1 Participatory Framework for Building Resilient Social-Ecological Pastoral Systems

Julia A. Klein, María E. Fernández-Giménez, Han Wei, Yu Changqing, Du Ling, D. Dorligsuren and Robin S. Reid

Introduction

Ecosystems and humans around the world are experiencing a time of great climatic, ecological, political and socio-economic change. One of the critical questions of our time is how to sustain people and the ecosystems on which they depend during this highly uncertain and fluctuating period in human history. Dryland[1] systems and their inhabitants, many of whom are pastoralists, have always existed on the margins, often facing extreme and highly variable conditions. However, the magnitude and complexity of changes and stresses that now face many pastoral societies is perhaps greater than ever before. Through a series of workshops and case study analyses, we asked whether pastoral ecosystems and peoples in North Asian drylands could maintain resilience in the face of current social and ecological changes. We surmised that community-based rangeland management was critical to maintaining resilience and reducing vulnerability under the current stresses. We define resilience as the amount of change a system can absorb without altering its essential structure and function and vulnerability as the degree to which a system is likely to experience harm due to exposure to a stress.

In April 2009, we held a workshop in Beijing, China on 'Poverty, Vulnerability and Resilience in North Asian Rangelands: Case Studies of Community-based Rangeland Management in China and Mongolia.' The workshop was attended by 45 scholars and practitioners working on seven community-based rangeland management projects in China and Mongolia. Participants included physical, social and biological scientists from China, Mongolia and the USA, with government, NGO and community practitioners. Each case-study team brought its experiences and lessons from community-based management projects and offered a place-based perspective as to the successes and challenges of their respective projects. We also discussed current paradigms in the field of rangeland ecology, community-based natural resource management, vulnerability and resilience theory, and global change ecology. We held a second workshop in Beijing in March 2010, where we collaboratively developed a conceptual framework to identify common attributes among the cases and to produce a construct through which all of the seven case studies could be examined.

The framework describes two different scenarios: a vulnerable social-ecological system and a resilient socio-ecological

(eds M.E. Fernández-Giménez *et al.*)

system. In both scenarios, there are local to global drivers acting on the system, including climate change and disasters, globalization, population changes and policy. The historical, geographic, ecological and cultural context in which these events unfold mediates how these factors play out within the system. In the 'vulnerable system' scenario, these drivers and their interactions with the social-ecological system lead to negative social and ecological outcomes, such as reduced grassland production and enhanced resource-based conflict. In the 'resilient system' scenario, various types of community-based natural resource management (CBNRM) practices interface with these dynamics to maintain a more resilient system with positive social-ecological outcomes, such as enhanced grassland health, livestock production and household incomes. Other factors, such as the ability to adapt and a flexible worldview in the face of novel conditions, also contribute to building resilient pastoral systems.

In this chapter, we describe the climatic, ecological, social and institutional features that characterize pastoral social-ecological systems and identify some of the recent pressures and changes that are occurring within them. We describe traditional and emerging paradigms in the field of rangeland management, the changing meaning and role of 'community' and the current understanding of vulnerability and resilience. The framework we present at the end of this chapter draws on these elements and highlights characteristics and linkages that can lead to resilient pastoral social-ecological systems and the role of community-based rangeland management in this process.

Pastoral Social-Ecological Systems

Nature's benefits and the physical and biological context

Drylands are water-limited ecosystems that include arid, semi-arid and dry sub-humid areas. They cover over 40% of the global land surface and are inhabited by close to 40% of the world's population (Reynolds *et al.*, 2007). Grazing is the primary land use in dryland regions and is the land use with the greatest areal extent on Earth (Asner *et al.*, 2004). Rangeland ecosystems provide many valuable products and essential services, including food, medicines, clothing, fuel and shelter. Freshwater resources, minerals, wildlife habitat and opportunities for recreation and tourism are additional benefits derived from rangelands. Rangeland systems also regulate climate, soil development and conservation, the quantity and quality of freshwater resources, and nutrient cycling and retention. Rangeland systems are highly valued for their cultural diversity and for their historic and aesthetic value (MEA, 2005; Havstad *et al.*, 2007). While most people living in dryland systems around the world directly depend on that system's ecological services for their livelihoods, the numerous people living downstream and in urban and peri-urban environments also rely on critical rangeland ecosystem services.

The climate in rangeland regions is characterized by low annual precipitation that varies over time and space. Intra-annual variability is typically high, with moisture deficits generally occurring during some part or all of the year. Precipitation variability increases as mean annual precipitation decreases, latitude decreases, and the effect of El Nino-Southern Oscillation (ENSO) increases (Nicholls and Wong, 1990). Precipitation events can be of short duration but high intensity, while diurnal, seasonal and annual temperature ranges can be large. Rangelands generally have a pronounced seasonality of precipitation, an abrupt transition between longer dry periods and a shorter wet season, and risk of both droughts and floods (Nicholson, 2002). While these are some of the general climatic features of rangelands, there is much diversity of climatic processes and variability across rangeland systems globally.

Rangelands occur on every continent, and include hot and cold deserts, tundra, scrub, chaparral, savannah and grassland systems (Easterling *et al.*, 2007). The amount, timing and variability in precipitation are all important drivers of dryland vegetative productivity, composition and diversity over a range of spatial and temporal scales. Overall precipitation amount is a primary determinant of system structure and function (Sala *et al.*, 1988; Huxman *et al.*, 2004). However, the seasonal distribution of precipitation, and whether it occurs during or is decoupled from the warmer vegetative growing season, is also an important system driver (Knapp *et al.*, 2006). Precipitation event size, frequency, timing and duration can also strongly influence rangeland properties and processes such as primary productivity (Boone *et al.*, 2002; Heisler-White *et al.*, 2009). The effect of precipitation is further modified by temperature (Oechel *et al.*, 1998), grazing (Schlesinger *et al.*, 1990; Collins *et al.*, 1998; Fernández-Giménez and Allen-Diaz, 1999), fire (Collins, 1987) and other properties such as soil texture (Noy-Meir, 1974) and competition (Callaway *et al.*, 2002; Berlow *et al.*, 2003). Moreover, system structure and function results from interactions among the primary system drivers (Shaw *et al.*, 2002; Klein *et al.*, 2007). Due to the variable nature of climate, the influence of patchy grazing by mobile animals and the heterogeneous nature of processes such as fire, resource availability in rangeland systems also tends to be highly variable across both space and time (Schlesinger *et al.*, 1990; Augustine and McNaughton, 2006).

Rangelands comprise approximately 40% of China's land area and represent a large and diverse biophysical and ethnic and cultural landscape that includes the alpine ecosystems of the Tibetan Plateau, the steppe and desert regions of Xinjiang Province, the subtropical regions of Yunnan Province and the semi-arid Loess Plateau, with an order of magnitude difference in mean annual precipitation across the driest (<100 mm) and wettest (>1000 mm) of the sites. While some form of animal husbandry is present across all of these regions, the ethnic diversity includes Mongols, Tibetans, Kazaks, Hui Muslims and Han Chinese (Ho, 2001).

Over 80% of Mongolia's 2.5 million km^2 are rangelands, which extend from the hyper-arid desert through the semi-arid desert-steppe or Gobi, the expansive grasslands of the true steppe in central and eastern Mongolia and the mountain and forest steppes of the central Khangai mountains and northern Mongolia. This gradient from desert to forest steppe represents wide variability in both mean annual precipitation and inter-annual variation in precipitation, with the desert and desert-steppe at one extreme of low rainfall (<100 mm/year) and high variability and the forest steppe at the other extreme, with much higher annual precipitation (300–400 mm/year) and lower variability among years (Hilbig, 1995; Fernández-Giménez and Allen-Diaz, 1999; Gunin *et al.*, 1999).

Pastoralism, pastoral strategies and the social and institutional context

Pastoralism is a land use characterized by extensive grazing on rangelands for livestock production. It occupies approximately 25% of the world's land area, produces approximately 10% of the meat used for human consumption, supports approximately 20 million pastoral households and is the dominant production system in drylands globally (FAO, 2001). Pastoralists around the world manage close to a billion camelids, cattle, sheep and goats and also raise yaks, horses, reindeer and other ungulates (FAO, 2001). In pastoral societies, livestock are a means of production and consumption, but they also maintain important socio-cultural value (Bonte *et al.*, 1996). Pastoralists generally keep enough livestock to meet subsistence needs and to sustain households through extreme events such as droughts. Pastoralism differs from commercial ranching because it is oriented first towards meeting subsistence needs and then produces for trade and the market, and also because it relies more on human labour, local knowledge, common tenure, and some type of mobility (BurnSilver, 2008). In contrast, ranching tends to be a more intensive, commodity-oriented production system where animal raising occurs primarily for monetary exchange rather than for direct consumption and where primary pasturelands are individually owned and fenced.

Pastoralists inhabit areas with variable and extreme climates, patchy resources, and often unstable political and economic environments, circumstances that expose pastoralists to the risk of losing part or even most of their herds. Even in the less extreme years, overall production is low and resource availability is highly spatially and temporally variable. Pastoralists have traditionally coped with these circumstances by using a wide range of strategies that allow maximum flexibility and manoeuvrability in dealing with these changing and uncertain conditions. These crucial and highly adaptive strategies include diversity, reciprocity and cooperation, reserves, flexibility, mobility, and shared access to resources (Fernández-Giménez and Le Febre, 2006), and are supported by pastoral social and institutional arrangements. Pastoralists embrace *diversity* through their livestock, products, livelihoods and institutions. Livestock diversity includes multispecies grazing (e.g. browsers, grazers and generalists) and herd diversification (e.g. age and reproductive structure), providing for maximum use of heterogeneous resources. Production of multiple livestock products and services is also a general characteristic of these systems. Pastoralists maintain diversified livelihood activities outside the pastoral sector, including agriculture, urban employment, engagement in medicinal plant enterprises and other cottage industries. Institutional diversity includes owner groups and ad hoc organizations. Another characteristic pastoral coping strategy involves *reciprocity* or moral economy, which refers to the practices and norms, such as social ties of mutual obligation, that serve to help the poorer members through hard times (Peluso, 1992; Neumann, 1998). For example, in the Tibet Autonomous Region, pastoralists with access to lower productivity pastures or who lack year-round access to critical resources such as water, obtain additional rangeland resources through inter-community resource-sharing arrangements (Bauer, 2006). Pastoralists also manage risk through the use of ecological and social *reserves*, such as setting aside land so it can be grazed in times of drought or snowstorms and maintaining a network of different groups of people in different areas for potential cooperation and reciprocity. *Flexibility* of management decisions, movements, livelihoods and institutions is another key strategy for the persistence of pastoral systems globally (Bonte *et al.*, 1996; Fernández-Giménez and Swift, 2003). *Mobility* and *common property regimes* are two features germane to pastoral survival that are today under great pressure; we therefore discuss these two characteristics in more detail below. In describing the fluidity of pastoral mobility and land tenure we also highlight the importance of flexibility to pastoral livelihoods.

Mobility is one of the most important and characteristic trademarks of pastoralism worldwide. Since conditions among and within pastoral regions are highly heterogeneous, the number, distance and duration of movements similarly varies. Pastoralists living in the most extreme environments can be highly mobile and nomadic, moving opportunistically and without regular pattern in response to stochastic weather events. Transhumance occurs in highly seasonal temperate environments where herders move either vertically or horizontally between relatively fixed points to exploit seasonal availability of resources. Seasonal aggregation and dispersal occurs when animals cluster during one season, such as the dry season when water is scarce, and then disperse in another season when resource availability changes, such as the wet season when resources are more abundant (Fernández-Giménez and Le Febre, 2006).

Mobility confers multiple benefits to pastoralists, their grazing animals and dryland ecosystems. Through mobility, pastoralists take advantage of changing climatic conditions and corresponding spatiotemporal resource patchiness so their animals have access to the most nutritious forage available. Mobility can provide access to key resources, such as water and salt, and also reduce competition for resources (Fernández-Giménez and Le Febre, 2006). Livestock survival is enhanced when pastoralists can move out of areas affected by insects and disease or those that have been exposed to extreme weather events, such as droughts or severe storms (Niamir-Fuller, 1999; Fernández-Giménez and Le Febre, 2006; Davies, 2008). Mobility can also confer social, political and economic benefits. For example, by moving, herders can market livestock and their products, engage in trade, avoid conflict, access health and education and other social services, and pursue diverse income opportunities. Mobility also provides opportunities to enhance social relationships that underlie reciprocity, an important aspect of pastoral persistence (Fernández-Giménez and Le Febre, 2006). The movement of grazing animals across a landscape also provides an opportunity for the grazed landscape to rest and recover.

Common property regimes are another adaptive feature of many pastoral systems worldwide. Shared resources are often referred to as common pool resources or CPRs. Extensive rangelands are considered CPRs because it is difficult to exclude potential grazers from these areas, and use by one individual reduces the amount of forage available for other herders. In an open access situation, where the absence of any rules that limit who may graze and place restrictions on the amount, timing or spatial distribution of grazing, there is a risk of overuse and subsequent degradation of the forage resource. Most rangelands are not open access, however. Instead, they are subject to some type of property regime, that is, a set of formal or informal rules that define the rights and obligations of specific individuals or groups with respect to access, use and management of the resource in question. Private property gives an individual the exclusive right to use and manage a resource, and the right to sell, lease, or transfer their property to another person. State property is owned and managed by the government on behalf of a nation's citizens. Common property, a characteristic of many pastoral systems, occurs where a group of resource users collectively holds the rights to use and manage a specific resource, including the right to exclude non-members from use.

In pastoral social-ecological systems, the absence of property rights and rules for pasture use can lead to an undesirable open access situation where there is no incentive for individual herders to conserve forage and overuse and degradation ensue, although this rarely occurs in traditional pastoral systems (McCabe, 1990). On the other hand, privatization and allocation of pasture to individual households in extensive semi-arid grazing systems can also have negative social and ecological consequences, as has occurred in some grassland areas of China under the household responsibility system (Williams, 1996). Even attempts to provide groups of pastoralists with exclusive rights to large areas of rangeland, such as the group ranches of Kenya, have not been entirely successful (Kimani and Pickard, 1998; Mwangi, 2007). The group ranches were initially effective in providing group members with secure and exclusive rights to their allocated land, but it proved impossible to regulate stocking rates within the group, herders still needed to migrate outside the ranch boundaries in times of drought, and eventually many of the ranches were subdivided and fragmented into many small parcels of land owned by individuals and are no longer viable for sustainable livestock production (Kimani and Pickard, 1998; Mwangi, 2007; Thompson *et al.*, 2009).

Under common property arrangements, a well-defined group of pastoralists collectively hold the rights to use and manage a defined rangeland (Ostrom and Mwangi, 2008). A tenure system where large expanses of land are collectively held and managed is well-suited to common pool resources such as rangelands, where it is hard to exclude others and where there is competition for a limited resource. This arrangement is also advantageous in systems where productivity is low and where resource distribution is heterogeneous (Ostrom and Mwangi, 2008). Multiple property rights are generally associated with different rangeland resources (e.g. reserve pastures and water resources) and with different agents holding those rights (e.g. individuals, households, traditional user groups or administrative units). Banks (2001) found that the tenure arrangement in a pastoral region of northwest China demonstrated elements of private rights, including close kin networks and long-term inheritable use rights; common rights with multiple household participation and community-based rangeland management institutional arrangements; and co-management, with the state also participating in aspects of rangeland management. There is often a disjuncture between official property rights policy and actual practice and implementation, as was found in Kazak and Tibetan pastoral regions of Western China (Banks, 2003; Banks *et al.*, 2003), in Ningxia Province in nothwest China (Ho, 2000) and in the Maqu case study presented in this book. Researchers describe rangeland boundaries as fuzzy and fluid with flexible and dynamic governance, institutions and rules for managing rangelands (Banks, 2003; Galvin, 2009; Beyene, 2010).

Many of the traditional strategies pastoralists have employed in the past are in flux and may not be readily available to them in the future. This will probably have profound effects on pastoralists' ability to adapt to their changing climatic, ecological and socio-economic situation and may impair their resilience in dealing with these ongoing pressures. The poor are more vulnerable to external stresses and shocks than the wealthy because they have fewer assets to help them survive difficult times, especially if they lose their traditional ways of coping with risks and loss. Pastoralists are generally considered among the poorest populations according to some measures, although this characterization may not be accurate. We discuss this paradox below.

The paradox of pastoral poverty

While pastoralists often rank poorly on standard measures of wealth (such as expenditures, education and access to markets), it may be incorrect to assume that they are some of the poorest people in the world (Little *et al.*, 2008). This arises because pastoral production and consumption are not properly valued. To understand pastoral wealth, livestock must be valued not only as a source of food and cash, but also as breeding capital, the value of herd reproduction each year (Little *et al.*, 2008). For example, the poorer pastoralists are those who do not have livestock and live around towns and have a higher proportion of their income in cash and not herds. Herders who still move and live with their herds can be better off, with less cash but more potential income in the form of the reproductive potential of their livestock. Thus, cash expenditures may be poor measures of poverty among pastoralists.

Another source of wealth for pastoral families is the social networks that holding livestock can provide (Dyson-Hudson and Dyson-Hudson, 1980). In many pastoral systems, herders create complicated sets of relations with kin and friends by exchanging livestock and forming informal bonds through this exchange. Those families with stronger 'safety nets' in the form of these bonds are less vulnerable in the face of drought or market failure. And families that lose all their livestock are highly vulnerable and often have to leave pastoralism for periods of time until they build up herds again and can revive obligations to other pastoralists through their animals.

Poverty is also assumed to be higher for people who live in remote areas, far from markets. By contrast, it is quite possible that geographic isolation may be associated with wealth for pastoralists because there is a trade-off between increased access to markets to sell livestock products and acquire necessities, and the crowded and over-grazed pastures near markets (Little *et al.*, 2008). Thus families that live far from markets, visiting only occasionally, may well be better off than those who live closer to markets. This links to an allied paradox: access to more education and social services may make pastoral families more vulnerable because these services are usually linked to living near towns and becoming sedentary (Little *et al.*, 2008). The disadvantage of living near towns can tip toward an advantage if the educational opportunities are sufficient enough for graduates to access good salaries from employment and can support the pastoral family through remittances. In Mongolia in the early 1990s, the government's attempt to rapidly modernize and improve the economic status of the rural pastoral population actually led to a decline in their standard of living (Marin, 2008).

These paradoxes suggest that it is important to question the conventional wisdom about pastoral development in relation to pastoral poverty and vulnerability. Later in this chapter we describe some of the contemporary changes pastoral societies are experiencing, including intensification and resettlement, policies that are at least partly driven by poverty alleviation goals. This chapter and some of the case studies from this book illustrate how top-down poverty alleviation policies meant to enhance well-being may be contributing to enhanced vulnerability in pastoral social-ecological systems.

Pastoralists and the state

Throughout the world, mobile pastoralists are often at odds with nation states and frequently are politically, economically or socially marginalized (Lane, 1998; Scott, 1998; Fratkin and Moir, 2005). By states, we mean the agents that hold power over defined national territories and organize, control and/or serve societies through organized government. The nature of states varies widely from highly organized, authoritarian entities that control most aspects of daily life, to much more loosely constituted democratic structures (Fratkin and Moir, 2005). Few pastoral societies have organized their own nation states. Instead, pastoral tribes or groups are more often dispersed subjects or citizens within states dominated by more sedentary agricultural or urban populations (Fratkin and Moir, 2005).

Pastoralists present a conundrum for economic development, because prosperity in agricultural systems is often advanced by private or quasi-private property regimes that are thought to provide incentives for investment in productivity-enhancing technologies and practices and the means to obtain credit to implement these technologies (collateral), resulting in increased productivity and prosperity. Mobility also creates challenges for social development because it is difficult for governments to provide public services such as health care and education to populations that are constantly on the move. Further, mobile populations are difficult to count, control and tax. In short, mobile pastoralists present problems for states that seek to impose order and regularity on social and physical landscapes, and in so doing to enhance the states' power and prosperity, as well as the well-being of their people. Moreover, the aims of states are often well intentioned, but may ignore the logic of locally adapted practices and ways of life, sometimes resulting in reduced well-being of pastoralists and rangeland landscapes, instead of improved conditions.

These conundrums have led to policies in many regions that are unfavourable to the maintenance of mobile livestock husbandry as a production system and way of life. Key state policies aimed at or affecting pastoralists and their rangelands include nationalization of pastoral lands; policies that force or encourage settlement of mobile groups; land titling and formal management plans; and privatization of common grazing lands (Lane, 1998). Dalintai *et al.* describe historic and current rangeland policy across China and Mongolia in Chapter 3 of this book. These policies may sometimes be well intentioned, but Lane (1998, pp. 14–15) argues 'centralized, uniform and imposed land tenure reform is not likely to succeed in enabling pastoral land users to produce more and protect their lands from overuse. On the contrary, it is likely to heighten insecurity, undermining productive potential and putting land at risk of degradation – the result that is hoped to be avoided.' The frequently negative outcomes of these types of policies are in part due to their disruption of often complex, diverse and locally adapted systems of customary pastoral land use and land tenure. State schemes to regularize pastoral land tenure often result in a loss of access to diverse resources and heterogeneous landscapes, and limit herders' ability to cope with climatic risks and weather disasters (Lane, 1998).

Policies to reform pastoral land tenure are often coupled with other technical interventions such as livestock breed improvement and intensification of production through fodder cultivation and stall feeding. James Scott (1998) exposes the rationale behind and devastating consequences of state projects to render landscapes 'legible' through formalized and centrally controlled land tenure systems, management planning, and technological improvements that simplify the inherent complexity and diversity of locally adapted and resilient management systems. Without romanticizing custom or underestimating the potential inequities within traditional societies, Scott (1998, pp. 34–35) argues that 'because [customary systems of tenure] are strongly local, particular, and adaptable, their plasticity can be the source of micro adjustments that lead to shifts in prevailing practice.' Scott attributes the failure of many top-down state schemes to their incompatibility with locally-evolved practical knowledge, and suggests that successful institutions will make room for and support the development and use of dynamic and accumulating local knowledge and experience, and will, indeed, continually change and adapt in response to this knowledge. Community-based rangeland management institutions may represent an example of this type of dynamic and locally adaptive approach to governance, economic and social development. One common weakness of top-down state policies has been poor understanding of rangeland dynamics and management, a field of study that continues to develop.

Managing Pastoral Social-Ecological Systems: Past and Current Paradigms

Are rangelands equilibrium or non-equilibrium systems?

Over the past few decades there have been several different paradigms for sustainably managing pastoral systems. Before the late 1980s, the mainstream view in range ecology was that livestock and vegetation were in equilibrium, and too much grazing (overstocking) caused the rangeland to lose productivity. This, in turn, meant fewer livestock could be supported on those pastures in the future. Thus, management practices and policy focused on making sure that livestock numbers stayed below the carrying capacity of the rangeland to prevent overstocking. Carrying capacity is the upper limit of forage that livestock can graze and still sustain the productivity of the range for the subsequent seasons. Much of China's rangeland policy is based on the carrying capacity concept and the perception of equilibrium rangeland dynamics (Ho, 2001).

Our thinking about rangeland ecology and the dynamics of pastoral ecosystems has undergone a major intellectual change in the last three decades (Sandford, 1983; Ellis and Swift, 1988; Westoby *et al.*, 1989). Based on work in arid rangelands with highly variable rainfall, research in the 1980s and 1990s showed that rainfall has more impact on rangeland productivity than livestock grazing. In these dry rangelands, frequent drought and long dry/cold seasons mean that livestock populations rarely become so high to exceed the ability of the vegetation to support them. If herders have no access to external sources of feed and water, they either move their livestock to wetter pastures or, if this is not possible, livestock die and herds are smaller after drought for a period of time. This movement of smaller herds allows the vegetation to recover. Here, there is no fixed carrying capacity; instead, the maximum number of animals a pasture can support fluctuates with rainfall. These are termed 'non-equilibrium systems'. Researchers contrast them to 'equilibrium systems' where overstocking of rangelands is possible, livestock can damage the production of vegetation, and where carrying capacity can be a useful concept. While the non-equilibrium rangeland concept was developed in arid rangelands, subsequent research demonstrates that other abiotic factors, such as large snowstorm events, can also produce non-equilibrium conditions. If climate is more important than grazing and livestock do not damage vegetation, then it does not matter how many livestock graze a rangeland, because livestock cannot degrade rangelands (Ellis and Swift, 1988). In this case, policy and management should support pastoral families to move to new pastures during drought or unfavourable climatic conditions, but there should be no limitation on animal numbers. On the other hand, if livestock can damage vegetation by grazing, it is critically important that policy and management focus on the numbers of animals that graze a rangeland, as well as supporting families during dry or harsh climatic periods. Ho (2001) describes a rangeland system in northwest China where the precipitation patterns suggest it should be a non-equilibrium system. However, he found no relationship between animal herd dynamics and climate variables such as precipitation and suggests the need to consider other factors, such as socio-economic and institutional factors. While acknowledging the important role of density-independent factors on livestock populations in other regions of the Tibetan Plateau, Cincotta *et al.* (1992) found that equilibrium conditions dominated at a relatively mesic, lower elevation site on the northeastern Tibetan Plateau.

Subsequent research shows that both equilibrium and non-equilibrium rangelands exist (Briske *et al.*, 2003), and sometimes they are mixed in the same landscape or shift from one type to the other during different seasons (Vetter, 2005). Generally, arid rangelands are non-equilibrium and semi-arid rangelands follow equilibrium patterns. But even in arid rangelands, areas like wetlands and river-edge vegetation, often called key resources, can be in equilibrium with livestock populations, while the wider rangeland is not (Illius and O'Connor, 1999, 2000). In Mongolia, Fernández-Giménez and Allen-Diaz (1999) found that the mountain steppe site demonstrated primarily equilibrium dynamics, the desert-steppe non-equilibrium dynamics and the steppe a combination of characteristics of both types of systems. In a dry steppe site in the Gobi Altay of Mongolia, Stumpp *et al.* (2005) also found no significant association between species composition and distance from water, a proxy for grazing pressure.

In conclusion, drier rangelands tend to have non-equilibrium dynamics, wetter ones have more equilibrium dynamics, and there is significant mixing of these dynamics. However, it is important not to assume that livestock have no impact on arid, non-equilibrium rangelands (Fernández-Giménez and Allen-Diaz, 1999); there is evidence that heavy grazing can degrade even very dry rangelands (Todd and Hoffman, 1999). Rather, it is important to monitor and assess the health of all rangelands over time. And as we measure change, we need to distinguish among the effects of livestock, land use and climate, so that we know what types of policy or management will be effective in addressing problems as they arise. It is also critical to understand how reversible change is, so that we can avoid damaging change that is very costly to reverse or even irreversible. These ideas suggest that rangeland degradation results from the interplay between climate, grazing and other factors, with dynamic spatial and temporal components.

Dryland dynamics: state and transition models

There has also been a shift in the way we think about how rangelands change over time. Previous to 1989, the traditional rangeland model, following principles of equilibrium dynamics, described rangeland change as a linear process along one predictable pathway that could be reversed. Since then, significant research shows that any particular rangeland can follow several different pathways of change to different vegetation states, depending on the amount of grazing or other dominant processes, such as fire. A vegetation state is a recognizable, resistant and resilient complex of soil and vegetation structure that is distinguished from other states by relatively large differences in plant functional groups and ecosystem processes, and therefore in vegetative structure, biodiversity and management requirements. State changes may not be linear; there may be abrupt changes that occur as the system passes over a threshold (Friedel, 1991; Bestelmeyer *et al.*, 2003; Briske *et al.*, 2006). These thresholds may make the change from one vegetation state to another hard to reverse or irreversible, unless there is significant labour and capital to restore the vegetation to the original state. Non-linear change happens, for example, when livestock grazing pushes vegetation structure or productivity over a threshold, with a major change from a grassland (the first 'state') to a shrubland (the second 'state').

These ideas led to the development of state-and-transition models that help us describe changes in rangeland vegetation and estimate future changes. These models (Westoby *et al.*, 1989; Bestelmeyer *et al.*, 2003; Stringham *et al.*, 2003; Briske *et al.*, 2006) describe non-linear changes across thresholds from one state (such as stable soil) through a transition to another state (such as eroded soil). These models are particularly useful because they can integrate local, traditional and scientific knowledge together to describe alternative states of rangelands. Rangeland managers find these models useful in describing change in rangelands. The models focus much attention on identifying thresholds, periods of 'hard-to-reverse' change and can be applied to non-equilibrium or equilibrium rangelands.

Drylands as social-ecological systems

In recent years, scholars and practitioners have been examining linked social and ecological systems to address today's most pressing environmental challenges. Research within the emerging programmes of 'sustainability science', 'land change science', 'coupled human–natural systems science' and 'ecosystem stewardship' (Clark and Dickson, 2003; Liu *et al.*, 2007; Turner *et al.*, 2007; Chapin *et al.*, 2010) is often focused on dryland systems. Some of the guiding principles behind these new approaches are illustrated by the dryland development paradigm (Reynolds *et al.*, 2007): (i) drylands are coupled human–natural systems; they are dynamic and result from complex interactions between bio-physical and human subsystems; (ii) critical dynamics are determined by socio-ecological variables that change slowly or 'slow variables' (e.g. water holding and cation exchange capacity of soils) and solutions need to focus on these variables; (iii) if slow variables cross thresholds, the system can move into a new state or condition; (iv) the involvement of multiple stakeholders, with different objectives and perspectives, is critical and illustrates the multi-scaled and interacting nature of these systems; and (v) healthy coupled systems use environmental knowledge integrating local management and policy experience with science-based knowledge, all of which must be mediated through an effective institutional framework. The drylands development paradigm shares many ideas with those of resilience theory. The framework we describe at the end of this chapter is strongly influenced by the non-equilibrium, state and transition thresholds, resilience, and linked social-ecological systems thinking.

Changes in Pastoral Ecosystems

A suite of climatic, biological, socio-economic and political drivers are changing the fundamental nature of pastoral systems. Climate change, intensification of pastoral production, privatization, sedentarization, land fragmentation, livelihood diversification, resettlement, economic change and degradation are exerting tremendous pressures on pastoral ecosystems and peoples worldwide. There is some discussion as to whether these changes reflect the highly adaptive and resilient nature of pastoral social-ecological systems, or whether they are moving these systems towards undesirable thresholds (Galvin, 2009). These drivers and their resulting impacts are both external and internal stressors in our conceptual framework, which suggests that community-based rangeland management can foster resilience in the face of some of these uncertain and changing conditions.

Climate change

Climate change is affecting pastoral social-ecological systems globally. Over the past century, global mean temperatures have increased by 0.74°C and mountain glaciers and snow cover have declined globally. Changes in precipitation have also occurred, with overall increases in wetter regions and overall decreases in arid regions, including drylands (IPCC, 2007). North Asian rangelands, including the rangelands of China, Mongolia and the Tibetan Plateau, have also experienced climate change and are predicted to experience greater changes than the global average. The Intergovernmental Panel on Climate Change (IPCC) provides regional climate change predictions for the years 2080–2099 relative to 1980–1999 and divides the Asian continent into five different regions (Christensen *et al.*, 2007). Chinese and Mongolian rangelands are covered in the East Asia, North Asia and Tibetan Plateau regions of the IPCC regional analysis (Christensen *et al.*, 2007). Mean annual warming for East Asia, the Tibetan Plateau and North Asia is predicted to be 3.3°C, 3.8°C and 4.3°C, respectively. In North Asia, winter warming is predicted to surpass warming in other months (6.0°C in December, January, February); monthly warming is predicted to be more uniform for East Asia and the Tibetan Plateau. Small increases in precipitation are predicted (15% in North Asia, 9–10% in East Asia and the Tibetan Plateau) with greater increases in winter than other seasons. The net effect of the temperature and precipitation changes will probably be a decrease in soil moisture availability during the summer growing season in these regions.

Climate change is not only leading to warmer temperatures, but is also predicted to increase the frequency and intensity of extreme weather events. In Mongolia and on the Tibetan Plateau, extreme weather events consist of snowstorms (referred to as *dzud* in Mongolia) that cover the vegetation and prevent animals from accessing their primary food source. During the winter months, livestock tend to survive on intact but senesced vegetation on the rangelands. Therefore, large snowstorms that cover the vegetation for extended periods of time can cause high livestock mortality rates. Snowstorms are not a new phenomenon in this region of the world, but their frequency and intensity may be increasing. Folland and Karl (2001) conclude it is likely there has been a widespread increase in heavy and extreme precipitation events in the mid- and high latitudes of the northern hemisphere. Regionally, there is evidence for increasing spring snow accumulation over the past few decades on the Tibetan Plateau (Niu *et al.*, 2004; Zhang *et al.*, 2004). The observed changes in extremes are qualitatively consistent with changes in model simulations of future climate (Cubasch and Meehl, 2001).

Changes in CO_2 and climate will affect rangeland quantity and quality through changes in production, species composition and plant tissue chemistry. Enhanced CO_2, in combination with up to 2°C warming, is predicted to have positive effects on production in humid temperate rangelands and negative effects in arid and semi-arid rangelands (Easterling *et al.*, 2007). Climate change may decrease rangeland quality through CO_2-induced decreases in plant protein content or through climate-induced increases in shrub production, but enhance rangeland quality through CO_2 induced increases in N fixers or C_3 relative to C_4 plants (Easterling *et al.*, 2007). Warmer and drier conditions increase animal thermal stress with resultant declines in meat output, dairy production and measures of animal success, such as conception rates (Easterling *et al.*, 2007). Rangelands and livestock are likely to be susceptible to extreme weather events, such as floods and droughts.

North Asian rangelands are predicted to experience increased exposure to climate stressors and have also demonstrated high sensitivity to climate changes, with both factors increasing the vulnerability of these systems to climate change. In Mongolia, field and satellite observations of vegetation change that control for grazing suggest that Mongolian rangelands may be negatively impacted by climate change, with the forest steppe and steppe regions of the country particularly sensitive to warming and drying (Angerer *et al.*, 2008). Warming experiments on the Tibetan Plateau decreased rangeland health through reductions in vegetative productivity, forage availability, species diversity and the delivery of key ecosystem services (Klein *et al.*, 2004, 2007, 2008). Warming reduced forage quality through plant tissue chemistry changes and through the replacement of more digestible graminoids with less digestible shrubs (Klein *et al.*, 2007).

The vulnerability of North Asian social-ecological pastoral systems to climate change will be mediated by the other ongoing changes to this social-ecological system. For example, Klein *et al.* (2004, 2007, 2008) found that moderate amounts of grazing could partially mediate the negative effects of warming. Current rangeland policies, which are removing grazing altogether from some regions of the Plateau, may therefore increase the sensitivity and overall vulnerability to climate change (Klein *et al.*, 2011). There is also evidence that political economic transformations and other pressures across North Asian rangelands have altered herding and pasture management strategies that developed as means to survive in a variable and unpredictable environment; this may result in reduced resilience of pastoral systems to stresses and shocks. For example, on the Tibetan Plateau, because temporary migration is one of the only options available for coping with severe snowstorms in the short term, the current trend toward hardened property boundaries and fencing could exacerbate future vulnerability to severe storms (Klein *et al.*, 2011). Finally, while we discuss system vulnerability in broad terms, the reality is that individuals and households will probably be differentially affected by climate change, due to different entitlements (e.g. rights to land) and social/biophysical endowments (e.g. the quality of the land to which they have rights). These differential vulnerabilities may be further exacerbated by other trends, such as increased socio-economic disparities among herders.

Intensification

Intensification of pastoral production systems is a global phenomenon whose goal is to ‘modernize’ traditional livestock operations, which have been perceived by outsiders as economically irrational and inefficient. Intensification refers to an increase in the unit of livestock production (meat, wool, dairy) relative to a given amount of inputs (labour, land area, water, feed, medicines). The intensification process typically involves increasing livestock off-take rates and acquiring additional inputs (e.g. improved livestock breeds, veterinary medicines) to both bolster outputs and to increase the efficiency of production (BurnSilver, 2008). Access to credit is generally required to purchase the inputs. Intensification also involves activities such as water development projects and market development. Formalizing rights and privatizing land are often part of the intensification process, as it is thought that ownership will enhance investments and lead to greater outputs. This is often also accompanied by sedentarization as described below. The motivation for intensification is economic, but it is also intended to address degradation and is inspired by political and other reasons. It is not always planned, and does not necessarily lead to positive outcomes. While the pressure from policy makers is toward intensification, economic factors, such as limited access to credit, can hinder this process. Moreover, climatic and ecological factors, which favour mobility, are also in conflict with this top down approach (BurnSilver, 2008).

Privatization

As discussed in the above paragraph, the trend towards intensification of pastoral production systems typically includes top-down policies converting previously community-managed rangelands into private property ownership with the perception that individuals will invest and steward land better than groups. These development policies often focus on converting land and/or livestock and sometimes water to private ownership. For example, in the 40% of China's land that is rangeland, current policy favours individualization of land tenure, so that households have long-term rights to use pasture. In some instances and regions of the world, pastoralists are initiating this process themselves to gain control over their land to prevent other interests, such as agriculture and land conservation organizations, from obtaining title to the land (Galvin, 2009). In northern Kenya, land privatization had no impact on household wealth (Lesorogol, 2008). However, as we discussed in previous sections, communal tenure arrangements are part of a suite of pastoral coping strategies to deal with environmental risk (Banks, 2003) and thus reduce vulnerability. These land tenure changes across Chinese and Mongolian rangelands may therefore limit options for pastoral systems and make them more vulnerable to ongoing and future change.

Sedentarization

In pastoral systems globally, there is a dramatic transformation in land use toward reduced mobility and sedentarization. The process of sedentarization is inextricably linked to the privatization and overall intensification process. Factors contributing to this trend include the conversion of rangelands to other uses, such as agriculture or parks and protected areas; establishment and enforcement of political and administrative boundaries; disruption of local institutional control and rangeland practices; increased labour costs associated with movement; and the development of stationary goods and services such as schools and medical facilities (Fernández-Giménez and Le Febre, 2006; Galvin, 2009). Local and national governments encourage pastoralists to settle because they can control less mobile pastoral populations, tax them and provide social services to them. Moreover, decreased mobility can result from pastoral institution adaptations whereby pastoralists are trying to regain local control over their resources by making permanent claims to grazing lands through permanent structures or establishing year-long residence on grazing lands by all or part of the household. Both privatization and sedentarization have been shown to increase rangeland degradation (Williams, 1996) and may decrease pastoral identity and control over or access to critical resources. Vulnerability to livestock losses under drought or storms increases with sedentarization (McPeak and Little, 2005; Galvin, 2009; Klein *et al.*, 2011). Moreover, the loss of movement must be compensated by economic inputs or livelihood diversification (Galvin, 2009).

Fragmentation

Changes in land tenure, agricultural expansion, sedentarization and institutions have led to rangeland fragmentation, the dissection of previously intact, extensive grazing lands into spatially isolated parts (Hobbs *et al.*, 2008; Galvin, 2009). This phenomenon was common in US and Australian rangelands in the last century, and in Africa and Asia in this century (Galvin, 2008). As land tenure changes and use of rangelands intensifies, herders often settle and convert rangelands into other uses or fragment them, often with fences (Reid *et al.*, 2008). Globally, 35–50% of semi-arid and subhumid rangelands and 10% of arid rangelands are now used for urban areas, cropland or conservation areas (MEA, 2005). Fragmented rangelands can support fewer livestock per hectare because herders and their livestock can no longer move freely and thus do not have easy access to different types of vegetation that are important for grazing in different seasons and during drought (Boone, 2007; Hobbs *et al.*, 2008). While it appears that herders in the driest lands are largely immune to the impacts of this conversion, this is often an illusion: incoming farmers plough the small pockets of wetlands, river edges or bottomlands that are critical to livestock during drought (Campbell *et al.*, 2000). Fragmentation not only affects livestock and rangeland resources, but it also affects pastoral social networks and the use of reciprocal rights and obligations (Galvin, 2009), factors that are important in fostering resilience under change.

Livelihood diversification

Diversification beyond livestock rearing is also occurring in many regions of the world. For example, in Tanzania and Kenya, Maasai pastoralists also engage in agricultural activities (Galvin, 2009; Homewood *et al.*, 2009). As Mongolia increasingly exploits its mineral wealth, herders are diversifying in a variety of ways, including engaging in unregulated 'artisanal' mining for precious metals, going to work in commercial mines, and providing services to emerging mining towns and camps of artisanal miners. Factors encouraging herders to diversify out of the pastoral sector include population growth, loss of land, drought, livestock raiding, lack of services, agriculture, wage labour and other economic opportunities (Galvin, 2009). While some would argue that movement away from the pastoral sector demonstrates crossing a threshold in terms of being a pastoralist, others contend that diversifying income sources is an adaptive feature of a resilient pastoral system (McCabe, 2003a, b).

Resettlement

Environmental resettlement of pastoral populations includes both conservation resettlement and ecological migration. The former involves moving people out of newly established protected areas, while the latter refers to resettlement to restore fragile or environmentally damaged areas (Rogers and Wang, 2006). In China, where the stated goals of environmental resettlement are to both reduce environmental degradation and also to achieve poverty alleviation, environmental resettlement is becoming widespread in pastoral regions such as Inner Mongolia, Tibet, Qinghai and Gansu Provinces (Yeh, 2005, 2009; Rogers and Wang, 2006). For example, in Gansu Province (see Cao *et al.*, Chapter 8, this volume), 20,000 herders were relocated in response to severe soil erosion. On the Tibetan Plateau, *tuimu huancao* or 'converting pastures to grasslands' policy calls for fencing off areas of purportedly degraded rangelands, some for a few months at a time, and others for several to 10 years, or even permanently, depending on location. There are various ways in which this policy is playing out across the region. In its most dramatic form, found in the 150,000 km^2 Sanjiangyuan nature reserve in Qinghai Province, it is being implemented in conjunction with ecological migration and plans for the resettlement of 100,000 nomads to towns (Foggin, 2008). While participants in this programme receive varying amounts of financial compensation and grain subsidies, Yeh (2009) found that participants cite social and economic problems associated with the programme. Rogers and Wang (2006) found that resettlers in Inner Mongolia suffered from economic deprivation, unfavourable construction, uncertain futures, and spatial ambiguity and uncertainty. However, the community was highly adaptable and able to maintain social cohesion, networks and behaviour, despite the large-scale dislocation.

Economic changes

In many parts of the world, pastoral families have growing aspirations that require cash to purchase commodities, pay for health care for livestock and people and to send children to school. These growing cash needs can increase the perception of poverty among pastoralists. Many families in pastoral societies around the globe hold fewer livestock today than they did in the past (Reid *et al.*, 2008). In parts of Africa and Asia, human populations are growing but livestock populations remain stagnant (Neupert, 1999; Fratkin and Roth, 2005). In parts of Central Asia, recent changes from collectivized to market economies led livestock populations to fall (Behnke, 2003). Galvin (2009) cites a widening gap between 'rich and poor' where income stratification is increasing in some pastoral regions of Africa. In Mongolia, poverty rose and the gap between rich and poor widened following the adoption of a market economy (Nixon and Walters, 2006). Globalization of markets means that pastoral families, even in remote areas, are more connected to markets far away, and when changes in distant markets affect local prices for livestock products or commodities, these long-distance connections can make families more vulnerable. However, when talking about pastoral populations, we should be cautious in talking about traditional measures of poverty.

Land degradation

Land degradation in dryland systems, also referred to as desertification, is characterized by a reduction in the biophysical potential of a system and translates into the reduction in the ability of the land to support human populations, livestock and wild herbivores (Reynolds and Smith, 2002). In 1992, the United Nations adopted the Convention to Combat Desertification (CCD) in order to highlight the severity and extent of dryland degradation and to initiate a coordinated effort to address this global phenomenon. The UNCCD defines desertification as 'land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities'. Some of the biological indicators of degradation include reductions in plant productivity, cover and diversity; reduced soil fertility and soil organic matter content; increases in soil salinization and enhanced erosion. Severe land degradation may directly affect 250 million people in the developing world (Reynolds *et al.*, 2007). The UNCCD asserts that desertification potentially affects one-fifth of the world's human population.

There is much debate about the extent and causes of degradation as well as its potential solutions. Some reports assert that up to 70% of all drylands are desertified, while others suggest this number is closer to 17% (Reynolds and Smith, 2002). There is discussion as to whether the drivers of degradation are 'natural' (e.g. climatic) or 'anthropogenic' (e.g. overgrazing) and whether there is a single cause or too many factors to identify the underlying drivers (Geist and Lambin, 2004). There is also disagreement as to whether degradation is reversible and whether the solution is technological and scientific or socio-economic and institutional (Reynolds and Smith, 2002). Geist and Lambin (2004) conducted a meta-analysis of 132 sub-national case studies on the proximate and underlying causes of dryland degradation. They found that the underlying drivers were generally climatic, economic, institutions, national policies, population growth and remote influences. The proximate drivers included cropland expansion, overgrazing and infrastructure extension. While there were usually multiple factors involved, there were a limited number of common pathways of degradation across the 132 cases. In an effort to explain why drylands may be particularly vulnerable to degradation, Reynolds *et al.* (2007) discuss the 'drylands syndrome' concept. This syndrome describes unique features of drylands, including high variability in climatic conditions, low soil fertility, sparse populations, remoteness from markets, and populations that are physically distant from decision makers and whose interests are generally marginalized and of low priority to decision makers.

Rangeland degradation is an ongoing issue in Northern Asia, although the extent, causes and solutions remain a source of inquiry and debate. Deng (2005) reports (in Meyer, 2006) that severe degradation threatens one-third of all Chinese grasslands. Chinese scientists and government officials assert that 90% of the rangelands on the Tibetan Plateau are experiencing some degree of degradation, a phenomenon that is increasing at a rate of 200 km/year (State Council, 2002); however, substantial evidence supporting this assertion is lacking (Harris, 2010). Over 70% of steppe vegetation in the Xilin River Basin in Inner Mongolia was degraded, a number that had increased over a 14-year timeframe (Tong *et al.*, 2004). In Inner Mongolia, agriculture has greatly infringed on rangelands accompanied by an influx of Han Chinese farmers. Mongolians now comprise only 11% of the population of Inner Mongolia. As a consequence of this expanding agriculture, grazing occurs on increasingly marginalized land, with large implications for degradation. While overgrazing is often cited as the primary driver of degradation and this is motivating rangeland policy, research suggests that other factors, such as climate change (Klein *et al.*, 2004, 2007, 2008) and institutional and policy changes (Williams, 1996; Sneath, 2003; Li *et al.*, 2007) may be the underlying factors driving the observed degradation in regions of North Asian rangelands. Regardless of the drivers, an increasingly degraded system may be less resilient to the suite of stressors we have described in this section as well as to other unexpected system disturbances and shocks.

Vulnerability and Resilience

The fields of vulnerability and resilience science provide a conceptual understanding of a social-ecological system's susceptibility to damage and ability to maintain structure and function in the face of external shocks, disturbances and perturbations. Vulnerability and resilience are terms that are often considered to be inversely related and are two important and dynamic outcomes in our conceptual framework of North Asian rangelands. Below, we discuss the prevailing vulnerability and resilience frameworks that are most in use today.

According to Turner *et al.* (2003) vulnerability is the degree to which a system or subsystem is likely to experience harm due to exposure to a perturbation or stress. This definition of vulnerability requires identifying a threshold above or below which the system is said to be 'harmed or damaged' (Luers, 2005). Stresses or perturbations may arise from human driving forces, natural events, or combinations of the two. Turner *et al.* (2003) describe three components of vulnerability: exposure, sensitivity and resilience. Exposure refers to the contact between a system and a stress or perturbation. It is a function of both the characteristics of the stress or perturbation (e.g. frequency, magnitude and duration) and the components with which it comes into contact (e.g. individuals, households, flora/fauna). The second aspect of vulnerability is sensitivity, which refers to the degree to which an exposed system (or part of a system) is affected by exposure to any set of stresses or perturbations. These exposed units include individuals, groups, economic sectors, places, and parts of ecosystems. Sensitivity depends on the human and environmental condition, such as entitlements, institutions, soil, water, climate, and ecosystem structure and function. Resilience, the final component of Turner's concept of vulnerability, is the ability of the exposed system to resist or recover from damage associated with the stress or perturbation. Here, resilience is a function of the impact of the stress on the system as modified by coping, adjustment and adaptation responses. Thus, the level of vulnerability of a particular population depends on the magnitude of the stress or shock they are exposed to, how sensitive they are to that stress or shock and how well they can cope/adapt in response (Turner *et al.*, 2003).

Based on the work of Holling and others (Holling and Meffe, 1996; Gunderson and Holling, 2002), Walker and Salt (2006) advance a slightly different and broader definition of resilience, as the amount of change a system can absorb without altering its essential structure and function (Walker and Salt, 2006). In this view, resilience is not an inherently good or bad property. Undesirable system states such as totalitarian governments or highly degraded ecosystems are sometimes very resilient. A central tenet of resilience thinking is that change is constant and pervasive, and that learning to live with change is a more successful strategy than trying to control or limit it. Understanding resilience in the context of natural resource management requires that we consider ecosystems and the human societies that are part of and depend upon these systems as linked or coupled social-ecological systems.

A system is comprised of both its parts and the interactions among those parts. The parts are the system's structure and the interactions among those parts are its functions or processes. According to resilience theory, a system is comprised of interacting social and ecological components, as is illustrated by our conceptual framework. For example, a social-ecological system is composed of abiotic components such as soil, water, rocks and air, as well as biotic components such as plants, animals and people. The arrangement of these components in space and time is the structure of the ecosystem. The interactions among the parts include ecosystem functions such as nutrient and water cycling, which provide critical ecological services to humans, and human decisions about resource use, which affect the biological system components. Natural disturbances such as fire, grazing and droughts are important to the function of many ecosystems, and are part of the natural variability of these systems. Attempting to limit these natural disturbances through management may sometimes be successful in the short term, but may have long-term unintended and undesirable consequences (Holling and Meffe, 1996). Learning to live with change also entails anticipating and responding to other kinds of shocks such as economic changes and political transitions.

The concept of feedback loops is important to understanding system dynamics and resilience. A feedback loop is a set of interactions within a system that enables the system to self-regulate its behaviour (negative or stabilizing feedbacks) or that is self-reinforcing (positive or amplifying feedbacks) and may ultimately push a system into an alternative state. In equilibrium rangeland systems, herbivores and plants regulate each other's biomass in a self-correcting negative feedback cycle. Increased plant biomass leads to growing populations of grazing animals, which consume the biomass, eventually leading to food shortages, and decline in the population of animals that releases the plant populations and allows plant biomass to again accumulate. An example of a positive feedback is the relationship between melting snow in alpine zones triggered by climate change. Warming leads to increased melting, which lowers reflectivity of the earth's surface (albedo) thereby increasing heat absorption, which in turn accelerates warming. Managing for a sustainable and resilient system often involves reinforcing the negative or stabilizing feedbacks within that system (Chapin *et al.*, 2010).

A complex adaptive system is a system with many parts and interactions that is capable of learning from and adapting to change (Lynam and Smith, 2004). Because of their complexity, complex adaptive systems behave in unpredictable ways and are subject to uncertainty and surprise. Sources of uncertainty and surprise include natural physical and biological phenomena as well as attributes of human society and behaviour (Meffe *et al.*, 2002). The behaviour of complex systems often results in unanticipated effects of events and interactions, such as those that can result from removal of a key species (such as a predator) in an ecosystem. Synergistic effects occur when multiple events happen together and their effects are multiplicative rather than additive, such as the coincidence of drought and disease. Cumulative effects happen when small events that may have little effect individually, combine over space or time to create large effects. For example, the loss of vegetation cover on a small patch of ground may be insignificant by itself, but when many small patches grow larger and more connected over time, bare areas eventually exceed areas covered with plants, and soil is lost to erosion, potentially leading to desertification.

A key idea in resilience thinking is that a given system may exist in one or more alternative stable states. The shift from grasslands to shrublands in the southwestern USA over the past century is an example of a switch from one stable state to another. The feedbacks that maintained the grassland state were altered, probably due to a combination of prolonged severe drought and excessive livestock grazing, resulting in an increase in shrub cover and loss of most or all perennial grass species (Schlesinger *et al.*, 1990). This change in ecosystem structure also led to a change in the function of the system with altered nutrient distributions and erosion dynamics. Alternative states are separated by thresholds, defined as the point in space and time where the feedbacks that maintained a system in one state are changed such that the system enters a different state that is maintained by a different set of feedbacks. Another way to think about thresholds is as a set of conditions that changes a system's structure and function so much that it loses resilience and enters an alternative state. Once a system crosses a threshold, this change is difficult or impossible to reverse without significant inputs of energy and time.

If change is a constant feature of system behaviour, what enables a system to evolve and adapt, yet to remain recognizable as the same system, with the same essential parts and processes? In other words, what characteristics make a system resilient? How do we know when a system has changed irreversibly (crossed a threshold), or more important, when it is on the verge of an irreversible change? Resilience thinking moves us away from a mindset of controlling complex coupled human–natural systems towards an attitude of understanding, embracing and adapting to change as an integral aspect of system behaviuor. Gunderson and Holling (2002) proposed that ecosystems and by extension, social-ecological systems, undergo an ongoing adaptive cycle of change, called the 'adaptive cycle', whereby a system grows, conserves, collapses and reorganizes time and again. The ability to reorganize, adapt and learn as a system moves through this cycle repeatedly over time is the key to resilience.

To illustrate with an example from North Asia, pastoral systems in Mongolia have undergone several dramatic political economic and environmental shocks over the past century, yet despite these significant changes, basic features of this social-ecological system have remained constant over time – the system has thus far remained resilient. Over the past 100 years the pastoral political economy has transitioned from a quasi-feudal system to a collective system under a socialist command economy, to a newly emerging democracy and free market economy. Although each of these political and economic shifts brought significant changes to the economic structure of pastoral production, the basic characteristics of this social-ecological system remained the same – an extensive, mobile livestock production system relying on multiple species and habitats to support herding households. Over the same period, livestock populations have undergone repeated cycles of growth and decline. In this largely non-equilibrium rangeland system, sudden declines in the livestock population are usually caused by winter snow disasters, which are a naturally occurring density-independent limitation on livestock populations. The most recent such disasters occurred in 1999–2003 and 2010. In 1999, the livestock population in Mongolia had reached some 33 million head (conservation phase of the adaptive cycle), and the rangelands in some areas were at risk of severe degradation. Following several years of drought and two very harsh winters with severe storms, about a third of the 1999 livestock population had perished, and many families were left with no livestock at all (release phase of the adaptive cycle). The loss of livestock, while devastating in its human consequences, provided the opportunity for rangelands that retained their productive potential to recover their productivity, and by 2008, the national herd size had rebounded and reached 40 million animals (growth phase of the adaptive cycle). Another very cold winter in 2009/10 led to a loss of about 20% of the national herd.

Resilience thinking requires attention to the dynamics of cross-scale interactions – that is, the ways that processes and structures at one spatial or temporal scale affect those at scales above and below that focal scale (Peters *et al.*, 2004). Often we cannot understand the consequences of specific events or changes by focusing at a single scale. Processes that occur as broad spatial and long temporal scales often dominate those that occur at finer and faster scales. For example, broad patterns in geomorphology and climate determine the distribution of plant and animal species at more local spatial scales and shorter time periods. However, sometimes fine-scale dynamics may cascade upwards to alter broad-scale patterns. The conversion of grasslands to shrublands and subsequent desertification provides one example of this type of upward cascade, whereby patch-scale dynamics may eventually spread over broad areas, and create feedbacks to atmospheric conditions through the increased albedo associated with large areas of bare ground (Peters *et al.*, 2004).

In the context of Mongolian pastoral social-ecological systems, national-level political and economic changes in the early 1990s resulted in local-scale changes in herder communities across the country, however different communities responded and adapted differently to these changes. Similarly, national-level law relating to the management of pastures has been interpreted and implemented in different ways in different communities. These are examples of the effects of broad-scale changes on fine-scale dynamics. We have also observed examples of fine-scale processes influencing broad-scale events. One example of this in Mongolia may be the influence of many local-scale experiments in community-based rangeland management affecting the direction of national-level policies for pastureland tenure (Fernández-Giménez *et al.*, 2008).

The understanding of social-ecological systems as complex adaptive systems leads to the insight that progress towards solving or managing natural resource challenges requires an inherently interdisciplinary approach. Further, the sustainability and resilience of these complex coupled systems depends upon their ability to adapt and to maintain the self-regulating feedbacks within the system. Maintaining these feedbacks, in turn, requires attention to the 'slow variables' that underlie key processes. In social-ecological systems, the human ability to learn and act on the basis of new information can play a key role in adaptation and self-regulation within the system. This is one reason why various forms of ecological knowledge – local, traditional and scientific – as well as environmental monitoring are critical to the resilience of these systems. Social institutions (rules, norms, policies and laws) that are adaptive, flexible, locally responsive, multi-scale and diverse also promote resilience (Folke *et al.*, 2005). Successful adaptive governance institutions help maintain the resilience of desirable systems in the face of change, but also recognize the opportunity and need to transform systems in the face of crisis – to create new, more desirable systems. Work by Berkes *et al.* (2003) and Walker and Salt (2006) both highlight the importance for resilience of social and ecological diversity, variability and innovation; ecological knowledge and the ability to observe and respond to key slow variables that produce changes in ecosystems and the services they provide; and strong social relationships and institutions that foster learning and adaptation.

Resilience and Adaptation in Pastoral Systems

Adaptation is the set of actions, attitudes, activities and decisions that maintain the capacity to deal with current or future change or shocks to a social-ecological system (Nelson *et al.*, 2007; Agrawal, 2008). Agrawal (2008) argues that livelihood adaptation to climate change among the rural poor requires strong local institutions as well as improved cross-scale interactions among institutions operating at different levels, and identifies five key strategies for adaptation employed by the rural poor: mobility, storage, diversification, resource pooling and exchange. Agrawal asserts that 'adaptation to climate change is inevitably local,' (p. 3), that institutions shape adaptation in critical ways, and that 'the gap in current knowledge about the role of institutions in adapting to climate change is remarkably large' (p. 1).

We have discussed in previous sections the strategies pastoralists have developed over centuries and millennia that have enabled them to deal with the inherent variability in their biophysical and socioeconomic environments. Many traditional pastoral systems thus already have incorporated many of the principles of resilience, and this in turn has enhanced their ongoing ability to adapt to and cope with an environment of constant change. However, the magnitude of changes and stresses that now face many pastoral societies is perhaps greater than ever before, calling into question the continued resilience and adaptive capacity of such systems. Resilience thinking is an emerging field of study and practice. In this book we explore what resilience means in pastoral social-ecological systems, how resilience of positive states can be strengthened, and how community-based rangeland management influences resilience of these systems.

Changing Meaning and Role of 'Community' in Rangeland Management

The term 'community' has multiple and sometimes conflicting meanings when applied to human populations. Community is often defined by geography (a group of co-located individuals circumscribed by geographic boundaries); common interests (regardless of residence); or shared values and norms related to livelihoods, culture or occupation. Community may be defined in opposition to the state ('community-based' in contrast to 'state-run') or individual actors (as in collective versus individual action). Wilkinson (1991), a rural sociologist, defines community as a 'field of action' and community development as collective efforts by community members to have a positive impact on this social field. Agrawal and Gibson (2001) caution against over-simplication of community, especially in relation to the growing interest in community-based natural resource management, and advance a definition of community as a network of actors and institutions. Agrawal and Gibson (2001) question conventional definitions of 'community' as a territorially-defined social group bound by common values and norms. Such conventional notions of community often obscure profound differences within communities in resource access and use, voice, and power along gender, age, ethnic, religious, class, or caste lines (Agrawal and Gibson, 2001). They also fail to account for the temporally dynamic composition of many pastoral social groupings, where membership may shift annually, seasonally, or more often (Scoones, 1994; Niamir-Fuller, 1999; Fernández-Giménez, 2002). In this book, we do not attempt to impose our own definition of community, but rather to explore the diversity of meanings and manifestations of community that emerge from the experiences of the seven cases included here.

In addition to debating the meaning of community, since the mid-20th century, scholars have argued about the continued relevance of community in an increasingly urban, industrial and globalizing world. Putnam (2000) chronicled the decline of social capital in the USA, and many studies from the USA and elsewhere have documented changing demographic, economic and social structure of rural communities (Marin-Yaseli and Martinez, 2003; Lee and Field, 2005). Pronouncements about the demise of 'community' may be premature, given recent emergence of virtual communities that inhabit cyberspace, as well as the growing popularity of decentralized, 'community-based' institutions for natural resource governance. Indeed, place-based social networks and social relationships of trust and reciprocity embedded in them (also known as social capital) are increasingly recognized as critical to sustainable management for resilient social-ecological systems (Pretty and Ward, 2001; Plummer and FitzGibbon, 2006). Through the stories of seven grassland communities in different regions of China and Mongolia, this book narrates the trajectory of seven communities in transition, the emergent theme of which is the mutually reinforcing relationship between the well-being of communities of resource users and the health of the rangelands on which they depend.

Community-based natural resource management (CBNRM) has been defined as 'a process by which landholders gain access and use rights to, or ownership of, natural resources; collaboratively and transparently plan and participate in the management of resource use; and achieve financial and other benefits from stewardship' (Child and Lyman, 2005). Co-management refers to a management regime where decision-making authority is shared between local people and local, regional or national government (Pinkerton, 1989).

Benefits attributed to these related approaches include: (i) increased implementation of and compliance with management decisions (Ostrom, 1990; Western and Wright, 1994); (ii) application of diverse knowledge sources to management, including both local ecological knowledge and science (Berkes *et al.*, 2003); (iii) improved on-the-ground resource management; (iv) increased monitoring and adaptive management; (v) decreased conflict over resources; (vi) increased trust and strengthened relationships (social capital) within the community (Pretty and Ward, 2001); (vii) improved livelihoods; (viii) greater community capacity to mobilize resources for community benefit; (ix) improved environmental conditions; and (x) more resilient social-ecological systems (Walker and Salt, 2006). These claims lead to a suite of important questions, such as: Does community-based natural resource management live up to its promise? How should 'success' be defined and who should define it? What factors influence the process and outcomes of CBNRM? Are the outcomes of CBNRM really different or better than existing management regimes?

The central question driving much research on community-based natural resource management is: How can communities of resource users effectively organize themselves to self-regulate their use of shared resources? There is a long tradition of theory development and empirical research on CBNRM that spans many disciplines, including political science, anthropology, economics, sociology, and various natural resource sciences. Many of these investigations were launched in the 1970s, following the publication in 1968 of Garrett Hardin's famous article, 'The Tragedy of the Commons.' In response to Hardin's assertion that the only ways to overcome the incentive for individuals to overuse shared resources were either privatization or government regulation, many scholars set out to document successful common property regimes and identify design principles for successful community-based resource management (McCay and Acheson, 1987; Ostrom, 1990; Bromley, 1992; Wade, 1994; Baland and Platteau, 1996). Studies of common pool resource (CPR) management regimes include investigation of institutions for the management of forests (Gibson *et al.*, 2000), fisheries, rangelands (Lane, 1998) and irrigation systems (Lam, 1998), among other resources. Most research was based on qualitative case studies, and the most influential of these works is Elinor Ostrom's (1990) book *Governing the Commons*, in which Ostrom analysed 14 long-standing common property management systems and proposed eight key design principles for successful CPR management institutions (Ostrom, 1990). More recently, drawing on Ostrom's work and that of others, Agrawal (2002) distilled this vast literature to identify six categories and 35 specific variables that help explain the performance of common property management regimes.

In addition to research on long-standing and traditional community-based management institutions, there has been growing interest in understanding the process and performance of contemporary, recently-established CBNRM, community-based conservation and co-management institutions. Initially, much of the work on CBNRM and its sister institutions was positive and somewhat promotional in nature, featuring many success stories (Western and Wright, 1994; Child and Lyman, 2005). More recently, much of the scientific literature has been more critical of CBNRM and questioned whether the promise of these approaches can really be achieved (Kellert *et al.*, 2000; Brosius *et al.*, 2005). An important contribution of this work is the growing awareness of the importance of understanding the historical, cultural and social-political contexts of specific CBNRM cases.

Research on CBNRM is challenging. Although many participants in CBNRM are convinced of its benefits, there is still relatively little empirical scientific evidence to support claims that CBNRM improves environmental or social conditions. Changes in environmental conditions are difficult to measure directly, in part because the environment may respond slowly to changes in management. Even when changes are documented, variation in environmental conditions among case study sites and other confounding factors make it difficult to establish causal links between an improved environment and community-based management. Social outcomes can also be difficult to measure. For example, how do we measure intangible variables such as trust and hope? It may also be difficult to identify the appropriate spatial scale or level of social organization for measuring social outcomes. Are household-level indicators sufficient? Is it possible to detect community- or regional-level changes in economic and social conditions related to implementation of CBNRM? Finally, a basic principle of scientific research is the use of 'control' or 'comparison' cases where the intervention under study does not occur. This enables the researcher to determine whether the observed changes would have happened anyway, even in the absence of the intervention. Very few studies of community-based resource management effectively compare CBNRM to alternative management regimes in similar social and environmental settings. In addition, there are few longitudinal studies of CBNRM, especially longitudinal studies of multiple cases, which would help clarify the pathways and causal mechanisms for success or failure.

On the frontier of research on CBNRM are investigations of the relationship between CBNRM and the resilience of social-ecological systems. Resilience thinking is a cutting-edge area of interdisciplinary theory-building and research. Most of the work on resilience is still largely conceptual, and where empirical studies have been done, they are primarily descriptive and qualitative (Walker and Salt, 2006). The existing literature on resilience in social-ecological systems strongly suggests that community-based institutions may play a key role in fostering the resilience of communities and the ecosystems they inhabit. Davidson-Hunt and Berkes (2003) suggest that local management institutions enhance resilience because: (i) management practices are locally adapted and based on local ecological knowledge; (ii) local institutions are 'close to the ground' and able to observe and adapt rapidly, making and learning from small mistakes where centralized bureaucracies make large ones; and (iii) there is a tremendous diversity of institutional arrangements among local CBNRM groups, and such diversity increases the likelihood of learning what works. In addition, we propose that (iv) CBNRM has been shown to strengthen social capital (Wagner and Fernández-Giménez, 2008), and social capital, in turn, is thought to be a key to adaptive capacity in communities (Adger, 2003; Berkes *et al.*, 2003; Walker and Salt, 2006). Further, we suggest that (v) some CBNRM organizations promote social learning, an intentional process of collective self-reflection through interaction and dialogue among diverse participants (Keen and Mahanty, 2006). Social learning is promoted in part due to CBNRM groups' attention to monitoring and adaptive management, and their emphasis on learning and education. We suggest that this attention to monitoring and collective learning through adaptive management strengthens feedbacks between social and ecological systems. As rural communities face increasing environmental stresses as well as unpredictable economic and political shocks, the ability to learn and adapt is critical to their sustainability and resilience.

The Framework

We developed a conceptual framework based on the seven case studies presented in this book through a series of workshops on 'Poverty, Vulnerability and Resilience in North Asian Rangelands: Case Studies of Community-based Rangeland Management in China and Mongolia'. The workshops were each attended by 45 scholars and practitioners working on seven community-based rangeland management projects in China and Mongolia. Developing the framework through a collaborative process entailed a series of steps with different groups of participants. Each case-study team first identified keywords and themes that were most descriptive of their individual case studies. A sub-set of the workshop participants then compiled all of the keywords and categorized them into broader themes, which ranged from topics such as 'main drivers of change', to 'different types of community-based rangeland management approaches' and 'outcomes'. A group of participants then identified the main categories and linked them in a diagram, which illustrated the key drivers of change, the factors that mediated how the system responded to the drivers, the way in which community-based rangeland management interfaced with the system and its changes, and shared lessons and outcomes. This general framework was then presented to the whole group and a discussion ensued, after which further refinements were made to the framework based on the whole group's input. Through this iterative process, a conceptual framework emerged to illustrate the case studies presented in this book. The framework helps to illustrate some of the common properties and processes of the individual case studies; to identify key drivers, interactions and feedbacks among the various system components; to demonstrate how certain underlying factors can mediate how the system responds; and to illustrate the role of community-based rangeland management in enhancing system resilience.

The framework describes two different scenarios: a vulnerable social-ecological system (Fig. 1.1) and a resilient socio-ecological system (Fig. 1.2). The central focus of both scenarios is the strongly linked social-ecological system. This linked system is represented by the central, main box in each figure (A) with the two social and ecological sub-components (B1 and B2, respectively, in both Figs 1.1 and 1.2) connected by arrows. Decisions and actions emerging from the social system, and services and benefits from the ecological system, maintain a continuous and dynamic interplay between these tightly coupled systems. For example, the household responsibility system in China reduced pastoral mobility, with detrimental effects on the rangelands. The resulting degradation led to grazing removal policies and also sparked the formation of community-based rangeland management (CBRM) practices that, in turn, affected grazing resources and the quantity and quality of rangeland products.

In both the resilient and less resilient scenarios, we consider a suite of local to global drivers that are acting on the system (boxes C1–4). These drivers fall into four broad categories: climate changes and disasters, globalization, population changes and policy.

The first set of drivers includes climate changes (e.g. gradual warming/drying, drought, and snow disasters) and natural disasters (e.g. earthquakes), powerful forces that are currently shaping and transforming North Asian rangeland systems. *Climate changes and disasters* can lead to enhanced degradation and a system that is more vulnerable to physical, biological and socio-economic perturbations. In Huolonggou village, Sichuan, China, for example, earthquakes and landslides destroyed stone fences, which increased conflicts over land, fostered inappropriate grazing practices and contributed to rangeland degradation. Drought and water shortages were also common problems experienced across most of

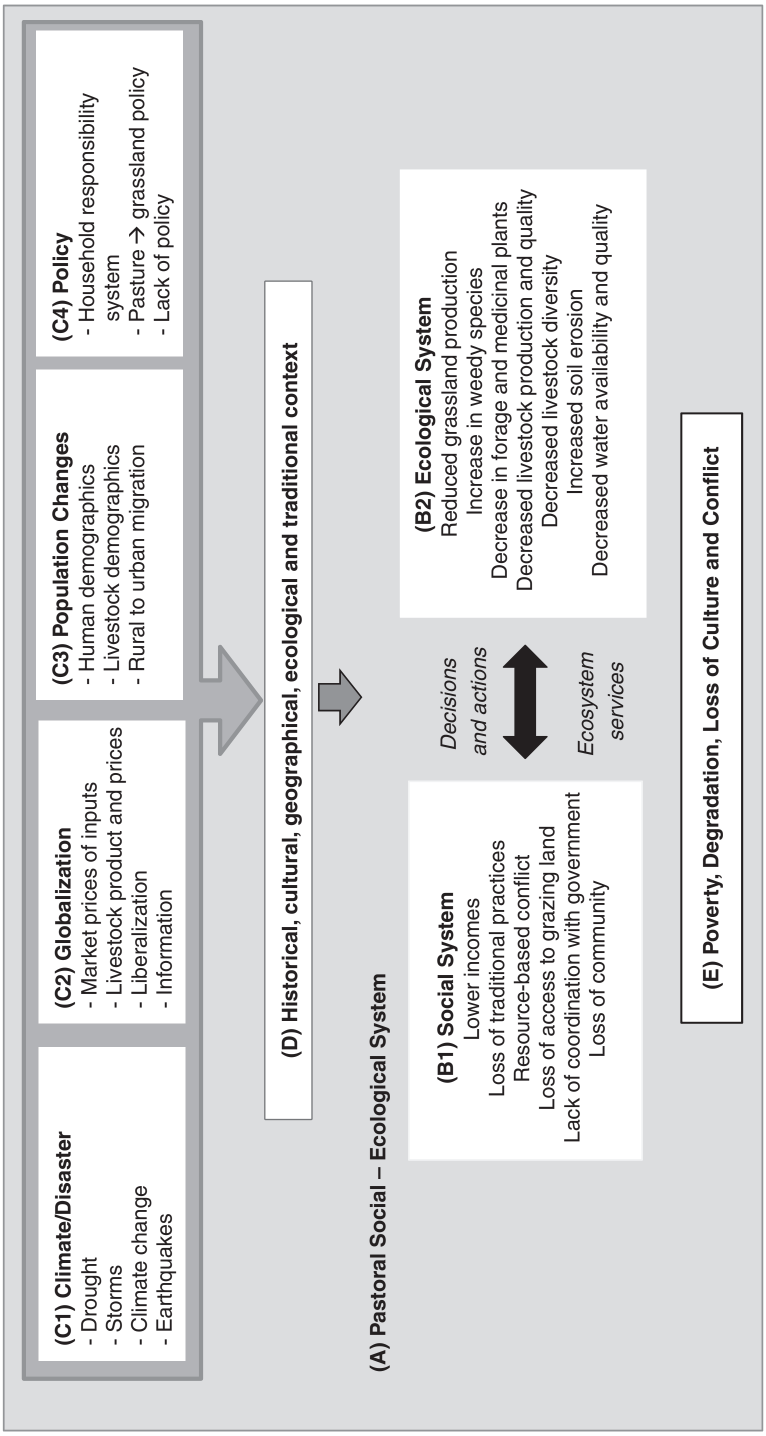

Fig. 1.1. A vulnerable social-ecological pastoral system.

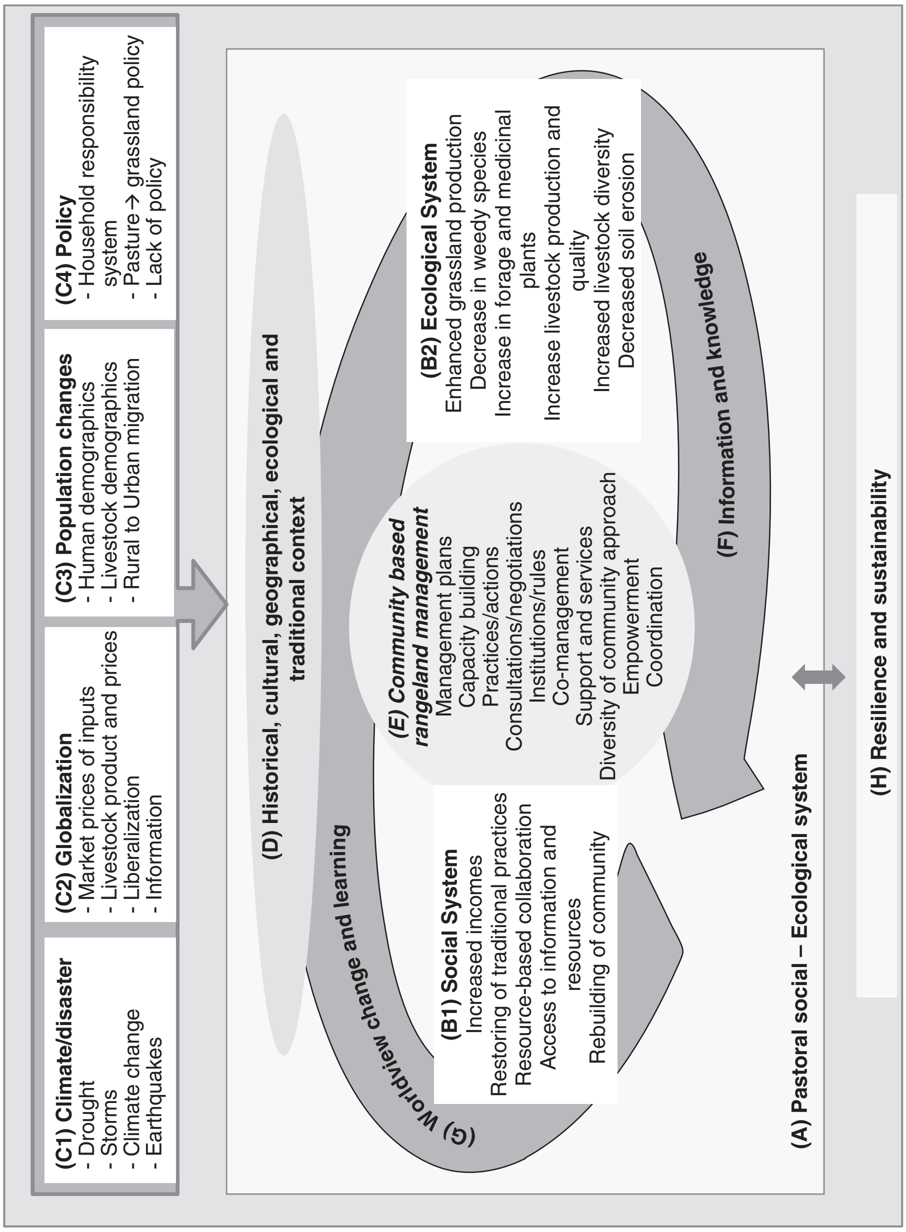

Fig. 1.2. A resilient social-ecological pastoral system.

the seven cases studies presented in this book. In Ikhtamir, Mongolia, water shortages and dry rivers altered grazing patterns so that herders concentrated animals in areas where water was available. Episodic events and disasters can themselves be the shocks that reveal the underlying vulnerability of a social-ecological system as occurred during the Mongolian *dzud* or snow disasters from 1999 to 2002, which resulted in large negative and direct impacts on pastoral livelihoods.

The second driver in the framework, *globalization*, influences the market prices of goods, which can affect pastoral production decisions. For example, in the Jinst, Mongolia and Pifang, China cases the high price of cashmere in the 1990s led to a dramatic increase in goats, which then caused overgrazing and subsequent degradation. Ironically, the opposite market trend had a similar ecological outcome later in Jinst, when the 2007 price reduction in cashmere and all livestock products led to an increase in overall herd size and further contributed to overgrazing. In Sichuan, overharvesting of certain medicinal plants, for which there is now a global market, further contributed to degradation in the region. Globalization also changes the flow of information and influences economic and other policies, such as economic liberalization and movement away from a planned to a market-oriented economy, with the cascading effects that accompany these policies as experienced in both China and Mongolia. Moreover, the Chinese state's insertion of a more 'Western style' rangeland management model, as compared to the traditional pastoral models in place, emerged in part from the global exchange of ideas and information.

Population dynamics are the third type of driver in the conceptual framework as they greatly alter the dynamics of the coupled social-ecological system. Population dynamics can include not only increasing human and livestock population pressures, but also reduced herder populations due to patterns of rural to urban migration. In both the China and Mongolian cases, the economic opportunities and the social and other services that cities provide are attracting the younger generation of pastoralists who are leaving their traditional homes and livelihoods, with important implications for the resilience and functioning of these systems.

The fourth system driver, *policy changes*, are having transformative effects on pastoral social-ecological systems in North Asian rangelands, as Dalintai *et al.* describe in Chapter 3 of this book. These policies include the household responsibility system and the pasture to grasslands policy in China. The lack of policy and the resulting management vacuum can also be a strong system driver, as is illustrated by the Mongolian cases where the lack of an enabling legal framework for rangeland tenure and ineffective enforcement of rangeland laws led to negative social and ecological outcomes.

The way in which these ongoing changes impact the social-ecological system depends in part on the historical, geographic, ecological and cultural/traditional context in which these events unfold (D). The context is the lens through which the perturbations are focused and the background that mediates how these factors play out within the system. For example, sites that are drier may be more resilient to changing grazing pressures and more adapted to deal with drought than more mesic sites, whereas sites with a history of conflicts over access to pastures are more likely to experience this type of conflict again. Moreover, as the theme of 're-creating community' in the synthesis chapter of this book suggests, CBNRM arrangements may be more likely to form in regions where there is a history and tradition of community use and management of natural resources. For example, in many of the seven case studies of this book, the newly formed CBNRM programmes were founded on existing kinship relationships or traditional social networks and community organizations. The overarching outcomes of these system dynamics (E in Fig. 1.1, H in Fig. 1.2) differ between the resilient and vulnerable systems, as discussed below.

In the vulnerable social-ecological system (Fig. 1.1), the interactions of the drivers and the existing context result in negative changes to the social-ecological system, as described in boxes B1 and B2. The Huolonggou village case study, for example, describes increasing conflicts with neighbouring communities, negative vegetation changes, and decreases in yak size and milk production. These negative outcomes are summarized in E (Fig. 1.1). A less resilient system is characterized by large-scale rangeland degradation, poverty, conflict over resources, and loss of culture and identity among the community. This was generally the situation in the case study sites prior to the implementation of the CBNRM projects, although the degree of conflict, poverty and degradation varied from case to case. For example, of the sites that quantified the amount of degradation that had occurred prior to CBNRM and the development of collective action, the estimates ranged from 33% in Pifang, 59% in Sonid Left Banner and >90% in Maqu. Moreover, poverty was a major feature of Sichuan's Huolonggou village and the two Mongolian cases and less prominent in some of the other cases.

The resilient social-ecological system (Fig. 1.2) shares the same overall structure (A, B) and the same drivers (C) and context (D) as the vulnerable pastoral social-ecological system. However, in this scenario, the social and ecological conditions (B1 and 2) improve as compared to the less resilient scenario. For example, the Maqu, China case documents the transition from a less productive and lower quality vegetation to improved forage quantity and quality with the multi-household grazing arrangement. The resilient system scenario reveals intact ecological and social systems that are able to withstand external stresses, can handle surprises, and can use multiple sources of knowledge and learning to adapt to changing conditions. These social and ecological features result in a more resilient system that can maintain its overall structure and function, can adapt to new and uncertain conditions, and is sustainable (H) despite the external shocks, the uncertainties, and the overall dynamic nature of the system and the drivers that influence it. These outcomes were achieved, at least in the short-term, following the implementation of the CBNRM programmes. A key distinguishing feature of the resilient system is the prominent role of community-based rangeland management (E). Here, local communities and resource users are empowered to organize, make decisions, and to manage and monitor their resources. There is a high degree of coordination and cooperation within and between user groups and other stakeholders. There are opportunities for capacity building at the local level and for practices and actions to be implemented on the ground according to local context and changing conditions. Participants and governments embrace a suite of community approaches, including co-management, depending on historic, geographic, political, economic and other contextual factors. The process of developing and implementing management plans, and monitoring their outcomes, involves consultation with and negotiations among various stakeholder groups. Institutions and rules are developed to formalize these community-based arrangements and communities receive the support and services necessary to successfully implement all stages of their plans. In Chapter 12 of this book, Fernández-Giménez *et al.* provide a detailed account of the types of community, the different forces leading to their formation, common and different features of the community practices across the seven case studies, and the lessons learned to date from these different cases.

Another distinguishing feature of the resilient system is the ability to make adjustments and to adapt over time to changing conditions. This process is facilitated by the constant acquisition of different types of knowledge and information about the system, the driving factors, and the use of both traditional and indigenous knowledge supplemented by outside technology and information where appropriate (F). In Sonid Left Banner and Jinst, technical training and on-site demonstrations on rangeland management are part of their ongoing rangeland improvement programmes. In Huolonggou village, herders are using digital photography to monitor changes in grassland condition over time. The Jinst case asserts that 'Learning is promoted when herders are mobilized to do something collaboratively and are supported with technical expertise.' Thus, the technical input is only effective when combined with local expertise and a favourable and collaborative framework for implementing change. In the resilient scenario, users and managers constantly adjust and learn from their own experiences and those of others. Stakeholders carry an open and flexible world-view (G) to address existing and novel situations.

This framework allows us to identify some of the common features across the case studies that are the subject of this book. By presenting this framework, we are not suggesting that all cases fit exactly within each aspect of the framework. Rather, each case study has its unique and illustrative attributes that make place-based experiences distinct, richly woven, and beyond generalization. That is why each case has its own chapter in this book. However, we can abstract some common features that allow us to draw conclusions about community-based rangeland management under change so that we can provide a way to investigate these issues beyond the boundaries of these case studies. The characteristics and linkages described in our conceptual framework demonstrate how community-based rangeland management can enhance the resilience of coupled systems to change and may provide a guiding framework for understanding and managing change in other social-ecological systems around the world.

Note

[1] In this chapter, we use the words drylands and rangelands interchangeably. The term 'drylands' reflects the defining climatic characteristic of these systems, while the word 'rangelands' describes the predominant land use in these systems.

References

Adger, W.N. (2003) Social capital, collective action, and adaptation to climate change. *Economic Geography* 79(4), 387–404.

Agrawal, A. (2002) Common resources and institutional sustainability. In: Ostrom, E. (ed.) *The Drama of the Commons*. National Research Council, Washington, DC, pp. 41–85.

Agrawal, A. (2008) The Role of local institutions in adaptation to climate change. *Ifri Working Paper* W08i-3, 45.

Agrawal, A. and Gibson, C. (2001) *Communities and the Environment: Ethnicity, Gender, and the State in Community-Based Conservation*. Rutgers University Press, New Brunswick, New Jersey.

Angerer, J., Han, G., Fujisaki, I. and Havstad, K.M. (2008) Climate change and ecosystems of Asia with emphasis on Inner Mongolia and Mongolia. *Rangelands* 30(3), 46–51.

Asner, G.P., Elmore, A.J., Olander, L.P., Martin, R.E. and Harris, A.T. (2004) Grazing systems, ecosystem responses, and global change. *Annual Review of Environment and Resources* 29, 261–299.

Augustine, D.J. and McNaughton, S.J. (2006) Interactive effects of ungulate herbivores, soil fertility, and variable rainfall on ecosystem processes in a semi-arid Savanna. *Ecosystems* 9(8), 1242–1256.

Baland, J. and Platteau, J. (1996) *Halting Degradation of Natural Resources: Is There a Role for Rural Communities?* Clarendon Press, Oxford, UK.

Banks, T. (2001) Property rights and the environment in pastoral China: evidence from the field. *Development and Change* 32(4), 717–740.

Banks, T. (2003) Property rights reform in rangeland China: Dilemmas on the road to the household ranch. *World Development* 31(12), 2129–2142.

Banks, T., Richard, C., Ping, L. and Zhaoli, Y. (2003) Community-based grassland management in Western China – rationale, pilot project experience, and policy implications. *Mountain Research and Development* 23(2), 132–140.

Bauer, K.M. (2006) Common property and power: insights from a spatial analysis of historical and contemporary pasture boundaries among pastoralists in Central Tibet. *Journal of Political Ecology* 13, 24–47.

Behnke, R.J. (2003) Reconfiguring property rights and land use. In: Kerven, C. (ed.) *Prospects for Pastoralism in Kazakstan and Turkmenistan: From State Farms to Private Flocks*. Routledge Curzon, New York, pp. 75–107.

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

Berlow, E.L., D'Antonio, C.M. and Swartz, H. (2003) Response of herbs to shrub removal across natural and experimental variation in soil moisture. *Ecological Applications* 13(5), 1375–1387.

Bestelmeyer, B.T., Brown, J.R., Havstad, K.M., Alexander, R., Chavez, G. and Herrick, J. (2003) Development and use of state-and-transition models for rangelands. *Journal of Range Management* 56(2), 114–126.

Beyene, F. (2010) Customary tenure and reciprocal grazing arrangements in Eastern Ethiopia. *Development and Change* 41(1), 107–129.

Bonte, P., Guillaume, H. and Zecchin, F. (1996) Nomads: changing societies and environments. *Nature & Resources* 32(1), 2–10.

Boone, R.B. (2007) Effects of fragmentation on cattle in African Savannas under variable precipitation. *Landscape Ecology* 22, 1355–1369.

Boone, R.B., Coughenour, M.B., Galvin, K.A. and Ellis, J.E. (2002) Addressing management questions for Ngorongoro conservation area, Tanzania, using the Savanna modelling system. *African Journal of Ecology* 40(2), 138–150.

Briske, D.D., Fuhlendorf, S.D. and Smeins, F.E. (2003) Vegetation dynamics on rangelands: a critique of the current paradigms. *Journal of Applied Ecology* 40(4), 601–614.

Briske, D.D., Fuhlendorf, S.D. and Smeins, F.E. (2006) A unified framework for assessment and application of ecological thresholds. *Rangeland Ecology & Management* 59(3), 225–236.

Bromley, D. (1992) *Making the Commons Work: Theory, Practice, and Policy*. ICS Press, San Francisco, California.

Brosius, P.J., Tsing, A.L. and Zerner, C. (2005) *Communities and Conservation: Histories and Politics of Community-Based Natural Resource Management*. Alta Mira Press, Walnut Creek, California.

Burnsilver, S. (2008) Grassland/rangeland intensification vs extensification debate. In: *People and Policy in Rangeland Management – a Glossary of Key Concepts*. Science in China Press, Bejing, China, pp. 150–158.

Callaway, R.M., Brooker, R.W., Choler, P., Kikvidze, Z., Lortie, C.J., Michalet, R., Paolini, L., Pugnaire, F.I., Newingham, B., Aschehoug, E.T., Armas, C., Kikodze, D. and Cook, B.J. (2002) Positive interactions among Alpine plants increase with stress. *Nature* 417(6891), 844–848.

Campbell, D.J., Gichohi, H., Mwangi, A. and Chege, L. (2000) Land use conflict in Kajiado district, Kenya. *Land Use Policy* 17(4), 337–348.

Chapin, F.S., Carpenter, S.R., Kofinas, G.P., Folke, C., Abel, N., Clark, W.C., Olsson, P., Smith, D.M.S., Walker, B., Young, O.R., Berkes, F., Biggs, R., Grove, J.M., Naylor, R.L., Pinkerton, E., Steffen, W. and Swanson, F.J. (2010) Ecosystem stewardship: sustainability strategies for a rapidly changing planet. *Trends in Ecology & Evolution* 25(4), 241–249.

Child, B. and Lyman, M.W. (2005) *Natural Resources and Community Assets: Lessons from Two Continents*. Sand County Foundation and The Aspen Institute, Madison, Wisconsin and Washington, DC.

Christensen, J.H., Hewitson, A., Busuioc, A., Chen, X., Gao, I., Held, R., Jones, R.K., Kolli, W.T., Kwon, R., Laprise, V., Magaña Rueda, L., Mearns, C.G., Menéndez, J., Räisänen, A., Rinke, A., Sarr, A. and Whetton, P. (2007) Regional climate projections. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. and Miller, H.L. (eds) *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK and New York.

Cincotta, R.P., Zhang, Y. and Zhou, X. (1992) Transhumant Alpine pastoralism in northeastern Qinghai province: an evaluation of livestock population response during China's agrarian economic reform. *Nomadic Peoples* 30, 3–25.

Clark, W.C. and Dickson, N.M. (2003) Sustainability science: the emerging research program. *Proceedings of the National Academy of Sciences of the United States of America* 100(14), 8059–8061.

Collins, S.L. (1987) Interaction of disturbances in Tallgrass Prairie – a field experiment. *Ecology* 68(5), 1243–1250.

Collins, S.L., Knapp, A.K., Briggs, J.M., Blair, J.M. and Steinauer, E.M. (1998) Modulation of diversity by grazing and mowing in native Tallgrass Prairie. *Science* 280(5364), 745–747.

Cubasch, U. and Meehl, G.A. (2001) Projections of future climate change. In: Houghton, J.T., Griggs, D.J., Noguer, M., Van Der Linden, P.J., Dai, X., Maskell, K. and Johnson, C.A. (eds) *Climate Change: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. University of Cambridge Press, Cambridge, UK.

Davidson-Hunt, I.J. and Berkes, F. (2003) Nature and society through the lens of resilience: toward a human-in-ecosystem perspective. In: Berkes, F., Colding, J. and Folke, C. (eds) *Navigating Social-Ecological Systems: Building Resilience for Complexity and Change*. Cambridge University Press, Cambridge, UK, pp. 53–82.

Davies, J. (2008) Pastoralism and pastoralists. In: *People and Policy in Rangeland Management – a Glossary of Key Concepts*. Science in China Press, Beijing, China, pp. 64–74.

Deng, X.Z. (2005) Some issues on population, resources, and environment in the developing China. Center for Chinese Agricultural Policy, Chinese Academy of Sciences, Beijing, China.

Dyson-Hudson, R. and Dyson-Hudson, N. (1980) Nomadic pastoralism. *Annual Review of Anthropology* 9, 15–61.

Easterling, W.E., Aggarwal, P.K., Batima, P., Brander, K.M., Erda, L., Howden, S.M., Kirilenko, A., Morton, J., Soussana, J.F., Schmidhuber, J. and Tubiello, F.N. (2007) Food, fibre and forest products. In: Canziani, O.F., Parry, M.L., Palutikof, J.P., Van der Linden, P.J. and Hanson, C.E. (eds) *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK, pp. 273–313.

Ellis, J.E. and Swift, D.M. (1988) Stability of African pastoral ecosystems – alternate paradigms and implications for development. *Journal of Range Management* 41(6), 450–459.

FAO (2001) Pastoralism in the New Millennium. *FAO Animal Production and Health Paper* 150, Rome, Italy.

Fernández-Giménez, M.E. (2002) Spatial and social boundaries and the paradox of pastoral land tenure: a case study from postsocialist Mongolia. *Human Ecology* 30(1), 49–78.

Fernández-Giménez, M.E. and Allen-Diaz, B. (1999) Testing a non-equilibrium model of rangeland vegetation dynamics in Mongolia. *Journal of Applied Ecology* 36, 871–885.

Fernández-Giménez, M.E. and Le Febre, S. (2006) Mobility in pastoral systems: dynamic flux or downward trend? *International Journal of Sustainable Development and World Ecology* 13, 1–22.

Fernández-Giménez, M.E. and Swift, D.M. (2003) Strategies for sustainable grazing management in the developing world. In: Allsopp, N. *et al*. (eds) *Proceedings of the VIIth International Rangelands Congress,* 26 July–1 August 2003, Durban, South Africa, pp. 821–831.

Fernández-Giménez, M.E., Kamimura, A. and Batbuyan, B. (2008) *Implementing Mongolia's Land Law: Progress and Issues*. Final Report to the Central Asian Legal Exchange. Nagoya University, Nagoya, Japan.

Foggin, J.M. (2008) Depopulating the Tibetan grasslands: national policies and perspectives for the future of Tibetan herders in Qinghai province, China. *Mountain Research and Development* 28(1), 26–31.

Folke, C., Hahn, T., Olsson, P. and Norberg, J. (2005) Adaptive governance of social-ecological systems. *Annual Review of Environment and Resources* 30, 441–473.

Folland, C.K. and Karl, T.R. (2001) Observed climate variability and change. In: Houghton, J.T., Griggs, D.J., Noguer, M., Van Der Linden, P.J., Dai, X., Maskell, K. and Johnson, C.A. (eds) *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK, pp. 99–181.

Fratkin, E. and Moir, A. (2005) Pastoralists and the state: an editorial introduction. *Geography Research Forum* 25, 1–13.

Fratkin, E. and Roth, E.A. (2005) *As Pastoralists Settle: Social, Health, and Economic Consequences of Pastoral Sedentarization in Marsabit District, Kenya*. Kluwer Academic Publishers, Dordrecht, the Netherlands.

Friedel, M.H. (1991) Range condition assessment and the concept of thresholds – a viewpoint. *Journal of Range Management* 44(5), 422–426.

Galvin, K.A. (2008) Responses of pastoralists to land fragmentation: social capital, connectivity and resilience. In: Galvin, K.A., Reid, R.S., Behnke, R.H. and Hobbs, N.T. (eds) *Fragmentation in Semi-arid and Arid Landscapes*. Springer, Dordrecht, the Netherlands, pp. 369–389.

Galvin, K.A. (2009) Transitions: pastoralists living with change. *Annual Review of Anthropology* 38, 185–198.

Geist, H.J. and Lambin, E.F. (2004) Dynamic causal patterns of desertification. *Bioscience* 54(9), 817–829.

Gibson, C., McKean, M.A. and Ostrom, E. (2000) *People and Forests: Communities, Institutions and Governance*. Cambridge University Press, New York.

Gunderson, L.H. and Holling, C.S. (2002) *Panarchy*. Island Press, Washington, DC.

Gunin, P.D., Vostrokova, E.A., Dorofeyuk, N.I., Tarasov, P.E. and Black, C.C. (1999) *Vegetation Dynamics of Mongolia*. Kluwer Academic Publishers, Dordrecht, the Netherlands.

Harris, R.B. (2010) Rangeland degradation on the Qinghai-Tibetan Plateau: a review of the evidence of its magnitude and causes. *Journal of Arid Environments* 74(1), 1–12.

Havstad, K.M., Peters, D.P.C., Skaggs, R., Brown, J., Bestelmeyer, B., Fredrickson, E., Herrick, J. and Wright, J. (2007) Ecological services to and from rangelands of the United States. *Ecological Economics* 64(2), 261–268.

Heisler-White, J.L., Blair, J.M., Kelly, E.F., Harmoney, K. and Knapp, A.K. (2009) Contingent productivity responses to more extreme rainfall regimes across a grassland biome. *Global Change Biology* 15(12), 2894–2904.

Hilbig, W. (1995) *The Vegetation of Mongolia*. SPB Academic Publishing, Amsterdam, the Netherlands.

Ho, P. (2000) China's rangelands under stress: a comparative study of pasture commons in the Ningxia Hui Autonomous Region. *Development and Change* 31(2), 385–412.

Ho, P. (2001) Rangeland degradation in north China revisited: a preliminary statistical analysis to validate non-equilibrium range ecology. *Journal of Development Studies* 37(3), 99–133.

Hobbs, N.T., Galvin, K.A., Stokes, C.J., Lackett, J.M., Ash, A.J., Boone, R.B., Reid, R.S. and Thornton, P.K. (2008) Fragmentation of rangelands: implications for humans, animals, and landscapes. *Global Environmental Change – Human and Policy Dimensions* 18(4), 776–785.

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

Homewood, K., Trench, P.C. and Kristjanson, P. (2009) Staying Maasai? Pastoral livelihoods, diversification and the role of wildlife in development. In: Homewood, K., Kristjanson, P. and Trench, P.C. (eds) *Staying Maasai? Livelihoods, Conservation and Development in East African Rangelands*. Springer, New York, pp. 369–408.

Huxman, T.E., Smith, M.D., Fay, P.A., Knapp, A.K., Shaw, M.R., Loik, M.E., Smith, S.D., Tissue, D.T., Zak, J.C., Weltzin, J.F., Pockman, W.T., Sala, O.E., Haddad, B.M., Harte, J., Koch, G.W., Schwinning, S., Small, E.E. and Williams, D.G. (2004) Convergence across biomes to a common rain-use efficiency. *Nature* 429, 651–654.

Illius, A.W. and O'Connor, T.G. (1999) On the relevance of nonequilibrium concepts to arid and semiarid grazing systems. *Ecological Applications* 9, 798–813.

Illius, A.W. and O'Connor, T.G. (2000) Resource heterogeneity and ungulate population dynamics. *Oikos* 89, 283–294.

IPCC (2007) Summary for policymakers. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., Van Der Linden, P.J. and Hanson, C.E. (eds) *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK, pp. 7–22.

Keen, M. and Mahanty, S. (2006) Learning in sustainable natural resource management: challenges and opportunities in the Pacific. *Society & Natural Resources* 19(6), 497–513.

Kellert, S.R., Mehta, J.N., Ebbin, S.A. and Lichtenfeld, L.L. (2000) Community natural resource management: promise, rhetoric, and reality. *Society & Natural Resources* 13(8), 705–715.

Kimani, K. and Pickard, J. (1998) Recent trends and implications of group ranch sub-division and fragmentation in Kajiado district, Kenya. *Geographical Journal* 164(2), 202–213.

Klein, J.A., Harte, J. and Zhao, X.Q. (2004) Experimental warming causes large and rapid species loss, dampened by simulated grazing, on the Tibetan Plateau. *Ecology Letters* 7(12), 1170–1179.

Klein, J.A., Harte, J. and Zhao, X.Q. (2007) Experimental warming, not grazing, decreases rangeland quality on the Tibetan Plateau. *Ecological Applications* 17(2), 541–557.

Klein, J.A., Harte, J. and Zhao, X.Q. (2008) Decline in medicinal and forage species with warming is mediated by plant traits on the Tibetan Plateau. *Ecosystems* 11(5), 775–789.

Klein, J.A., Yeh, E.T., Bump, J.K., Nyima, Y. and Hopping, K.A. (2011) Coordinating environmental protection and climate change adaptation policy in resource-dependent communities: a case study from the Tibetan Plateau. In: Ford, J.D. and Ford, L.B. (eds) *Climate Change Adaptation in Developed Nations*. Springer, New York, in press.

Knapp, A.K., Burns, C.E., Fynn, R.W.S., Kirkman, K.P., Morris, C.D. and Smith, M.D. (2006) Convergence and contingency in production–precipitation relationships in North American and South African C-4 Grasslands. *Oecologia* 149, 456–464.

Lam, W.F. (1998) *Governing Irrigation Systems in Nepal: Institutions, Infrastructure, and Collective Action*. ICS Press, San Francisco, California.

Lane, C.R. (1998) *Custodians of the Commons: Pastoral Land Tenure in East and West Africa*. Earthscan Publications, London, UK.

Lee, R.G. and Field, D.R. (2005) *Communities and Forests: Where People Meet the Land*. Oregon State University Press, Corvallis, Oregon.

Lesorogol, C.K. (2008) Land privatization and pastoralist well-being in Kenya. *Development and Change* 39(2), 309–331.

Li, W.J., Ali, S.H. and Zhang, Q. (2007) Property rights and grassland degradation: a study of the Xilingol pasture, Inner Mongolia, China. *Journal of Environmental Management* 85, 461–470.

Little, P.D., McPeak, J., Barrett, C.B. and Kristjanson, P. (2008) Challenging orthodoxies: understanding poverty in pastoral areas of East Africa. *Development and Change* 39, 587–611.

Liu, J.G., Dietz, T., Carpenter, S.R., Alberti, M., Folke, C., Moran, E., Pell, A.N., Deadman, P., Kratz, T., Lubchenco, J., Ostrom, E., Ouyang, Z., Provencher, W., Redman, C.L., Schneider, S.H. and Taylor, W.W. (2007) Complexity of coupled human and natural systems. *Science* 317(5844), 1513–1516.

Luers, A.L. (2005) The surface of vulnerability: an analytical framework for examining environmental change. *Global Environmental Change – Human and Policy Dimensions* 15(3), 214–223.

Lynam, T.J.P. and Smith, M.S. (2004) Monitoring in a complex world – seeking slow variables, a scaled focus, and speedier learning. *African Journal of Range and Forage Science* 21(2), 69–78.

Marin, A. (2008) Between cash cows and golden calves: adaptations of Mongolian pastoralism in the 'Age of the Market'. *Nomadic Peoples* 12, 75–101.

Marin-Yaseli, M.L. and Martinez, T.L. (2003) Competing for meadows – a case study on tourism and livestock farming in the Spanish Pyrenees. *Mountain Research & Development* 23(2), 169–176.

McCabe, J.T. (1990) Turkana pastoralism: a case against the Tragedy of the Commons. *Human Ecology* 18, 81–103.

McCabe, J.T. (2003a) Sustainability and livelihood diversification among the Maasai of Northern Tanzania. *Human Organization* 62(3), 100–111.

McCabe, J.T. (2003b) Disequilibrial ecosystems and livelihood diversification among the Maasai of Northern Tanzania: implications for conservation policy in Eastern Africa. *Nomadic Peoples* 7(1), 74–91.

McCay, B. and Acheson, J. (1987) *The Question of the Commons: the Culture and Ecology of Communal Resources*. University of Arizona Press, Tucson, Arizona.

McPeak, J. and Little, P.D. (2005) Cursed if you do, cursed if you don't: The contradictory processes of sedentarization in northern Kenya. In: Fratkin, E. and Roth, E.A. (eds) *As Pastoralists Settle*. Kluwer Academic Publishers, Dordrecht, the Netherlands, pp. 87–104.

MEA (Millennium Ecosystem Assessment) (2005) Ecosystems and Human Well-being: *Desertification Synthesis*. World Resources Institute, Washington, DC.

Meffe, G.K., Nielson, L.A., Knight, R.L. and Schenborn, D.A. (2002) *Ecosystem Management: Adaptive, Community-Based Conservation*. Island Press, Washington, DC.

Meyer, N. (2006) Desertification and restoration of grasslands in Inner Mongolia. *Journal of Forestry* 104(6), 328–331.

Mwangi, E. (2007) The puzzle of group ranch subdivision in Kenya's Maasailand. *Development and Change* 38(5), 889–910.

Nelson, D.R., Adger, W.N. and Brown, K. (2007) Adaptation to environmental change: contributions of a resilience framework. *Annual Review of Environment and Resources* 32, 395–419.

Neumann, R.P. (1998) *Imposing Wilderness: Struggles over Livelihood and Nature Preservation in Africa*. University of California Press, Berkeley, California.

Neupert, R.F. (1999) Population, nomadic pastoralism and the environment in the Mongolian Plateau. *Population and Environment* 20(5), 413–441.

Niamir-Fuller, M. (1999) *Managing Mobility in African Rangelands: the Legitimization of Transhumance*. Intermediate Technology Publications, London, UK.

Nicholls, N. and Wong, K.K. (1990) Dependence of rainfall variability on mean rainfall, latitude, and the southern oscillation. *Journal of Climate* 3(1), 163–170.

Nicholson, S.E. (2002) What are the key components of climate as a driver of desertification? In: Reynolds, J.F. and Smith, S.M. (eds) *Global Desertification – Do Humans Cause Deserts?* Dahlem University Press, Kothen, Germany, pp. 41–55.

Niu, T., Chen, L.X. and Zhou, Z.J. (2004) The characteristics of climate change over the Tibetan Plateau in the last 40 years and the detection of climatic jumps. *Advances in Atmospheric Sciences* 21(2), 193–203.

Nixon, F. and Walters, B. (2006) Privatization, income distribution and poverty: the Mongolian experience. *World Development* 34, 1557–1579.

Noy-Meir, I. (1974) Desert ecosystems: higher trophic levels. *Annual Review of Ecology & Systematics* 5, 195–214.

Oechel, W.C., Vourlitis, G.L., Hastings, S.J., Ault, R.P. and Bryant, P. (1998) The effects of water table manipulation and elevated temperature on the net CO2 flux of wet sedge Tundra ecosystems. *Global Change Biology* 4(1), 77–90.

Ostrom, E. (1990) *Governing the Commons: the Evolution of Institutions for Collective Action*. Cambridge University Press, Cambridge, UK.

Ostrom, E. and Mwangi, E. (2008) Common pool resources and rangelands. In: *People and Policy in Rangeland Management – a Glossary of Key Concepts*. Science in China Press, Beijing, China, pp. 160–168.

Peluso, N.L. (1992) *Rich Forests, Poor People: Resource Control and Resistance in Java*. University of California Press, Berkeley, California.

Peters, D.P.C., Pielke Sr, R.A., Bestelmeyer, B.T., Allen, C.D., Munson-McGee, S. and Havstad, K.M. (2004) Cross-scale interactions, nonlinearities, and forecasting catastrophic events. *Proceedings of the National Academy of Sciences* 101(42), 15130–15135.

Pinkerton, E. (1989) *Cooperative Management of Local Fisheries: New Directions for Improved Management and Community Development*. University of British Columbia Press, Vancouver, Canada.

Plummer, R. and FitzGibbon, J. (2006) People matter: the importance of social capital in the co-management of natural resources. *Natural Resources Forum* 30(1), 51–62.

Pretty, J. and Ward, H. (2001) Social capital and the environment. *World Development* 29(2), 209–227.

Putnam, R. (2000) *Bowling Alone: the Collapse and Revival of American Community*. Simon and Schuster, New York.

Reid, R.S., Galvin, K.A. and Kruska, R.L. (2008) Global significance of extensive grazing lands and pastoral societies: an introduction. In: Galvin, K.A., Reid, R.S., Behnke, R.H. and Hobbs, N.T. (eds) *Fragmentation in Semi-Arid and Arid Landscapes: Consequences for Human and Natural Systems*. Springer, Dordrecht, the Netherlands, pp. 1–24.

Reynolds, J.F. and Smith, S.D.M. (2002) Do humans cause deserts? In: Reynolds, J.F. and Smith, S.D.M. (eds) *Global Desertification: Do Humans Cause Deserts?* Dahlem University Press, Berlin, Germany, pp.1–21.

Reynolds, J.F., Smith, S.D.M., Lambin, E.F., Turner, B.L., Mortimore, M., Batterbury, S.P.J., Downing, T.E., Dowlatabadi, H., Fernandez, R.J., Herrick, J.E., Huber-Sannwald, E., Jiang, H., Leemans, R., Lynam, T., Maestre, F.T., Ayarza, M. and Walker, B. (2007) Global desertification: building a science for dryland development. *Science* 316(5826), 847–851.

Rogers, S. and Wang, M. (2006) Environmental resettlement and social dis/re-articulation in Inner Mongolia, China. *Population and Environment* 28(1), 41–68.

Sala, O.E., Parton, W.J., Joyce, L.A. and Lauenroth, W.K. (1988) Primary production of the central grassland region of the United States. *Ecology* 69, 40–45.

Sandford, S. (1983) *Management of Pastoral Development in the Third World*. Wiley, Chichester, UK.

Schlesinger, W.H., Reynolds, J.F., Cunningham, G.L., Huenneke, L.F., Jarrell, W.M., Virginia, R.A. and Whitford, W.G. (1990) Biological feedbacks in global desertification. *Science* 247, 1043–1048.

Scoones, I. (1994) *Living with Uncertainty: New Directions in Pastoral Development in Africa*. Intermediate Technology Publications, London, UK.

Scott, J. (1998) *Seeing Like a State: How Certain Schemes to Improve the Human Condition Have Failed*. Yale University Press, New Haven, Connecticut.

Shaw, R.M., Zavaleta, E.S., Chiariello, N.R., Cleland, E.E., Mooney, H.A. and Field, C.B. (2002) Grassland responses to global environmental changes suppressed by elevated CO_2. *Science* 298, 1987–1990.

Sneath, D. (2003) Land use, the environment and development in post-socialist Mongolia. *Oxford Development Studies* 31(4), 441–459.

State Council (2002) Some Suggestions Regarding Strengthening Grassland Protection and Construction. *State Council Circular 19*.

Stringham, T.K., Krueger, W.C. and Shaver, P.L. (2003) State and transition modeling: an ecological process approach. *Journal of Range Management* 56(2), 106–113.

Stumpp, M., Wesche, K., Retzer, V. and Miehe, G. (2005) Impact of grazing livestock and distance from water sources on soil fertility in Southern Mongolia. *Mountain Research and Development* 25(3), 244–251.

Thompson, D.M., Serneels, S., Kaelo, D. and Trench, P.C. (2009) Maasai Mara – land privatization and wildlife decline: can conservation pay its way? In: Homewood, K., Kristjanson, P. and Trench, P. (eds) *Staying Maasai: Livelihoods, Conservation and Development in East African Rangelands*. Springer, London, UK, pp. 77–114.

Todd, S.W. and Hoffman, M.T. (1999) A fence-line contrast reveals effects of heavy grazing on plant diversity and community composition in Namaqualand, South Africa. *Plant Ecology* 142(1), 2169–2178.

Tong, C., Wu, J., Yong, S., Yang, J. and Yong, W. (2004) A landscape-scale assessment of steppe degradation in the Xilin River Basin, Inner Mongolia, China. *Journal of Arid Environments* 59(1), 133–149.

Turner, B.L., Kasperson, R.E., Matson, P.A., McCarthy, J.J., Corell, R.W., Christensen, L., Eckley, N., Kasperson, J.X., Luers, A., Martello, M.L., Polsky, C., Pulsipher, A. and Schiller, A. (2003) A framework for vulnerability analysis in sustainability science. *Proceedings of the National Academy of Sciences of the United States of America* 100(14), 8074–8079.

Turner, B.L., Lambin, E.F. and Reenberg, A. (2007) The emergence of land change science for global environmental change and sustainability. *Proceedings of the National Academy of Sciences of the United States of America* 104(52), 20666–20671.

Vetter, S. (2005) Rangelands at equilibrium and non-equilibrium: recent developments in the debate. *Journal of Arid Environments* 62(2), 321–341.

Wade, R. (1994) *Village Republics: Economic Conditions for Collective Action in South India*. ICS Press, San Francisco, California.

Wagner, C.L. and Fernández-Giménez, M. (2008) Does collaboration increase social capital? *Society & Natural Resources* 21, 324–344.

Walker, B. and Salt, D. (2006) *Resilience Thinking: Sustaining Ecosystems and People in a Changing World*. Island Press, Washington, DC.

Western, D. and Wright, R.M. (1994) *Natural Connections*. Island Press, Washington, DC.

Westoby, M., Walker, B. and Noy-Meir, I. (1989) Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* 42, 266–274.

Wilkinson, K.P. (1991) *The Community in Rural American*. Greenwood Press and the Rural Sociological Society, Westport, Connecticut.

Williams, D.M. (1996) Grassland enclosures: catalyst of land degradation in Inner Mongolia. *Human Organization* 55(3), 307–313.

Yeh, E.T. (2005) Green governmentality and pastoralism in Western China: converting pastures to grasslands. *Nomadic Peoples* 9(1), 9–29.

Yeh, E.T. (2009) Greening Western China: a critical view. *Geoforum* 40(5), 884–894.

Zhang, Y.S., Li, T. and Wang, B. (2004) Decadal change of the spring snow depth over the Tibetan Plateau: the associated circulation and influence on the East Asian Summer Monsoon. *Journal of Climate* 17(14), 2780–2793.