



REVIEW

Strategies for global rangeland stewardship: Assessment through the lens of the equilibrium–non-equilibrium debate

David D. Briske¹  | D. Layne Coppock² | Andrew W. Illius³ | Samuel D. Fuhlendorf⁴

¹Department of Ecosystem Science & Management, Texas A&M University, College Station, TX, USA

²Department of Environment and Society, Utah State University, Logan, UT, USA

³School of Biological Sciences, University of Edinburgh, Edinburgh, UK

⁴Natural Resource Ecology and Management, Oklahoma State University, Stillwater, OK, USA

Correspondence

David D. Briske

Email: dbriske@tamu.edu

Handling Editor: Kechang Niu

Abstract

1. This Review assesses the adequacy of current stewardship strategies to address the accelerating challenges confronting global rangeland systems. The assessment was conducted through the lens of the rangeland ecology debate that was initiated in the late 1980s to determine whether equilibrial or non-equilibrial models more accurately represented the ecological dynamics of rangeland systems.
2. The following lessons have emerged from this prolonged debate: (a) equilibrial and non-equilibrial dynamics among plants and herbivores always coexist in individual systems; (b) herbivore persistence is strongly influenced by functional resource heterogeneity; (c) biotic feedbacks are highly dependent upon spatial and temporal scale; (d) multiple stable states separated by nonlinear trajectories may coexist on individual ecological sites and (e) management and policy decisions can have important consequences for rangeland systems despite high environmental stochasticity.
3. These lessons demonstrate that scale and functional heterogeneity were critical 'blind spots' within the traditional rangeland profession. In contrast, pastoral strategies recognized these system variables as essential for exploitation of heterogeneous resource distribution, rather than assuming stable, uniform resource distribution. These divergent perceptions of scale and heterogeneity in grazed ecosystems largely explain the disparate interpretations of ecological carrying capacity that was central to the debate.
4. Programmes supporting supplemental feeding and infrastructure development have created an anthropogenic category of non-equilibrial dynamics by sustaining large numbers of livestock during droughts. These programmes decouple livestock and rangeland resources to create an additional stewardship dilemma regarding complex trade-offs between livestock production and the capacity of systems to provision a broad array of global ecosystem services.
5. *Synthesis and applications.* The fundamental challenge facing the global rangeland community may not be identification of a unified model of rangeland ecology—as assumed during the rangeland debate. Rather, the challenge may be how to best transform rangeland social–ecological systems to provide optimal combinations of ecosystem services to meet the needs of global citizens, while improving the well-being of millions of rangeland residents who are highly dependent upon

provisioning services. A comprehensive accounting of rangeland ecosystem services, supported by institutional governance and delivered as state–community partnerships, may provide the foundation for an alternative stewardship strategy to pursue this critical goal.

KEY WORDS

carrying capacity, drylands, equilibrium–non-equilibrium debate, functional resource heterogeneity, grazed ecosystems, pastoralism, rangeland ecology, rangeland management

1 | INTRODUCTION

Drylands are a major land cover type that have global implications for the provision of diverse ecosystem services and human well-being. Drylands, 69% of which are rangelands, occupy 41% of the Earth's land area (6 billion hectares) and support 2 billion people (Millennium Ecosystem Assessment, 2005). Global rangelands supply numerous provisioning, regulating, supporting and cultural services, including the production of 50% of all livestock, storage of 30% of the global C pool and eight of 25 biodiversity hot spots, including a high diversity of large mammal species; numerous World Heritage sites, and 24% of global languages (Millennium Ecosystem Assessment, 2005). Despite the existence of diverse ecosystem services, the human inhabitants of drylands lag far behind the rest of the world in terms of socio-economic well-being, especially in the developing world.¹

Global rangelands are undergoing accelerated ecological and social changes in the 21st century. This is especially evident for the developing world, where the most important drivers of change include: (a) human population growth and the associated unsustainable use of natural resources (Holechek, Cibils, Bengaly, & Kinyamario, 2017; Petz et al., 2014); (b) climate-induced warming and drying with loss of surface water (Huang, Yu, Guan, Wang, & Guo, 2016; Koutoulou, 2019); (c) accelerated land conversion and fragmentation (Reid, Fernández-Gímenez, & Galvin, 2014) and (d) reduced effectiveness of traditional commons governance (Coppock et al., 2017; Godber & Wall, 2014). Rangelands in the developed world are also confronting major challenges, with priority issues including: (a) increasing atmospheric warming and climate variability (Huang et al., 2016; Koutoulou, 2019); (b) exurban development (Hansen et al., 2005); (c) societal preference for non-provisioning services (Holmes, 2002; Yahdjian, Sala, & Havstad, 2015); (d) ageing and slow intergenerational replacement of ranching populations (Peterson & Coppock, 2001) and (e) low profit margins for rangeland livestock production (Fitzhardinge, 2012; Holmes, 2015).

Effective strategies are needed to promote wise stewardship of global rangelands, enhance well-being of rangeland residents, and to provision the full suite of rangeland ecosystem services for societal

benefit. This raises a series of critical questions for the professional rangeland community. Do we have sufficient capacity to effectively address the pressing challenges identified above? If not, where do the greatest deficiencies exist? How can these deficiencies be corrected, and who is responsible? We explore these questions through the lens of the rangeland equilibrium–non-equilibrium debate that began in the late 1980s. This debate was focused on identifying the appropriate ecological model that could successfully guide rangeland management and policy.

This Review is organized around four broad goals. First, the rangeland ecology debate is briefly summarized to identify the lessons learned. Second, we examine the adequacy of these lessons to inform the stewardship strategies required to address the accelerating challenges confronting global rangelands. Third, an anthropogenic category of non-equilibrium dynamics between livestock and rangeland resources is identified. Finally, an alternative stewardship strategy is presented to contend with the scope and complexity of 21st century challenges.

2 | THE RANGELAND ECOLOGY DEBATE

2.1 | Debate origin

The rangeland ecology debate is grounded in a colonial narrative stating that pastoralism is backward, inefficient and environmentally destructive (Roe, Huntsinger, & Labnow, 1998; Warren, 1995). This perception of 'irrational pastoral behavior' contributed to implementation of 'science-based' pastoral development programmes in Africa beginning in the late 1950s to remedy these inefficiencies (Sayre, 2017). The debate emerged in the 1980s from a conceptual analysis of a pastoral ecosystem in South Turkana, Kenya, that sought to determine why these development programmes had failed (Ellis & Swift, 1988). This alternative interpretation of pastoral system function, which challenged major tenets of traditional range ecology, directly contributed to this ardent and prolonged debate.

Multi-year droughts are frequent in the South Turkana region and severe drought can reduce livestock numbers >50%. Consequently, Ellis and Swift (1988) hypothesized that drought, especially multi-year drought, kept livestock populations well below an ecological carrying capacity to maintain this system in

¹In this paper we broadly refer to pastoralism as the traditional or transitioning lifestyles found among rangeland residents in the developing world. We broadly refer to ranching as the commercialized lifestyles found among rangeland residents in the developed world.

a persistent state of non- or disequilibrium (NEP). The livestock population appeared capable of removing only 10%–12% of annual herbaceous production, and the study area lacked overt signs of livestock-induced environmental degradation. It was reasoned that highly dynamic patterns of herbaceous production were the result of inter-annual precipitation variation, rather than livestock grazing (Box 1).

A second landmark paper introducing the state-and-transition model (STM) framework was published the following year by Westoby, Walker, and Noy-Meir (1989). These authors advocated that rangeland management should be oriented to opportunistically respond to event-driven, nonlinear vegetation dynamics. They argued that shifts in the composition of plant communities could be driven by combinations of external events (i.e. fire, drought, invasive species, flooding, etc.) often unrelated to livestock grazing (Box 1). Consequently, the importance of stochastic abiotic events undermined the equilibrial argument that grazing alone was the primary contributor to vegetation change. This conclusion, however, had been recognized decades earlier by ecologists in the American southwest, but it was dismissed by the administrative authority of the U.S. Forest Service following adoption of the Clementsian successional model for rangeland assessment

(Sayre, 2017). The contribution of multiple abiotic drivers to vegetation dynamics became more plausible in the 1980s in response to growing criticism of ecosystem stability, and recognition of the existence of alternative stable states (Holling, 1973; May, 1977). In this context, the rangeland equilibrium–non-equilibrium debate, as introduced to the rangeland profession by the NEP and STM models, represented a subset of this broader ecological debate (Briske, Illius, & Andries, 2017).

Taken together, the implications of the NEP and STM models were antithetical to traditional range science by challenging the foundational concepts of ecological carrying capacity, and linear plant–herbivore dynamics (Briske, Fuhlendorf, & Smeins, 2003; Vetter, 2005). The NEP paradigm rapidly gained credence among scholars of pastoralism who were seeking insights to explain failures of pastoral development projects that had been based on traditional range science principles (Behnke, Scoones, & Kerven, 1993). It also gained considerable momentum as a paradigm to guide pastoral policy in the form of the ‘new rangeland ecology’. The STM paradigm rapidly gained acceptance among range scientists and range management agencies during the 1990s—especially in the United States and Australia—as an alternative model to Clementsian succession (Friedel, 1991; Laycock, 1991). The NEP and STM paradigms

BOX 1 Interpretive summary of the key concepts associated with the rangeland ecology debate

State-and-transition model (STM)—Stochastic abiotic variables, in addition to grazing, may cause abrupt, nonlinear changes in plant community composition. Sufficient disturbance may result in persistent transitions of existing stable states (i.e. plant communities) to alternative stable states on specific ecological sites (e.g. conversion of grassland to woodland). STM models are equilibrial because each stable state is organized around an individual equilibrium point, even though multiple stable states may potentially exist. The STM model, unlike the following NEP model, does not explicitly address forage production, herbivore numbers or their interaction.

Non-equilibrium persistent model (NEP)—Forage production and herbivore numbers do not appear to be directly coupled in grazed ecosystems with highly variable inter-annual rainfall because forage production recovers more rapidly than herbivore numbers following severe drought. Yet, in spite of this apparent mismatch (i.e. non-equilibrium) herbivores persist in these environments because they are coupled to ‘key resources’ during the dormant season, and not to the abundant forage during the growing season. All grazed ecosystems contain equilibrial and non-equilibrial components because herbivore diets reflect differential dependence on the diverse resources available.

Functional resource heterogeneity (FRH)—Herbivore population dynamics are critically dependent on selecting adequate resources during key life stages, such as reproduction, lactation and in particular, surviving the winter or dry season. Herbivores shift their diets seasonally according to the abundance and diversity of resources at local to landscape scales. Functional resource heterogeneity is the spatio-temporal variation in accessible resources that supports the vital rates determining herbivore population change. FRH distinguishes between resources that matter from the remaining heterogeneity that does not.

Carrying capacity (CC)—The fundamental question of how many animals an area can support for a specific management outcome is greatly complicated by inter-annual variation in primary production, as well as by the degree and scale of spatial heterogeneity. Herbivore population dynamics are disturbed by temporal variation, but buffered by spatial heterogeneity. In highly variable environments, CC can only be defined as a long-term mean, thereby limiting the practical applicability of the concept for managing rangeland systems. CC does, however, have heuristic value in the exploration of how herbivore populations exploit and depend on functional resource heterogeneity. The key challenge is to distinguish among resources that support herbivore vital rates and those which do not. Previous misapplications of CC should not obscure the dependence of herbivore populations on a subset of select forage resources, despite their inter-annual variability.

subsequently evolved in parallel in the rangeland profession, with NEP deemed most relevant to pastoral systems in developing nations while STMs were initially applied in developed nations where commercial ranching prevailed.

2.2 | Debate progression and resolution

In the three decades following inception of the debate, the NEP model has encountered major conceptual challenges and has been reinterpreted as the coexistence of equilibrium and non-equilibrium dynamics in individual systems (Briske et al., 2017). The STM framework has subsequently been recognized as an equilibrium model, and by 2000 it was widely implemented as a rangeland management tool in the United States and it has since been implemented in other nations (Bestelmeyer, Ash, Brown, Densambuu, & Fernández-Giménez, 2017).

2.2.1 | Non-equilibrium persistent model

A major ecological criticism of the NEP model is that it did not sufficiently consider the spatial and temporal heterogeneity of resource use by grazing animals (Illiis & O'Connor, 1999, 2000; Vetter, 2005). Large herbivores in most rangeland systems are regarded as being dynamically coupled to a small subset of 'key' resources in dry seasons (Illiis & O'Connor, 2000; Ngugi & Conant, 2008; Scoones, 1995); correspondingly, animals are largely uncoupled from abundant forage in wet seasons. These key resources are the critical parameters determining animal survival over periods of forage scarcity during dry seasons, winter or prolonged drought (i.e. dormant seasons), rather than forage abundance during the growing season (Box 1).

The contribution of resource heterogeneity to herbivore survival and persistence is now widely recognized (Fryxell, Wilmshurst, Sinclair, Haydon, & Abrams, 2005; Fynn, 2012; Owen-Smith, 2014). Resource heterogeneity is both inherent to rangeland ecosystems and generated by disturbances (Fuhlendorf, Fynn, McGranahan, & Twidwell, 2017). Inherent resource heterogeneity is expressed temporally via seasonality and climatic variability, and spatially among landscapes and topo-edaphic features, plant communities and species and individual plants and their tissues (Buttolph & Coppock, 2004; Orians & Jones, 2001). Disturbance-driven heterogeneity is established by grazing and fire patterns that often interact with inherent heterogeneity (Fuhlendorf & Engle, 2001; Fuhlendorf, Engle, Kirby, & Hamilton, 2009; Wang et al., 2019).

The key resource concept was strongly supported in a study of semi-arid South African rangeland (Hempson, Illius, Hendricks, Bond, & Vetter, 2015). Livestock (goats) body condition followed a density-dependent depletion of the very limited dry-season riverine vegetation—a key resource area—and annual demographic parameters of the animals tracked dry-season forage availability. The main

determinants of animal population trajectory were animal population size at the onset of the dry season and dry season duration. There was no clear evidence of biotic feedbacks between forage availability and animal populations during the wet season.

These investigations corroborate the existence of two broad categories of forage resources definable in terms of herbivore population dynamics: those resources that underpin key demographic transitions of survival and possibly recruitment, and those that largely do not (Illiis & O'Connor, 2000). The key resource concept applies to all critical resources, in any season, that support key processes of animal population performance (Parker, Barboza, & Gillingham, 2009; Pettorelli et al., 2003; Wang et al., 2006) and has been generalized as functional resource heterogeneity (Fynn, 2012). The exploitation of resource heterogeneity by herbivores is a ubiquitous feature of grazed ecosystems, all of which contain a critical subset of resources tending to regulate animal numbers, while the remaining resources lack feedbacks on demographic parameters. Therefore, grazed ecosystems are not distinguished by whether they are equilibrial or non-equilibrial, as was widely assumed, but by the spatial and temporal scale of resource heterogeneity and the relative abundance of these two forage categories. The dichotomous interpretation of systems as being either equilibrial or non-equilibrial diverted attention from the realization that these two components always coexist, and may have unnecessarily prolonged the debate.

Another major claim of the NEP model was that herbivores were unable to cause rangeland degradation because the overriding effects of periodic drought kept animal numbers below an ecological carrying capacity (Ellis & Swift, 1988).² This claim is difficult to corroborate given abundant evidence indicating that pastoral systems are often degraded to varying degrees, although the effects of livestock can be conflated with those of other human-related disturbances such as cultivation or wood harvest (Coppock et al., 2017). For example, approximately 73% of dryland soils have been environmentally degraded according to Millennium Ecosystem Assessment (2005). Other modelling work has substantiated these conclusions by indicating that chronic livestock grazing on the world's rangelands can decrease biodiversity, increase carbon emissions and enhance soil erosion (Petz et al., 2014). A global meta-analysis, however, indicated that zonal degradation (i.e. degradation independent of water and key resources) was absent in arid systems having an annual rainfall CV >33%; degradation was more common in mesic systems with low CVs or in situations where animals impacted sites in the vicinity of water or other key resources (von Wehrden, Hanspach, Kaczensky, Fischer, & Wesche, 2012). It is clear that all rangelands are susceptible to some degree of human- or livestock-induced degradation, and a

²Unique attributes also contribute to the resistance of the South Turkana rangelands to livestock impacts. For example, the soils are heavily dominated by sandy or rocky volcanic surfaces, while the herbaceous plants are largely annuals (Coppock, 1993). It has also been hypothesized that some plant communities in northern Kenya represent alternative stable states where herbaceous annuals have replaced herbaceous perennials in response to a long history of intensive livestock use in conjunction with extended droughts (Herlocker, 1999).

focus on the CV of annual rainfall does not account for spatial patterns of vegetation types, soils, distributions of key resources and other disturbances that underpin all contemporary forms of rangeland degradation.

2.2.2 | State-and-transition model

The STM framework was initially envisioned as non-equilibrium because '*...rangelands not at equilibrium*' appeared in the title of the Westoby et al. (1989) paper. The concept of multiple stable states for individual range sites represented a substantial departure from the emphasis previously placed on the dynamics of individual stable states. However, the authors of the original paper stated that the objective of STMs was identification of *multiple equilibria* to describe discrete stable states, and the possible transitions between them, for specific rangeland sites (Westoby et al., 1989).

In this framework, each stable state is organized around a single equilibrium point, with thresholds representing boundaries between alternative equilibria (Bestelmeyer et al., 2017; Briske et al., 2017; Petraitis, 2013). For example, woody plant encroachment into grasslands and savannas represents the conversion of one stable state to another that has organized around an alternative equilibrium point or basin of attraction (Ratajczak, Nippet, Briggs, & Blair, 2014). In some cases, management interventions may be able to re-establish the self-organizing capacity of the former stable state. The STM framework is able to account for the numerous pathways exhibited by vegetation dynamics, in addition to linear directional change, and to address drivers of vegetation change other than grazing (Bestelmeyer et al., 2017). They are widely envisioned as a tool to link science and management, rather than an ecological theory.

2.3 | Lessons learned

The rangeland ecology debate represented a comprehensive assessment of interrelationships among vegetation, herbivores, functional resource heterogeneity and spatial and temporal scales in climatically stochastic environments (Briske et al., 2017). The following lessons have emerged: (a) equilibrium and non-equilibrium dynamics among plants and herbivores always coexist in individual systems; (b) herbivore persistence is strongly influenced by functional resource heterogeneity; (c) biotic feedbacks are highly dependent upon spatial and temporal scale; (d) multiple stable states separated by nonlinear trajectories may coexist on individual ecological sites and (e) management and policy decisions can have important consequences for rangeland systems despite high environmental stochasticity.

These lessons have collectively contributed to a hybrid ecological model in which equilibrium and non-equilibrium dynamics simultaneously operate over various spatial and temporal scales. This outcome had been foreshadowed by both theoretical and empirical assessments throughout the debate. DeAngelis and Waterhouse (1987) posited that equilibrium was a property asymptotically derived from

extrapolation to successively larger scales. Sullivan (1996) emphasized the need to minimize duality between biotic and abiotic drivers of rangeland dynamics, as did Walker and Wilson (2002). Upon finding evidence of equilibrial dynamics in semi-arid grasslands of Mongolia, Fernández-Giménez and Allen-Diaz (1999) expressed concern that non-equilibrium models were being embraced with such enthusiasm that they were in danger of being misapplied just as equilibrium models had been.

These lessons demonstrate that the importance of scale and heterogeneity were unrecognized or undervalued within the traditional rangeland profession. Sayre (2017) refers to such deficiencies as 'blind spots' that emerged with the authoritarian (e.g. top down) approach that characterized the origins of range science in the United States. In contrast, pastoral strategies recognized these system variables as being essential for exploitation of heterogeneous resource distribution, rather than assuming the existence of stable, uniform resource distribution. In other words, pastoral strategies focused on system variability across large scales, rather than average conditions at small scales (Behnke et al., 1993; Kratli & Schareika, 2010). These divergent perceptions of scale and heterogeneity in grazed ecosystems may largely explain the disparate interpretations of ecological carrying capacity—the number of grazing animals that a range is able to sustainably support over the long-term—that was central to the debate (Behnke et al., 1993; Ellis & Swift, 1988).

Carrying capacity may represent the central blind spot of traditional range management because it was implicitly founded on the assumptions of mean annual NPP in landscapes characterized by homogenous vegetation cover—conditions that seldom exist in rangeland systems (Fuhlendorf et al., 2017; Sayre, 2008). Carrying capacity is clearly related to the availability of rangeland resources as shown by the positive correlation between animal abundance and primary production across productivity gradients (Fritz & Duncan, 1994; Oesterheld, DiBella, & Kerdiles, 2008). However, this broad equilibrial relationship belies the complexity associated with intra- and inter-annual precipitation variability and landscape heterogeneity to directly and substantially affect livestock production and persistence, especially over large spatial and temporal scales. Consequently, even though the concept of carrying capacity exists, temporal variability and resource heterogeneity—two defining characteristics of rangelands—make it impractical to apply to grazing management (Box 1).

These lessons may partially reflect the deficiencies that Ellis and Swift (1988) sought to identify as responsible for the failure of pastoral development programmes in Africa. The inability of traditional range science to improve pastoral systems, however, went far beyond insufficient ecological understanding. This failure has been described as the 'misapplication of developed world technology, management, and practices that had been injected into pastoral systems without questioning their need or value' (Poulton, 1984; Sayre, 2017). In many respects, traditional subsistence pastoralism and commercial livestock enterprises are diametrically opposed in terms of cultural values, production systems and objectives for natural resource management (Behnke, 1983; Coppock, 1994).

Progressive pastoral development projects today, in contrast to those of the past, emphasize more bottom-up participatory approaches with a priority on improving human welfare, diversifying livelihoods and facilitating livestock marketing (Coppock et al., 2017).

3 | LESSON APPLICATION TO RANGELAND STEWARDSHIP

3.1 | Scale and functional heterogeneity are critical

Functional heterogeneity has proven to be critical to livestock production and the provision of diverse ecosystem services, including biodiversity, and it arguably serves as the foundation for conservation and management of rangeland ecosystems (Fuhlendorf & Engle, 2001; Fuhlendorf, Engle, Elmore, Limb, & Bidwell, 2012; Fuhlendorf et al., 2017). It is widely acknowledged that most ecological investigations have been conducted at relatively small scales with the intent of limiting heterogeneity among and within sampling units (Wiens, 1989). This was clearly the case for U.S. range science which was based on quadrat and plot methods throughout much of the 20th century (Sayre, 2017). Plot data were then linearly scaled to larger areas without question or evaluation. Consequently, the importance of coupling heterogeneity to the scale of observation has gained broad acceptance within the ecological community over the past several decades.

The provision of rangeland ecosystem services is a function of heterogeneity in space and time to a greater extent than it is of average conditions across landscapes (Fuhlendorf et al., 2017). Management strategies designed to promote spatial heterogeneity have been demonstrated to stabilize cattle production during dry periods by establishing vegetation patches of varying species composition and forage maturity and quality (Allred, Scasta, Hovick, Fuhlendorf, & Hamilton, 2014; McGranahan et al., 2012, 2016). Multiple patches that have variable structure allow grazing animals to select the most suitable foraging patch for the existing conditions, rather than forcing animals to forage on homogeneous patches under all conditions. Yet, the rangeland profession continues to struggle with the ecological interpretation and management application of scale and heterogeneity. For example, the STM framework partitions these variables into specific ecological sites on the basis of topo-edaphic, climatic and vegetation homogeneity (Bestelmeyer et al., 2017), even though the value of spatially explicit models has been recognized (Bestelmeyer, Goolsby, & Archer, 2011).

Recognition that heterogeneity underpins herbivore persistence reaffirms the importance of key resource areas—in relation to the size and proximity of wet season forage resources—as being critical to sustainable production systems (Fynn, 2012; Ngugi & Conant, 2008). It is essential that the value of key resource areas be recognized because their loss exceeds the proportion of land area removed due to critical forage or water resources provided

(Buttolph & Coppock, 2004; Galvin, 2009; Scoones, 1995). For example, rangeland fragmentation, privatization, human overpopulation and conversion to alternative uses pose severe threats to the stability of rangeland production systems by reducing key resource areas (Clover & Eriksen, 2009; Coppock, 2016; Rohde et al., 2006). These accelerating drivers identify the need for substantial management and policy reform—for both pastoralism and ranching systems—that emphasize development of forage reserves through the creation of landscape heterogeneity to better support livestock during drought (Coppock, 2011, 2016).

The lesson that rangelands can exhibit multiple stable states has become widely accepted as evidenced by numerous examples of grasslands and savannas that have ostensibly undergone nonreversible transitions to woodlands on several continents (Archer et al., 2017; Eldridge et al., 2011). The concept of alternative stable states, and the thresholds that separate them, are effectively represented by the STM framework to support management decision-making. The fundamental challenge is the ability to distinguish between transient and permanent vegetation change—managerially unacceptable long return times (Bestelmeyer et al., 2017). In spite of wide adoption of STMs, management perceptions and actions often continue to adhere to the traditional reference of linear successional dynamics. For example, livestock grazing is perceived to be the dominant driver of state transitions, with the abiotic disturbances of fire and drought referenced much less frequently in STMs developed in the United States (Twidwell, Allred, & Fuhlendorf, 2013).

The coexistence of equilibrial and non-equilibrial dynamics in rangeland systems reaffirms that management and policy decisions can have important consequences in arid and semi-arid environments despite the occurrence of high environmental stochasticity. For example, degradation of pastoral systems in the arid African Sahel during droughts of the 1970s and 1980s has been partially attributed to pastoral sedentarization and provision of livestock supplementation during drought (Ward, Saltz, & Ngairorue, 2004). Increasing forage deficits and high cattle mortality in eastern Africa have been attributed to increasing drought frequency, but evidence suggests that accelerated grazing pressure or loss of forage reserves are also direct contributors to both outcomes (Coppock, 2016; Desta & Coppock, 2002; Western, Mose, Worden, & Maitumo, 2015). Similarly, large increases in both people and livestock over the past several decades in Inner Mongolia have ostensibly had adverse impacts on grassland vegetation and soils, in addition to the severe abiotic conditions imposed by the high-latitude, continental climate (Briske et al., 2015; Han et al., 2008). In each of these cases, adverse consequences resulted from interactions of management and policy actions (i.e. sedentarization, stocking rates, supplementation, land tenure) with environmental stochasticity.

3.2 | An anthropogenic non-equilibrial dynamic

Management strategies devised in the 20th century have contributed to the development of an alternative non-equilibrial dynamic

between livestock and rangeland resources. Large-scale livestock supplementation intended to sustain pastoral livelihoods during and after droughts is a growing trend, especially in developing nations (Müller, Schulze, Kreuer, Linstädter, & Frank, 2015; Schulze, Frank, & Müller, 2016). Programmes to sustain large numbers of humans and livestock in environments characterized by limited and heterogeneously distributed resources increase the potential for trade-offs between livestock production and alternative categories of ecosystem services. Provisioning services are often maximized at the local scale to sustain human livelihoods, while decreasing the supporting, regulating and cultural services that benefit humans globally (Davies, Ogali, Laban, & Metternicht, 2015; Favretto et al., 2016; Petz et al., 2014). Supplementation of range livestock may continue to increase world-wide in response to improved transportation infrastructure, fragmentation of pastoral lands, greater climate-induced variability of rangeland forage production and heightened global demand for livestock products (Boone, Conant, Sircely, Thornton, & Herrero, 2018; Herrero et al., 2016; Sloat et al., 2018).

Similarly, recent efforts to implement index-based, livestock mortality insurance programmes to mitigate poverty could also incentivize pastoralists to maintain animal numbers deep into multi-year droughts and reduce inclinations to destock via commercial or government-supported marketing options (Coppock, Bailey, Ibrahim, & Tezera, 2018; John, Toth, Frank, & Muller, 2019). These programmes could further contribute to rangeland degradation from overgrazing, creating complex trade-offs—with sobering ethical implications—between individual beneficiaries of provisioning services and ecosystem capacity to provision diverse ecosystem services for global benefit (Kubiszewski, Costanza, Anderson, & Sutton, 2017).

Drought management programmes in the developed world create similar challenges, although with reduced long-term risks to natural resources and human well-being because of social safety nets. For example, emergency cattle supplementation and financial bailout packages have been regular features of federal drought-response programmes in the United States (Coppock, 2011; Wilhite, Sivakumar, & Pulwarty, 2014) and Australia (Stafford Smith et al., 2007). While such efforts can indeed be vital to support ranching livelihoods, they have also been criticized as undermining the adoption of appropriate drought risk-management practices within the ranching community as well as contributing to unsustainable herd numbers and potential over grazing (see review in Coppock, 2011).

4 | A 21ST CENTURY STEWARDSHIP STRATEGY

The narrowly focused stewardship strategy of the 20th century that emphasized sustainable forage and livestock production desperately needs to be supplanted by one that conveys ecosystem services to humanity at large, including both rangeland

residents and non-residents (Reed et al., 2015; Sayre, McAllister, Bestelmeyer, Moritz, & Turner, 2013; Verstraete, Scholes, & Stafford Smith, 2009). A new stewardship strategy could be organized around a more complete accounting of ecosystem services with the explicit goal of provisioning optimal combinations of ecosystem services to support global human well-being (Dougill et al., 2012; Millennium Ecosystem Assessment, 2005; Reed et al., 2015). In a global context, the aggregate value of diffuse non-provisioning services may be of equal or greater value than those of the select provisioning services that are currently emphasized (Sayre et al., 2013; Sutton, Anderson, Costanza, & Kubiszewski, 2016). Revenue derived from heritage tourism may also represent a means to diversify the livelihoods of rangeland residents and reduce reliance on provisioning services (Timothy & Nyaupane, 2009). The demand for rangeland services in some developed nations has been transitioning from an exclusive production emphasis to a multifunctional rural land use driven by agricultural overproduction, alternative amenity-oriented uses and changing social values (Holmes, 2002, 2015).

A fundamental challenge for rangeland stewardship—perhaps more so than other land cover types—is that the vast majority of ecosystem services are not inherently perceived to have value, both within and beyond the market economy (Davies et al., 2015; Herrick et al., 2012; Kratli & Schareika, 2010). The extent to which rangelands are marginalized is inherent in the terms used to describe their deficiencies for providing provisioning services; that is, unpredictability, resource scarcity, sparse human populations and remoteness—collectively termed the drylands syndrome (Foran et al., 2019; Hoover et al., 2020; Reynolds et al., 2007). Similarly, Sayre et al. (2013) refer to rangelands as lands that have not yet been converted to other uses with higher rates of economic production and return. Consequently, the collective value of global rangeland services are often marginalized or unrecognized, and many non-provisioning ecosystem services become externalities associated with market transactions of the few valued provisioning services (Davies et al., 2015; de Groot et al., 2012). For example, land privatization has been identified as a primary contributor to declining wildlife populations on African rangelands (Homewood, 2004; Niamir-Fuller, Kerven, Reid, & Milner-Gulland, 2012).

A further challenge to the valuation of rangeland ecosystem services is that they are diffuse and broadly distributed throughout remote regions—a condition that is counter to the development of effective payment for ecosystem-service schemes (Sayre et al., 2013). This challenge could be addressed with the promotion of state–market–community hierarchies and relationships to reduce transaction costs and enhance cooperation associated with the provision of ecosystem services (Vatn, 2009, 2010). The objective would be to provide sufficient societal payment for non-provisioning services to minimize the need for provisioning services by local inhabitants, especially those with a high degree of resource dependency (Dougill et al., 2012). Greater international cooperation would further support implementation of global rangeland stewardship given that rangelands are present in numerous countries and all continents

(Biermann, Abbott, Andresen, Backstrand, & Bernstein, 2012; van Kerkhoff & Szlezak, 2016; Walker et al., 2009). For example, global monitoring of forest degradation and restoration potential to mitigate climate change is a priority for international agencies such as FAO, but a similar set of priorities specific to rangelands are not evident.³

A 21st century stewardship strategy must explicitly address the marginal socioeconomic status of the human inhabitants of rangelands (Millennium Ecosystem Assessment, 2005; Schild, Vermaat, de Groot, Quatrini, & van Bodegom, 2018; United Nations, 2015). Accelerating global drivers have seriously challenged the sustainability of pastoral systems, as evidenced by declining ecological and social conditions (Boone et al., 2018; Coppock, 2016; Galvin, 2009). This raises the critical questions, 'at what point are pastoral systems unsustainable, and how is sustainability assessed?' Recognizing and evaluating the need for transformation, as well as developing policies and programmes to determine when and how to implement transformational change—represents a critical and immediate challenge for the global rangeland community (Colloff et al., 2017; Reed et al., 2015). The need for system transformation may initially be forced by climate-induced warming and drying and the associated decrease in surface water (Godber & Wall, 2014; Huang et al., 2016; Sloat et al., 2018) that may spur a global exodus of climate refugees (Biermann & Boas, 2010). Institutional governance, policies and financial resource availability, delivered as state–community partnerships, are essential requirements to convert a vision for transformation into action and successful outcomes (van Kerkhoff & Szlezak, 2016).

5 | FUTURE PERSPECTIVES

The fundamental intellectual challenge facing the global rangeland community may not be identification of a unified model of rangeland ecology as assumed during the rangeland debate; rather, the challenge may be how to best transform rangeland social–ecological systems to provide optimal combinations of ecosystem services to meet the needs of all citizens, while improving the well-being of millions of rangeland residents who are highly dependent upon provisioning services (Colloff et al., 2017; Safriel & Adeel, 2008). The pivotal challenge in this transformation will be navigating the complex trade-offs that exist among individual beneficiaries of provisioning services and ecosystem capacity to provision diverse ecosystem services to benefit society. A conceptual prerequisite to successful transformation may be replacement of the marginalization narrative that has long-characterized rangelands, with one that explicitly recognizes

³The FAO Department of Forestry oversees global forest monitoring (<http://www.fao.org/forestry/en/>). Rangelands are covered via the FAO Pastoralist Knowledge Hub (<http://www.fao.org/pastoralist-knowledge-hub/en>) that networks stakeholders on many issues pertaining to pastoral development. The United Nations Convention to Combat Desertification (UNCCD) is another entity that has dealt with the promotion of sustainable dryland management using a series of strategic frameworks (<https://www.unccd.int/convention/about-convention>). The effectiveness of global institutions may be questioned, however. Davies et al. (2015) note that while many rangelands world-wide are significantly affected by environmental degradation, measures to address problems are typically weak or absent.

the aggregate value of rangeland ecosystem services (Hoover et al., 2020; Verstraete et al., 2009).

Even though the challenges confronting global rangelands have been recognized, institutional programmes and policies to assess and correct them remain limited or absent (Davies et al., 2015; Dougill et al., 2012). This is in part a consequence of an insufficient framework to identify and interpret these complex challenges and effectively prioritize actions and investments to address them. A comprehensive accounting of ecosystem services may provide a framework to gauge the global benefits of rangelands and serve as a benchmark to assess appropriate stewardship goals and strategies (Guuroh et al., 2018; Kubiszewski et al., 2017). This framework will require interaction among national and multinational institutions given the global distribution of rangelands and the ecosystem services that they provision (Biermann et al., 2012; Walker et al., 2009). Innovative institutional designs will be required to confront the social, political, scientific and technical complexity that characterize social–ecological rangeland systems in the 21st century (van Kerkhoff & Szlezak, 2016).

It is somewhat ironic that the rangeland profession, which emerged in response to a crisis of natural resource degradation in the United States early in the 20th century (Sayre, 2017), is now confronted with an even greater global dilemma at the dawn of the 21st century. This clearly establishes that stewardship strategies of the 20th century are insufficient to effectively address the emerging challenges in this century; in fact, the previous strategies may have contributed to some of the current and emerging challenges. Future strategies should strive to more effectively incorporate scale and heterogeneity and develop a more complete accounting of the diverse ecosystem services provided by global rangelands. This will require the redesign of current institutions or the development of new ones to develop and implement policies and programmes appropriate for 21st century stewardship.

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AUTHORS' CONTRIBUTIONS

D.D.B. framed the review and led manuscript writing. A.W.I. and S.D.F. developed the scale and functional heterogeneity content and D.L.C. prepared the pastoral development and social–ecological systems content. All authors contributed to the article and gave approval for publication.

DATA AVAILABILITY STATEMENT

Data have not been archived because this Review was not derived from original data.

ORCID

David D. Briske  <https://orcid.org/0000-0001-9487-0506>

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