Viewing Woody-Plant Encroachment through a Social–Ecological Lens

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Grasslands and savannas worldwide have been dramatically altered by woody-plant encroachment (WPE). Maintaining remnant grasslands and restoring degraded grasslands for the people and animals that depend on them will require a new paradigm for WPE, one that views WPE as a complex social–ecological system. Here, we examine WPE in this light, using a conceptual framework designed to bridge the biophysical and social domains. On the basis of this press–pulse WPE framework, we develop a set of integrative hypotheses and identify key knowledge gaps using the Southern Great Plains as a case study. An alternative—and potentially complementary—approach to the press–pulse WPE framework is that of classical dynamic systems modeling, which has been widely adopted in ecology and economics. The explicit coupling of the press–pulse WPE framework with dynamic systems modeling has the potential to yield new insights for understanding the local- to regional-scale processes that drive and constrain changes in grass–woody plant abundances and for predicting the socioeconomic and ecological consequences of these changes.

Keywords: disturbance ecology, grasslands, interdisciplinary science, range science, woodland ecology

The human imprint on grasslands and savannas is enormous and has been throughout human history. Through the eons, humans, largely through the application of fire, have created and expanded grassy biomes at the expense of woodlands and forests (Sauer 1950, Pyne 2001, 2016). Grasslands and savannas worldwide have undergone substantial changes in recent decades and currently face a host of threats (e.g., overgrazing, alteration of fire regimes, and invasion by nonnative species). Many of these threats are exacerbated by climatic changes and are directly or indirectly related to the widespread proliferation of shrubs and trees (Parr et al. 2014). This transformation, known as *woody-plant encroachment* (WPE), has affected biotic climate zones worldwide, including temperate, tropical, and even arctic (Archer et al. 2017). The shift from a grass-dominated to a tree- or shrub-dominated landscape results in potentially profound changes to ecosystem function and processes, biogeochemical and energy budgets, and provisioning of ecosystem services (Barger et al. 2011, Eldridge et al. 2011). These changes, in turn, have important implications for the sustainability of pastoral societies and commercial livestock production systems, which are the foundation of rural economies and cultures (Archer and Predick 2014).

Grasslands have historically existed in semiarid and subhumid climates that are wet enough to support trees. The upper limit of woody-plant cover is dictated by mean annual precipitation but is constrained by disturbances (e.g., fire, browsing, and brush control) or soil properties (e.g., depth and texture; Archer et al. 2017). Grasslands and open savannas in these climates have been maintained largely by fire and, to some extent, herbivory (Bond and Parr 2010). Under European colonization during the mid- to late 1800s, the fire regimes in many grassy biomes in the Americas, Africa, and Australia were dramatically altered by overgrazing and active fire suppression (Walker and Janssen 2002), setting the stage for WPE in many rangelands across the globe. Other factors may be exacerbating WPE—including the dissemination of woody-plant seeds by livestock, eradication of native browsers (e.g., prairie dogs), increased atmospheric CO_2 concentrations, and a warming climate—but these are secondary to changes in fire regime (Archer et al. 2017).

Historically, attempts to control the proliferation of woody plants have relied on a variety of approaches collectively known as *brush management* (Archer and Predick 2014). During the 1960s and 1970s, brush management became a virtual "war on shrubs," aimed at eradication through mechanical and chemical methods. Enormous amounts of money, effort, and time were spent on reversing WPE (Briske et al. 2016), but consistent with state-change theory, the results were short lived (5–10 years) and therefore ineffective (Archer et al. 2017). Since the 1980s, it has gradually been recognized that if management interventions to conserve and restore grasslands are to be economically feasible and ecologically effective, they must (a) be implemented before critical state-change thresholds are crossed, (b) be based on decadal planning horizons that include follow-up

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interventions, and (c) explicitly take into account the provision of ecosystem services (Hamilton et al. 2004, Noble and Walker 2006, Archer et al. 2011). Despite this evolution toward broader and more informed perspectives, however, managers and society have yet to devise any mechanism for their practical application—at least on large scales.

In general, the grasslands present today have persisted in those areas where there has been explicit recognition that some semblance of pre-European-settlement fire regimes is critical to maintaining the structure and function of these ecosystems (Lehman and Parr 2016, Twidwell et al. 2016). Their continued maintenance depends on the recognition that humans and society are the primary drivers of fire regimes, either directly or indirectly. In the absence of fire, our remaining grasslands will almost certainly be invaded by shrubs and trees and transformed into woodlands. Fire, wild and prescribed, has potential ecological benefits, but mitigating circumstances related to fuel loads, smoke hazards, climatic conditions, and risk to infrastructure and life may constrain the use of prescribed fire and preclude letting wild fires burn unchecked.

Maintaining remnant grasslands and restoring degraded grasslands for the people and animals that depend on them will require a new paradigm for WPE, one that views this phenomenon as a complex social–ecological system within which coupled biophysical, social, and cultural processes operate and dynamically interact. To establish this new paradigm, frameworks will need to be developed that explicitly recognize nonlinear responses and feedback loops, time-lag and legacy effects, and ecosystem traits that can forecast transitions to alternative states brought about by interactions among natural and anthropogenic factors (Liu et al. 2007a, 2007b, Seneviratne et al. 2010, Virapongse et al. 2016).

Case study: The Southern Great Plains

There are few other regions that have been as significantly altered by WPE as the Southern Great Plains (SGP) of the United States. Rates of expansion of woody plants in this region are five- to sevenfold greater than in other regions of the United States (Barger et al. 2011). As we show in figure 1, much of the SGP region, especially in Texas and Oklahoma, is now covered by trees and shrubs; intact grasslands, although still existing in pockets, have largely disappeared. Although this transition is widely acknowledged in scientific and management circles, WPE in the SGP has occurred so gradually that its extent is not universally appreciated. Even contemporary narratives of the SGP describe it as predominantly a grassland (Cunfer 2005).

The SGP comprises a number of ecoregions, each having a distinct social, cultural, and ecological history and a distinct trajectory with respect to WPE and society's response to it. For example, at the SGP's southern endpoint is the Edwards Plateau, which is now characterized mostly by juniper or oak woodlands (Diamond and True 2008). At the northern endpoint is the Tallgrass Prairie, which remains mostly grassland with inclusions of woodland stands that are expanding (Ratajczak et al. 2016). Between these endpoints (both biogeographically and with respect to WPE) are the mixed-grass prairies of the Rolling Red Plains, where juniper woodlands are rapidly expanding but significant areas of open grasslands still exist (Barger et al. 2011).

Within the SGP, the timing and particulars of the grassland-to-woodland conversion differ across the region. In Texas, the conversion began in the early twentieth century, when these regions were being settled and livestock grazing was unregulated (Box 1967). At present, although still dynamic with respect to woody-plant cover, the conversion has likely reached the maximum that can be supported by the climatic conditions.

In Oklahoma and Kansas, WPE is more recent, probably because much of the area was cultivated following settlement and then eventually returned to grassland when cultivation proved unsustainable. In the past few decades, WPE has been advancing at an accelerated rate and currently represents a serious threat to the remaining tall- and mixed-grass prairies, including the plants and animals endemic to them (Briggs et al. 2005, Wang et al. 2018). In particular, eastern red cedar (*Juniperus virginiana*) has expanded steadily northward (Engle et al. 2008, Twidwell et al. 2016).

The dynamics of land-cover change in the Edwards Plateau exemplify those that have taken place in much of the SGP and in semiarid rangelands globally (Walker et al. 1981). These dynamics are most clearly revealed by the interactions among grazing, fire, and woody-plant cover (figure 2). Historically, plant communities in the region were typical of fire-maintained open savanna, with broadleaf evergreen (live oak) and deciduous arborescents (less than 10% cover), scattered individually or in clusters, in a matrix of midheight C_4 grasses. Both fire and herbivory were endogenous processes controlled by internal stabilizing feedback processes until the introduction of enormous herds of domestic livestock (figure 2). Although the region evolved with grazing pressure (e.g., by bison), domestic livestock were not transient or migratory; their numbers were maintained year round, year in and year out, in high numbers, and their movement was constrained by fences. In a relatively short period (between about 1875 and 1895), the landscape was converted from open parkland to a highly degraded landscape of exposed soil and rock with a few scattered trees (Box 1967). This process, which occurred across the southwestern United States (Bahre 1991), eventuated in the elimination of fire, allowing the proliferation of *Juniperus* spp. (scale-leaf evergreens) and *Prosopis glandulosa* (an N_2-fixing deciduous shrub; Fuhlendorf et al. 2008).

This human-mediated disturbance, which dramatically increased selective utilization of grasses by grazers, was not subject to internal ecological feedback loops (e.g., high densities of livestock were artificially maintained by supplemental feeding, and new animals were added to replace those that died). As a consequence, the fine fuel–fire feedback was disrupted. These degraded conditions persisted until the advent of conservation management

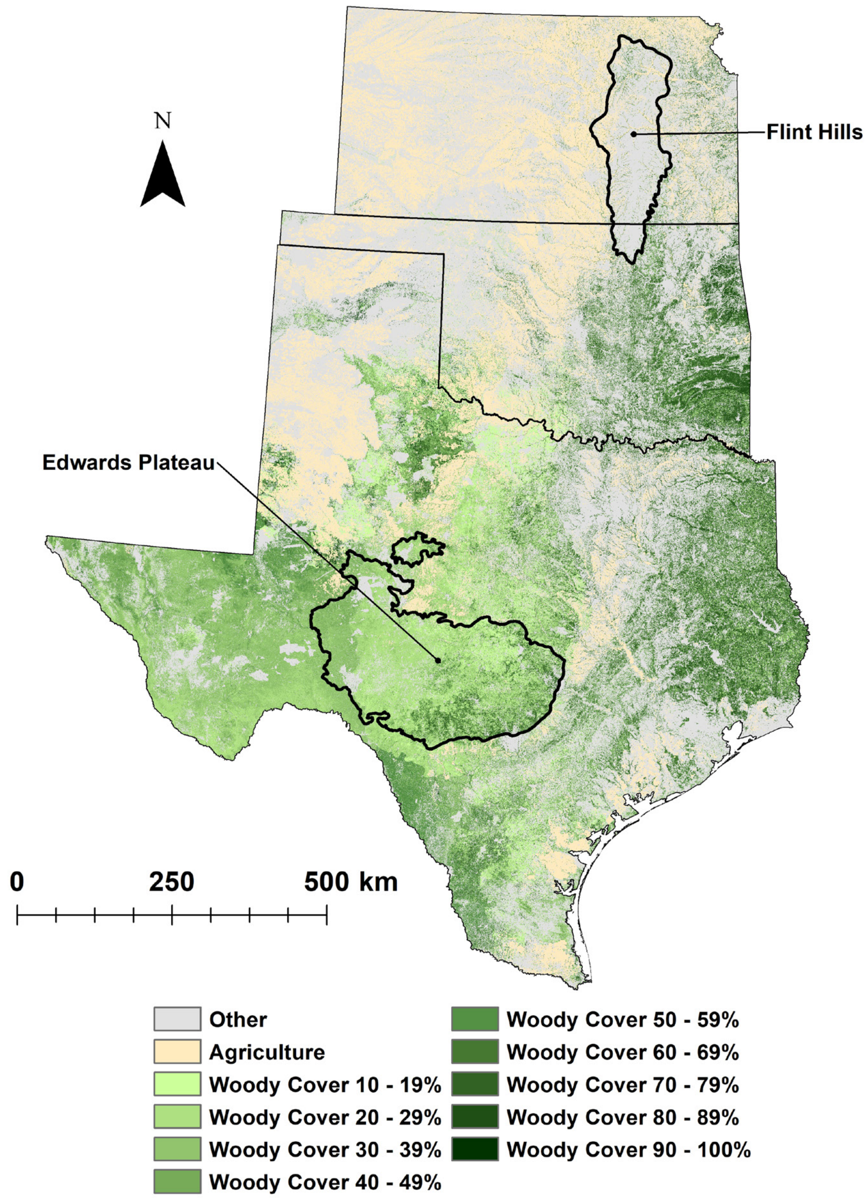

Figure 1. The extent of woody cover over the Southern Great Plains states of Texas, Oklahoma, and Kansas. Much of the present-day woody-plant cover within this region has developed over the past 100 years as shrubs and trees have proliferated in areas historically classified as grassland and open savanna (Barger et al. 2011). The "Other" theme consists mostly of herbaceous cover and some urban areas. Data source: LANDFIRE 2014.

in the mid-1960s; since then, with a reduction in stocking rates and the proliferation of woody plants, the area has gradually become a savanna parkland or woodland (Wilcox et al. 2008a, 2012). However, the fine fuel–fire feedback has not been reestablished because (a) herbaceous vegetation is dominated by species with inherently low productivity, (b) fire suppression is common, and (c) trees have gained ascendancy over herbaceous biomass and have grown too large to be controlled by surface fires (Fuhlendorf et al. 2008).

With respect to the SGP, we argue that prescribed fire is the linchpin for managing the balance between grassland and woodland states. The maintenance and restoration of grasslands and open savannas require the re-establishment of fire and grazing as endogenous processes, creating anew the fine fuel–fire feedback that maintained these landscapes for centuries. Societal perspectives on prescribed fire differ substantially across the SGP. For example, in the Tallgrass Prairie, its utility as a management tool is culturally ingrained; in contrast, in other parts of the region, people historically hold an antifire perspective. However, in the latter areas, a "burning culture" has begun to emerge as the adverse effects of WPE become more and more evident (Twidwell et al. 2013) and as successes with prescribed fire programs become known (Taylor et al. 2012).

Fundamentally, understanding and addressing WPE mean confronting the WPE paradox: Although ecologists appreciate the critical role of fire in maintaining many grassland and savanna ecosystems, there are serious social, economic, legal, and policy impediments that must be articulated and overcome for fire to be a viable and widely used management tool in the twenty-first century. One way to confront this science–policy disconnect is by conceptualizing WPE as a social–ecological system.

The press–pulse framework for evaluating woody-plant encroachment

The term *social–ecological system* (SES) has been used synonymously with *coupled human and natural systems* (CHANS). According to Liu and colleagues (2007a, 2007b), SES are integrated systems within which people reciprocally interact with natural components, and these interactions produce complex feedback loops with time lags and legacy effects. As a result, relationships in SES are often nonlinear and exhibit threshold behavior between two alternative stable states. Surprises often follow. Subtle losses of resilience can lead to rapid shifts (surprises) that are difficult to reverse (Standish et al. 2014, Bowman et al. 2015). Untangling the complex interactions across multiple temporal and spatial scales is the essence of SES research and is essential for developing effective policies for social–ecological sustainability.

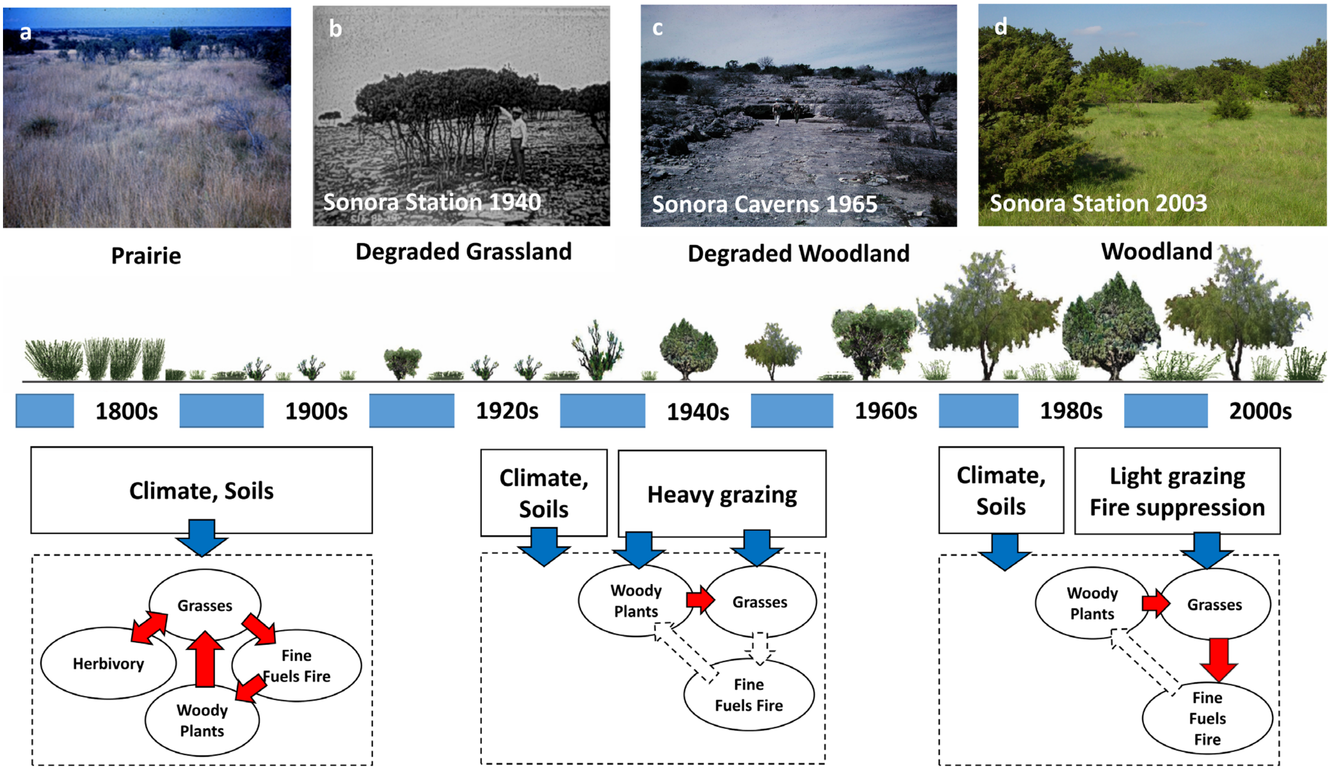

Figure 2. Prior to settlement, native grazers and browsers influenced grass–woody plant abundance within constraints imposed by soils and climate. In the late 1800s, overgrazing by domestic livestock (i.e., cattle and sheep) on the Edwards Plateau, Texas, set in motion a broadscale regime shift from largely open savannas (a) to the woodlands that dominate many of the landscapes today (d). The intermediate stage of this shift was a highly degraded landscape (b and c) that did not recover until grazing pressure relaxed during the 1970s and 1980s. The box and arrow diagrams below the photos show the key exogenous (solid box) and endogenous (dashed box) biophysical processes driving these transitions.

An essential but challenging aspect of the SES approach is developing an appropriate framework for understanding the problem, one that can guide us in both identifying the social and the ecological factors at play and determining how they interact (Hruska et al. 2017). The frameworks that have been proposed to date (Binder et al. 2013) have in common the assumption that the biophysical components are linked to the social components via the provision of ecosystem services. In other words, human decisions are governed by the effects they are expected to have on the receipt or not of key, valued ecosystem services.

One such framework, known as the *press–pulse dynamics framework*, was developed by Collins and colleagues (2011) to guide integrative and long-term interdisciplinary research aimed at bridging the biophysical and social domains. The expectation is that research based on this framework will lead to a deeper understanding of social–ecological systems and build a knowledge base sophisticated enough to address persistent environmental challenges. Collins and colleagues (2011) defined *press dynamics* as drivers of change that are extensive, pervasive, and subtle and *pulse dynamics* as drivers of change that are sudden and episodic. Examples of press disturbances would be gradual changes in temperature or atmospheric gases, whereas discrete events such as fire and drought would exemplify pulse disturbances. Admittedly, the lines between the two can blur depending on the temporal and spatial scales considered. Grazing, for example, could be a press disturbance (if it is long-term, continuous overgrazing) or a pulse disturbance (if it is transient, short-term, high-intensity grazing). Nevertheless, the press–pulse concept is a useful one, facilitating explicit consideration of the timing, frequency, and extent of disturbances or drivers of change.

We have adapted Collins and colleagues' (2011) framework to conceptualize WPE as a social–ecological problem, using the SGP as a case study (figure 3). Consistent with their original terminology, we refer to our framework as the *press–pulse WPE framework*. It links the social and biophysical domains through press and/or pulse disturbance influences on the provision of ecosystem services. More specifically, the press–pulse WPE framework conceptualizes a system driven by interactions among a number of external variables: press disturbances such as temperature, rainfall patterns, and anthropogenically manipulated grazing and pulse disturbances such as periodic drought, flood, fire, and brush management. Many of these disturbances are

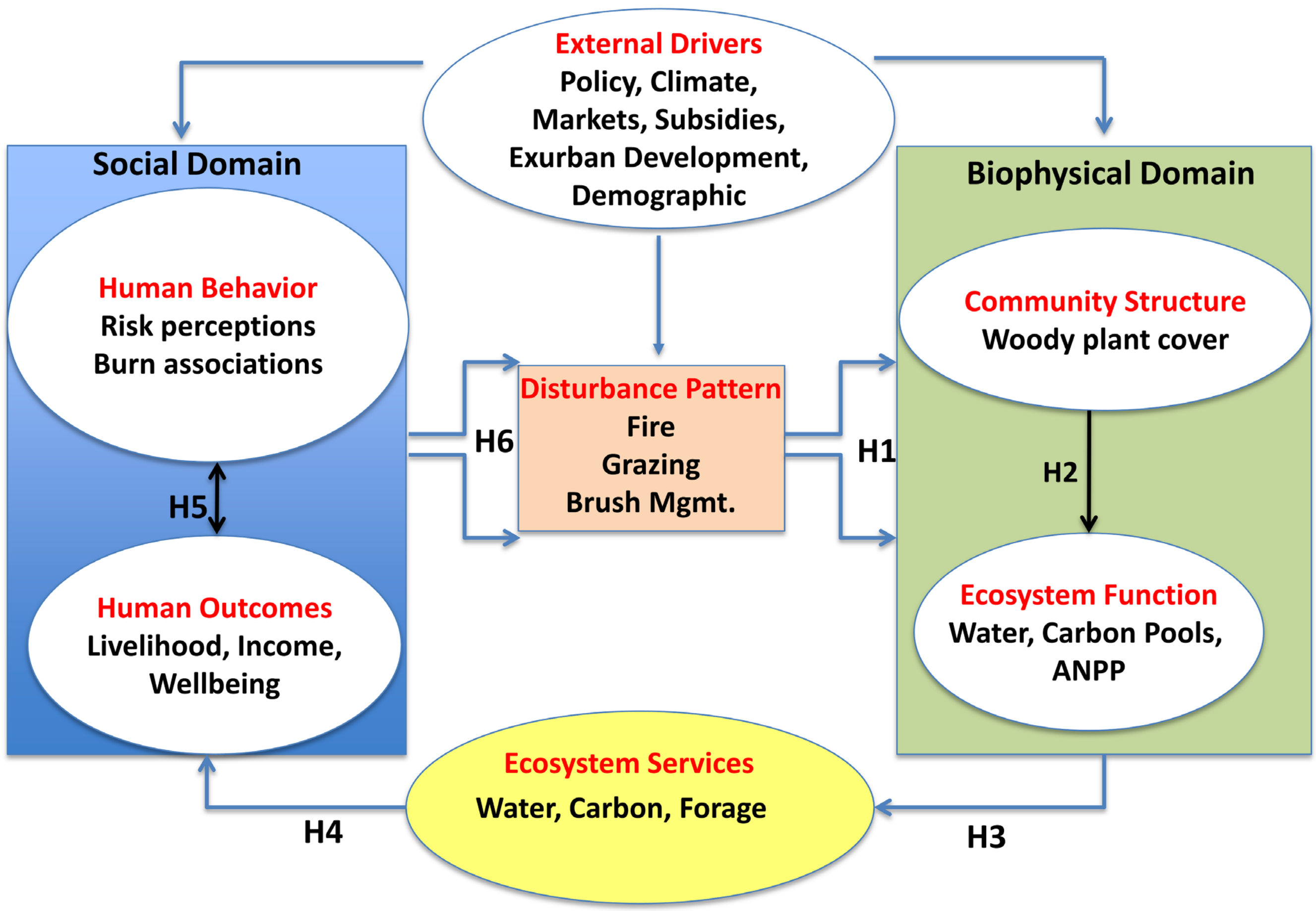

Figure 3. The press–pulse WPE framework, adapted from Collins and colleagues' (2011) press–pulse conceptual model. The social and biophysical domains are each influenced by various external drivers and are linked by anthropogenic-mediated disturbance (management) and the provision of ecosystem services.

triggered by human decisions that subsequently amplify or dampen their impacts. These decisions are, in turn, influenced by the ecosystem services provided by the biophysical landscape and by externalities such as urban expansion and fluctuation in the economic value of and demand for goods and services. Our framework acknowledges that rangelands are "open" systems but also systems in which human decision-making is integrated into endogenous variables such that the behavior of those variables cannot be reliably predicted or explained without consideration of the human dimension. We revisit this concept more formally in the latter part of the article.

The various components of the framework are connected by integrative hypotheses (H1–H6) that help direct long-term research agendas. The explicit formulation of these hypotheses not only enables more accurate identification of knowledge gaps but also aids the development of integrative tools for (a) better understanding WPE as a complex adaptive system with feedback loops, thresholds, nonlinear behavior, and emergent properties; (b) forecasting the extent and rate of spread of WPE under contrasting scenarios; and (c) evaluating how different policy options influence landowner willingness to use prescribed fire as a management tool. Although developed using the SGP as a case study, these hypotheses are relevant to many other grassland and savanna systems in which WPE is prevalent (the application of the framework to other locations will obviously need to incorporate relevant place-based particularities).

The integrative hypotheses, along with some of the key knowledge gaps related to each, are listed below. This list is not meant to be exhaustive but rather illustrative of the utility of the Collins and colleagues (2011) framework for conceptualizing WPE as a social–ecological problem.

Hypothesis 1: Land-use patterns drive and constrain woody-plant encroachment. There is general agreement among ecologists that WPE in semiarid and subhumid drylands has been driven largely by the elimination of fire from the system, both because overgrazing has diminished the amount and continuity of the fine fuel needed to propagate fire and because of active fire suppression (Walker and Meyers 2004). It follows that if grasslands and savannas are to be maintained and/or restored, fire must be incorporated as an endogenous part of the system. Although other factors are most certainly exacerbating WPE—such as the dissemination of woody-plant seeds by livestock, the eradication of native browsers (e.g., prairie dogs), increased atmospheric CO_2 concentrations, and a warming climate—these are secondary to the dramatic changes in fire regime (Archer et al. 2017).

With respect to the SGP, it is clear that woody-plant coverage has increased. Surprisingly, however, it is not well understood to what extent and at what rate this change has occurred. These two questions constitute the principal knowledge gap related to hypothesis 1.

Knowledge gap 1: At a regional scale, how much of the Southern Great Plains is currently under woody-plant coverage, and at what rate is coverage increasing? The increase in woody plants across the SGP has been well documented at a number of locations within the Edwards Plateau (Smeins and Merrill 1988, Diamond and True 2008), the Texas Rolling Plains (Ansley et al. 2001, Asner et al. 2003), central Oklahoma (Wang et al. 2018), and the Kansas Tallgrass Prairie (Briggs et al. 2002). However, as was highlighted by Barger and colleagues (2011), what is missing is a comprehensive assessment of total woody-plant coverage and the rate of spread across the region. Most of what we know is based on fine-scale studies carried out in specific areas where woody plants have been aggressively encroaching. Extrapolations based on such studies, therefore, will almost certainly overestimate both the rate and extent of regional changes. They will also fail to take into account constraints imposed by landforms (parent material and surface age), soils (primarily texture and depth), grazing intensity, and brush management, each of which is known to mediate the grass–woody plant ratio (Archer et al. 2011). Therefore, what will be needed to fill this knowledge gap are studies designed to quantify the rate and extent of changes in woody-plant cover across the region in a manner that considers variations in these factors (spatial and temporal) and in their interactions.

Hypothesis 2: Changes in ecosystem structure alter ecosystem processes. A shift from a grassland to a woodland state, or vice versa, fundamentally alters ecosystem processes—including water, energy, and biogeochemical cycles—which in turn brings about changes in productivity, biodiversity, and carbon storage (Eldridge et al. 2011, Archer and Predick 2014). At the same time, contrary to common perceptions, WPE does not necessarily lead to declines in functions related to productivity, biodiversity, and carbon storage and is not always synonymous with degradation or desertification (Eldridge et al. 2011, Archer et al. 2017). This is particularly true in the SGP, where rainfall is high enough to support good vegetation cover in both the canopy and intercanopy patches.

Knowledge gap 2: What are the long-term implications of woody-plant encroachment for soil organic carbon? Although many knowledge gaps remain with respect to how WPE alters ecosystem function (e.g., productivity and nutrient pools and fluxes), some consistent patterns have emerged (Archer et al. 2011). For example, WPE generally leads to higher aboveground productivity, higher evapotranspiration, and lower diversity of plant species. Much less is known, however, about the extent to which WPE affects belowground organic carbon pools (which in drylands dwarf aboveground pools). This knowledge gap is particularly important because it is the key to determining whether WPE-affected landscapes become carbon sinks or carbon sources.

In rangeland systems, the lion's share of carbon by far is in the soils. But unlike aboveground carbon storage, which consistently increases with WPE, trends in soil organic carbon (SOC) are highly variable, likely because of differences in shrub attributes (evergreen versus deciduous, N-fixing versus non-N-fixing, shallow- versus deep-rooted), climate, soil properties, and past land management (Archer et al. 2011). To address this knowledge gap, studies must be devised that will provide us with a mechanistic understanding of how WPE alters the inputs, chemical composition, and stability of SOC (DeMarco et al. 2016).

Hypothesis 3: Woody plant encroachment–driven alterations in ecosystem structure and function will alter the provision of ecosystem services. Rangelands in the SGP provide a broad suite of ecosystem services that directly benefit not only the people who derive their livelihood from these lands but the general public as well. Given the importance of these ecosystem services, learning how and to what extent WPE might alter them is a critical part of formulating land management policies (Archer and Predick 2014).

Knowledge gap 3: What are the social, economic, and ecological tradeoffs in the provision of forage, water, and carbon resources under various WPE and management scenarios? Forage production and wildlife habitat are among the most important of the various ecosystem services that directly affect livelihoods. Other important services are recreation potential and aesthetic qualities. As we noted above, WPE does not necessarily lead to a degradation of ecosystem function. Certainly, in arid areas, WPE often represents a form of desertification, but in areas where rainfall is higher, an increase in shrubs and trees may enhance primary production and nutrient cycling (Archer 2010). Furthermore, in some cases, an increase in woody plants has helped to restore watershed function by both improving infiltration capacity and decreasing erosion (Wilcox et al. 2008b). In other words, WPE does alter ecosystem services, but the desirability of these changes depends on the perspectives and the priorities of stakeholders. It is for this reason that we must develop a much better understanding of the tradeoffs involved under different scenarios of WPE-driven changes in ecosystem services.

Hypothesis 4: The well-being of rural landowners is affected by changes in ecosystem services. To motivate adaptive behaviors, we will need to understand landowners' perspectives regarding the risks associated with a given adaptive behavior. These include both readily identifiable risks (e.g., how WPE might affect forage production, how the cost of reducing woody-plant abundance might affect profitability, and what the liability risks associated with the use of fire are) and risks that are less obvious or more contentious (e.g., how WPE affects water supply). For private lands, individual landowner behaviors based on these perceptions of risk will aggregate to determine the patterns of disturbance that characterize a landscape (Kreuter et al. 2004).

Knowledge gap 4: To what extent are perceptions of vulnerability influenced by perceptions of biophysical processes and the benefits they provide? One way of assessing perceived vulnerability to WPE is to examine landowners' mental models concerning ecological processes and the risks associated with them (Santo et al. 2015). Although measured at the level of the individual, mental models provide insight into shared beliefs and knowledge about risk at the group and community levels. In the case of WPE, mental models shed light on when, how, and to what extent people recognize changes in ecosystem services as affecting human outcomes (e.g., livelihoods) and to what extent they are likely to take action to sustain livelihoods, either by altering the disturbance dynamics that lead to those changes or by guiding transformation to a "new" stable state.

A deeper understanding of landowner mental models will enable us to (a) identify beliefs about the risks associated with WPE and (b) compare and contrast lay and expert beliefs as a basis for communicating information that addresses unscientific beliefs, reinforces scientifically accurate beliefs, and minimizes beliefs that are correct but peripheral to the issue (Gentner 2002). This approach will help us understand not only how laypeople perceive biophysical processes and the role of management interventions but also their level of trust in governing institutions (Jones et al. 2011).

Hypothesis 5: Individual and collective behavior with regard to land use will be guided by effects on livelihood. Land-cover change is occurring against the backdrop of rural restructuring (Gosnell and Abrams 2009), wherein the perception of stable cultural norms and values (i.e., the rural lifestyle) drives immigration to rural areas from urban areas (Brown and Kandel 2006). Immigration of this kind increases the proportion of landowners who do not depend on the land for their livelihood. These landowners are less likely to have land management skills (Gill et al. 2010), and their perspectives about the value of grasslands versus that of woodlands often differ from those of traditional ranchers.

Knowledge gap 5: To what extent is the use of prescribed burning positively correlated with the level of dependency on grassland resources for generating income? In the SGP, the distribution of rural property sizes is increasingly bimodal. On one hand, both the number of large properties formed by the amalgamation of smaller tracts and the number of smaller properties resulting from land subdivision are increasing, resulting in, on the other hand, a decrease in the number of medium-sized family farms and ranches (Kjelland et al. 2007). Whereas more resource-dependent owners of larger properties may prefer open grasslands for livestock and native wildlife populations, landowners who have little reliance on their grasslands to support their livelihood may prefer the hunting opportunities and seclusion provided by a higher density of trees and may be less aware of the ramifications of changes in plant composition for other ecosystem services. These shifts in land ownership and in land-use preferences are likely to directly and strongly influence the extent to which prescribed fire is adopted as a conservation management tool, even though it may be the sole economically feasible means of restoring and maintaining existing remnants grasslands. In other words, the sensitivity of resource users to changes in resource conditions is critical for understanding perceptions regarding WPE and the use of prescribed fire (Marshall 2011).

Hypothesis 6: Patterns of landscape disturbance are influenced by the extent of landowner adoption of prescribed burning. Predictable and unpredictable human behavioral responses influence the frequency, magnitude, and form of press and pulse disturbance regimes. Landowner decisions regarding prescribed fire are influenced by three key factors. First, landowners are concerned about legal liability in the event of prescribed fire accidents. Recent studies in Texas and Oklahoma demonstrated that liability concerns are a major reason for the reluctance of private landowners to use prescribed fire on their land (Kreuter et al. 2008). More landowners elect to use prescribed fire in states with gross negligence liability standards than in states with simple negligence standards (Wonkka et al. 2015). The second key factor in decision-making is landowner attitudes toward fire. In many areas, landowners are influenced by social pressures (subjective norms) that discourage the use of fire (Toledo et al. 2014). The third key factor is knowledge of how to use prescribed fire. Prescribed burn associations (PBAs) are networks of landowners that provide members with fire safety training, equipment, and experienced assistance to safely carry out prescribed burns on their land. Since their inception in Texas in 1997, PBAs have proliferated across many Great Plains states to the north (Twidwell et al. 2013).

Knowledge gap 6a: To what extent is the decision to use prescribed burning as a standard management tool mediated by personal and/or social norms that alter perceptions of risks and benefits? The regular use of fire requires acceptance of a certain level of risk—namely, that the hazards can be safely controlled and that the benefits of the action outweigh the risks (Toledo et al. 2012). We propose that an individual's willingness to employ fire on a regular basis is primarily motivated by a

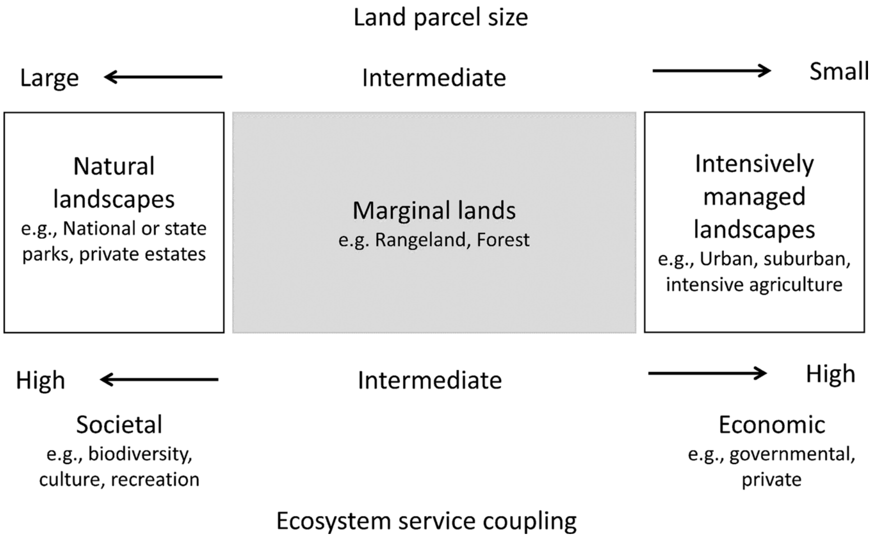

Figure 4. The coupling between ecosystem services and human decision-making is weaker on rangelands and other marginal landscapes than on (a) more intensively managed landscapes that provide a high economic return or (b) publicly or privately held natural landscapes wherein society has placed a high value on noneconomic ecosystem services, such as open space, biodiversity conservation, wildlife, and watershed functioning.

personal conviction that it is "the right thing to do" (Harland et al. 1999). This personal norm is activated by an awareness of the consequences of burning (and of not burning) and by a perception that burning effectively reduces a threat (Stern 2000). The norm is then reinforced by a perceived high level of personal control over the behavior, social pressure to adopt fire as a management tool (Ajzen 1991), affect (in this case, the positive feelings associated with the behavior; Finucane et al. 2000), and direct experience (Heberlein 2012). We expect that personal norms will explain variances in risk perceptions on the part of landowners having previous experience, whereas for landowners with little to no previous experience, the willingness to adopt fire as a practice is mainly influenced by affect (in this case negative feelings) and social norms rather than by a personal norm. That is, inexperienced landowners use their feelings as a way of simplifying the decision-making context. Normative pressures to engage in prescribed burning are more likely to come from outside sources (social norms) than from internal ones (personal norms) (Finucane et al. 2000).

Knowledge gap 6b: To what extent is the adoption of prescribed burning facilitated through voluntary organizations (prescribed burn associations) that influence personal and/or social norms of behavior? The extent to which PBAs can be effective tools for changing attitudes and behavior is uncertain, although we expect that their influence is important. We hypothesize that PBAs lower the social threshold concerning the adoption of fire as a management tool and engender a culture of burning by influencing perceptions of risk and benefits through the construct of social trust (Cvetkovich and Löfstedt 1999). Group membership creates a shared identity that fosters pride in the group, trust, and reciprocity, leading individuals to prioritize group interest over self-interest (De Cremer and Van Vugt 1999) and to behave in ways that coincide with group standards (Van Vugt and Hart 2004). Prescribed burn associations take a participatory, democratic approach that focuses on shared ownership and responsibility (Wondolleck and Yaffee 2000) and provides peer-to-peer learning opportunities (Kreuter et al. 2008).

Assessing the press–pulse framework for evaluating woody-plant encroachment as a social–ecological system

Our adaptation of the press–pulse dynamics framework was useful in identifying the social–ecological interactions that are key drivers of WPE. It has also proved useful in the development of integrative hypotheses and identification of key knowledge gaps.

For example, application of the press–pulse WPE framework to the problem of WPE has made clear that our knowledge and understanding of the social domain are lagging far behind those of the biophysical domain. This is true with respect to not only the SGP but also other grasslands and savannas undergoing WPE. Particularly challenging in this regard is the difficulty of recognizing clear and direct links between ecosystem services and human decision-making, which we attribute to several factors. For one, rangelands are often economically, politically, and ecologically marginal landscapes relative to more intensively managed landscapes (figure 4). Rangelands are marginal in terms of traditional goods and services, whereas nontraditional goods and services (e.g., water, biodiversity, open space, and wildlife) remain undervalued (Sayre et al. 2012, 2013). Furthermore, because an increasing number of landowners do not rely on their land for income generation, they lack the traditional economic incentive to manage woody plants for livestock forage production (Sorice et al. 2014, Hurst et al. 2017).

Further weakening the coupling between management action and ecological response is the reality that rangelands change in response to slow variables, which means that time lags may be long, especially if post-treatment environmental conditions (e.g., rainfall) are unfavorable. As a consequence, it may be years before landowners experience the ecological effects (be they positive or negative) of land management decisions. The combination of slow driving variables and lagged feedback processes results in a system that is highly nonlinear and challenging to evaluate using nondynamic frameworks—even though these characteristics also make these rangelands a very interesting subject for SES research.

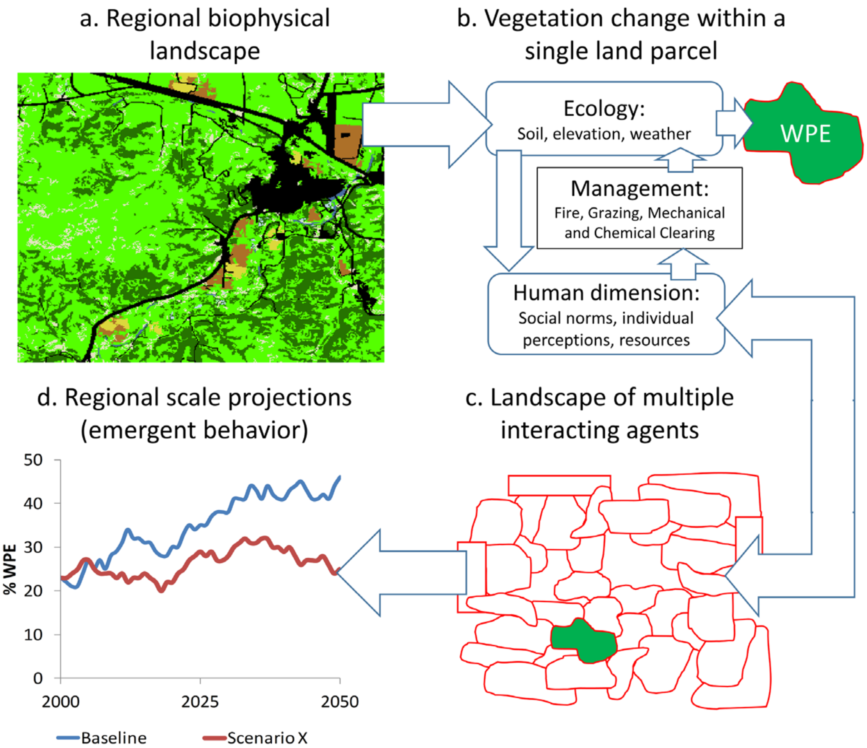

Figure 5. A conceptual diagram of agent-based models (ABM). (a) The model simulates landscapes at the regional scale using land-use, soil, elevation, and weather data. (b) The biophysical and social components of the model are linked by the management actions of fire, grazing, and brush clearing. (c) Interactions between agents drive both the ecological and socioeconomic decisions of each agent. (d) Multiple interacting agents generate emergent, regional-scale patterns for features of interest (e.g., WPE, carbon, and hydrology). The effects of perturbations to the system can be explored through scenario analysis.

Like other SES frameworks, therefore, the press–pulse WPE framework has three limitations: (1) It is not particularly helpful in identifying and clarifying the nature and strength of reciprocal feedback loops between social and ecological actors (Hruska et al. 2017). (2) It does not include a mechanism for assessing the influence of social and ecological processes on (a) the rate and consequences of WPE or (b) the effectiveness and results of management activities aimed at reducing WPE. And (3) it does not identify the emergent properties of the SES. To overcome these limitations, complementary approaches that are more quantitative and dynamic will need to be developed.

Integrative approaches for linking the social and biophysical domains

A particular challenge for social–ecological research is quantitatively linking the social and biophysical domains. Success in meeting this challenge would enable us to (a) better understand WPE as a complex adaptive system with feedback loops, thresholds, nonlinear behavior, and emergent properties; (b) forecast the extent and rate of spread of WPE under contrasting scenarios; and (c) evaluate how different policy options influence landowner willingness to use prescribed fire as a management tool.

On the biophysical side, we have a rich assortment of models and modeling approaches for exploring and extrapolating a range of processes (e.g., carbon models, hydrology models, and land-cover models). But until recently, modeling approaches for helping us understand how social decision-making relates to these processes have been rather limited.

Agent-based modeling. Over the last decade, agent-based models (ABMs) have increasingly proved to be a powerful approach for modeling human decision-making, through dynamically linking social and environmental processes (Groeneveld et al. 2017). Might an ABM then unlock new insights into the WPE phenomenon? And if so, how would it be configured?

To our knowledge, no attempt has yet been made to develop an ABM for WPE. Such a model would need two core components: (1) a biophysical one capable of simulating changes in woody-plant cover and (2) a socioeconomic one capable of simulating human decision-making processes in a way that translates the current perceptions, beliefs, and resources of each landowner into management actions that affect the biophysical variables. With this approach, owners of individual land parcels would be the logical “agents” (figure 5). Decision-making, in other words, would be at the scale of an individual land parcel, and interactions between agents would drive both the ecological and human-dimension decisions of each agent. In this way, interacting agents could potentially generate emergent, regional-scale patterns for woody plants.

A preeminent challenge in developing a credible ABM for understanding WPE is how to model human decision-making. The socioeconomic component would need to incorporate three central considerations: (1) If landowners do not understand the ecological role of fire in maintaining grasslands, this lack of knowledge could lead to decisions that promote WPE. (2) Land managers’ perceptions and their economic resources are affected by positive and negative feedback processes, including personal experience with prescribed burns, regional social norms, availability of economic resources for prescribed burning, and logistic or legal support for burning (e.g., access to PBAs).

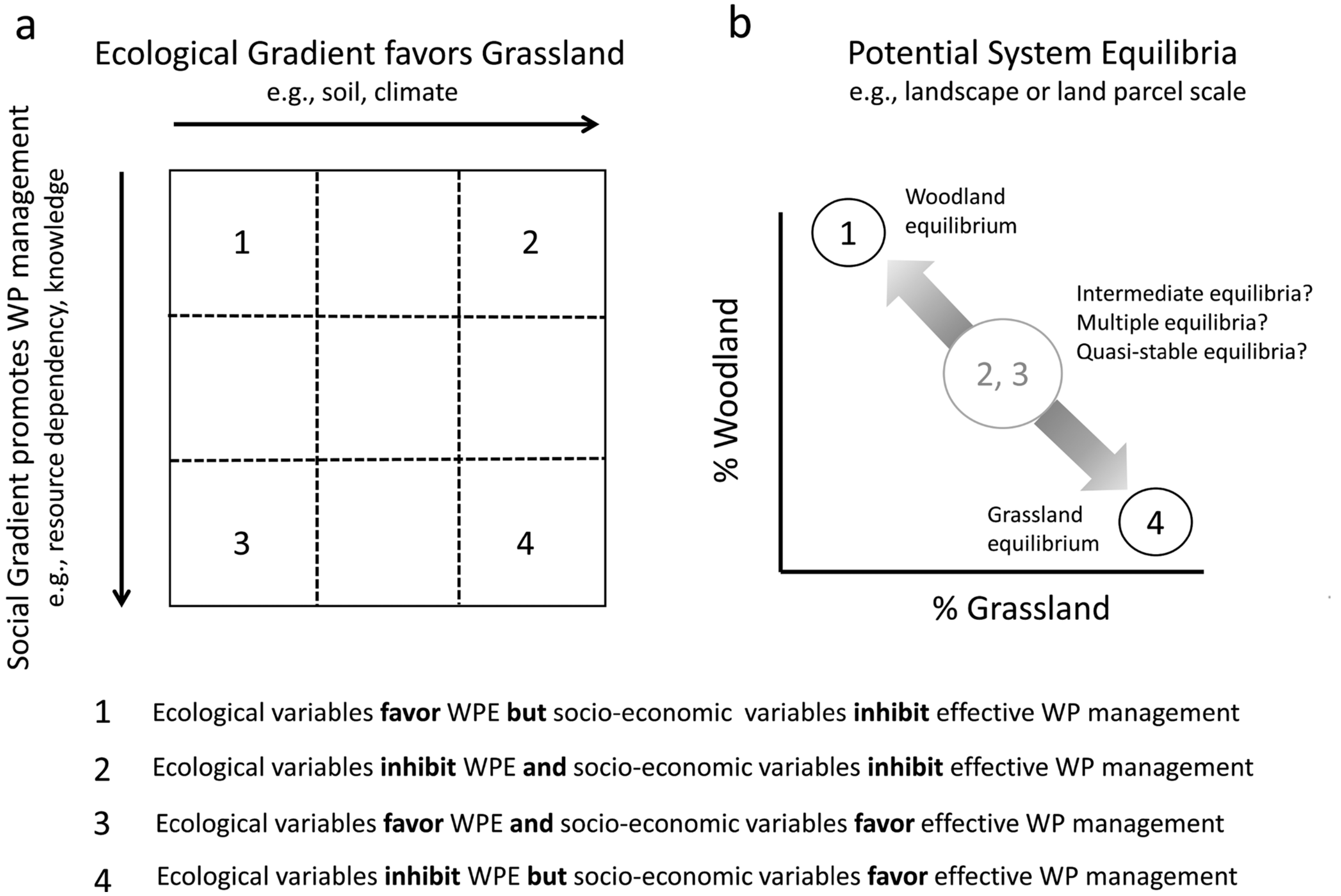

Figure 6. Rangelands undergoing WPE are social–ecological systems in which interactions between social and ecological variables determine the extent of woody-plant cover and the state of the ecosystem. (a) The numbers 1–4 represent the potential ecological states for different combinations of management and ecological variables that favor either woodlands or grasslands. (b) The resulting management and ecological constraints will determine where on the grassland or woodland continuum a given rangeland may lie.

(3) Individuals make decisions on the basis of perceptions and motives, and on the "visibility" of information. The information most "visible" to land managers is the current economic and ecological state of their operation, and that information will likely drive the majority of their decisions. However, informational feedback loops over larger spatial and temporal scales (driven by motives such as conservation of ecosystem resources, sustainability of an operation, and adherence to social norms) also play an important role in decision-making. We believe that the ABM approach may be useful for gaining deeper insights into how WPE can be understood as a social–ecological system.

Systems modeling. An alternative or even complementary approach for linking the social and biophysical domains is that of systems modeling, with its capability to incorporate reciprocal feedback loops between ecological variables and human decision-making. For example, as is highlighted in figure 6, the extent of woody-plant cover is a function of both the ecological capacity to support woody plants and the propensity of land managers to promote woody-plant management. In other words, both the equilibria and trajectories of WPE are driven by the reciprocal interaction between biophysical and social factors. Two systems can be similar with respect to woody-plant cover but for entirely different social and ecological reasons.

The immediate problem confronting social–ecological research is that of bridging the fields of ecology, psychology, and economics. We propose that the immediate goal of SES research should be to develop models based on sound scientific theories derived independently from the different disciplines. Headway can be made only if these theories can be described simply and with a generality that can be grasped by collaborators in each field. The key to developing a dynamic framework for evaluating social–ecological systems is formulation of a simplistic representation of an ordered but coupled system.

Figure 7 illustrates different ways in which rangelands could be modeled via a systems-modeling approach. The process consists of parsimoniously identifying features of

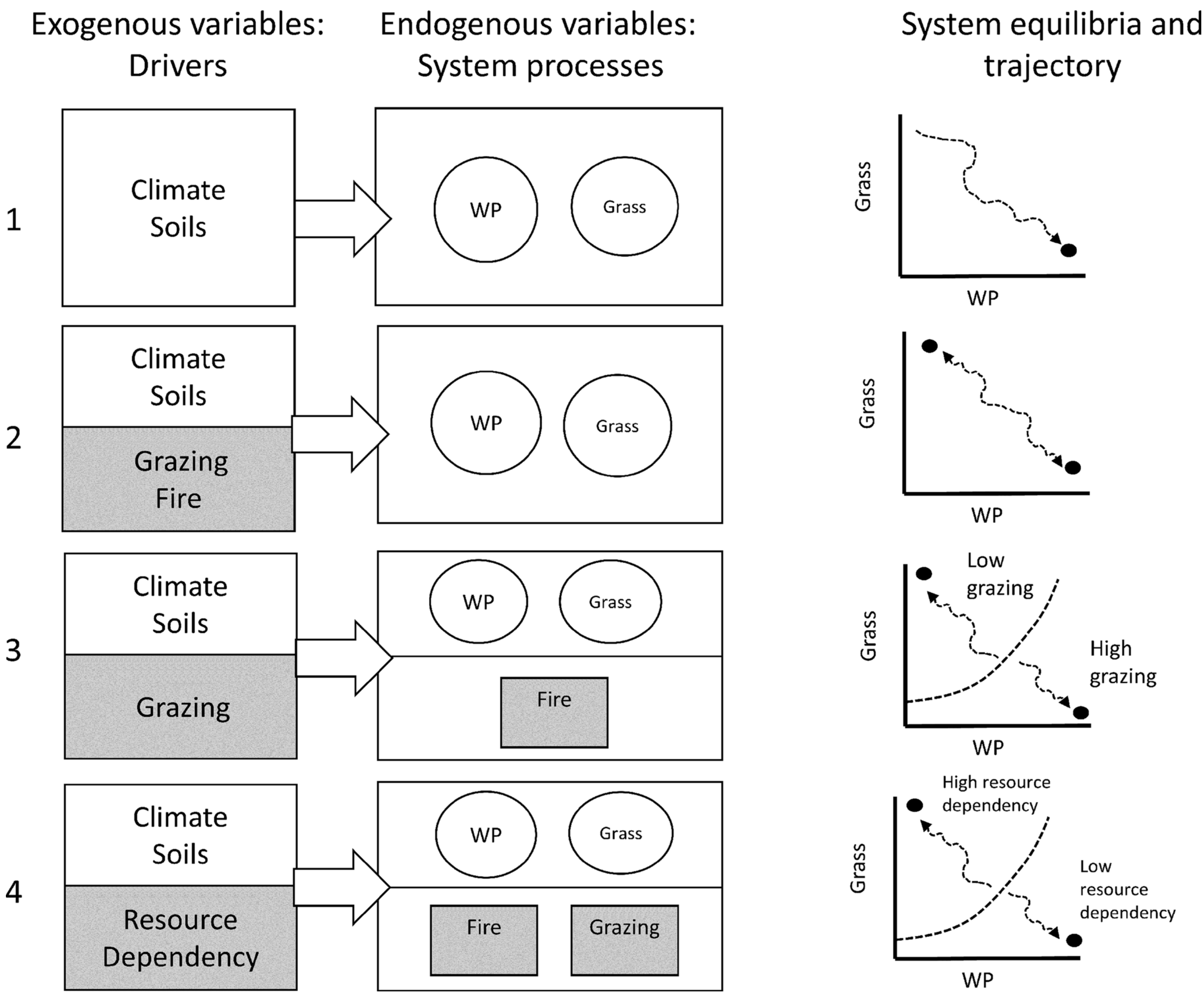

Figure 7. The phases of the systems-modeling approach for conceptualizing rangelands as a social–ecological system. This approach categorizes variables as either exogenous or endogenous. In phase 1, climate and soil variables are modeled as exogenous (immutable) properties of the system that drive the interaction or competition between woody plants and grasses. In phase 2, grazing and fire are added as exogenous drivers. In phase 3, fire is modeled as an endogenous process that is driven by grazing and influences the interaction or competition between grasses and woody plants. Phase 4 represents an explicit social–ecological system in which fire and grazing are conceptualized as endogenous variables driven by an exogenous social driver, such as the dependency of a land manager on resources from the land.

the system that are of interest—that is, as being exogenous and endogenous state variables—effectively "bounding" the system on the basis of questions the model is designed to address. The next step is characterizing these state variables either as *exogenous* (fixed and immutable variables that drive the system but are not affected by it) or as *endogenous* (variables that are internal to the system, reciprocally interacting with other endogenous variables and with exogenous variables).

At the most basic level (phase 1), an ecological system can be modeled as having no social component; it is modeled and/or conceptualized solely as interactions between ecological entities and external abiotic drivers. There are thus no reciprocal links between the state of the system and decision-making. Traditionally, conceptualizations of this kind have been used by ecologists to understand a system in biophysical terms alone. For example, climate and soils are modeled as exogenous factors (immutable properties of the system) that drive the competition between woody plants and herbaceous vegetation. The exogenous drivers are "fixed" in the sense that although they may vary with time (e.g., weather conditions may change on a daily basis) or with location, the model formulation contains no mechanisms by which they can be altered by endogenous state

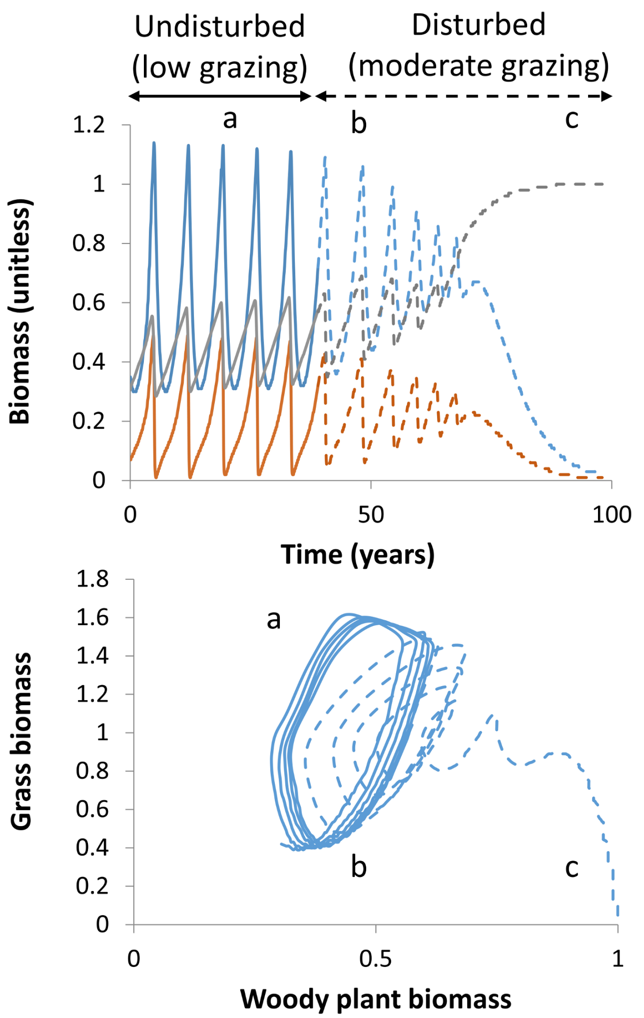

Figure 8. Output of the Anderies and colleagues (2002) model, showing alternative stable states for a savanna system. The top panel illustrates the temporal dynamics predicted by the model before and after a disturbance (solid and dashed lines, respectively), the disturbance being an increase in grazing: (a) the dynamic grassland state; (b) the transition triggered by the disturbance; (c) the stable shrubland state. The bottom panel shows a phase plane plot of the same data.

variables (e.g., woody plants and herbaceous vegetation). As it stands, phase 1 modeling is useful for understanding how the competition between woody and herbaceous vegetation—and therefore the dynamics of the system—might be influenced by soil and/or weather.

In phase 2 (figure 7), the ecological state of the system may be conceptualized as driven by factors that are within the control of humans but are still "external" or exogenous drivers. This phase does not take into account reciprocal interactions between the biophysical and social domains but does acknowledge that humans can have direct effects on the ecological state of the system. For example, phase 2 includes grazing and fire as two of these exogenous drivers—both potential results of human decision-making and as such influencing the dynamics of the system. The output of the model might clearly show that manipulation of grazing and fire in the system could greatly affect its long-term dynamics (the relative abundances of grasses and woody plants). But common sense would suggest that although incorporating elements that are human driven to some extent, phase 2 modeling does not simulate a true social–ecological system.

In phase 3, the system is conceptualized as having a single exogenous human driver. For example, grazing has complex and indirect effects on the system and is modeled as an exogenous process (one that could be altered by a land manager), but fire is modeled as an endogenous process that is driven by grazing and by reciprocal interactions between grasses and woody plants. Anderies and colleagues (2002), for example, have developed a simple phase 3–type model whereby fire is driven by the accumulation of fine litter in the system, which in turn is driven by both the relative abundances of woody plants and grasses and the intensity of grazing (heavy grazing prevents this accumulation). The outputs of the model, then, could be used to understand the trajectory of the system on the basis of a single management factor, such as grazing intensity (figure 8).

Phases 2 and 3 can be used to understand how humans can influence ecological systems but do not represent an explicit social–ecological system. A model that does would need to simulate a rangeland system in which fire and grazing are conceptualized as endogenous variables that are driven by a "fixed" exogenous social driver, such as the dependency of a land manager on resources from the land, as is depicted in phase 4 (figure 7). Phase 4 represents the most integrated social–ecological state, wherein the same biophysical variables drive the system along with a key exogenous social variable (e.g., resource dependency). This social driver determines ecologically relevant, endogenous management variables, such as fire and grazing, which are reciprocally related to the abundances of woody plants and grasses. To date no such model has been developed, but the output of such a model could help us understand how grazing, fire, woody plants, and grasses function as properties of a system under different conditions of resource dependency.

Conclusions

Our adaptation of the press–pulse framework has proved useful not only for identifying the social–ecological interactions that are key drivers of WPE but also for developing integrative hypotheses and identifying key knowledge gaps. For example, application of the press–pulse WPE framework to the problem of WPE has made clear that our knowledge and understanding of the social domain are lagging far behind those of the biophysical domain. Particularly challenging in this regard is the task of making clear and direct links between ecosystem services and human decision-making.

To advance our understanding of WPE as a social–ecological problem, we need to improve our ability to quantitatively link social and ecological processes using models. Both agent-based modeling and more classical systems modeling are potential approaches for accomplishing this aim, although much work remains. Agent-based models have been effectively used to address a wide array of SES-type problems (An 2012, Rammer and Seidl 2015, Groeneveld et al. 2017). Classical systems modeling has been used to address WPE, as was shown by the work of Anderies and colleagues (2002). However, we believe that the Anderies approach can be taken further by more clearly identifying key exogenous social drivers that influence the system (in this case, how “resource dependency” may influence endogenous variables such as grazing intensity and fire, and how these variables in turn influence woody plant–grass dynamics).

A key challenge lies in reconciling and integrating key theories and ideas from both ecology and economics. Unless this challenge can be overcome, both quantitative and qualitative models of SES can become complex, difficult to analyze, and therefore of limited use for developing and testing novel theories. In our view, the language and methods of classical systems modeling facilitate effective collaboration between researchers from both fields, enabling ideas and theories to be shared and providing insight into how social–ecological systems operate over long temporal and spatial scales.

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