Chapter 12
Managing the Livestock–Wildlife Interface on Rangelands

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Abstract On rangelands the livestock–wildlife interface is mostly characterized by management actions aimed at controlling problems associated with competition, disease, and depredation. Wildlife communities (especially the large vertebrate species) are typically incompatible with agricultural development because the opportunity costs of wildlife conservation are unaffordable except in arid and semiarid regions. Ecological factors including the provision of supplementary food and water for livestock, together with the persecution of large predators, result in livestock replacing wildlife at biomass densities far exceeding those of indigenous ungulates. Diseases are difficult to eradicate from free-ranging wildlife populations and so veterinary controls usually focus on separating commercial livestock herds from wildlife. Persecution of large carnivores due to their depredation of livestock has caused the virtual eradication of apex predators from most rangelands. However, recent research points to a broad range of solutions to reduce conflict at the livestock–wildlife interface. Conserving wildlife bolsters the adaptive capacity of a rangeland by providing stakeholders with options for dealing with environmental change. This is contingent upon local communities being empowered to benefit directly from their wildlife resources within a management framework that integrates land-use sectors at the landscape scale. As rangelands undergo irreversible changes caused by species invasions and climate forcings, the future perspective favors a proactive shift in attitude towards the livestock–wildlife interface, from problem control to asset management.

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12.1   Introduction

We write this chapter in a time of increasing recognition of the value of rangelands as providers of ecosystem services, broadening the traditionally focused view of rangelands as areas for the production of commodities from free-ranging livestock (Havstad et al. 2007). It is also a time in which ecologists are calling for conventional production-maximizing management approaches to be transformed into a "resilience framework" for the stewardship of social–ecological systems (Chapin et al. 2009). We are living in the Anthropocene epoch, and have been for quite a long time already (Balter 2013), during which we have unwittingly changed just about everything on our planet, including the climate. In this time of self-awareness of our environmental responsibility, rangelands provide a stage on which new approaches to natural resource management can be developed and implemented. On a global scale, rangelands are particularly important for wildlife conservation and there is urgency in the search for effective ways of reconciling conservation with livestock production (du Toit et al. 2010). This is a vast topic with many facets that we cannot comprehensively cover and so we focus on what we consider to be the key issues, with our treatment of wildlife applying mainly to large (>5 kg) free-living mammals. We have structured this chapter to first review some of the main conceptual advances over the past 25 years or so, as we see them, at the livestock–wildlife interface on rangelands. We then suggest the social implications of translating those concepts—when and where possible—into applications that could contribute to rangeland resilience.

12.2   Conceptual Advances at the Livestock–Wildlife Interface

Reconciling wildlife conservation with livestock production on rangelands requires a departure from the conventional "either/or" model in which conservation and agriculture are represented by separate, competing sectors of society and governmental administration. Integrated approaches are required at the landscape scale (Sayer et al. 2013) with local communities empowered to benefit from wildlife and livestock together, and with management agencies geared for enhancing the resilience of entire social–ecological systems (Biggs et al. 2012). Resilient systems can maintain their function, structure, identity, and feedbacks by absorbing disturbances and reorganizing within continually changing environments (Walker et al. 2004; Chap. 6, this volume). But achieving an integrated

approach requires an understanding of the reasons why the sectoral management approach persists across most rangelands, together with an understanding of how it can be mitigated.

Fundamentally, the practice of livestock production stems from the domestication of wild species, which has been accomplished through the removal of "wildness" over centuries. That process has built a culture of livestock husbandry that sequesters and protects livestock from the ever-present forces of wildness, which is obviously necessary in times and places where wildness is overwhelmingly intractable. Now, however, the wildness-to-tameness ratio has long since flipped on most of the world's rangelands and conceptual advances point towards a new and very different model for rangeland management, or stewardship (Chapin et al. 2009; Walker 2010). Here we review those advances and consider how science and management might interact to better understand and work with, rather than against, the main features that define the livestock–wildlife interface on rangelands.

12.2.1 Competition

Livestock–wildlife competition operates through two sets of processes within the social–ecological systems we call rangelands: economic processes influence agricultural and wildlife-based enterprises as sources of income for producer communities; ecological processes influence the relative efficiencies of livestock and wildlife species in utilizing the food and water resources occurring in their shared range.

At the global scale, economic processes generally result in agricultural returns outcompeting wildlife returns and the patchwork of land use within rangelands intensifying towards croplands and fragmented rangelands (Hobbs et al. 2008). As markets, technology, and infrastructure develop, the position of a rangeland on its production possibility frontier (PPF) changes (Bastian et al. 1991; Smith et al. 2012) with agricultural production becoming specialized, driving down the possibilities for wildlife production (Fig. 12.1). The transition begins with a fully intact wildlife community (Point A) as still occurs in wildlife reserves, game ranches, and areas where diseases (e.g., trypanosomiasis in Africa) exclude livestock. Eventually livestock production is so specialized (Point C; irrigated and fertilized pastures, winter supplementation, fenced paddocks, etc.) that wildlife production is impossible. In some cases, well-regulated hunting for trophies and meat can add to the rangeland's production potential from livestock (Point B), and infrastructure (waterpoints, access roads, etc.) provided for livestock production can also be beneficial to the sustainable utilization of wildlife. However, in most rangelands the transition has proceeded directly to maximizing livestock production (Point C) and restoring the wildlife community would necessitate a disproportionate pull-back in agricultural production, with unaffordable opportunity costs. Avoiding or overcoming those costs—incurred by foregoing land-use opportunities that are incompatible with wildlife—requires innovative policies to enable competitive and sustainable returns from wildlife to local communities and private landowners (Norton-Griffiths and
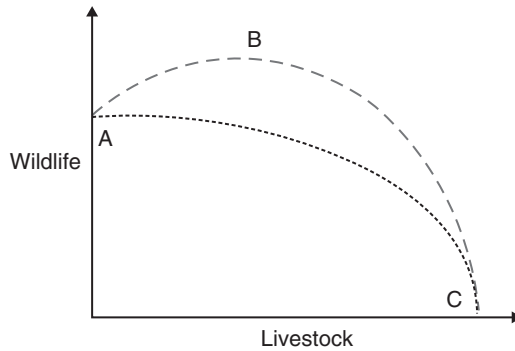
Fig. 12.1 Production possibility frontiers (PPFs) are represented by hypothetical curves describing the maximum possible output of goods from wildlife relative to each possible level of output of livestock from a shared rangeland. The curve between points A and C is typical, with intensifying livestock production forcing a decline in wildlife resources due to persecution and competition for habitats and food. Raising the PPF to include point B is an option where the back-and-forth transmission of diseases between livestock and wildlife is not a major concern
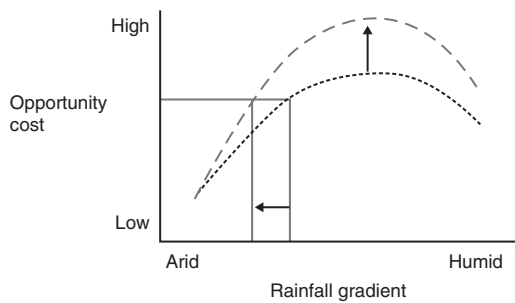

Fig. 12.2 The opportunity cost of conserving wildlife on a rangeland increases across the rainfall gradient, with mesic areas being valuable for agriculture and therefore unavailable as habitat for wildlife. In very humid areas the agricultural potential declines but various land-use options (e.g., logging followed by cattle production) can still be more lucrative than conserving wildlife for ecotourism. Developments in agricultural technology and changing market forces can raise the relative value of non-wildlife options and thus inflate the opportunity cost of conserving wildlife (from *dotted* to *dashed line*, *vertical arrow*). Then, without a commensurate increase in the value of wildlife, the areas in which any particular opportunity cost can be met by wildlife are reduced towards the arid end of the gradient (*horizontal arrow*)

Said 2010). Wildlife resources on rangelands are especially vulnerable to increasing opportunity costs driven by technological advances in exploration for, and exploitation of, groundwater, natural gas, and oil (e.g., Altchenko and Villholth 2013; Copeland et al. 2013; Northrup and Wittemyer 2013). Increasingly, arid rangelands are being converted to supply agricultural commodity markets (Fig. 12.2) at the expense of wildlife habitat. In addition, even the most remote rangelands are influenced by globalization, which can drive price anomalies in certain livestock products. Examples include cashmere wool from goats in Central Asia (Berger et al.

2013) and cattle dung used as fertilizer for organically grown coffee in India (Madhusudan 2005). In such cases the producer communities overstock, resulting in indigenous herbivores being outcompeted and predators persecuted with added intensity because the opportunity costs of tolerating them are hyper-inflated.

Despite the global trend towards degradation and fragmentation of wildlife habitats on rangelands, wildlife and mixed wildlife–livestock enterprises can be successful in semi-arid rangelands where opportunity costs are low (du Toit 2010). Commercial wildlife management ("game ranching") now predominates over cattle ranching on private land in the dry "bushveld" regions of South Africa where climatic, political, and economic changes have increased the relative profitability of wildlife enterprises (Carruthers 2008). Elsewhere in the world, some socioeconomic transitions have caused sharp declines in agricultural production resulting in land abandonment, most notably in post-Soviet Eastern Europe (Prishchepov et al. 2012) and also in Oceania and parts of North America (Queiroz et al. 2014). Where land abandonment occurs, low opportunity costs to wildlife conservation allow the resilience of social–ecological systems to be rebuilt with projects that restore biodiversity and ecological processes.

In the Great Plains of North America, for example, shrinking rural economies in the 1980s prompted a proposal that the USA's federal government should buy back failed farmlands and create a vast "buffalo commons" (Popper and Popper 1987). Since then, despite political and cultural resistance to that proposal from local communities, and without government buy-backs, the agrarian downtrend has opened new opportunities for wildlife restoration. Coordinated initiatives involving the nonprofit sector, private landowners, Native American tribes, and federal and state agencies are currently using American plains bison (*Bison bison bison*) as the flagship species for restoring ecologically significant expanses of prairie (Freese et al. 2007). Similar opportunities exist in Eastern Europe, where abundant habitat is now available for European bison (*Bison bonasus*) in depopulated areas of former farmland and rangeland in which livestock numbers have dropped by 60–75 % since the early 1990s (Kuemmerle et al. 2011).

The outlook is mixed, however, because in the western USA there are some areas that were once rangeland but are being—or already have been—transformed into exurban and suburban housing developments. This has been facilitated by the rise of the Internet, which enables executives to live in comparatively remote areas where they can work from home while enjoying environmental recreation opportunities. The situation is exacerbated in areas with access to both public land and a regional airport (Rasker et al. 2013; Fig. 12.3). Thus, whereas livestock ranching can be somewhat detrimental to wildlife it does still conserve habitats that are lost when converted into housing developments (Hansen et al. 2005).

As for ecological processes, the widespread competitive success of livestock *vs.* wildlife depends on the facilitation of favorable conditions for livestock by humans. That is how a small number of livestock species—usually fewer than five in any one area—has come to overwhelmingly dominate the herbivore biomass of rangelands that were formerly the natural habitats of diverse assemblages of coevolved wild species (du Toit and Cumming 1999). The global trend towards sedentary pastoralism on

Fig. 12.3 Examples of urban (*above*) and exurban (*below*) encroachment on rangelands in the western United States. Aerial photographs show the boundary between the town of Jackson, Wyoming, and the USFWS National Elk Refuge (*above*), and a recent development outside of Pinedale, Wyoming (*below*). Photo credits: USGS

rangelands (Hobbs et al. 2008; Western et al. 2009; Groom and Western 2013) is linked as both cause and effect with habitat fragmentation, supplementary fodder production, predator eradication, and water provision. In combination those factors typically maintain artificially high livestock densities and chronic overutilization of the remaining rangeland, outcompeting indigenous herbivores (e.g., Mishra et al. 2004; Ogutu et al. 2010). Wild ungulates can only coexist with cattle in the few subsistence pastoral systems in which watering points are widely distributed and wildlife species are not intensively persecuted (Georgiadis et al. 2007). Wild ungulates generally do not share watering points with livestock (Sitters et al. 2009) probably because of the activities of herders (and their dogs) around watering points. Overall, therefore, the combined effect of all aspects of livestock production across most rangelands worldwide is that indigenous large herbivores and their predators get ratcheted back to isolated habitat refugia and protected reserves. That process is, however, being at least partially reversed (large predators excluded) in some areas such as in the western USA, where valued game populations are increasingly being allowed to recover on rangelands under mixed livestock–wildlife management.

There is growing recognition that some plant–herbivore interactions specific to non-livestock large herbivores are potentially valuable and worth conserving, restoring, or introducing to rangelands. For example, where cattle graze

together with wild ungulates in East Africa during the wet season they achieve higher weight gains than in foraging areas from which wild ungulates are excluded (Odadi et al. 2011). That is probably because zebras (*Equus quagga*), being comparatively large hind-gut fermenters, reduce the stem:leaf ratio in the sward and thereby facilitate conditions for grazing ruminants such as cattle. Even though competition occurs in the dry season, weight gains from the facilitative effect in the wet season are sufficient that a net benefit might be achievable with management. In theory, managers could impose seasonal shifts in both the foraging areas used by livestock (currently common) and the mixes of livestock and wildlife species foraging together in each season and area (currently uncommon). In practice, however, there are few areas of rangeland in which livestock and wildlife can be actively managed together in a commercial operation at the landscape scale. Yet there are mounting scientific arguments in favor of selectively exploiting the diverse functional properties of wildlife species (du Toit 2011), despite the barriers of command-and-control laws and societal inertia.

The megafaunal extinctions that occurred on all continents other than Africa in the late Quaternary Period are attributed to a suite of factors that include overhunting by humans (Brook and Bowman 2004). Now, introducing morphologically and ecologically similar surrogates—mostly from Africa—has been proposed to restore the ecosystem processes once driven by those now-extinct megaherbivores in North America (Donlan et al. 2005) and Australia (Bowman 2012). Those "rewilding" proposals are contentious because benchmark conditions of archaic ecosystems are unknown, the global climate is continually changing, and the practicalities are prohibitive (Rubenstein et al. 2006). Also, sociopolitical acceptance of the rewilding argument requires a leap of faith in the net benefits of quasi-Pleistocene assemblages replete with large predators and megaherbivores (Soulé and Noss 1998). Nevertheless, "hybrid" or "novel" ecosystems (Hobbs et al. 2009) are emerging all the time mainly as a result of unintended species invasions and climate change, as well as the intended engineering activities of humans. For example, in the Netherlands 6000 ha of grazing land was created from seabed less than 50 years ago and is now a popular wildlife refuge called Oostvaardersplassen (Marris 2009). The bold approach adopted there was to stock the area with relict breeds of cattle and horses, together with indigenous cervids, lagomorphs, and waterfowl that moved in, and allow competitive and facilitative interactions to play out through time. With minimal intervention from managers, Oostvaardersplassen now supports one of Western Europe's richest terrestrial faunas occupying a patchwork of grassland, woodland, and wetland habitats. Those plant communities were established naturally, mainly from seeds brought in by birds, and the landscape's heterogeneity is maintained by interactions among top-down and bottom-up processes operating through herbivory, the climate, and soil.

The ongoing global changes that define the Anthropocene epoch (Zalasiewicz et al. 2010) mean that the reference state of an ecosystem cannot be an historic condition, but is the most desired of the potential alternative states at some

future stage along that ecosystem's trajectory of change. Some rangelands (e.g., in the Great Basin of the western USA) are now so radically altered by invasive plants and the fire regimes they support, on top of past overgrazing, changing land use, and ongoing climatic changes, that managers cannot restore them to their former states (Davies et al. 2012). They are either drifting through hybrid states towards thresholds of irreversible change or are reorganizing as novel ecosystems and so historic benchmark conditions are irrelevant and misleading. Within these novel ecosystems the basis for classifying species as "native," "exotic," or "feral" has become unclear, apart from legal mandates to conserve native threatened and endangered species.

Conserving ecological resilience involves conserving the full suite of functional types within a species assemblage, which entails developing a functional typology (cataloging the key functional types and quantifying their equilibrium biomass densities) for each target ecosystem. This is an emerging challenge for wildlife ecologists, many (perhaps most) of whom are reluctant to shift from their professionally trained focus on the taxonomic typology of the pristine state of the ecosystem at some benchmark stage in history. Nevertheless, while conservation biologists debate whether it is right or wrong (e.g., Doak et al. 2014; Marvier and Kareiva 2014), biodiversity is inevitably managed more for what it does than for what it is. Building resilience in transformed rangelands would involve finding the "right" combination of large herbivores to represent the particular mix of functional types (e.g., large-, medium-, and small-bodied grazers and browsers) needed to reach a feasibly desired state of the rangeland in question. This would enhance heterogeneity at the plant–herbivore interface, facilitate processes such as nutrient cycling and seed dispersal, and diversify the portfolio of options for stakeholders in the social–ecological system. If the need for a functional type could not be filled by species native to the geographical area, or by local livestock breeds, then it would be pragmatic to trial selected exotic species within an adaptive management framework. A key management consideration would, however, be the feasibility of controlling introduced species where they are desired and preventing them from invading where they are not, which could be difficult or impossible.

12.2.2   Disease

The impacts of pathogens and parasites were historically ignored by wildlife ecologists who mostly considered disease as a compensatory form of mortality. That was probably because, with the exception of acute disease outbreaks, infectious diseases were not easily observed and those disease outbreaks that did result in large-scale die-offs were perceived as random one-time events. Only by knowing the disease status of individuals, and following them through time, do some of the underlying processes become apparent. Due to these observational challenges the importance of disease to the functioning of ecosystems was undervalued.

12.2.2.1   Development of Disease Ecology as a Discipline

One of the key developments in disease ecology was when May and Anderson (1979) explored how disease dynamics might change when the host population is considered to be dynamic rather than assumed to be constant. Then, depending on the specifics of how the parasite is transmitted, disease can emerge as a strong factor regulating the host population. This development probably sounds obvious to many ecologists, but at that time much of disease ecology was borrowed from studies on humans, in less variable populations. Many principles of wildlife disease ecology continue to be derived from human systems, where the datasets tend to be richer (Grenfell et al. 2002). However, there are several important ways in which wildlife populations are likely to differ from human or livestock populations: wildlife populations fluctuate more; reproduction is more closely tied to resources and population density; movements can be more localized; and predation can interact with disease. These differences, independently or in combination, can have important management implications. For example, "critical community size" is the population size required for a disease to persist. The concept is implicit in the rationale of those managers who are inclined towards reducing a wildlife population's size as a disease control strategy. It originated from analyses of measles, which dies out in cities smaller than about 200,000–300,000 people (Bartlett 1957; Grenfell et al. 2002). With humans the number of new susceptible individuals recruited into a population tends to be highly correlated with population size. This is also to be expected with livestock, but wildlife populations near carrying capacity have much-reduced recruitment and so the number of new susceptibles required for the disease to persist might not be correlated with total population size (Lloyd-Smith et al. 2005).

Even when episodic, diseases can have long-term effects on ecosystems (Dobson and Hudson 1986). For example, rinderpest, a morbillivirus of artiodactyls (Plowright 1962), was introduced to Africa in the late nineteenth century and caused massive die-offs across sub-Saharan Africa before being eradicated in the early part of the twenty-first century. This outbreak and subsequent die-off resulted in a large-scale release of herbivory on trees and shrubs by ungulates, triggering long-term disturbances across the African savanna biome, such as in northern Botswana (Vandewalle and Alexander 2014). Similar ecosystem-level effects, this time acting through release of predation on insects, might occur as a result of the ongoing epidemic of a fungal pathogen (*Geomyces destructans*) causing white-nose syndrome in bats in eastern North America (Frick et al. 2010).

Developments in disease ecology have followed a similar progression as those in general ecology. First there was a focus on population dynamics of the host and/or pathogen and issues of density dependence for each. This was followed by work on spatial structure and metapopulations (Hess 1994, 1996a, b). More recently, disease ecology has branched out to multi-host/multi-pathogen interactions (Jolles et al. 2008; Viana et al. 2014), community-level interaction networks (Lafferty et al. 2006), and the effects of biodiversity on disease dynamics (Johnson and Thieltges 2010; Johnson et al. 2013).

Research on the effects of biodiversity on disease dynamics has been motivated, in part, by the example of Lyme disease (Ostfeld and Keesing 2000). The causative bacterium *Borrelia burgdorferi* is hosted by white-footed mice (*Peromyscus leucopus*) that reach higher densities in less diverse ecosystems where mammalian mesopredators and avian raptors are rare or absent. An ensuing debate has concentrated on whether this represents a general relationship between higher biodiversity and lower disease or if this is idiosyncratic and system-specific (Lafferty 2012). Yet for parasites with complex lifestyles—such as trematodes, cestodes, nematodes, acanthocephalans, chytridiomycetes, oomycetes, and myxosporeans that move between multiple host species to complete their life cycles—there are multiple different mechanisms by which "dilution" or "decoy" effects could moderate disease prevalence (Johnson and Thieltges 2010). The debate over the impact of biodiversity on disease is similar to the debate over the mechanisms by which biodiversity may affect ecosystem processes (Loreau et al. 2001; Hooper et al. 2005).

Traditionally parasites and pathogens were, and mostly still are, viewed as things to be controlled or eradicated where possible. More recently, however, researchers have been investigating the effects of parasites on ecosystem function and whether a stable and resilient ecosystem has a rich parasite assemblage (Hudson et al. 2006; Vannier-Santos and Lenzi 2011; Hatcher et al. 2012). Similarly, there is an explosion of research on the microbiome, or the microbial community within an individual, and how that community composition affects nutrition and obesity, immune function, disease risk, cancer, and so forth (Kau et al. 2011; Vannier-Santos and Lenzi 2011). Healthy wildlife populations are not necessarily devoid of pathogens (Stephen 2014) but have a mix they coevolved with and which protect them from invasions of novel types.

12.2.2.2   Management of Wildlife Diseases

In rangelands, various zoonoses (rabies, bovine tuberculosis, brucellosis, etc.) can be hosted by wildlife species but are most commonly transmitted to humans through their domesticated animals. For example, brucellosis, which affects most pastoral societies, is one of the most common zoonotic infections worldwide with more than 500,000 new cases annually (Pappas et al. 2006). Also, because human populations in rangelands depend to greater or lesser degrees on domesticated animals for their livelihoods, nonzoonotic diseases with wildlife reservoirs, such as foot-and-mouth disease, are also of concern. Political and social controversies at the interface between domesticated and wild animals in rangelands are thus likely to include disease issues of some type (Kock et al. 2010).

Such controversies are heightened by the poor success rate of campaigns to eradicate diseases in wildlife reservoirs, with the notable exception of fox rabies in Europe where a safe and effective vaccine bait was available (Brochier et al. 1991). In systems where diseases co-circulate in livestock and wildlife, control efforts can be successful when they are targeted in livestock, such as with rinderpest for example (Mariner et al. 2012; Roeder et al. 2013). Similarly, in Spain,

brucellosis in red deer (*Cervus elaphus*) declined as a result of control efforts in livestock (Serrano et al. 2011). In these cases the wildlife populations were not competent disease reservoirs, failing to sustain the infection without co-circulation through livestock. In cases where wildlife populations are competent reservoir hosts, control efforts are complicated by social, logistical, and ecological factors (e.g., Donnelly et al. 2006).

Culling and increased hunting of wildlife are often proposed to control wildlife diseases but they are seldom effective. First, it might be difficult to achieve the necessary hunter participation to create large reductions in host density (e.g., Heberlein 2004). Many hunters will want to maintain high densities to maximize their future hunting opportunities. Second, the cost of culling wildlife, particularly in a test and cull strategy, can be very high and so is only applicable to localized operations (Wolfe et al. 2004). One of the rationales for culling wildlife is that disease transmission might be positively correlated with host density (Lloyd-Smith et al. 2005). Although plausible, this assumption is not always supported by data and even if transmission is density-dependent, culling can have adverse consequences. For example, culling badgers (*Meles meles*) to control bovine tuberculosis in Britain causes the disease to spread even further due to social disruption, with surviving badgers roaming more widely to find mates (Bielby et al. 2014). In addition, increased hunting in regions of easy access might locally aggregate wildlife in areas of limited hunter access even if the regional density declines.

Part of the problem, at least for diseases that can be transmitted back and forth between livestock and wildlife, is that disease control measures used on wildlife are based on those developed for livestock. Therein lies a mismatch in spatial scale, because livestock can be intensively managed within fences at the ranch scale but wildlife populations are free-ranging at the landscape scale (Bienen and Tabor 2006). There is also a mismatch in temporal scale because disease eradication campaigns are typically continuous for livestock but in wildlife they can be tactically scheduled to take advantage of episodic natural disturbances. For example, in the Kruger ecosystem of South Africa, buffalo (*Syncerus caffer*) in herds with a higher prevalence of bovine tuberculosis lose condition faster during dry seasons (Caron et al. 2003). It follows that any attempts to minimize the disease in the buffalo population should be reserved for immediately after droughts, which occur about once per decade. That is when die-offs of 50 % or more leave a smaller population size to contend with, and presumably a much reduced proportion of infected animals (du Toit 2010).

For rangelands in particular, progress in reducing—or at least accommodating—the risks of wildlife-borne diseases to livestock will depend upon the better integration of veterinary practice and epidemiology with disease ecology. As a case in point, before brucellosis was virtually eradicated in cattle in the USA it was transmitted to elk (*Cervus canadensis*) and bison in the Greater Yellowstone Area, where it has repeatedly been transmitted back to cattle from elk (Rhyan et al. 2013). Free-ranging elk are now increasing in the area in which brucellosis is endemic and so the veterinary achievements in fighting the disease in livestock over the past 75 years must now be followed by ecologically based adaptive

management approaches to control the spread of the disease in elk (Rhyan et al. 2013). Devising such approaches for any wildlife host of a livestock disease will depend upon studies at the landscape scale and population level to understand such issues as: seasonal movements between feeding areas; joining-and-leaving behavior of contagious individuals moving between social groups; use of habitats at key times for disease transmission; effects of climate on the dynamics of populations and the aggregation and dispersal of groups and individuals; and responses to fear induced by hunting and predation. In addition, there are calls for trade policy reforms to relax stringent veterinary regulations such as those responsible for veterinary cordon fences in southern Africa (Fynn and Bonyongo 2011; McGahey 2011; Thomson et al. 2013). There, the fences have decimated migratory wildlife populations that, in the longer term, could be more valuable to local communities than export-quality beef. At a minimum, certain wildlife species should be destigmatized as disease vectors, such as bison in North America and buffalo in South Africa, because disease-free herds can be established and used to restock areas where they can safely comingle with cattle.

12.2.3   Predation

Considering the obvious risks that large predators impose on humans and the animals they depend upon for their welfare, it is not surprising that large predators are now threatened worldwide (Ripple et al. 2014). The geographic expansion of human activities results in large predator populations becoming increasingly fragmented (Woodroffe 2000), for example the Iberian lynx *Lynx pardinus* (Rodriguez and Delibes 1992) and the African lion *Panthera leo* (Riggio et al. 2013). Most species have become extirpated from parts of their range with, for example, the African lion having lost ~85 % of its range in the last 500 years (Morrison et al. 2007). This global erosion of large predator guilds raises concerns about the loss of natural predation as an ecosystem process. In a visionary article, Hairston et al. (1960) suggested "the world is green" because herbivore populations are limited by their predators and so major ecosystem-level effects should result from the dwindling abundance and distribution of apex predators. Fifty years later, evidence is mounting that apex predators represent a functionally important component of stable ecosystems (Sergio et al. 2008; Ritchie and Johnson 2009; Estes et al. 2011; Ripple et al. 2014). Their effects flow through trophic cascades, in which lethal and nonlethal interactions between predators and prey drive sequential responses down the food chain. Artificially replicating predation is problematic because sport hunting or culling is selective and episodic, being a pulse disturbance instead of the press disturbance imposed by an intact, coevolved, predator guild. As is characteristic of a press disturbance, the ecosystem-level effects of predators are most apparent when they are either removed or reintroduced. For instance, in some regions of the world the eradication of large predators has resulted in problematically high densities of native and feral herbivores, with associated impacts on biodiversity (Terborgh et al.

2001). Examples include feral horses *Equus caballus* in the western USA (Garrott 1991), wild boar *Sus scrofa* in western Europe (Sáez-Royuela and Tellería 1986), and white-tailed deer *Odocoileus virginianus* in eastern North America (Côté et al. 2004), for which the ecological effects of their population irruptions are similar to those following the introduction of large herbivores to predator-free islands (Allombert et al. 2005a, b; Martin et al. 2010). Conversely, some evidence suggests that the reintroduction of wolves *Canis lupus* to Yellowstone National Park in 1995 triggered a trophic cascade that is ultimately contributing to changes in plant communities as different as riparian thickets and montane grasslands (White et al. 2013). The interactions are, however, complex and cannot be predicted from trophic dynamics alone (Marshall et al. 2014).

Prey populations are directly influenced by the consumptive effects of predation but there are also indirect, nonlethal influences arising from fear-driven behavioral responses of prey to their risk of predation. The "ecology of fear" (Lima 1998; Brown et al. 1999; Laundré 2010) is now an accepted subdiscipline focusing on how prey populations respond to predation risk across a suite of response variables including spatial movement and habitat use (Creel et al. 2005; Valeix et al. 2009a; Courbin et al. 2015), temporal niche (Valeix et al. 2009b), vigilance level (Laundré et al. 2001), group size (Creel and Winnie 2005), and so forth. Spatial heterogeneity in predation risk and corresponding behavioral adjustments of prey give rise to "a landscape of fear" (Laundré et al. 2001, 2014), which might then influence ecosystem structure and function (Ripple et al. 2001; Ripple and Beschta 2006; Kuijper et al. 2013). Predator-induced behavioral adjustments by prey might involve energetic costs and physiological responses (Creel et al. 2007; Barnier et al. 2014) that can ultimately affect prey demography (Creel and Christianson 2008; Christianson and Creel 2010; Zanette et al. 2011). Indeed, predators could have a greater effect on prey demography through fear than through direct consumption of individual prey (Preisser et al. 2005). For livestock on rangelands the indirect effects of predators are likely to include decreased conception rates and weaning weights (Howery and DeLiberto 2004; Steele et al. 2013). On the other hand, apex predators dominate intraguild relations (Palomares and Caro 1999; Caro and Stoner 2003) such that a collapse in an apex predator population typically results in the phenomenon of mesopredator release (Prugh et al. 2009; Ritchie and Johnson 2009), which can have negative implications for biodiversity conservation (Johnson et al. 2006).

Overall, by promoting biodiversity and trophic web integrity, apex predators contribute to the resilience of ecosystems challenged by biological invasions (Wallach et al. 2010), disease outbreaks (Pongsiri et al. 2009), and climate change (Wilmers et al. 2006). Conservation of apex predators is now considered a worldwide priority and, as a result, large carnivores have been reintroduced to several ecosystems (Hayward and Somers 2009). Because large carnivores roam in the matrix outside protected areas (Woodroffe and Ginsberg 1998; Elliot et al. 2014a), people in rangelands have an especially important role to play in their conservation. The livestock production that defines these social–ecological systems is inevitably associated with an entrenched antipathy towards large predators across all continents. Evidence nevertheless indicates, at least in the USA, that feasible management options exist for

Fig. 12.4 Apex predators
can influence rangeland
ecosystems in ways that
are far more complex than
their best-known role as
problem animals
responsible for livestock
depredation. Interactions
can be direct (*solid arrows*)
and indirect (*dashed*)
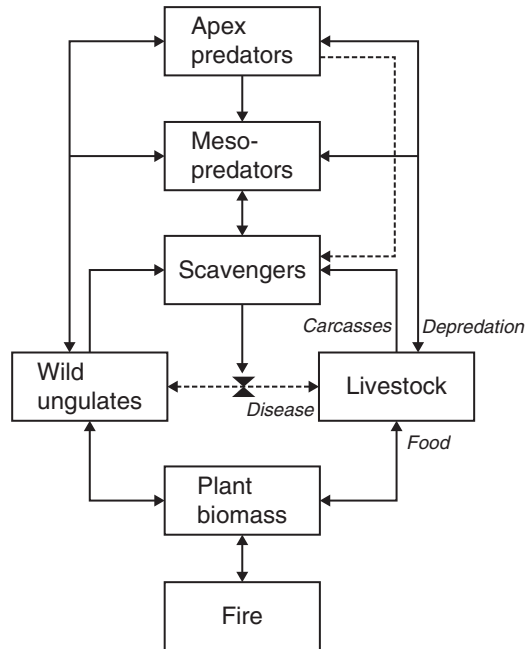
conserving large predators on rangelands without compromising the economic viability of the ranching lifestyle (Shivik 2014). Solutions require an understanding of how predators directly and indirectly influence livestock production either positively or negatively (Fig. 12.4). First, overabundant wild ungulate populations are associated with an increased risk of disease transmission to domesticated animals and an increased risk of zoonosis. Predators regulate prey populations and also comb out the sick and weak individuals, thereby contributing to animal health in rangelands. Additionally, they provide carrion for scavengers such as vultures (Wilmers et al. 2003), which have a controlling influence on the spread of diseases (Sekercioglu 2006). Some apex predators are facultative scavengers and so they interact indirectly with obligate scavengers by competing not only at their own kills, but also at carcasses they have stolen or found. Second, the dynamics of herbivory and fire are tightly coupled (Gill et al. 2009; Holdo et al. 2009) and fire is fundamental to rangeland ecology (Bond et al., this volume), so predators and diseases can influence a rangeland's structure and function indirectly via the fire regime. Third, abundant wild ungulate populations can create political tensions and socioeconomic costs on rangelands, as demonstrated by elk and feral horses in the western USA, which could be reduced by apex predators (Beschta et al. 2013). Finally, numerous studies have highlighted the perverse consequences of apex predator persecution, which include mesopredator release. Sheep ranchers in the Rocky Mountains of the USA, for example, might come to appreciate the drop in coyote density caused by the return of wolves (Berger and Gese 2007), given that depredation by coyotes is commonly perceived as an important factor in the declining sheep industry (Berger 2006).

Creativity is needed in employing effective tools and management practices to mitigate the negative effects of predators because the problem is globally urgent. Despite drastically declining populations in most large carnivore species, human–carnivore conflicts are steadily increasing (Treves and Karanth 2003; Woodroffe et al. 2005) with the most common reason being depredation on livestock (Sillero-Zubiri and Laurenson 2001; Thirgood et al. 2005). Examples of "problem" predators include lions, leopards *Panthera pardus*, spotted hyaenas *Crocuta crocuta*, cheetahs *Acinonyx jubatus* and wild dogs *Lycaon pictus* in Africa (e.g., Ogada et al. 2003), wolves and lynx in Europe (e.g., Sunde et al. 1998), wolves, coyotes *Canis latrans* and grizzly bears *Ursus arctos* in North America (e.g., Knowlton et al. 1999), pumas *Puma concolor* and jaguars *Panthera onca* in South America (e.g., Palmeira et al. 2008), and tigers *Panthera tigris* and snow leopards *Panthera uncia* in Asia (e.g., Bagchi and Mishra 2006).

Lethal control of predator populations has been the common rule for centuries but applied research over recent decades has developed a diverse "toolbox" for effective conflict mitigation (Breitenmoser et al. 2005; Thirgood et al. 2005; Shivik 2014). Technological advances have brought new methods of nonlethal deterrence (Shivik 2006) and predator-proof fencing, which can be used with or without changes in husbandry practices and guarding (Ogada et al. 2003, Woodroffe et al. 2005). Additionally, financial instruments can offset the costs of the conflicts and ameliorate human–carnivore coexistence (Dickman et al. 2011). With human–wildlife conflict, and especially depredation, being one of the most widespread and urgent issues facing conservation biologists today (Inskip and Zimmermann 2009), the publication rate in this field has steadily increased over the past 25 years (Dickman 2010). Emerging from this burgeoning literature are three branches of research into the prevention or mitigation of human–carnivore conflicts:

12.2.3.1 Consequences of Lethal Control

Lethal control, whether nonselective population reduction, illegal persecution of the species (snaring, poisoning), or retaliatory killing in the context of problem-animal-control policy, has been practiced for centuries. However, in territorial species (as most carnivores are), removal of a territory holder creates a vacuum that is rapidly filled by neighboring individuals or dispersers. This has been demonstrated for lions in Zimbabwe, where some territories outside the protected area of Hwange National Park were successively filled as its occupants were, one after another, removed by sport hunters (Loveridge et al. 2007; see also van de Meer et al. 2014 for territorial drift in wild dogs). Because of reduced levels of intraspecific competition, vacant territories are particularly attractive to dispersing subadults, which are often less efficient hunters than adults and less able to compete for occupied territories. However, compared with residents, dispersers are more daring and thus more likely to use human-dominated landscapes (Elliot et al. 2014a) and kill livestock (Patterson et al. 2004). Hence, the vacuum effect caused by indiscriminate retaliatory killing might not only compromise the viability of some carnivore populations (Woodroffe
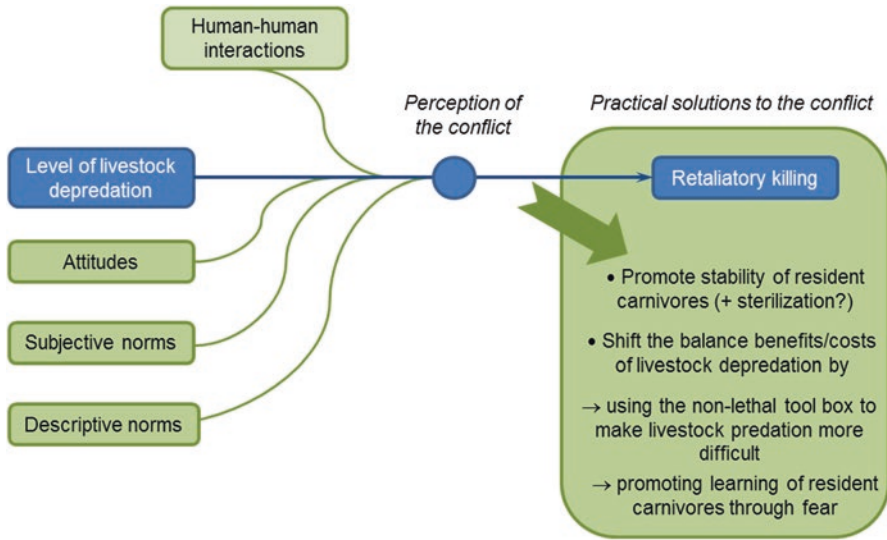
Fig. 12.5 In its most simple form, conflict between humans and large predators on rangelands is driven mainly by some level of depredation on livestock resulting in a negative perception that leads to retaliatory killing (*blue boxes*). However, large predators have important functional properties at the ecosystem level and so there is a mounting need to address the conflict. Recent research reveals that local people's perceptions of the conflict are influenced by a variety of factors that can be channeled into finding smarter solutions (*green boxes*)

and Ginsberg 1998) and disrupt their social stability (e.g., increased infanticide, Swenson et al. 1997; Loveridge et al. 2010—and delayed dispersal, Elliot et al. 2014b), but also be counterproductive for the mitigation of the human–carnivore conflicts. Hence, maintaining resident carnivores without disrupting their social structure is likely the most effective way forward to efficiently control livestock depredation (Fig. 12.5). Indeed, Bromley and Gese (2001a, b) showed that sterilization successfully reduced coyote depredation on sheep by removing the seasonal spike in nutritional demands imposed by provisioning pups, while territorial and social behaviors were maintained. Hence, resident ("better-behaved") sterile coyotes can be used as a management tool to exclude itinerant sheep-killing coyotes.

12.2.3.2 Carnivore Behavioral Ecology in Human-Dominated Landscapes

It is now recognized that carnivores create a landscape of fear for their prey (Laundré et al. 2001). What is less recognized is that carnivores themselves live in landscapes of fear too. People kill carnivores when they are perceived as a threat and so the spatial ecology of carnivores is influenced by their fear of humans. Studies on wolves (Theuerkauf et al. 2003) and lions (Valeix et al. 2012; Oriol-Cotterill et al. 2015a) have shown changes in the behavior of carnivores in the vicinity of human-dominated areas, as indicated by spatial avoidance, temporal shift, change of travel

speed, use of suboptimal habitats, and so on. Carnivores appear to balance the benefits of accessing livestock with the costs associated with livestock raiding, supporting the hypothesis that fear influences the behavioral ecology of carnivores in human-dominated landscapes (Oriol-Cotterill et al. 2015b). This raises new possibilities for the tactical use of fear in managing human–carnivore conflicts (Fig. 12.4) just as hunting for fear has been proposed to induce strong behavioral responses in ungulates as a way of diverting them from areas where their impacts are undesired (Cromsigt et al. 2013).

12.2.3.3 Human Attitudes

Traditionally, research on human–carnivore conflicts has focused on the numerically tractable aspects of ecology and economy. However, the killing of predators is not always simply retaliatory and so the social drivers of human attitudes have to be identified (Lindsey et al. 2005; Dickman 2010; Macdonald et al. 2010; Guerbois et al. 2012). The difference between expected and observed outcomes of mitigation measures is likely to lie in the mismatch between common assumptions made by conservation practitioners and the perceptions and attitudes held by local people (Calvacanti et al. 2010; Dickman 2010; Marchini and Macdonald 2012). This can explain why one conflict mitigation effort might be successful in one area but fail elsewhere. In a study inspired by social sciences, Marchini and Macdonald (2012) revealed that the intention of ranchers to kill jaguars in Brazil is influenced not only by the perceived impact of jaguars on livestock but also by attitudes (an individual's personal feelings about killing a jaguar), subjective norms (an individual's perception of whether important people would approve), and descriptive norms (an individual's perception of whether other people would do the same thing). Human–human antagonisms about response options can lie at the core of a human–wildlife conflict and so the relevant social issues have to be teased out (Fig. 12.5) before workable mitigation measures can be devised (Marshall et al. 2007; Peterson et al. 2010).

12.3 Societal Implications of Integrating the Management of Wildlife and Livestock

Globally, rangelands are crucial for wildlife conservation and much has been written about why and how to conserve wildlife while maintaining livestock in semiarid ecosystems (see du Toit et al. 2010). Yet implementation is the crux of the problem because rangelands, which comprise about 40% of global land cover, include countries with the lowest standards of governance that are home to the most impoverished and war-torn segments of humanity. Responsibility thus rests with societies occupying rangelands in those other parts of the world where reconciling agriculture with wildlife conservation is an achievable goal. Conventional

approaches to natural resource management place social–ecological systems on disappointing trajectories because the economically (and therefore politically) dominant sectors of agriculture and mining outcompete the environmental sector, commonly with negative implications for the sustainable livelihoods of local communities. As an alternative to this "sectoral approach," a review conducted for the UN's Convention on Biological Diversity distilled out a "landscape approach" (Sayer et al. 2013). The principles of that approach strive for the integration of sectoral priorities for the long-term benefit of human livelihoods within defined landscapes. In a western USA rangeland, for example, a ranching landscape might comprise cattle production on private and public land, mineral extraction, wind and solar energy production, irrigated cultivation, hunting and other recreational activities, and small-scale commerce and industry in rural towns. The landscape approach would aim to muddle through land-use conflicts by fostering an integrated long-term perspective by, and for, the diverse stakeholders across all active sectors in that ranching landscape. In some cases this is beginning to happen where agencies are realizing that building local support is the key to cooperative efforts that minimize the potential for legal challenges.

Competition between wildlife and livestock can be both actual and perceived, with perceived competition prevailing especially where livestock owners—and sometimes land owners—derive no tangible benefits from wildlife (Ranglack et al. 2015). The absence of benefits is likely because they are not allowed by command-and-control laws administered by government agencies. For example, the North American model of wildlife management rests on the doctrine that wildlife is held in trust by the government for the public and therefore cannot be owned or marketed. That model has resulted in many conservation successes but is at odds with global evidence of bottom-up models being more effective for ecosystem conservation, especially in rangelands (Victurine and Curtin 2010). Now, because ranchers in the USA still control access to their land, they can sell temporarily exclusive access to outfitter operations whose clients have acquired hunting licenses from the state. This inclusion of outfitting operations into a mixed income system does not happen easily, because it represents a new way of doing business on ranchland and a break from the historic community service provided by ranchers to local hunters (Haggerty and Travis 2006).

In the USA, state fish and wildlife management agencies depend on fishing and hunting license revenue and so they are most responsive to the demands of their hunter constituency. This commodity orientation discounts ecosystem services and brings wildlife management into direct competition with livestock production for three main reasons: (1) high-value game species include ungulates (such as elk) that share food and diseases with livestock; (2) revenue from wildlife goes to the relevant management agency whereas livestock supports local ranching livelihoods; (3) ranchers with permits to graze on public land are required (by a federal agency) to immediately destock if range quality deteriorates, but without commensurate reductions (by a state agency) of wild ungulate populations. Even where wild ungulate population densities are not high enough to measurably reduce range quality for livestock, negative effects are perceived and such beliefs are entrenched in ranching

communities that have railed against officialdom for generations (Ranglack et al. 2015). Now, the natural resource management paradigm of the twenty-first century is all about "resilience thinking" (Walker and Salt 2006; Allen et al. 2011; Curtin and Parker 2014) within a landscape approach (Sayer et al. 2013). For the USA this implies the empowerment of local institutions (counties, landowner associations, grazing associations, etc.) and landowners to economically benefit from, and actively participate in, the management of the local wildlife resource (threatened and endangered species aside), requiring substantial changes to the prevailing North American model of wildlife management.

An example to illustrate the motivation for the above changes can be found in the intermountain rangelands of the western USA. There, quaking aspen (*Populus tremuloides*) forests are vulnerable to chronic browsing of sprouts from the rootstock and when aspen stands die off they are typically replaced by coniferous woodland. This aspen-to-conifer shift and associated changes to ecosystem function exert negative impacts on biodiversity (Seager et al. 2013). There is also a catchment-wide decline in water yield for runoff and groundwater recharge (LaMalfa and Ryel 2008). Major browsers of aspen sprouts are elk and deer, for which state wildlife agencies set population objectives to satisfy the hunting lobby, which in turn generates important revenue for the agencies through license fees. To achieve their population objectives for elk and deer, state wildlife agencies typically suppress predator populations (Beschta et al. 2013). Hunting "tags" are a valuable commodity but local communities earn no share of the hunting revenue from public lands. Furthermore, around the Greater Yellowstone Area, expanding elk herds occurring at higher densities are boosting the spread of elk-borne brucellosis that is now being transmitted back to cattle (Rhyan et al. 2013). The costs of elevated elk and deer densities in upland catchments thus ultimately settle on local communities that incur negative impacts on their ranching enterprises and on the downstream availability of water for agricultural and urban needs. Finally, integrated management of herbivory (by wildlife and livestock), forest dynamics, hydrology, and animal health is bedeviled by the fact that each falls within the domain of a separate state or federal government agency. Meanwhile, in the absence of an integrated approach, the biodiversity of these rangelands is being eroded by exurban housing developments (Hansen et al. 2005).

Wildlife has to contribute meaningfully to community-level economies and become integrated with livestock and other land uses to enhance adaptive capacity at the landscape scale. Although still anathema to many in the North American wildlife profession, such concepts have been tested elsewhere and lessons can be learned. For example, since the latter decades of the twentieth century, integrated conservation and development projects (ICDPs) have been the cornerstone of most schemes to alleviate concurrent crises in human welfare and biodiversity conservation in developing countries. Success is contingent upon government hierarchies passing down appropriate authority to lower levels, or at least allowing revenue sharing, so that rural communities may develop a proprietary interest in their local wildlife resources. One such concept emerged in Zimbabwe in the late 1980s in the form of CAMPFIRE, a program empowering peasant farmers in communal lands to

benefit directly from the wildlife around them. The rapid success of CAMPFIRE in strengthening rural communities and fostering bottom-up conservation has made it an object lesson in the international development sector (Frost and Bond 2008). Unfortunately, ICDP schemes are vulnerable to autocracy and institutionalized corruption (Garnett et al. 2007) yet that does not diminish the concept's potential in countries with better governance. It offers an example of how a tightly legislated top-down wildlife management model can be transformed into a bottom-up model, enabling wildlife to contribute to the resilience of social–ecological systems, with particular application in rangelands (Ranglack and du Toit 2016).

12.4 Future Perspectives

Achieving the effective integration of livestock and wildlife management at the landscape scale requires the negotiation of multiple social and political barriers, and for many rangelands the opportunity might be lost before adequate change can occur. Nevertheless, there is mounting evidence that rangelands can be managed for both livestock and wildlife where the costs of competition, predation, and disease can be offset by mixed revenue streams and facilitated grazing. In affluent countries there are additional possibilities including marketing strategies, such as labeling livestock products as "wildlife-friendly," which might allow for additional premiums to be charged to help offset costs. In developing countries, integrated conservation and development projects (ICDPs) are advocated, funded, and technically supported by foreign aid agencies and international conservation organizations (Garnett et al. 2007). Corruption, poverty, and weak institutions hamper conservation on the rangelands of developing countries, but the flip side has problems too. Paradoxically, the developed countries that support ICDPs in developing countries have poor records of bringing back and adopting the best practices that have emerged. Tightly legislated and compartmentalized government agencies that exert centralized top-down control are not conducive to the emergence of an integrated landscape-scale approach. Local communities on rangelands have to muddle through the process of building their own management frameworks, for which they need technical support from extension services and cooperation from government agencies.

Completely overcoming a "wicked" problem such as livestock–wildlife conflict is virtually impossible because it is too complex to be clearly defined and so there is no clear solution. Nevertheless, progress towards at least a partial solution should be possible with the coordination of efforts in an integrated approach at the landscape scale. The key change for rangelands will be a shift in policies and incentives to sustain ecosystem services despite the pressure for commodity production (Havstad et al. 2007; Norton-Griffiths and Said 2010). Wildlife communities are integral to such services and so wildlife and livestock have to be, and can be, woven together into an integrated management framework if resilient rangelands are to be sustained for the long-term benefit of the people who live off them. But how might these changes come about and what forces will drive them?

In the context of social–ecological systems, it appears that transformations—whether intended or unintended, desirable or undesirable—are most likely to occur when crises force stakeholders to find new ways of doing business (Chapin et al. 2009). The future perspective for rangelands, already affected by the increasing frequency and duration of drought conditions associated with global climate change, is not lacking in crisis scenarios. In some areas, such as privately owned commercial ranchland in southern Africa, drought crises have already driven transitions from cattle-based to wildlife-based enterprises that have transformed the local ranching culture (Carruthers 2008). Yet a transformation of that type is only possible where indigenous wildlife communities and their habitats, and therefore the adaptive capacity of the system, remain sufficiently conserved. The lesson for global rangelands is twofold: (1) drought-driven transformation is to be expected; (2) the outcome could be more desirable than the alternatives if the management objective for the wildlife–livestock interface is proactively shifted from problem control to asset management.

12.5   Summary

Integrating wildlife conservation with livestock production is implicit in the paradigm shift from production maximizing to resilience building in the social–ecological systems known as rangelands. Globally, rangelands are especially important areas for wildlife conservation in return for which wildlife can provide benefits to local communities, both directly through consumptive and nonconsumptive uses and indirectly through the facilitation of ecosystem services. The main issues to accommodate are human–wildlife conflicts arising from competition, disease, and predation. Economic competition tends to relegate wildlife conservation to the arid side of the rainfall gradient for rangelands, where opportunity costs are low. Elsewhere, wild ungulates are generally outcompeted because humans facilitate conditions for livestock, persecute wildlife, and fragment and transform natural habitats.

Diseases that co-circulate through livestock and wildlife populations are difficult to control because veterinary protocols developed for livestock at the ranch scale are seldom effective on free-ranging wildlife populations at the landscape scale. Consequently there have been few successes in effectively controlling "wildlife diseases," many of which were transmitted to wildlife from livestock in the first place. Discussions about controlling diseases that affect livestock on rangelands tend to focus on imposing a spatiotemporal separation at the livestock–wildlife interface. That is simply necessary in certain circumstances but a pragmatic view is needed of the trade-offs associated with blanket animal health policies that enforce such separation. Innovative trade policy reforms are needed before intact wildlife communities can share rangelands with commercially produced livestock, adding adaptive capacity to their social–ecological systems.

Just as with diseases, pragmatic approaches to conserving large predators on rangelands require that societal stigmas be overcome. Apex predators have been extirpated from most of their former ranges but a growing literature demonstrates their diverse and previously underestimated effects at the ecosystem level. This is despite the inevitable association between large carnivores and livestock depredation, so achieving their effective conservation is an immense challenge. Recent methodological advances do nevertheless enable some innovative approaches to mitigating human–carnivore conflicts. These include the use of fear as a tactic to promote learning among resident carnivores without disrupting their social structures. Additionally, the development of a nonlethal toolbox of deterrence methods and changes in livestock husbandry practices should be helpful in shifting the balance between persecution and acceptance of large predators in rangeland ecosystems. Retaliatory killing can be counterproductive albeit necessary in certain circumstances as a short-term response to placate affected livestock owners. Sterilization is a management-intensive option to stabilize localized populations of territorial predators and is likely to have longer-term effectiveness than lethal control. Finally, carnivore conservation cannot be successful without the support of local communities and so the key to coexistence is an understanding of the drivers of human attitudes to large carnivores. Conflict mitigation requires a balance of practical solutions, outreach, and the best available information on both the ecology of the carnivore species concerned and the human dimensions of the problem.

Mounting evidence confirms that functionally intact wildlife assemblages have properties of importance at the ecosystem level. To conserve and restore such functional properties, policy changes and extension programs are needed for local communities to become proprietors of at least a segment of the local wildlife resource. The global experience is that livelihoods on rangelands are most likely sustained in the face of externally driven challenges if communities can self-organize within resilient social–ecological systems. Resilience can be enhanced by weaving wildlife into the frameworks of those systems, which requires that centralized wildlife management agencies adapt to "resilience thinking." Proactive management of the livestock–wildlife interface is integral to the process of strategizing for climate-driven transformations of global rangelands.

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