users in this system are tour operators and independent guides who bring visitors to the park and thus derive their livelihoods from the resource system. Tourists, as in the conservancy case, are exogenous drivers. These tourists can have a big impact on the park both through their gate and accommodation fees that fund a large portion of the park's running costs (Chidakel et al., 2020), but also through lobbying the public infrastructure when they disagree with how it is managing large charismatic wildlife like lions and elephants (e.g. there have been outcries in the past of culling to regulate wildlife numbers) (Venter et al., 2008). Visitors can thus influence links 3, 4, 5 and/or 6.

Private landowners in the conservancies (resource users) benefit from the dramatically increased land area associated with dropping fences with KNP which enables them to benefit from the charismatic biodiversity (e.g., lion, elephant) that occur in KNP (resource system) (Child et al., 2012). They also benefit from the infamous 'brand' of KNP, in attracting visitors (Chidakel et al., 2020). According to SANParks, removing fences between KNP and these conservancies meets their objective "to secure and improve ecosystem processes and associated socio-economic benefits through the consolidation of vast landscapes" (SANParks, 2018). The conservancies increase the size of the connected resource system and buffer its boundary from human pressure, as well as diversifying and increasing the accommodation options offered by the park. The private infrastructure in the conservancies generally provides higher-end accommodation options that attract international visitors and create a disproportionately high number of employment opportunities relative to their size (Kruger et al., 2019; Chidakel et al., 2020). They thus make an important socio-economic contribution, amidst growing pressure that the public infrastructure providers and public infrastructure in KNP deliver benefits to communities living on its boundary (Swemmer et al., 2017).

While there are clear benefits to this public-private partnership, it also places additional coordination (soft public infrastructure) burdens on SANParks. While wildlife are not hunted in the KNP, they are hunted in some of the conservancies, meaning that if hunting quotas are not set or administered correctly (via link 5), the conservancies could be a sink for KNP's wildlife, or alter population dynamics (e.g., of lion, Maputla et al., 2015). The conservancies coordinate their own lower-level management through their own shared public infrastructure, which can have significant consequences for animal movement between KNP and the conservancies, and thus vegetation dynamics in the connected resource system, particularly in droughts (Mwakiwa et al., 2013). Because of the lack of internal fencing, conservancies may benefit from KNP's considerable anti-poaching efforts (via link 4), as wildlife moves into the vacuum created by poaching in the conservancy. By contrast, if conservancies invest considerably in their own anti-poaching infrastructure and good external fencing (public infrastructure), this benefits KNP (Massé and Lunstrum, 2016).

There are many interesting questions that could be modeled in this multi-tenure conservation system, including how to balance the benefits of a state PA dropping its fence with neighbouring conservancies (e.g.,

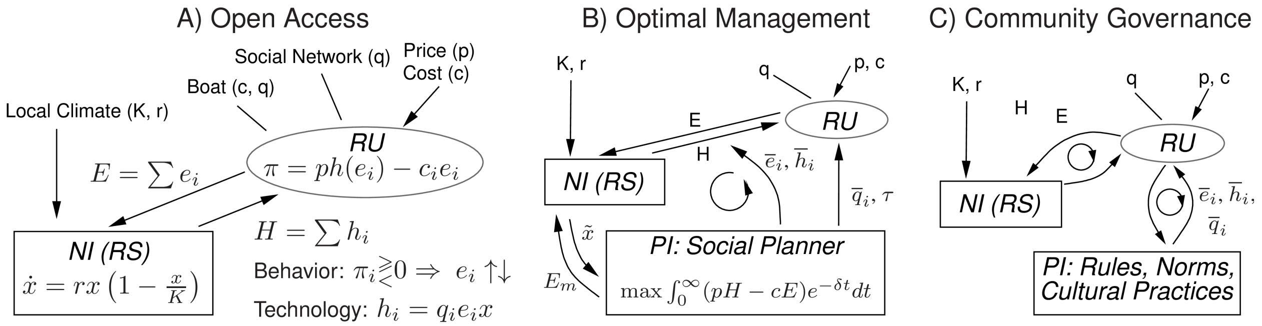


Figure 3: The basic bioeconomic model of resource management as special cases of the CIS Framework. Elements are instantiated with stocks (state variables), links are instantiated with constraints, information flows, and material flows. The relative size of the elements represents the relative importance off the elements in the model treatment. In open access, the RU and NI dominates. In the social planner problem, the PI dominates. See text for further discussion.

more biodiversity, more jobs, less boundary fence to maintain), with the increased costs associated with coordination and infrastructure to manage ecological spill-overs, the different user rights (e.g., hunting in conservancies but not in the KNP), strategic behaviours, etc. Multi-tenure PAs are promoted as a means of increasing the extent and resilience of PA systems around the world (Fitzsimons and Wescott, 2008; Clements et al., 2019; De Vos and Cumming, 2019), but managing the additional dynamics associated with increased actors and their coordination requires recognition of possible system feedbacks and ways to manage those.

Toward richer representations of behavior and politics in models of SESs

Modeling research on the economics of natural resource management initially focused on fisheries (e.g. Gordon, 1953), the over-exploitation of large marine mammals, and the possibility of extinction (e.g. Clark, 1973). Figure 3 illustrates 3 stages of resource management modeling from roughly the 1950's through the early 2000's. Panel A) illustrates the open access harvest of a logistic renewable resource. This model provides the basis of hundreds of variations on a theme. Resource users are rational actors who adjust harvesting effort up or down if profit, π is positive or negative. This simple rule is the only element of human 'behavior' captured in the model. The harvest process is captured with a simple mass-action model in which the harvest is the product of the effort, e_i (e.g. harvesting days per year), 'catchability (or harvestability) coefficient', q_i (proportion of standing stock harvested per unit effort - (e.g. 0.01% per harvest day), and the standing stock x (e.g. megatonnes). The main conclusion from this simple model is that the long run behavior of the system, the so-called 'bionomic equilibrium' is determined by the interaction between *external* relatively slow processes (e.g. weather patterns, soil processes, preferences, technology) and fast processes *endogenous* to the resource system and resource users (e.g. monthly or annual population dynamics, daily decisions about harvesting effort). The former are represented by fixed parameters. Regional scale ecological and climatological conditions set r and K. Preferences and technology drive market prices p and c. Technology and knowledge set q. Given these constraints, harvesters mobilize tools, (e.g. a boat) and harvest the resource (e.g. a fish stock). The resource (e.g. fish) stock is reduced from harvest and replenished based on logistic growth. Fishers adjust their effort accordingly. The bionomic equilibrium occurs when the harvest balances the natural growth *and* entry and exit from the resource system is balanced (e.g. when $\pi = 0$).

Even this very simple class of models can be challenging to analyze analytically. For example, if we assume that all harvesters are identical ($q_i = q,\ c_i = c\ \forall i$) and they operate in competitive markets (can't influence p), are perfectly rational, have perfect information, etc. we can treat the harvesters as a lumped effort, E. This simplifies the model significantly, allowing us to calculate the bionomic equilibrium (the resource stock and harvest effort are not changing) by solving $rx(1 - x/K) - qEx = 0$ and $pqEx - cE = 0$ for x and E. This couplet (x, E) constitutes the bionomic equilibrium. If we allow for any real-world deviations from this simple model, such as heterogeneity across resource users (different technology, opportunity costs, etc.) or imperfect information (not knowing the stock size, $x(t)$ exactly), analytical challenges mount quickly.

The most studied extension of the open access bioeconomic model is shown in Figure 3, Panel B). The essential feature is the insertion of public infrastructure in the form of a benevolent, omniscient social planner whose objective is to maximize the value of the natural infrastructure for society. The typical approach is maximize the sum of the discounted revenues generated by harvesting activities as shown in the public infrastructure element in Panel B. The simplifications in the open access model discussed above are typically carried over into the benevolent social planner problem to simplify the mathematics. Since each resource user is identical, maximizing social welfare is equivalent to setting total effort, E to maximize the value of the discounted (at rate δ) revenue stream generated by the total harvest, H. The social planner may control e_i (and thus E) by setting a harvesting season length to $\overline{e}_i$ or h_i with a harvest quota $\overline{h}_i$. They may control q_i by restricting harvest technology to $\overline{q}_i$ (e.g. vessel size, horsepower, net mesh size in a fishery) or the cost of harvesting effort through a license or use fee (τ). In practice, regulations are comprised of a combination of these policy instruments. It is worth noting that even in this simple model the possible combinations of policy interventions is large. Choosing the right mix is quite difficult. No matter the policy combination, the social planner creates a regulatory feedback system that can be used (at least in theory) to direct the state of the system (stock and effort levels) to any feasible value. Given the state of the system when the planner intervenes, the planner can choose a transient path and long run equilibrium that maximizes the total value of the asset.

The problem with this treatment is that in order to calculate and execute the optimal plan, the social planner needs accurate measurements of the parameters and system state as well as some level of certainty that the underlying model of the system (i.e. logistic growth) is correct. This should give the reader pause - it is very difficult to implement such models with 'real-world' knowledge infrastructure (e.g. Anderies et al., 2019b). The main contribution of this model set up is to illustrate that depending on the relative rate of growth of the resource and of other investments (which determines the discount rate in practice) it may, in fact, be 'optimal' to 'liquidate' the natural resource stock - i.e. extinction may be optimal (Clark, 1973). This finding contradicted existing belief in economics that rational actors would never destroy a valuable resource. The use of the social planner model illustrated that they would, and we should worry about it. In terms of practical management, on the other hand, this view of the optimal planner and the worldview based on western scientific knowledge that supports it has often not been very successful (Wilson et al., 1994; Hughes et al., 2005).

Panel C) summarizes work on bottom-up governance pioneered by Ostrom (1990). This view is a stark counterpoint to the top down technocratic management process in Panel B). In this case, resource users gain knowledge about the resource via a learning loop involving effort (E) and harvest (H). The local knowledge is used by the community to craft norms, shared strategies, and, less commonly, formal rules about resource use activities such as when, where, and what species to harvest with what technology. These guidelines, taboos, and ritual practices are represented by $\overline{e}_i$, $\overline{h}_i$, and $\overline{q}_i$, respectively. Note that there is no global feedback loop that connects the resource system, users, and management infrastructure as in the optimal management case in Panel B). The feedback among these elements is created by a *network of regulatory feedbacks* that are connected through the resource user element. In such systems, communities tend not to develop shared infrastructure to measure population sizes, effort levels, harvest amounts as in western scientific management approaches. Rather, they develop subtle knowledge about how their actions impact their environment and craft rituals, norms, and strategies that guide *how* individuals should interact with the world around them (both in terms of use and nurturing) rather than how much *stuff* they can *take* from it (Acheson and Wilson, 1996). The problem with such governance arrangements is that they are *fragile*. They rely on the resource users believing in, accepting, and behaving consistently with shared cultural practices. Such systems are very vulnerable to exogenous shocks to beliefs, worldviews, crowding-out, and opportunities of resource users. Further, they are sensitive to changing dynamics in the resource system as it may take too long for the learning feedback to work - i.e. people may not correctly attribute causal factors to realized dynamics.

Modeling the impact of community norms, rituals, etc. on the dynamics of a SES is straightforward. That is, it is simply a matter of assuming the norms, rules, ritual etc. exist and studying their impact (e.g. Anderies, 1998; Foin and Davis, 1984; Cifdaloz et al., 2010; Lansing, 1991; Lansing and Kremer, 1993). The larger challenge is understanding how successful rules emerge and, if we want to take lessons from traditional systems, how to design such rule systems. That is, what are the processes by which the arrows from the resource system, through the users, to the cultural system and clusters of rules and norms work? Trial and error? What is the ‘rule crafting’ process? This is an extremely difficult question about *cultural selection*. This process has been mathematically formalized using ideas from cultural group selection theory (Waring et al., 2017) that show promise for understanding how effective governance regimes emerge.

The last issue we wish to note is that none of the systems in Figure 3 contain the public infrastructure provider (PIP) element. One reason is the fact that it is extremely difficult to model strategic political behavior. One view has PIPs ‘selling’ policies to voters in political competition. PIPs propose projects funded through link 3 that ostensibly will benefit citizens (resource users) through link 4 (roads, dams, levees, canals), link 5 (coordination of activities) or link 6 (education, health care, welfare). Such political competition has been modeled in democratic capitalist societies. That is, PIPs propose a number of projects that, if elected to office, they will enact through link 3 (CW). In the best circumstances, PIPs will consult with professionals who work in the public infrastructure sector to accurately portray the potential feasibility, costs, and benefits of the projects they propose through link 3 (CCW). PIPs advertise their project portfolios to resource users (voters) through link 2 (CCW). Citizens express their preference for public works projects through their votes (link 2, CW). Weingast et al. (1981) take a neo-classical economic model in which PIPs objective is to maximize net benefits to their constituents to gain political support and explore the implications (e.g. projects are larger than they should be from an economic efficiency point of view) of such political competition. There are several generalizations that rely on models of rational choice and political competition to model the political-economy (e.g. Shepsle and Weingast, 1981; Magee et al., 1989) but we are unaware of cases where models in this research tradition have been linked to the management loop.

The only models we are aware of that provide a full, rigorous analysis of the full system with coupled political and economic loops are those of Muneepeerakul and Anderies (2018, 2020). The first explores the case where PIPs strategically set a tax or user fee, pay themselves with a portion of this fee and invest the rest in providing public infrastructure. If the tax is too high, resource users exit the system (e.g. farmers exiting irrigation systems to work in urban areas). If the ‘rent’ the PIPs extract for themselves is too low, they exit in search of better opportunities (Olson’s 1993 notion of the ‘roving bandit’). This model uses replicator dynamics to determine under what conditions agricultural systems might collapse. It does not, however, explicitly treat collective action problems. Muneepeerakul and Anderies (2020) explores the case in which the RU and PIP are the same individuals, e.g. villagers who sit on a water board, who make decisions about whether to use the resource, serve in the role of PIP to administer the PI, or leave the system. This model allows for governance to *emerge* endogenously as “co-management” and is the first of its kind that we are aware of. We present the full details of this model in the Supplementary Materials. Combining such models of feedbacks on both sides of Figure 1 at the right level of complexity is critical for addressing real-world conservation problems.

Conclusions

To make progress in improving the management of shared resources and the conservation of ecosystems, we argue that governance should be viewed not as something we *do*, but as an emergent feature of coupled infrastructure systems. This distinction is important because we must take a holistic view of our actions *within* a system rather than an isolated view of our actions *on* a system we erroneously believe we can control. The latter view invariably leads to unintended consequences of policy actions that focus narrowly on single issues or subsystems. This phenomenon is widely understood in economic systems (great examples of self-organizing coupled infrastructure systems) as the theory of the second best (Lipsey and Lancaster, 1956). Put simply, this theory suggests that a policy intervention that improves performance in one area (subsystem) may not imply a global improvement and could actually make overall performance worse. The details of every situation matter.

While it is practically impossible to capture all relevant details in a model, we suggest that we need to at least attempt to capture sufficient information about the entire system in question to have more informed discussions about potential unintended consequences and essential trade-offs that must be faced in any policy choice context. The CIS Framework provides a platform for systematic discussions of these issues, model building, and reflection. The foundational principle for effective policy captured in the CIS Framework is the notion of *regulatory feedbacks* of two types: 1) management feedbacks that drive day-to-day operations and 2) political feedbacks that drive longer term visioning and investments. These two feedback structures must be harmonized through shared infrastructures in order to achieve lasting conservation outcomes.

Looking ahead, the future success of conservation biology as a discipline will depend heavily on its ability to match the increasing sophistication of its ecological understanding with a corresponding understanding of the human societies and economies that have become the dominant influences on ecosystems across the globe. As our ability to describe the complexities of social-ecological dynamics starts to exceed the capacity of the human brain to qualitatively evaluate chains of interactions and their likely outcomes, decision makers are becoming increasingly dependent on a combination of heuristics and advanced computational models. In the same way that global climate models cannot reasonably ignore societal trends in the use of fossil fuels, conservation models can no longer afford to relegate the impacts of people on ecosystems to a single 'manager' box. Our proposed framework, while far from perfect, offers a starting point for attaining the level of complexity and sophistication that we believe is needed to support the continued growth and relevance of conservation efforts in today's highly interconnected world.

References

Acheson, J. M. and Wilson, J. A. (1996). Order out of chaos. *American anthropologist*, 98(3):579–594.

Anderies, J. M. (1998). Culture and human agro-ecosystem dynamics: the Tsembaga of New Guinea. *Journal of Theoretical Biology*, 192(4):515–530.

Anderies, J. M. (2003). Economic development, demographics, and renewable resources: a dynamical systems approach. *Environment and Development Economics*, 8(2):219–246.

Anderies, J. M. (2006). Robustness, institutions, and large-scale change in social-ecological systems: the Hohokam of the Phoenix Basin. *Journal of institutional economics*, 2(2):133–155.

Anderies, J. M., Barreteau, O., and Brady, U. (2019a). Refining the robustness of social-ecological systems framework for comparative analysis of coastal system adaptation to global change. *Regional Environmental Change*, 19(7):1891–1908.

Anderies, J. M., Janssen, M. A., and Ostrom, E. (2004). A framework to analyze the robustness of social-ecological systems from an institutional perspective. *Ecology and Society*, 9(1):18.

Anderies, J. M., Mathias, J.-D., and Janssen, M. A. (2019b). Knowledge infrastructure and safe operating spaces in social–ecological systems. *Proceedings of the National Academy of Sciences*, 116(12):5277–5284.

Anderies, J. M., Rodriguez, A., Janssen, M., and Cifdaloz, O. (2007). Panaceas, uncertainty, and the robust control framework in sustainability science. *Proceedings of the National Academy of Sciences USA*, 104:15194–15199.

Arias, A. (2015). Understanding and managing compliance in the nature conservation context. *Journal of environmental management*, 153:134–143.

Axelrod, R. (1984). *The Evolution of Cooperation*. Basic Books, New York.

Barnes, M. L., Lynham, J., Kalberg, K., and Leung, P. (2016). Social networks and environmental outcomes. *Proceedings of the National Academy of Sciences*, 113(23):6466–6471.

Barreteau, O., Bousquet, F., Attonaty, J.-M., et al. (2001). Role-playing games for opening the black box of multi-agent systems: method and lessons of its application to senegal river valley irrigated systems. *Journal of artificial societies and social simulation*, 4(2):5.

Beltratti, A. (1997). Growth with natural and environmental resources. In Carraro, C. and Siniscalcoa, D., editors, *New Directions in the Economic Theory of the Environment*. Cambridge University Press, Cambridge, U.K.

Bennett, N. J., Roth, R., Klain, S. C., Chan, K., Christie, P., Clark, D. A., Cullman, G., Curran, D., Durbin, T. J., Epstein, G., et al. (2017). Conservation social science: Understanding and integrating human dimensions to improve conservation. *biological conservation*, 205:93–108.

Berkes, F., Colding, J., and Folke, C. (2003). *Navigating Social-Ecological Systems: Building Resilience for Complexity and Change*. Cambridge University Press, Cambridge.

Bodin, Ö. and Norberg, J. (2005). Information network topologies for enhanced local adaptive management. *Environmental management*, 35(2):175–193.

Bonté, B., Therville, C., Bousquet, F., Abrami, G., Dhenain, S., and Mathevet, R. (2019). Analyzing coastal coupled infrastructure systems through multi-scale serious games in languedoc, france. *Regional Environmental Change*, 19(7):1879–1889.

Bousquet, F. and Le Page, C. (2004). Multi-agent simulations and ecosystem management: a review. *Ecological modelling*, 176(3-4):313–332.

Brander, J. A. and Taylor, M. S. (1998). The simple economics of Easter Island: A Ricardo-Malthus model of renewable resource use. *American Economic Review*, 88(1):119–138.

Bunnefeld, N., Hoshino, E., and Milner-Gulland, E. J. (2011). Management strategy evaluation: a powerful tool for conservation? *Trends in ecology & evolution*, 26(9):441–447.

Büscher, B. and Fletcher, R. (2019). Towards convivial conservation. *Conservation & Society*, 17(3):283–296.

Carlson, J. and Doyle, J. (2002). Complexity and robustness. *Proceedings of the National Academy of Science*, 99(suppl. 1):2538–45.

Carpenter, S. R., Ludwig, D., and Brock, W. A. (1999). Management of eutrophication for lakes subject to potentially irreversible change. *Ecological applications*, 9(3):751–771.

Chidakel, A., Eb, C., and Child, B. (2020). The comparative financial and economic performance of protected areas in the greater kruger national park, south africa: functional diversity and resilience in the socio-economics of a landscape-scale reserve network. *Journal of Sustainable Tourism*, 28(8):1100–1119.

Child, B. and Barnes, G. (2010). The conceptual evolution and practice of community-based natural resource management in southern africa: past, present and future. *Environmental Conservation*, pages 283–295.

Child, B. A., Musengezi, J., Parent, G. D., and Child, G. F. (2012). The economics and institutional economics of wildlife on private land in africa. *Pastoralism: Research, Policy and Practice*, 2(1):18.

Child, M. F., Peel, M. J., Smit, I. P., and Sutherland, W. J. (2013). Quantifying the effects of diverse private protected area management systems on ecosystem properties in a savannah biome, south africa. *Oryx*, 47(1):29–40.

Cifdaloz, O., Regmi, A., Anderies, J. M., Rodriguez, A. A., et al. (2010). Robustness, vulnerability, and adaptive capacity in small-scale social-ecological systems: The Pumpa Irrigation System in Nepal. *Ecology and Society*, 15(3):39.

Cinner, J. E., McClanahan, T. R., Daw, T. M., Graham, N. A., Maina, J., Wilson, S. K., and Hughes, T. P. (2009). Linking social and ecological systems to sustain coral reef fisheries. *Current Biology*, 19(3):206–212.

Cinner, J. E., Zamborain-Mason, J., Maire, E., Hoey, A. S., Graham, N. A., Mouillot, D., Villéger, S., Ferse, S., and Lockie, S. (2022). Linking key human-environment theories to inform the sustainability of coral reefs. *Current Biology*.

Clark, C. W. (1973). The economics of overexploitation. *Science*, 189:630–634.

Clark, C. W. (1976). *Mathematical Bioeconomics: The Optimal Management of Renewable Resources*. J. Wiley, N.Y., U.S.A., 1 edition.

Clark, C. W. (2010). *Mathematical bioeconomics: the mathematics of conservation*, volume 91. John Wiley & Sons.

Clark, C. W., Mangel, M., et al. (2000). *Dynamic state variable models in ecology: methods and applications*. Oxford University Press on Demand.

Clausen, R. and York, R. (2008). Economic growth and marine biodiversity: influence of human social structure on decline of marine trophic levels. *Conservation Biology*, 22(2):458–466.

Clements, H. S. and Cumming, G. S. (2017). Positives and pathologies of natural resource management on private land-conservation areas. *Conservation biology*, 31(3):707–717.

Clements, H. S. and Cumming, G. S. (2018). Traps and transformations influencing the financial viability of tourism on private-land conservation areas. *Conservation Biology*, 32(2):424–436.

Clements, H. S., Kerley, G. I., Cumming, G. S., De Vos, A., and Cook, C. N. (2019). Privately protected areas provide key opportunities for the regional persistence of large-and medium-sized mammals. *Journal of Applied Ecology*, 56(3):537–546.

Cousins, J. A., Sadler, J. P., and Evans, J. (2010). The challenge of regulating private wildlife ranches for conservation in south africa. *Ecology and Society*, 15(2).

Crona, B. and Bodin, Ö. (2006). What you know is who you know? communication patterns among resource users as a prerequisite for co-management. *Ecology and society*, 11(2).

Csete, M. E. and Doyle, J. C. (2002). Reverse engineering of biological complexity. *Science*, 295:1664–1669.

Cumming, G. S. and Allen, C. R. (2017). Protected areas as social-ecological systems: perspectives from resilience and social-ecological systems theory. *Ecological applications*, 27(6):1709–1717.

Cumming, G. S., Allen, C. R., Ban, N. C., Biggs, D., Biggs, H. C., Cumming, D. H., De Vos, A., Epstein, G., Etienne, M., Maciejewski, K., et al. (2015). Understanding protected area resilience: a multi-scale, social-ecological approach. *Ecological Applications*, 25(2):299–319.

Cumming, G. S. and Epstein, G. (2020). Landscape sustainability and the landscape ecology of institutions. *Landscape Ecology*, pages 1–16.

Daw, T. M., Coulthard, S., Cheung, W. W., Brown, K., Abunge, C., Galafassi, D., Peterson, G. D., McClanahan, T. R., Omukoto, J. O., and Munyi, L. (2015). Evaluating taboo trade-offs in ecosystems services and human well-being. *Proceedings of the National Academy of Sciences*, 112(22):6949–6954.

De Vos, A. and Cumming, G. S. (2019). The contribution of land tenure diversity to the spatial resilience of protected area networks. *People and Nature*, 1(3):331–346.

DeAngelis, D. L. (2018). *Individual-based models and approaches in ecology: populations, communities and ecosystems*. CRC Press.

del Mar Mancha-Cisneros, M., Suárez-Castillo, A. N., Torre, J., Anderies, J. M., and Gerber, L. R. (2018). The role of stakeholder perceptions and institutions for marine reserve efficacy in the Midriff Islands Region, Gulf of California, Mexico. *Ocean & Coastal Management*, 162:181–192.

Dyball, R. and Newell, B. (2014). *Understanding human ecology: A systems approach to sustainability*. Routledge.

Epstein, G., Gurney, G., Chawla, S., Anderies, J. M., Baggio, J., Unnikrishnan, H., Villamayor Tomas, S., and Cumming, G. S. (2021). Drivers of compliance monitoring in forest commons. *Nature Sustainability*, 4(5):450–456.

Epstein, G., Vogt, J., Mincey, S., Cox, M., and Fischer, B. (2013). Missing ecology: integrating ecological perspectives with the social-ecological system framework. *International journal of the Commons*, 7(2).

Fitzsimons, J. A. and Wescott, G. (2008). The role of multi-tenure reserve networks in improving reserve design and connectivity. *Landscape and Urban Planning*, 85(3-4):163–173.

Foin, T. C. and Davis, W. G. (1984). Ritual and self-regulation of the tsembaga maring ecosystem in the new guinea highlands. *Human Ecology*, 12(4):385–412.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., et al. (2005). Global consequences of land use. *science*, 309(5734):570–574.

Gintis, H., Bowles, S., Boyd, R. T., Fehr, E., et al. (2005). *Moral sentiments and material interests: The foundations of cooperation in economic life*, volume 6. MIT press.

Gintis, H., Henrich, J., Bowles, S., Boyd, R., and Fehr, E. (2008). Strong reciprocity and the roots of human morality. *Social Justice Research*, 21(2):241–253.

Gordon, H. (1954). The economic theory of a common property resource: The fishery. *Journal of Political Economy*, 62:124–142.

Gordon, H. S. (1953). An economic approach to the optimum utilization of fishery resources. *Journal of the Fisheries Board of Canada*, 10(7):442–457.

Green, R. E., Cornell, S. J., Scharlemann, J. P., and Balmford, A. (2005). Farming and the fate of wild nature. *science*, 307(5709):550–555.

Grossman, G. M. and Krueger, A. B. (1995). Economic growth and the environment. *The quarterly journal of economics*, 110(2):353–377.

Hinkel, J., Bots, P., and Schlüter, M. (2014). Enhancing the ostrom social-ecological system framework through formalization. *Ecology and Society*, 19(3).

Holland, J. (1992). Complex adaptive systems. *Daedalus*, 121:17–30.

Holland, J. H. (1995). *Hidden order: How adaptation builds complexity*. Perseus Books, New York.

Holling, C. (1973). Resilience and stability of ecological systems. *Annual Review of Ecology. and Systematics*, 4:2–23.

Holling, C. S. (2001). Understanding the complexity of economic, ecological, and social systems. *Ecosystems*, 4(5):390–405.

Holling, C. S. and Meffe, G. K. (1996). Command and control and the pathology of natural resource management. *Conservation biology*, 10(2):328–337.

Homayounfar, M., Muneepeerakul, R., Anderies, J. M., and Muneepeerakul, C. P. (2018). Linking resilience and robustness and uncovering their trade-offs in coupled infrastructure systems. *Earth System Dynamics*, 9(4):1159–1168.

Hughes, T. P., Bellwood, D. R., Folke, C., Steneck, R. S., and Wilson, J. (2005). New paradigms for supporting the resilience of marine ecosystems. *Trends in ecology & evolution*, 20(7):380–386.

Janssen, M. (2002). *Complexity and ecosystem management: the theory and practice of multi-agent systems*. Edward Elgar Publishing.

Kahneman, D. and Tversky, A. (1979). Prospect theory: An analysis of decision under risk. *Econometrica*, 47(2):263–291.

Keane, A., Jones, J. P., Edwards-Jones, G., and Milner-Gulland, E. J. (2008). The sleeping policeman: understanding issues of enforcement and compliance in conservation. *Animal conservation*, 11(2):75–82.

Kingsford, R. T. (2000). Ecological impacts of dams, water diversions and river management on floodplain wetlands in australia. *Austral Ecology*, 25(2):109–127.

Kiser, L. L. and Ostrom, E. (1982.). The three worlds of action: A metatheoretical synthesis of institutional approaches. In Ostrom, E., editor, *Strategies of Political Inquiry*, pages 179–222. Sage, Beverly Hills, CA.

Kruger, M., van der Merwe, P., Saayman, M., and Slabbert, E. (2019). Understanding accommodation preferences of visitors to the kruger national park. *Tourism and hospitality research*, 19(2):170–185.

Lansing, J. (1991). *Priests and Programmers: Technologies of Power in the Engineered Landscape of Bali*. Princeton University Press, Princeton, NJ, USA.

Lansing, J. S. and Kremer, J. N. (1993). Emergent properties of balinese water temple networks: Coadaptation on a rugged fitness landscape. *American Anthropologist*, 95(1):97–114.

Larkin, P. A. (1977). An epitaph for the concept of maximum sustained yield. *Transactions of the American fisheries society*, 106(1):1–11.

Leménager, T., King, D., Elliott, J., Gibbons, H., and King, A. (2014). Greater than the sum of their parts: Exploring the environmental complementarity of state, private and community protected areas. *Global Ecology and Conservation*, 2:238–247.

Lieberman, M. D. (2013). *Social: Why our brains are wired to connect*. OUP Oxford.

Lindsey, P., Romanach, S., and Davies-Mostert, H. (2009). The importance of conservancies for enhancing the value of game ranch land for large mammal conservation in southern africa. *Journal of Zoology*, 277(2):99–105.

Lipsey, R. G. and Lancaster, K. (1956). The general theory of second best. *The review of economic studies*, 24(1):11–32.

Long, N. E. (1958). The local community as an ecology of games. *American Journal of Sociology*, 64(3):251–261.

Lubell, M. (2013). Governing institutional complexity: The ecology of games framework. *Policy studies journal*, 41(3):537–559.

Ludwig, D. (2001). The era of management is over. *Ecosystems*, 4(8):758–764.

Ludwig, D., Walker, B., and Holling, C. S. (1997). Sustainability, stability, and resilience. *Conservation Ecology*, 1(1):[online] URL: http://www.consecol.org/vol1/iss1/art7/.

Lynam, T., Bousquet, F., Le Page, C., d'Aquino, P., Barreteau, O., Chinembiri, F., and Mombeshora, B. (2002). Adapting science to adaptive managers: spidergrams, belief models, and multi-agent systems modeling. *Conservation Ecology*, 5(2).

Mace, G. M. (2014). Whose conservation? *Science*, 345(6204):1558–1560.

Magee, S. P., Brock, W. A., and Young, L. (1989). *Black hole tariffs and endogenous policy theory: Political economy in general equilibrium*. Cambridge University Press.

Malthus, T. R. (1798). *An essay on the principle of population, as it affects the future improvement of society. With remarks on the speculations of Mr. Godwin, Mr. Condorcet, and other writers*. J. Johnson, London.

Mangel, M. and Clark, C. W. (1988). *Dynamic Modeling in Behavioral Ecology*. Princeton University Press.

Maputla, N. W., Maruping, N. T., Chimimba, C. T., and Ferreira, S. M. (2015). Spatio-temporal separation between lions and leopards in the kruger national park and the timbavati private nature reserve, south africa. *Global Ecology and Conservation*, 3:693–706.

Marschoff, E. R., Barrera-Oro, E. R., Alescio, N. S., and Ainley, D. G. (2012). Slow recovery of previously depleted demersal fish at the south shetland islands, 1983–2010. *Fisheries Research*, 125:206–213.

Maskin, E. S. (2008). Mechanism design: How to implement social goals. *American Economic Review*, 98(3):567–76.

Massé, F. and Lunstrum, E. (2016). Accumulation by securitization: Commercial poaching, neoliberal conservation, and the creation of new wildlife frontiers. *Geoforum*, 69:227–237.

Muneepeerakul, R. and Anderies, J. M. (2017). Strategic behaviors and governance challenges in social-ecological systems. *Earth's Future*, 5(8):865–876.

Muneepeerakul, R. and Anderies, J. M. (2018). Strategic behaviors and governance challenges in social-ecological systems. *Earth's Future*, 5(8):865–876.

Muneepeerakul, R. and Anderies, J. M. (2020). The emergence and resilience of self-organized governance in coupled infrastructure systems. *Proceedings of the National Academy of Sciences*, 117(9):4617–4622.

Mwakiwa, E., de Boer, W. F., Hearne, J. W., Slotow, R., van Langevelde, F., Peel, M., Grant, C. C., Pretorius, Y., Stigter, J. D., Skidmore, A. K., et al. (2013). Optimization of wildlife management in a large game reserve through waterpoints manipulation: a bio-economic analysis. *Journal of environmental management*, 114:352–361.

Mwakiwa, E., Hearne, J. W., Stigter, J. D., De Boer, W. F., Henley, M., Slotow, R., Van Langevelde, F., and Prins, H. H. (2016). Optimization of net returns from wildlife consumptive and non-consumptive uses by game reserve management. *Cooperation or Competition*, page 71.

Olson, M. (1993). Dictatorship, democracy, and development. *American Political Science Review*, 87(03):567–576.

Ostrom, E. (1990). *Governing the commons: The evolution of institutions for collective action*. Cambridge university press.

Ostrom, E. (1998). A behavioral approach to the rational choice theory of collective action: Presidential address, american political science association, 1997. *American political science review*, pages 1–22.

Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. *Proceedings of the national Academy of sciences*, 104(39):15181–15187.

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science*, 325(5939):419–422.

Palomo, I., Montes, C., Martin-Lopez, B., González, J. A., Garcia-Llorente, M., Alcorlo, P., and Mora, M. R. G. (2014). Incorporating the social–ecological approach in protected areas in the anthropocene. *BioScience*, 64(3):181–191.

Parker, D. C., Manson, S. M., Janssen, M. A., Hoffmann, M. J., and Deadman, P. (2003). Multi-agent systems for the simulation of land-use and land-cover change: a review. *Annals of the association of American Geographers*, 93(2):314–337.

Pereira, H. M. and Daily, G. C. (2006). Modeling biodiversity dynamics in countryside landscapes. *Ecology*, 87(8):1877–1885.

Phillips, J. S., Guðbergsson, G., and Ives, A. R. (2022). Opposing trends in survival and recruitment slow the recovery of a historically overexploited fishery. *Canadian Journal of Fisheries and Aquatic Sciences*, 99(999):1–7.

Pollnac, R., Christie, P., Cinner, J. E., Dalton, T., Daw, T. M., Forrester, G. E., Graham, N. A., and McClanahan, T. R. (2010). Marine reserves as linked social–ecological systems. *Proceedings of the National Academy of Sciences*, 107(43):18262–18265.

Ricardo, D. (1817). *On the Principles of Political Economy and Taxation*. John Murray.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F., Lambin, E., Lenton, T., Scheffer, M., Folke, C., Schellnhuber, H., et al. (2009). A safe operating space for humanity. *Nature*, 461(7263):472–475.

Sabatier, P. A. (1988). An advocacy coalition framework of policy change and the role of policy-oriented learning therein. *Policy sciences*, 21(2-3):129–168.

Sanchirico, J. N. and Wilen, J. E. (2001). A bioeconomic model of marine reserve creation. *Journal of Environmental Economics and Management*, 42(3):257–276.

SANParks (2018). Kruger national park - park management plan for the period 2018 - 2028. Technical report, SANParks.

Schaefer, M. (1957). Some considerations of population dynamics and economics in relation to the management of the commercial marine fisheries. *Journal of the Fisheries Research Board of Canada*, 14(5):669–681.

Schill, C., Anderies, J. M., Lindahl, T., Folke, C., Polasky, S., Cárdenas, J. C., Crépin, A.-S., Janssen, M. A., Norberg, J., and Schlüter, M. (2019). A more dynamic understanding of human behaviour for the anthropocene. *Nature Sustainability*, pages 1–8.

Schlüter, M., Müller, B., and Frank, K. (2019a). The potential of models and modeling for social-ecological systems research. *Ecology and Society*, 24(1).

Schlüter, M., Orach, K., Lindkvist, E., Martin, R., Wijermans, N., Bodin, Ö., and Boonstra, W. J. (2019b). Toward a methodology for explaining and theorizing about social-ecological phenomena. *Current Opinion in Environmental Sustainability*, 39:44–53.

Schneider, M., Scholz, J., Lubell, M., Mindruta, D., and Edwardsen, M. (2003). Building consensual institutions: networks and the national estuary program. *American journal of political science*, 47(1):143–158.

Seppelt, R., Arndt, C., Beckmann, M., Martin, E. A., and Hertel, T. W. (2020). Deciphering the biodiversity–production mutualism in the global food security debate. *Trends in Ecology & Evolution*.

Seppelt, R., Manceur, A. M., Liu, J., Fenichel, E. P., and Klotz, S. (2014). Synchronized peak-rate years of global resources use. *Ecology and Society*, 19(4).

Shepsle, K. A. and Weingast, B. R. (1981). Political preferences for the pork barrel: A generalization. *American journal of political science*, pages 96–111.

Solomon, J. N., Gavin, M. C., and Gore, M. L. (2015). Detecting and understanding non-compliance with conservation rules.

Soulé, M. E. (1985). What is conservation biology? *BioScience*, 35(11):727–734.

Starfield, A. M. and Bleloch, A. L. (1986). *Building models for conservation and wildlife management*. Macmillan.

Stiglitz, J. (1974). Growth with exhaustible natural resources: efficient and optimal growth paths. In *Review of Economic Studies, Symposium on the Economics of Exhaustible Resources*, volume 41, pages 123–137, Edinburgh. Longman.

Swemmer, L., Mmethi, H., and Twine, W. (2017). Tracing the cost/benefit pathway of protected areas: A case study of the kruger national park, south africa. *Ecosystem services*, 28:162–172.

Taylor, W. A., Lindsey, P. A., Nicholson, S. K., Relton, C., and Davies-Mostert, H. T. (2020). Jobs, game meat and profits: The benefits of wildlife ranching on marginal lands in south africa. *Biological Conservation*, 245:108561.

Tellman, B., Bausch, J., Eakin, H., Anderies, J., Mazari-Hiriart, M., Manuel-Navarrete, D., and Redman, C. (2018). Adaptive pathways and coupled infrastructure: seven centuries of adaptation to water risk and the production of vulnerability in mexico city. *Ecology and Society*, 23(1).

Thaler, R. (1980). Toward a positive theory of consumer choice. *Journal of economic behavior & organization*, 1(1):39–60.

Thaler, R. H. and Sunstein, C. R. (2009). *Nudge: Improving decisions about health, wealth, and happiness*. Penguin.

Therville, C., Brady, U., Barreteau, O., Bousquet, F., Mathevet, R., Dhenain, S., Grelot, F., and Brémond, P. (2019). Challenges for local adaptation when governance scales overlap. evidence from Languedoc, France. *Regional Environmental Change*, 19(7):1865–1877.

Venter, F. J., Naiman, R. J., Biggs, H. C., and Pienaar, D. J. (2008). The evolution of conservation management philosophy: Science, environmental change and social adjustments in kruger national park. *Ecosystems*, 11(2):173–192.

Vitousek, P. M., Mooney, H. A., Lubchenco, J., and Melillo, J. M. (1997). Human domination of earth's ecosystems. *Science*, 277(5325):494–499.

Waring, T. M., Goff, S. H., and Smaldino, P. E. (2017). The coevolution of economic institutions and sustainable consumption via cultural group selection. *Ecological Economics*, 131:524–532.

Weingast, B. R., Shepsle, K. A., and Johnsen, C. (1981). The political economy of benefits and costs: A neoclassical approach to distributive politics. *Journal of political Economy*, 89(4):642–664.

Westoby, M., Walker, B., and Noy-Meir, I. (1989). Opportunistic management for rangelands not at equilibrium. *Journal of Range Management*, 42(4).

White, B. and Wadsworth, R. (1994). A bioeconomic model of heather moorland management and conservation. *Ecological Economics*, 9(2):167–177.

Wilson, J. A., Acheson, J. M., Metcalfe, M., and Kleban, P. (1994). Chaos, complexity and community management of fisheries. *Marine Policy*, 18(4):291–305.