

GRAZING SYSTEMS, ECOSYSTEM RESPONSES, AND GLOBAL CHANGE

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■ **Abstract** Managed grazing covers more than 25% of the global land surface and has a larger geographic extent than any other form of land use. Grazing systems persist under marginal bioclimatic and edaphic conditions of different biomes, leading to the emergence of three regional syndromes inherent to global grazing: desertification, woody encroachment, and deforestation. These syndromes have widespread but differential effects on the structure, biogeochemistry, hydrology, and biosphere-atmosphere exchange of grazed ecosystems. In combination, these three syndromes represent a major component of global environmental change.

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INTRODUCTION

Managed grazing occupies more than 33 million square kilometers or 25% of the global land surface, making it the single most extensive form of land use on the planet. Managed grazing systems are defined here as any geographically extensive operation designed for the production of animals for consumption, including for meat, milk, and any major animal products. Recent work indicates that managed grazing systems have increased more than 600% in geographic extent (from about 5.3 M km²) during the past three centuries (1). More than 1.5 billion “animal units” (AU)¹ were present in managed grazing systems on Earth in 1990 (2).

Despite these impressive statistics, there are surprisingly few synthetic reports on how managed grazing systems affect global ecological, atmospheric, or hydrological processes. The predominance of site-specific perspectives on ecosystem responses to managed grazing has led to a fragmented understanding of this important land use as a contributor to global environmental change. Given the importance of managed grazing systems to the subsistence of the human population, and given the increasing role that land degradation plays in determining the long-term sustainability of pastoral practices throughout the world (3), a global-scale overview based on available scientific data is overdue.

In this review, we develop a perspective on ecosystem responses to managed grazing. We employ a basic geographic information system (GIS) analysis and a literature review to determine the environmental “footprint” of grazing systems throughout the world. In this case, the footprint represents responses of ecosystems to managed grazing relative to global bioclimatic and edaphic variability. We find that managed grazing occupies bioclimatically and edaphically marginal lands throughout much of the world and that these conditions predispose current rangelands to three regional syndromes—desertification, woody encroachment, and deforestation. We use these syndromes as an organizing framework to synthesize

¹An animal unit (AU) is defined as the number of cattle, buffalo, sheep, goats, horses, and camels weighted by their relative size and growth rates [AU = n (cows + buffalo) + 0.2 n (sheep + goats) + 1.2 n (horses + camels)] (2).

the impacts of managed grazing on ecosystem structure, biogeochemistry, hydrology, and biosphere-atmosphere interactions. We contend that these syndromes, when taken in combination, represent a major component of global environmental change.

THE GLOBAL FOOTPRINT OF GRAZING SYSTEMS

Basic Demographic Patterns

Combining global grazing area in 1990 (1) with country-level stocking rates [AU per grazed area; (2)], we estimated the geographic extent and intensity of managed grazing systems worldwide (Figure 1). The five countries with most land area in grazing systems are Australia, 4.4 M km²; China, 4.0 M km²; United States, 2.4 M km²; Brazil, 1.7 M km²; and Argentina, 1.4 M km². However, based on the fraction of total land area each nation uses for grazing, Mongolia, Botswana, and Uruguay lead with 80%, 76%, and 76%, respectively. Countries with the highest stocking rates are Malaysia, 320 AU km⁻²; India, 272 AU km⁻²; N. Korea, 213 AU km⁻²; and Vietnam, 184; others are found in central Europe and the Middle East. Countries containing large tracts of dryland grazing systems, such as in Australia, Argentina, and the United States, have low stocking rates.

Our GIS analyses of grazing extent, grazing intensity, and human demographic statistics reveal very few correlates (Figure 2). Grazing land area is well correlated with total land area of each country ($r = 0.79, p < 0.05$). However, the total number of animal units per country is weakly correlated with both grazing area ($r = 0.50, p < 0.05$) and total land area ($r = 0.60, p < 0.05$). Grazing intensity—stocking rates at the country level—is not correlated with grazing area or total land area (Figure 2). By far the strongest relationship is found between human population and the number of grazing animals per country ($r = 0.91, p < 0.01$). Human population growth is likely to increase demand for meat and dairy products, which will have to be met by a combination of increasing intensification and continued extensification, as has been observed over the past 300 or more years. Evidence reported in the following sections suggests that grazing extensification cannot occur without continued major changes in global land cover, and that intensification will also have significant environmental impacts.

GIS Analysis of Bioclimatic and Edaphic Conditions

The global footprint of managed grazing, which implicitly represents human decisions on where to develop grazing systems, spans a gradient of identifiable environmental conditions presented in this section. We show that managed grazing dominates in the marginal bioclimatic and edaphic regions of drylands. Grazing occurs in the best bioclimatic areas of temperate forests and woodlands but is employed on marginal soils found throughout much of the humid tropics.

Country-level correlates

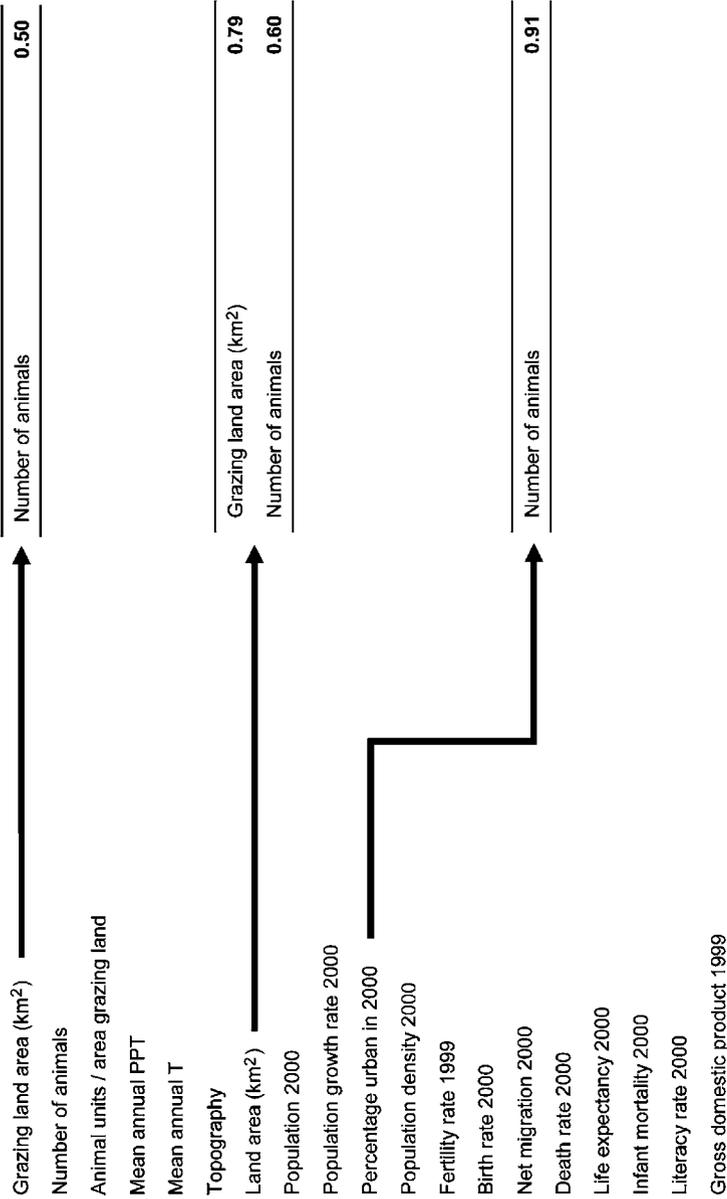


Figure 2 (left) Country-level variables used in a series of Pearson product-moment correlation analyses. (right) Correlations (*r*-values) of particular importance to determining linkages between land area, stocking rate, and human population size (all *p*-values < 0.05). All unreported correlations were weak and statistically insignificant. Derived by combining References 1, 2, and 8.

GIS analyses of global grazing by biome reveal that savannas, grasslands, shrublands, and deserts support the largest extent of managed pastoral systems (Figure 3). In combination, these dryland biomes cover more than 67 M km² of the ~132 M km² total global biome area (Table 1). Although these biomes contribute about 51% of the total land area on the planet, they support a disproportionately high 78% of the global grazing area.

Outside of these dryland systems, other biomes support substantial levels of managed grazing. Roughly 30% and 56%, respectively, of temperate deciduous and temperate evergreen broadleaf forests and woodlands are now supporting grazing systems (Table 1); however, some studies indicate that the extent of grazing systems is decreasing in temperate forests following historically higher levels (4). About 1.7 M km² (or 10%) of tropical evergreen broadleaf forests have been cleared for managed grazing, and this area is growing annually in regions such as the Amazon basin, Congo, and Southeast Asia (5). Many believe that humid tropical ecosystems represent the only viable way to expand global grazing systems beyond its current geographic extent (6).

We sought to uncover the environmental conditions under which managed grazing occurs globally. Actual evapotranspiration (AET) was selected as an integrating metric of bioclimatic stress for vegetation growth, because AET is low in cold and/or dry regions and high in warm and/or wet areas. We analyzed the spatial extent of managed grazing relative to the AET for each biome (Figure 4a). There is a clear bioclimatic footprint throughout global pastoral systems, evident even in using the coarse analyses afforded by available global GIS data (Table 1). In savannas, grasslands, and deserts, grazing systems persist in areas where AET is 20% to 25% lower than the average AET of each biome. Managed grazing is preferentially employed in areas that are much drier than the biome mean, as shown in Table 1, with the ratios of grazed biome:total biome annual precipitation ranging from 0.69 to 0.82 for savannas, grasslands, and deserts. These results suggest that the bioclimatically marginal portions of drylands service the global grazing enterprise and that other forms of land use (e.g., agriculture and urbanization) occupy the fraction of dryland biomes with less bioclimatic stress. Indeed, by overlaying the global extent of croplands (7) on the AET map (8) shown in Figure 4a, we found that agriculture persists on the fraction of dryland biomes with 47% to 203% higher AET than the biome average (map not shown).

In contrast to drylands, AET rates of grazing systems in temperate deciduous, evergreen, and mixed forests are 30% to 75% higher than their biome mean AET (Table 1). This trend persists in the extreme cold boreal and tundra regions. The managed grazing footprint is evident as grazing AET:mean biome AET ratios are from 1.20 to 1.75, most of which is explained by the preferential use of warmer regions for grazing practices in these cold ecoregions. Temperature conditions in the portion of boreal evergreen forests used for managed grazing are more than 300% warmer than the biome mean annual temperature. However, these are the extreme cases; boreal and tundra biomes support less than 1% of the global grazing enterprise (Table 1).

TABLE 1 Biome statistics of land area, managed grazing, and climatology

Biome ^a	Total area (M km ²)	Percent of global land area	Area grazed (M km ²)	Percent biome grazed	Mean grazing area AET	Mean biome AET	Grazing: biome AET	Grazing: biome MAP ^b	Grazing: biome MAT
Savanna	19.31	15	9.48	49.1	595	781	0.76	0.69	0.90
Grassland/steppe	14.22	11	7.68	54.0	321	401	0.80	0.75	0.59
Desert	15.45	12	1.97	12.8	71	88	0.81	0.82	0.95
Dense shrubland	6.01	5	2.73	45.4	314	339	0.93	0.90	0.96
Tropical evergreen forest/woodland	17.43	13	1.72	9.9	1114	1141	0.98	0.97	0.87
Temperate broadleaf evergreen forest/woodlands	1.26	1	0.71	56.0	821	818	1.00	1.00	1.00
Tropical deciduous forest/woodland	5.96	5	1.20	20.2	935	859	1.09	1.04	1.04
Boreal evergreen forest/woodland	6.36	5	0.08	1.2	424	354	1.20	0.78	3.03
Open shrubland	12.09	9	3.98	32.9	297	243	1.22	1.22	1.26
Boreal deciduous forest/woodland	2.18	2	0.02	1.1	435	352	1.24	1.45	0.45
Temperate deciduous forest/woodland	5.10	4	1.49	29.1	793	611	1.30	1.62	1.32
Temperate needleleaf evergreen forest/woodland	3.62	3	0.76	20.9	689	463	1.49	1.59	2.23
Evergreen/deciduous forest/woodland	15.68	12	1.26	8.0	642	369	1.74	1.50	1.67
Tundra	7.32	6	0.17	2.3	431	247	1.75	1.74	0.01

^aBiomes are ordered by the ratio of actual evapotranspiration (AET) in grazed portions of each biome compared to the biome mean AET. Source data: Ramankutty & Foley (160), Hearn et al. (8), and Goldewijk et al. (1).

^bOther abbreviations are MAP, mean annual precipitation, and MAT, mean annual temperature.

Mean annual AET is a metric of the average bioclimatic conditions of an ecosystem, but the manageability of grazing systems (and other forms of land use) is also largely determined by climate variability, which is not captured in a mean AET estimate. Regions with large interannual precipitation and temperature variation undergo climatic boom-bust cycles that strongly affect vegetation production, grazing capacity, and human living conditions. It is extremely difficult to assess the global footprint of grazing systems with respect to climate variability because long-term, spatially explicit climate records are not readily available. We used a 17-year satellite record of the normalized difference vegetation index (NDVI) (monthly temporal resolution; $1^\circ \times 1^\circ$ spatial resolution) to map the global interannual variability of vegetation greenness, which is highly correlated with primary production (9). We calculated absolute NDVI anomalies from the average annual climatic cycle, 1982–1999. We then compared this measure of vegetation variability to the global distribution of managed grazing (Figure 4b). This analysis shows that global grazing systems persist in nearly all regions with high NDVI variation (darker gray). Biomes with the highest NDVI variation are, in descending order of variance: savannas, temperate deciduous forest, shrublands, grasslands, and boreal systems. Within these biomes, grazing systems occur in zones that are 5% to 63% more variable in terms of vegetation cover and condition than the mean biome variability. Managed grazing is thus practiced in the biomes and within the regions of these biomes that experience substantial climatological and ecological variation.

Managed grazing systems also have a distinguishable global edaphic footprint. Soil types are shown by taxonomic order (10) along with global grazing extent in Figure 4c. We calculated the statistical mode of soil type presence by taxonomic order for grazed areas of each biome and compared it to the mode of soil presence for the entire biome (Table 2). In savanna, shrubland, and desert biomes, grazing systems are predominantly found on marginal soils, such as aridisols and entisols, relative to the most common soils, alfisols, found globally in these biomes. In the colder boreal biomes, grazing takes place preferentially on alfisols and spodosols but not on frozen gelisols, which are the most common soil order found in these regions.

There also exists a clear edaphic footprint of grazing in humid tropical regions. Ultisols dominate grazing systems found in the Amazon basin, Congo, and Southeast Asia (Figure 4c), yet the most common soils found in these regions are oxisols (Table 2). Oxisols are widely recognized as nutrient poor and thus marginal for managed grazing systems (11). Ultisols, are more manageable in terms of fertility but are also often considered biogeochemically marginal. This global footprint of grazing systems is probably not a coincidence because ranch managers often select the best available soils (ultisols) (6). It is nevertheless surprising to observe a global footprint of managed grazing on soils in the humid tropics, as this area represents the summed effect of millions of ranch managers operating at small geographic scales throughout the world. Tropical deforestation is largely driven by an increasing need for grazing land, with grazing systems expanding in the humid tropics at a rate of $>15,000 \text{ km}^2 \text{ year}^{-1}$ (12). The need for additional

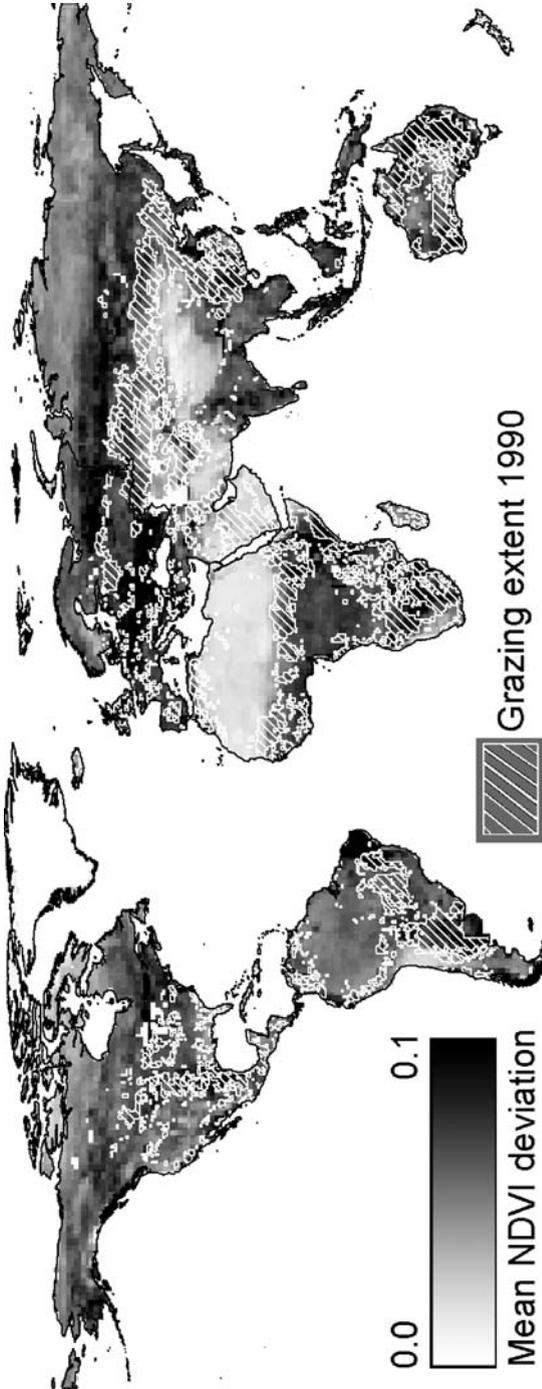


Figure 4b The global distribution of managed grazing systems and the interannual variability of vegetation production, as indicated by the satellite metric normalized difference vegetation index (NDVI). Mean NDVI deviation is the interannual variability of vegetation greenness, after accounting for mean monthly greenness. This record represents the period from 1982 to 1999. Derived from combined References 1 and 161.

TABLE 2 Global distribution of managed grazing systems by biome and taxonomic soil order

Biome ^a	Mode of soil order of grazing areas	Mode of soil order	
Tropical evergreen forest/woodland	Ultisols	Oxisols	} Grazing on the more fertile soil
Tropical deciduous forest/woodland	Ultisols	Oxisols	
Temperate broadleaf evergreen forest/woodlands	Ultisols	Histosols	
Temperate needleleaf evergreen forest/woodland	Alfisols	Alfisols	
Temperate deciduous forest/woodland	Inceptisols	Inceptisols	
Boreal evergreen forest/woodland	Alfisols	Gelisols	} Grazing on the unfrozen soils
Boreal deciduous forest/woodland	Spodosols	Gelisols	
Evergreen/deciduous mixed forest/woodland	Inceptisols	Gelisols	
Grassland/steppe	Mollisols	Mollisols	
Savanna	Entisols	Alfisols	} Grazing on the less fertile soils
Dense shrubland	Aridisols	Alfisols	
Open shrubland	Aridisols	Alfisols	
Desert	Aridisols	Entisols	

^aSource data: Klein Goldewijk et al. (1), Ramankutty & Foley (160), USDA (10).

grazing land results in part from human population growth but also from pasture degradation caused by the dominance of low fertility soils (see the deforestation section).

These GIS analyses illuminate the climatic and edaphic factors limiting the expansion of grazing systems. Considering the current extent of grazing and a growing population, expansion of grazing systems in arid and semiarid regions will require the conversion of cropland systems, a land-use change that is unlikely to occur given the pressure for grain production worldwide (13). Intensification of animal production and grazing systems is likely to continue, requiring expensive management or causing greater degradation of already marginal lands. Any further intensification of global grazing systems will likely occur through the conversion of forests to pastures, as is well under way in the humid tropics.

Regional Syndromes in the Global Grazing Footprint

At the global scale, endogenous environmental conditions set the stage for determining both the ecosystem responses to grazing and the limitations imposed by climate and soils on the expansion and intensification of grazing practices worldwide. The bioclimatically marginal nature of grazing lands in arid and semiarid

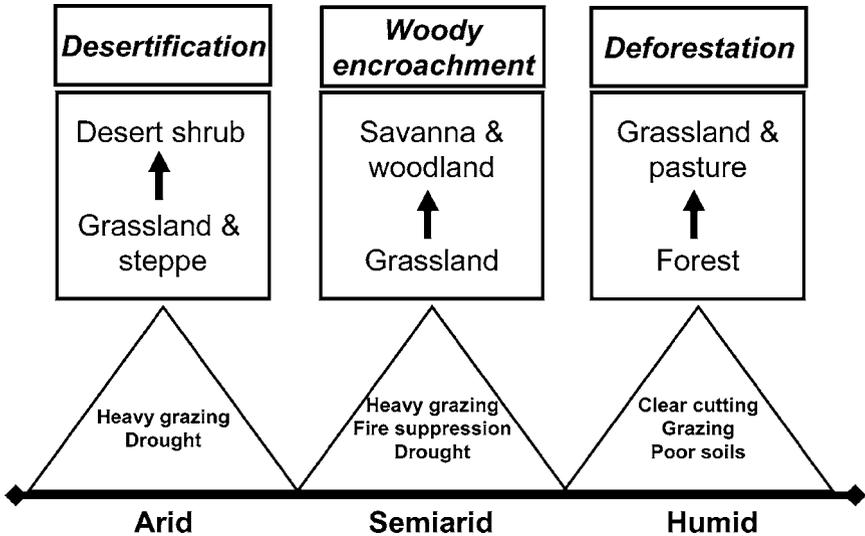


Figure 5 Three regional syndromes resulting from managed grazing practices across global-scale gradients of bioclimatic and edaphic conditions.

regions plays a key role in two regional syndromes widely reported throughout the literature—desertification and woody encroachment. The marginal biogeochemical nature of humid tropical soils accelerates the process of deforestation, a third regional syndrome (Figure 5).

One problem in describing these syndromes lies in the definitions of desertification, woody encroachment, and deforestation. The most difficult to define is desertification. In the past, a reduction in net primary productivity (NPP; vegetation growth) has been used as an indicator of desertification (14). Others have focused more on the composition and structural configuration of vegetation types (15), whereas the common observation of increased bare soil (exposed and eroded surfaces) has been the most definitive trait in yet other studies (16). Ash et al. (17) argue that desertification can best be analyzed in terms of secondary production losses (e.g., cattle, sheep, and human). In contrast, woody encroachment has been defined as the increased geographic extent of woody vegetation in ecosystems. Expansion of woody vegetation may or may not be a component of desertification; and when it is, the structural features of the woody cover changes are different from that of woody encroachment. Most notably, in woody encroachment, herbaceous cover in the intercanopy zones is typically left intact, whereas in desertification, these zones become bare soil surfaces with decreased soil resources (e.g., organic matter). Deforestation has been described as the conversion of forest landscapes to grazing systems (pastures), but this use is complicated by variation in the definition of forest. Some focus on primary or mature forest; others include areas of secondary or regrowing forest.

To present our synthetic perspective on the common syndromes inherent to grazing systems worldwide, we simply define all three processes, desertification, woody encroachment, and deforestation, in terms of changing ecosystem structure (Figure 5). *Desertification* is the replacement of herbaceous cover by shrub cover and bare soil. *Woody encroachment* is the addition of woody canopies without major losses of herbaceous cover, although herbaceous production may decrease. *Deforestation* is operationally defined as the replacement of forest cover with herbaceous pasture systems.

Independent of the precise definition of each regional syndrome, the bioclimatic and edaphic conditions under which managed grazing occurs have, to some degree, contributed to the development of these three syndromes. It is widely understood that desertification has occurred in arid regions of the world (e.g., southwest United States, Australia, South Africa, and Argentina) as a result of large-scale grazing and pronounced climatic variability (Figure 5) (3). Woody encroachment, as defined above, has occurred in semiarid to mesic environments as a result of large-scale grazing, fire suppression, and climatic variability (18). Deforestation continues to expand in the humid tropics (and elsewhere) in part because of grazing development on infertile soils that often cannot sustain large-scale managed pastoral operations (6). These three syndromes are regional in nature, but they are present throughout grazing systems on a global scale.

The remainder of this review provides a synthetic perspective on the ways that managed grazing has altered ecosystems across a bioclimatic gradient from arid to mesic to humid conditions. Synthesis of vegetation structural and the biogeochemical, hydrological, and atmospheric effects of grazing systems are presented using the syndromes as the organizing framework. In doing so, we demonstrate how managed grazing systems contribute significantly to global environmental change.

DESERTIFICATION

Ecosystem Structure

The myriad perspectives and intended audiences of studies on grazing systems lead to variation in vegetation classifications and, thus, in the observations, analyses, and conclusions in the literature. Most published studies use one or very few classifications to describe vegetation-grazing interactions. Common classifications in grazing studies are woody versus herbaceous (19) or perennial versus annual (20). Relatively few studies break the vegetation down into classes of C3 versus C4 physiology (21), evergreen versus deciduous life forms (22), or by nitrogen fixing abilities (23). Few studies have focused at the species level (24), and those that do often involve the introduction or spread of invasive plants (25, 26). Drawing from the scientific literature, managed grazing appears to play a central role in altering the biophysical structure of grazed ecosystems globally. We use *ecosystem structure* to discuss the spatial extent and configuration of major vegetation life-forms (trees, shrubs, and herbaceous cover). This is useful because the majority

of studies describe changes in ecosystem structure that appear to be caused or accelerated by grazing practices.

The literature highlights a consistent set of ecosystem structural changes involved in desertification. These changes can be described in three major features and one overarching pattern. The features include (a) increased bare soil surface area, (b) decreased herbaceous cover, and (c) increased cover of woody shrubs and shrub clusters. The overarching pattern is one of increased spatial heterogeneity of vegetation cover and a concomitant increase in the spatial variance of belowground resources, such as organic matter, nutrients (see the Biogeochemistry section), and soil moisture (see the Hydrology section).

Many grazing systems experiencing desertification in the southwestern United States, Australia, and Africa are now dominated by one or a few woody shrub species, with little herbaceous canopy remaining on the landscape (27–29). Okin et al. (16) suggested wind erosion removes soil nutrients and carbon from shrub interspaces. Once established, a combination of biogeochemical and hydrological feedbacks sustains these shrub systems in a new stable state, very different from the prior grassland (Figure 6) (15).

Desertification can also happen without a major increase in woody plant cover but rather as an increase in bare soil causing fragmented herbaceous cover (30). Van de Koppel et al. (31) suggested the following progression of events leading to a fragmented landscape. Herbaceous cover decreases in areas preferential to

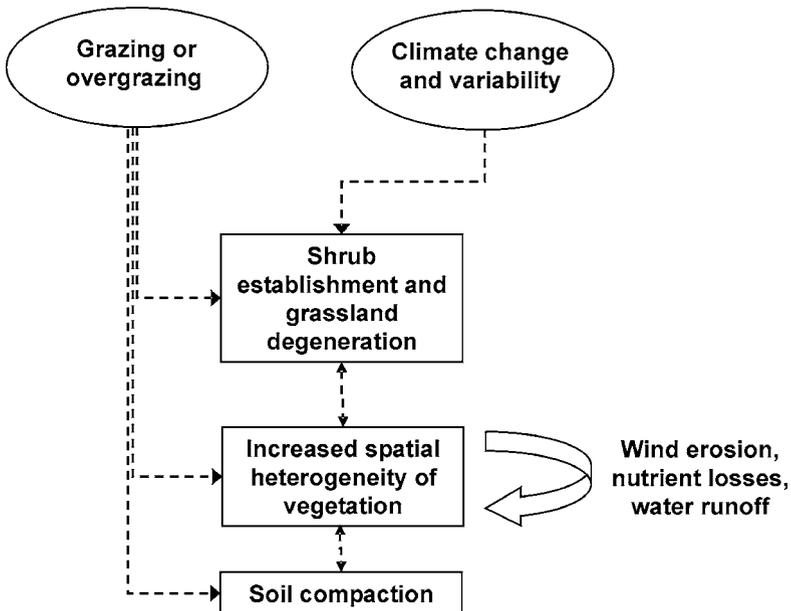


Figure 6 Processes mediating desertification in arid grazing systems.

grazers, leaving compacted bare soils that allow rainfall to run off into remaining vegetation patches. This increases the productivity of remaining patches, which stimulates increased grazing and, thus, loss of these patches, eventually leading to total ecosystem collapse. There is currently little empirical evidence to test this model, but another modeling study concurs that patch dynamics are mediated by grazing, climate variability, and surface hydrological transport (32).

A phenomenon related to managed grazing, land degradation, and desertification is the human-mediated dispersal of African grasses worldwide. Introduced African grasses have made their ecological mark in dryland (and tropical) systems in North America, Central and South America, Australia, and Oceania (33). These grasses compete effectively with native grass species and can alter nutrient cycling and other ecosystem processes (34–36). African grasses are typically fire tolerant and quite flammable, increasing fire frequency and promoting their further geographic expansion (26).

Biogeochemistry

Some changes in ecosystem structure (abundance, cover, and configuration of life-forms) described above are directly attributable to grazing. These structural alterations result in a cascade of change in other ecosystem processes, such as water drainage, wind and water erosion, species invasion, disturbance types and frequency (e.g., fire), carbon cycling, and the biophysical and biochemical characteristics of soils (21, 31).

Where desertification is occurring, degradation often results in reduced productivity or vegetative cover, which brings with it a change in the carbon (C) and nutrient stocks and cycling of the system (Figure 7) (37–43). The primary ecosystem response to desertification is an increase in the heterogeneity of vegetation cover, with concomitant increases in the spatial variability of soil C and nutrients (15, 54). Overall, reduced vegetative cover and total aboveground biomass seem to result in a small reduction in aboveground C stocks and a slight decline in C fixation, measured as NPP, but there is significant variability by vegetation type with topographic and edaphic factors (43). Because total nutrient pools are relatively small, any decline in total nutrient stocks has a significant impact on productivity.

Despite small, sometimes undetectable changes in aboveground biomass and NPP, both total soil C and nitrogen (N) usually decline (38). The reduced soil organic N may result from increased N lost in surface runoff, increased trace-gas flux, and vegetation removal by grazers. Decreased infiltration and increased runoff elevate losses of both inorganic and organic N in overland flow; however, these losses are smaller than inputs to the system from deposition in some regions (39, 55). Nitrogen trace-gas losses, particularly as nitrous oxide, are large relative to the total N pool (see the next section). Most studies on biogeochemical changes caused by desertification are from Northern Chihuahuan ecosystems in New Mexico, United States of America. A recent study by Asner et al. (37) in Argentina also found that desertification resulted in little change in woody cover, but there was

	Before	Change in flux	After	Net effect
Desertification	<p>C 2.5–40.0^{5*} N ?</p>  <p>C 0.7–1.2¹ 0.33–1.99%² N 0.07–0.16%²</p> <p>10cm depth</p>	<p>C slight decrease in ANPP⁵ 0.7–2.3 increase in N gas loss⁴</p> <p>Desertification Increased spatial heterogeneity of C and nutrients</p> <p>C ? N 0.15 increase in runoff³</p>	<p>C 3.3–23.0^{5*} N ?</p>  <p>C 0.6–0.8¹ 0.25–0.57%² N 0.07–0.10%²</p> <p>10cm depth</p>	↓
Woody encroachment	<p>C 400–3800^{6,8} N 1–68⁸</p>  <p>C 11650–22000^{8,9} 10cm depth N 910–2000^{8,9} C 33800–27800⁷ 3m depth N 4400–26400⁷</p>	<p>N 0–0.8 as NO¹¹</p> <p>C 0 – 1400 increase in ANPP^{8,10} N 9–40 increase in N production^{8,10} (large increases require N fixing shrub)</p> <p>Woody Encroachment Increased spatial heterogeneity of C and nutrients</p> <p>C ? N 36 increase in leaching⁸</p> <p>Erosion Losses ?</p>	<p>C 3000–21000^{6,8} N 40–536⁸</p>  <p>C 15000–23520^{8,9} 10cm depth N 1500–1920^{8,9} C 41700–21600⁷ 3m depth N 4400–23700⁷</p>	?
Deforestation (Wet tropical)	<p>C 130000¹² N 1370¹² P 41¹²</p>  <p>C 213000^{12,14} N 18180¹² P 12754¹²</p> <p>8m depth</p>	<p>C 88000¹³ N 1181¹³</p> <p>Fire & Conversion</p> <p>Leaching losses ? Erosion Losses ?</p> <p>Repeat Burning (each burn)</p>	<p>C 3900¹² N 35¹² P 1.9¹²</p>  <p>C 201000^{12,14} N 17639¹² P 13277¹²</p> <p>8m depth</p>	↓

Desertification

- ¹ Asner et al. (37)
- ² Gallardo & Schlesinger (38)
- ³ Schlesinger et al. (39)
- ⁴ Hartley & Schlesinger (40)
- ⁵ Schlesinger & Peterjohn (41)
- ⁶ Peterjohn & Schlesinger (42)
- ⁷ Huemneke et al. (43)
- * assumes that biomass is 50% C

Woody encroachment

- ⁶ Asner et al. (44)
- ⁷ Jackson et al. (45)
- ⁸ Hughes et al. (46)
- ⁹ Hibbard et al. (47)
- ¹⁰ Geesing et al. (48)
- ¹¹ Martin et al. (49,50)

Deforestation (Wet tropical)

- ¹² Markewitz et al. (51)
- ¹³ Guild et al. (52)
- ¹⁴ Fearnside & Barbosa (53)

a 25% to 80% decline in soil organic C and N storage in areas with long-term grazing.

Soil compaction reduces infiltration and increases runoff, resulting in faster and greater flow through waterways and greater channel and gully erosion (see the Hydrology section). As a result, soil is lost, and sediment loads increase in waterways (56). In systems where soil erosion is substantial, resulting losses of soil C, N, and P may also be important but have not been well quantified. Wind erosion could also be important and enhanced by increased bare soil in the case of desertification (16). Phosphorus in arid ecosystems is often bound to calcium carbonates and retained in the mineral soils; thus, erosion tends to mobilize C and N more so than it does P (57). Grazing also breaks up and reduces coverage of cryptobiotic soil crusts found in many arid regions. Although disturbing these crusts can increase infiltration of water, it also reduces inputs of C and N fixed by biological activity of the crusts (58).

Trace-Gas Emissions

Increases in greenhouse trace gases (CO_2 , CH_4 , N_2O , and O_3) since preindustrial times have led to a warming of the Earth's surface and other climate changes (59). Though anthropogenic emissions of trace gases from fossil-fuel burning and fertilizers account for the majority of trace-gas emissions, soils account for more than 30% of biogenic trace-gas emissions (60). Soil trace-gas production and consumption vary spatially and temporally, and they are governed by factors such as soil nutrient stocks and cycling rates, soil temperature and moisture content, and vegetation cover (61), all of which are changed by grazing (previous sections).

Nitric and nitrous oxide gases (NO and N_2O) are produced in the soil during the processes of nitrification and denitrification, and these fluxes are mediated by vegetation litter inputs. The partitioning of NO and N_2O fluxes at the soil-air interface is dependent upon the soil water content, with a shift from NO production to N_2O production as the soil water increases (62). Nitric oxide is a key component in regional-scale ozone regulation.

Woody vegetation cover changes associated with desertification have measurable impacts on soil NO emissions. Hartley & Schlesinger (40) found higher NO emissions from soils under woody canopies than in intercanopy zones. These findings support the concept of enhanced nutrient stocks and cycling under woody vegetation canopies, thereby resulting in enhanced N gas emissions at the plot scale. Although there were little to no N_2O emissions measured in the field from

Figure 7 Effects of three syndromes of managed grazing on biogeochemical properties of arid/semiarid, mesic, and humid tropical ecosystems. Carbon (C), nitrogen (N), and phosphorus (P) stocks and fluxes are taken from literature sources cited to the left. All pools are in kg/hectare (ha) except those noted as a percentage, and all fluxes are in kg/ha/year except for the tropical systems where they are kg/ha/burn. Values for tropical ecosystems are averages across multiple studies.

bare soils due to the aridity, laboratory studies indicated N_2O emissions from soils collected under shrub canopies can potentially be two times greater than those collected in a nearby grassland (63).

Methane (CH_4) is generated in soils during anaerobic respiration, and it has a global warming potential 24.5 times that of CO_2 . In some cases, soil compaction due to grazing limits soil aeration and stimulates CH_4 production (64). Grazing affects the atmosphere through the direct emission of CH_4 gas from ruminants and via increases in ammonia (NH_3) production from livestock excreta. Globally, approximately 54 terragrams (Tg) $N-NH_3$ are emitted each year, with the largest fraction (~40%) from animal waste (65). Excreta deposited on grassland by grazing animals stimulated N_2O production, contributing up to 22% of the total N_2O emission from a U.K. grassland (66). The current estimate of CH_4 from ruminant animals and animal waste is 100 Tg CH_4 , nearly a fifth of the total global emissions (59). CH_4 emissions from livestock have increased about fivefold over the last century in close step with the increasing rate of cattle production (67).

Hydrology

The balance between plant-available water and evapotranspiration (ET) regulates soil moisture and ultimately determines many characteristics of ecosystem structure and functioning on grazed lands (68). In grazing systems, ET (calculated as the sum of evaporation from bare soil and plant surfaces and from transpiration through plant stomata) is often the largest loss of water from the system (Figure 8) (56, 69, 70). Controls over this flux are the relative fractions of soil and vegetation cover, leaf area index (LAI), and plant-available soil moisture. Grazing influences each of these variables.

Vegetation cover and LAI decline as grazers remove plant matter. Reduced plant surface area results in lower transpiration and retention of soil moisture throughout the root zone (Figure 8) (71). Grazing thus leads to increased soil

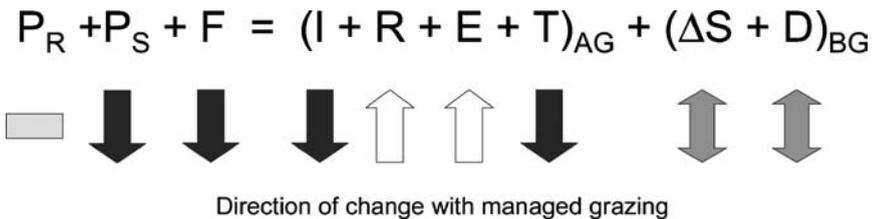


Figure 8 Components of the hydrologic cycle and reported directions of change with managed grazing. Processes are divided into aboveground (_{AG}) and belowground (_{BG}) components. Other abbreviations are P_R , rainfall; P_S , spring snow melt; F , fog or cloud condensation on aboveground plant matter; I , canopy interception; R , runoff; E , evaporation from soil surfaces; T , transpiration from the canopy; S , the change in soil moisture; and D , discharge through subsurface flow vertically and horizontally away from plant roots. P_S and F are not applicable in all environments.

moisture relative to nongrazed pastures (72–74). Grazing also increases bare soil surface area, resulting in greater radiative heating of soil surfaces and increased evaporation (71). Therefore, net changes in soil moisture can be assessed with regard to the balance between LAI and bare soil surface changes resulting from grazing. Reductions in LAI alone will increase soil moisture; increases in bare soil area will decrease soil moisture; and when both occur, the change in soil moisture is uncertain (75).

Grazing compacts soil and exposes soil surfaces, both of which lead to lower infiltration, increased runoff, and higher erosion rates (Figure 8) (56). Trimble & Mendel (76) found that infiltration decreased from approximately 50 mm h^{-1} on lightly grazed to 25 mm h^{-1} on heavily grazed land, but they also highlighted the large variance in field measurements. Infiltration is sensitive to many factors, including soil conditions at the time of grazing and the degree of bare soil exposed following grazing. The link between higher runoff and erosion rates follows logically because increased surface flow can carry larger sediment loads. Grazing reduces vegetation cover, leading to bare and unstable soil surfaces; therefore, the effects of grazing on erosion can be notably larger than the impact of climatic changes, such as increased precipitation (77). Conversely, the absence of grazing results in litter buildup, which has been found to reduce runoff and erosion (78). The balance between infiltration and runoff depends on the hydrologic conductivity and spatial heterogeneity of soil and vegetation surfaces (79).

Desertification involves specific hydrologic changes that include increased spatial heterogeneity of soil moisture and less effective transfer of precipitation to soil moisture (15). Net primary production per unit of precipitation may decrease (80), creating landscapes, with a low rain-use efficiency, that resemble deserts. There is evidence that grazing, combined with feedback mechanisms in the climate system, eventually makes these symptoms permanent (31). The process may begin in areas with vegetation cover removed and soils compacted, resulting in declining infiltration rates and moisture in surface soils. Deeper soil layers and soils where vegetation cover remains high may continue to receive recharge from large storm events (81). The landscape thus becomes more heterogeneous, with patches of vegetation helping to maintain higher infiltration, soil stability, and nutrient retention. Meanwhile, bare soil interspaces become increasingly depauperate in soil moisture and nutrient resources. With continued grazing and high climate variability, vegetated areas become increasingly rare, shifting systems even further toward a desert-like state.

Climate Interactions

Early studies of desertification suggested that changes in surface albedo caused by increased bare soil cover could have regional and even global climate effects. Charney et al. (82) predicted that increases in surface albedo due to Sahelian desertification would increase radiative heat losses from the Sahara, thus reducing rainfall. Additional studies incorporating albedo, transpiration, and roughness

found that a positive feedback reduced rainfall in the Sahel as well as on the Indian subcontinent (83).

In Sonoran drylands, surface temperatures were generally 2°–4° higher on the brighter, more heavily grazed Mexican side of the border than on the U.S. side (84). In this case, temperature differences were shown to impact soil moisture and cloudiness, but no changes in precipitation were apparent. A more recent study, using Landsat Thematic Mapper data, found only small Arizona/Sonora trans-border differences in albedo and radiant temperature along 25 one-kilometer transects (85).

WOODY ENCROACHMENT

Ecosystem Structure

There are hundreds of documented cases of increased woody plant cover in semiarid, subtropical rangelands of the world [(86); <http://cnrit.tamu.edu/>]. In North and South America, Africa, Australia, and elsewhere, woody vegetation cover has increased significantly in grazing systems during the past few decades. Cited causes of woody encroachment include overgrazing of herbaceous cover that reduces competition for woody seedlings, fire suppression that enhances woody plant survival, atmospheric CO₂ enrichment that favors C₃ (woody) plant growth, and nitrogen pollution that favors woody encroachment (Figure 9) (18, 87).

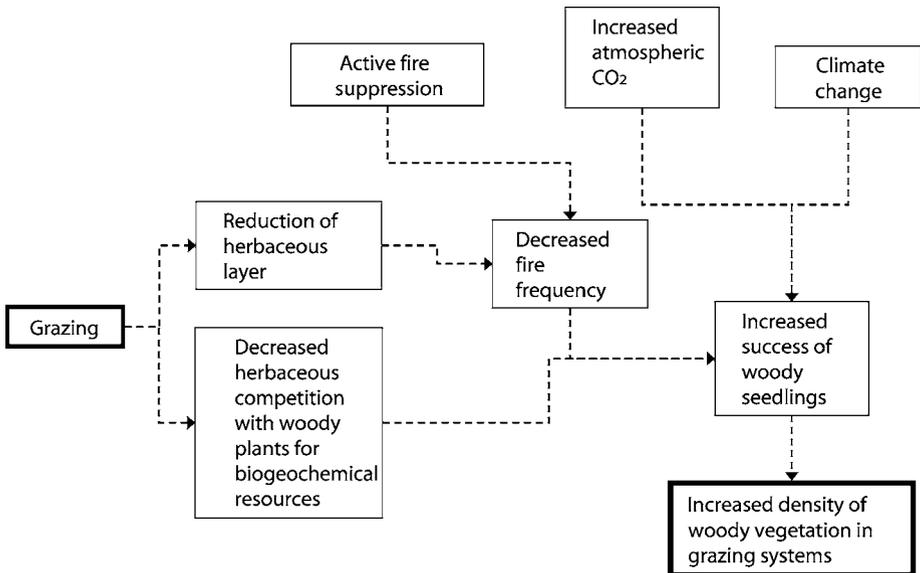


Figure 9 Processes mediating woody vegetation encroachment in semiarid and mesic grazing systems.

It is noted in most encroachment studies that the woody plants were present somewhere on the landscape prior to the installment of managed grazing. For example, in a southern Texas rangeland containing a diverse array of trees, shrubs, and subshrubs, heavy grazing caused increases in the cover of the nitrogen-fixing tree *Prosopis glandulosa* var. *glandulosa* (mesquite). Long-term records and aerial photographs indicate that mesquite encroachment then facilitated the establishment of other woody plants in its understory, which subsequently outcompeted mesquite for light and other resources (88). Mesquite remnants are commonly found among well-developed patches of woody vegetation known not to have existed a century ago (89). The same species of mesquite has increased dramatically in cover in a northern Texas rangeland during the past century (44), but there are very few other woody species established in this region. Most other species are confined to riparian zones; thus few woody plants can be found in association with the mesquite cover (90). Precipitation conditions are similar between the northern (650 mm) and southern (680 mm) Texas sites, but temperatures are substantially lower in the north, with values below freezing in many months (91). Low temperatures in the north likely preclude the presence of many warm-climate woody plants found in the south (*Acacia*, *Diospyros* spp.), and thus the ecological dynamics of woody encroachment are very different between sites.

There are some basic trends in vegetation-grazing interactions associated with woody encroachment in global drylands. Five vegetation properties are consistently highlighted in the literature as changing with respect to grazing and/or the release from grazing: (a) woody vegetation cover, (b) herbaceous vegetation cover, (c) surface litter cover, (d) dominance by perennial herbaceous plants, and (e) dominance by annual herbaceous plants (Figure 10).

At light grazing intensity, most studies indicate slight increases in woody cover if the woody plants are present in the area (Figure 10) (92). Somewhat independent of woody vegetation dynamics, most reports show a decrease in herbaceous vegetation biomass and/or cover and in surface litter cover/biomass (24, 93). In light grazing scenarios, some studies mention decreases in perennial grasses, although a clear trend among annuals is not evident (94). Changes are much more pronounced in cases of long-term, heavy grazing (Figure 10).

Many studies indicate dramatic increases in woody cover or biomass if the woody plants are already present or introduced to the region (86). Both herbaceous and surface litter (cover and biomass) are found to decrease under conditions of heavy grazing (37, 78). Shifts from perennial to annual grasses are more obvious in heavy grazing regimes (95, 96). Following heavy grazing, a release, or substantial rest period, woody cover often remains elevated (Figure 10); that is, there are few if any studies showing decreases in woody cover following release from heavy grazing, but herbaceous and surface litter cover and biomass typically do increase (97). A few studies indicate that annual grasses may initially increase following release from grazing, but often the grasses are replaced by perennials over periods of years or a few decades (98). A net outcome of heavy grazing, even following a release from such practices, can be an increase in woody cover regionally.

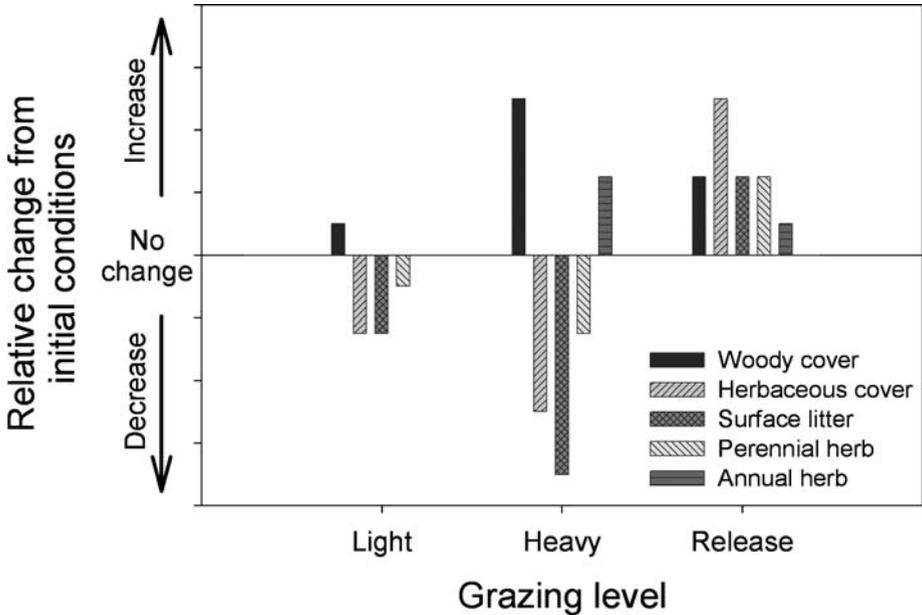


Figure 10 Commonly reported responses of five-dryland ecosystem structural properties to light and heavy grazing and to release from grazing. Bars show relative, directional responses as reported in the literature.

Biogeochemistry

Woody encroachment reduces the quality of land for animal production (39), yet in some cases, it enriches total ecosystem C and N stocks (Figure 7). As previously discussed, the shift from herbaceous to woody vegetation is different from desertification in that the woody cover increases without a major loss of herbaceous cover. With the shift to woody vegetation comes a large increase in aboveground NPP and C storage. Increases in aboveground NPP of up to $1400 \text{ kg C ha}^{-1} \text{ year}^{-1}$ have been observed when the dominant woody species is a nitrogen fixer (48, 90). Increases in the aboveground C pool can range from 300 to $44,000 \text{ kg C ha}^{-1}$ in less than 100 years of woody encroachment (44). When the dominant woody species is a nitrogen fixer, nitrogen accumulation can be $9\text{--}40 \text{ kg N ha}^{-1} \text{ year}^{-1}$ greater in the woody areas than the grasslands (48), with aboveground nitrogen increasing $39\text{--}468 \text{ kg N ha}^{-1}$ following encroachment (46).

Despite an increase in aboveground C and N with encroachment, the trends in soil organic C and N are highly variable. Measuring soil carbon to 3 m depth, Jackson et al. (45) found that woody encroachment increased soil C and N in drier grassland regions but reduced it in regions with mean annual precipitation greater than $\sim 500 \text{ mm}$. They found the decline in soil C in wetter ecosystems was sufficient to offset aboveground gains from woody encroachment, resulting in no net ecosystem C gain. In contrast, Boutton et al. (99) studied grasslands with

annual precipitation of 700 mm and found a 27% to 103% increase in soil C in the upper 10 cm. Likewise, Geesing et al. (48) measured soil organic C increases of 40% to 80% across a range of sites with mean annual precipitation of ~ 700 mm.

As with soil C, there is some uncertainty in how total soil N changes with woody encroachment. Jackson et al. (45) found trends similar to those observed for C, an increase in N stocks in drier regions and a decrease in wetter regions. In contrast, Hibbard et al. (47) found a clear increase in soil N in the top 10 cm of a wetter site, and Martin & Asner (49) measured an increase in soil N pools following 30–70 years of woody encroachment. Clearly, our understanding of how woody encroachment changes soil resources, and thus the long-term productivity and sustainability of semiarid regions, remains highly fragmented.

Trace-Gas Emissions

Woody encroachment has a measurable effect on soil nitrogen oxide emissions. In northern Texas rangelands, encroaching *Prosopis glandulosa*, a N-fixing species, caused C and N storage to increase in surface soils, which resulted in enhanced soil nitric oxide (NO) fluxes during nitrification (49, 50). Aboveground woody biomass was the best spatial predictor of NO emissions, with values increasing 20-fold ($0.04\text{--}0.78$ mg NO-N m^{-2} day^{-1}) across a 70-fold biomass gradient ($5\text{--}350$ g m^{-2}). Emissions also covaried with soil pH and clay content. Temporally, NO emissions and nitrification were positively correlated with temperature. Precipitation events elevated NO emissions fourfold over 24-hour periods and produced small amounts of N₂O. Overall, mesquite encroachment in these grasslands increased NO emissions in a spatially explicit manner determined by the woody biomass and soil type, which was then temporally mediated by temperature and secondarily by precipitation (49, 50).

At a regional scale, desertification and woody encroachment appear to have very different effects on the N status of dryland ecosystems. Desertification promotes nutrient accretion in soils under woody plant canopies, but the surrounding bare soil areas have very low-nitrogen contents. When the fractional covers of woody clusters with higher NO emissions and bare soils with low-NO emissions are taken into account, there is ~ 0.4 kg N ha^{-1} year^{-1} (53%) decrease in NO emissions regionally, in comparison to the preexisting grassland (40). In contrast, when woody encroachment involves an N-fixing species in a grassland, and when woody expansion does not decrease the herbaceous cover significantly, NO gas emissions increase beyond that of the original grasslands by ~ 1.0 kg N ha^{-1} year^{-1} or 29% (49).

Hydrology

Woody and herbaceous life-forms utilize soil moisture from different depths in the soil profile (100, 101). Woody plants take advantage of deeper soil moisture, and herbaceous plants access moisture only in the upper soil layers. It has often been theorized that variation in soil moisture, both vertically and horizontally, determines the relative fraction of woody and herbaceous cover (102). Grazing

affects soil moisture through compaction and reduced infiltration and through the exposure of bare soil surfaces. Increased evaporative losses from bare soils increase the disparity between shallow and deep soil moisture, and such losses support the notion that surface soil moisture is more sensitive to grazing intensity than deep soil moisture (103). These changes in soil moisture may drive changes in the relative balance between woody and herbaceous cover, favoring the deep-rooted shrub species.

Woody encroachment may be controlled by changes in the annual timing of precipitation (32) or the relative proportion of rainfall occurring in large precipitation events (100). Particularly in the presence of grazing, high-rainfall events or several months of elevated precipitation are effective in recharging deep soil layers, thereby creating soil moisture conditions that favor woody species. Modeling studies confirm the importance of temporal precipitation patterns in determining the relative abundance of herbaceous and woody plants, but grazing activity must be included for full transitions between plant life-forms in these models (32).

Climate Interactions

At the scale of individual canopies, shading, litter accumulation, and canopy interception of precipitation causes soils beneath woody vegetation to receive less solar irradiance and to have lower temperatures and water contents compared to inter-canopy areas (104, 105). However, recent work suggests that the albedo changes resulting from shrubland encroachment have been slight. For example, Grover & Musik (106) showed that mean daily albedo for a creosote bush shrubland and a grassland were not significantly different (0.2 and 0.27, respectively), and these did not differ from other creosote bush, mesquite, or grassland communities. They concluded that increasing spatial heterogeneity of woody cover does not significantly impact surface albedo and climate. In contrast, Hoffman & Jackson (107) attributed a decline in precipitation of approximately 10% to a reduction in roughness length and to an increase in albedo. They also found that deeper rooting had a small positive effect on latent heat flux, with a corresponding reduction in sensible heat.

In comparison to canopy-scale processes, the impacts of woody encroachment on regional climate variables are virtually unknown. It is thought that the effects of woody encroachment on surface energy fluxes, critical to the formation of clouds and precipitation, are small relative to evaporative fluxes from other forms of land use (e.g., irrigated croplands). However, the spacing of vegetation in semiarid regions can affect the development of local wind circulations that contribute to cumulus cloud formation and precipitation (108).

DEFORESTATION

Ecosystem Structure

At first glance, the ecosystem structural changes caused by forest-to-pasture conversion might seem obvious. Biologically diverse, large-stature forest is cut (and often burned) in geometric patterns easily discernable from satellite imagery. This

is true, and in humid tropical regions, the “installed” herbaceous species are predominantly African grasses (33, 109). However, tropical pasture development is carried out across an enormous range of environmental, social, and economic conditions, leaving the installed pasture in an equally broad number of biophysical states. In the humid tropics, the biophysical structure of pastures ranges from highly managed monospecific grasslands to savanna-like systems containing varying densities of palm and secondary forest species. In a study of 145 pastures in the central Amazon, Asner et al. (110) found that shrub and secondary forest vegetation cover ranged from 0% to 45%, and palms covered 20% to 60% of pasture areas. Although variation in pasture cover causes concomitant variation in biogeochemical, atmospheric, and hydrological processes (see below), vegetation structure is rarely quantified in published studies. Therefore, our synthesis implicitly incorporates the complicating effects of structural variability in pastures, but it does not determine the effects of this variation.

Biogeochemistry

In forests and woodlands, the largest impacts of grazing result from the conversion of the system to herbaceous cover, and deforestation for cattle pasture occurs across a wide range of climatic conditions. We discuss both dry and wet tropical forest conversion, where future conversion is most likely to take place on a large scale. With tropical dry forest conversion followed by repeated burning, up to 90% of aboveground C and N stocks and nearly 50% of the aboveground phosphorus (P) can be lost (111). C, N, and P are volatilized, whereas calcium (Ca) and the remaining C, N, and P are deposited as ash, much of which is lost in wind erosion immediately after the fire. Water erosion and soil loss can be significant for a few months after fire (112), and concentrations of dissolved mobile nutrients [e.g., NO_3 and potassium (K)] can be substantially elevated in overland flow (113). For a short period after burning, large increases in available soil N and nitrification rates are observed because NH_4 is mineralized during fire (114). Fire has little direct effect on total soil nutrients (C, N, and P); losses of these nutrients are mainly from the aboveground pools, and the size of these losses is small relative to the size of total soil nutrient pools (111, 115). However, it is estimated that it would take a century or more of recovery for a dry forest ecosystem to accumulate the nutrients lost during slash and burn (111).

With the global expansion of grazing systems, further conversion is most likely in humid forests. In contrast to dry forests, humid tropical forests are often found on highly weathered soils (ultisols and oxisols) rich in available N and poor in P and base cations [e.g., Ca and magnesium (Mg)], which are weathered and leached from the original rock (116, 117). Losses of P and Ca during deforestation are thus important to future pasture productivity (Figure 7). Many studies show that the initial burning of slashed primary forest results in combustion of ~48% of biomass, or 88 Mg C, 1181 kg N, and 107 kg sulfur (S) per hectare (52). The percentage of aboveground nutrients lost through combustion and transport of particulates average roughly 90% of N, 45% of P, and over 30% of Ca, Mg, and K (118). For

about a decade after conversion, pastures release significantly more N₂O (nitrous oxide) trace gas than the forest but then decline to background forest levels (see the Trace-Gas section). Following deforestation, soil pH and exchangeable cations remain elevated in pastures, but a recent synthesis showed that neither soil C nor N changed in a consistent manner (11, 119). However, a detailed reaccounting of changes in soil C that considered the effects of soil compaction and management practices suggests an average loss of 12 tons C ha⁻¹ in tropical lands maintained as pasture (53).

Repeated burning is often used as a management tool to remove woody regrowth and weeds and to renew nutrient availability in pastures (120). In the Brazilian Amazon, repeated burning of cattle pasture consumes up to 46% of aboveground biomass (slash, grass, and litter), with ~14 Mg C, 199 kg N, and 16 kg S per hectare lost to the atmosphere (52). Losses of P, Ca, and K in one experimental burn were nearly 33 kg ha⁻¹ (120). With repeated fires over a six-year period, over 1900 kg N ha⁻¹ is lost, which is equivalent to ~90% of the aboveground pool in a mature tropical forest. As a result, repeated burning can lead to N limitation even in previously nitrogen-rich tropical systems (121).

Results from the literature are inconsistent in describing how P changes with deforestation and repeating burning of pastures (11, 51, 118). Available P in ash after the initial fire may remain in the system, rapidly sorbing to soil minerals or consumed by vegetation and soil microbes. In many cases, available P seems to accumulate in soil organic material, which can remain enriched in pastures for years after conversion (11, 122–124). Others do not find the missing available P in the soil (51).

With long-term grazing (e.g., decades), total soil P usually declines; Ca, K, and Mg often decline as well; and N may also decline if there is frequent burning (44, 118, 125). The mechanisms for P loss are uncertain but include combustion, erosion of ash, and leaching to deep soils. Another possibility is that P is transferred into a bound form that is not easily detected with methods currently used. In any case, available phosphorus declines and grass productivity in pastures is often P limited. Phosphorus fertilization (~50 kg ha⁻¹ every 5–10 years) is thus becoming more common in some regions, such as the eastern Brazilian Amazon (121, 126). Despite these recent and more localized trends, many studies show that tropical pastures accumulate and cycle fewer nutrients than forest, redistribute cations from trees to soils, lose most of the C and N that was stored in aboveground forest biomass, and maintain reduced soil nutrient availability (Figure 7). These reported trends are induced and mediated by poor edaphic conditions inherent to many humid tropical forest regions.

Trace-Gas and Aerosol Emissions

Tropical forest soils are the largest biogenic source of N₂O, accounting for 25% to 50% of the global source (59). NO emissions from tropical forests are also significant, accounting for up to 20% of global emissions (127). The disturbance and initial volatilization of nutrients during burning associated with forest-to-pasture

conversion increases trace-gas emissions for months to years, but emissions often return to or drop below initial forest or savanna levels as pastures age. Studies in Costa Rica demonstrated that soil N_2O emissions and CH_4 production were higher for 15 years following conversion, but returned to background levels after 18 years (128). In a synthesis, Davidson et al. (129) found for a variety of tropical sites in Costa Rica, Puerto Rico, and Brazil that old tropical pastures produce consistently lower NO fluxes than old-growth tropical forests. The reasons for these differences were changes in the environmental factors that control N oxide emissions, such as soil water, temperature, nutrient status, pH, diffusion, and plant biomass.

Biomass burning is a common tool for the establishment and management of pastures for grazing, most recognized in tropical rain forests and savanna regions but also in grasslands worldwide (130, 131). Biomass burning is a significant source of globally relevant trace gases (CO_2 , NO_x , CO , and CH_4) and aerosols (130, 132). Climatological effects include the formation of photochemical smog, hydrocarbons, and NO_x that rapidly produce O_3 . Long-range transport of smoke plumes may be redistributed locally, transported throughout the lower troposphere, or entrained in large-scale circulation patterns in the mid and upper troposphere. The perturbation of these gases to the atmosphere is evident in satellite observations of high- O_3 and $-\text{CO}$ levels over large areas of Africa, South America, and the tropical Atlantic and Indian Oceans (133).

Pyrogenic aerosols from pasture biomass burning dominate the atmospheric concentration of aerosols over the Amazon basin and Africa (132, 134). Concentrations of aerosol particles are highly seasonal, with a clear maximum in the dry (burning) season, contributing to cooling both through increasing atmospheric scattering of incoming light and the supply of cloud condensation nuclei (CCN). High-CCN concentrations from biomass burning stimulate rainfall production and affect large-scale climate dynamics (135). The cooling effect of smoke alone may be minimal ($-0.3 \text{ watts m}^{-2}$) compared to the heating from anthropogenic greenhouse gases [$2.45 \text{ watts m}^{-2}$ globally (136)].

Hydrology

The conversion of forest to pasture is the primary driver of grazing-induced hydrologic change in mesic to wet climate zones (137). The majority of research conducted in tropical environments is in regions where deforestation has occurred recently and is likely to continue (5). Higher surface albedo, lower surface aerodynamic roughness, reduced LAI, and shallower rooting depths combine to reduce ET in pastures relative to forests. This model is supported by stable isotope studies, which reveal that water vapor above forest sites is derived from plant transpiration, in the eastern Amazon, whereas water vapor above pasture is derived primarily from surface evaporation (138). Lower ET and reduced infiltration rates within pastures culminate in increased average long-term discharge (139). Where changes in infiltration rates are modest, reduced ET alone can increase rainy season runoff (140). Decreases in dry-season flow are also theorized to result from deforestation,

but evidence of this effect is limited (141). Initial increases in stream discharge decline over time as forests regenerate (137, 142).

Climate Interactions

Forest-to-pasture conversion causes substantial decreases in land-to-atmosphere moisture transport. Early global-scale numerical simulations, using drastic levels of deforestation, predicted a 25% decrease in precipitation associated with a 30% decrease in evapotranspiration and a 2.5°C increase in surface temperature (143). However, recent mesoscale modeling studies suggest that, in spatially complex mosaics of forest and pasture lands, moisture fluxes from the land to the atmosphere can be enhanced, as can reciprocal fluxes of precipitation (144). Recycling of water by up to 25% to 35% was attributed to interactions between increased albedo, sensible heat flux, and mixed-layer height in pastures relative to forest (144, 145). Modeling studies also suggest that changes in surface energy dynamics can affect the upper atmosphere, perturbing tropical circulation patterns, shifting the position of the Hadley circulation, and altering planetary waves that propagate moisture to upper- and midlatitudes (146). Nonetheless, the effects of managed grazing and deforestation on climate, like that of desertification and woody encroachment, are poorly understood.

ECOSYSTEM RESPONSES TO MANAGED GRAZING

Response Typologies

Managed grazing has flourished for thousands of years, but the spatial extent and intensity (e.g., stocking rates) of grazing systems have increased substantially in the past several decades to centuries. The typologies of ecosystem response to managed grazing are regional in nature because they vary with bioclimatic and edaphic conditions. Not all grazing systems or practices lead to the syndromes of desertification, woody encroachment, and deforestation highlighted in this synthesis. We have a poor understanding of how much land has been affected by these phenomena. Nonetheless, scientific reports tend to address common environmental concerns, and these are the phenomena that often emerge as identifiable syndromes of managed grazing.

In arid regions, ecosystem responses to grazing practices are mediated by extreme climatic conditions combined with nutrient-poor soils. Low and highly variable precipitation causes ecological boom-bust cycles (Figure 4b), and yet managed grazing occurs on the most variable and climatologically marginal portions of these biomes (Table 1). When grazing systems are implicated in cases of desertification, thresholds in ecosystem resistance and resilience to drought are often crossed owing to the persistence of grazing at the worst of times climatologically (27, 147). The results are long-term losses of surface herbaceous cover, increases in bare soil extent, and, at times, increases in woody shrub cover (Table 3). A cascade of biogeochemical and hydrological feedbacks then takes place, such as nutrient

TABLE 3 Relative effects of land-cover change due to grazing on land surface properties that mediate biosphere-atmosphere interactions. Larger arrows indicate the dominant change within the ecosystem syndromes listed

	Deforestation	Woody proliferation	Desertification
Albedo	↑	↓	↑
Roughness length	↓	↑	↓
Turbulence	↑	↑	—
Vegetated fraction	↓	↑	↓
Evaporation	↓	—	↑
Transpiration	↓	↑	↓

losses via runoff and wind erosion, soil compaction, reduced soil water infiltration and increased patchiness of soil moisture (Table 4). Structural, biogeochemical and hydrological processes change to the point where an alternative “stable” state ecosystem then persists (15, 148). Additional feedbacks to the atmosphere result from changing albedo, surface temperature, and trace-gas emissions (Table 3). The scientific community is only beginning to understand the potential effects of desertification at the ecosystem-atmosphere interface. Whether these changes alter regional climate in ways that may enhance or dampen the effects of desertification on the global climate system is not known.

In semiarid and mesic biomes, woody encroachment is a widely reported ecosystem response to managed grazing (Table 4); however, other critically important cofactors are climate variability, fire suppression, and the presence of woody plant seed sources (Figure 9). Fire suppression favors woody seedling recruitment and survival, especially when grazers are actively consuming the herbaceous layer. In contrast to desertification, which appears to entail a combination of extreme climate variability and heavy grazing, even light-to-moderate grazing intensities can promote woody encroachment in semiarid and mesic environments (86, 149) (Figure 10). Like that of desertification, the spatial heterogeneity of aboveground and belowground resources such as vegetation cover and soil organic matter often increases. However, woody vegetation increases tend not to be well linked to large-scale losses of herbaceous cover. Biogeochemical responses include soil compaction, increased carbon storage and nutrient stocks (especially when nitrogen fixing woody species are implicated), and increased greenhouse gas emissions. The hydrology of these systems shows a typological response as well, with increased heterogeneity of soil moisture with depth, increased runoff and erosion, and decreased soil water infiltration (Table 3). Virtually nothing is known regarding the effects of woody encroachment on the climate system.

The major effects of managed grazing in humid forest regions are determined by deforestation and pasture maintenance (Table 4). Forests are replaced by herbaceous systems, which decrease ecosystem carbon storage and nutrient stocks

TABLE 4 Summary of most commonly reported changes in ecosystem properties for desertification, woody encroachment, and deforestation

Ecosystem properties	Regional syndrome		
	Desertification	Woody encroachment	Deforestation
Vegetation structure	<ul style="list-style-type: none"> • Decreased herbaceous, increased bare soil • Increased woody cover • Increased spatial heterogeneity 	<ul style="list-style-type: none"> • Increased woody cover • Increased spatial heterogeneity 	<ul style="list-style-type: none"> • Increased herbaceous • Decreased forest cover
Biogeochemistry	<ul style="list-style-type: none"> • Increased spatial heterogeneity of nutrients and carbon • Increased nutrient loss via runoff and erosion • Decreased soil nutrient stocks • Soil compaction 	<ul style="list-style-type: none"> • Increased carbon storage • Increased soil nutrient stocks, when N-fixing plants are present and active • Soil compaction 	<ul style="list-style-type: none"> • Decreased carbon storage • Decreased soil nutrient stocks over time • Nutrient losses via burning and aerosol
Biosphere-atmosphere exchange	<ul style="list-style-type: none"> • Increased albedo • Increased surface temperature • Increased ammonia and methane production 	<ul style="list-style-type: none"> • Increased N trace-gas production • Increased ammonia and methane production 	<ul style="list-style-type: none"> • Changed surface energy budget • Short-lived increase in soil N trace gas emission • Increased ammonia and methane production • Aerosol production
Hydrology	<ul style="list-style-type: none"> • Increased spatial heterogeneity of soil moisture • Increased runoff and erosion • Reduced infiltration 	<ul style="list-style-type: none"> • Increased vertical heterogeneity of soil moisture • Increased runoff and erosion • Reduced infiltration 	<ul style="list-style-type: none"> • Decreased transpiration • Decreased infiltration • Increased temporal variation in streamflow

(Table 3). Nutrient losses and greenhouse gas emissions may persist in cases of repeated burning of pastures. Substantial hydrologic changes occur via decreased plant transpiration, decreased soil infiltration, and increase variability in runoff and stream flow (Table 3). In humid tropical systems, the global dominance of nutrient-poor ultisol and oxisol soils (Figure 4c, Table 2) are widely implicated in the decline of productivity in ranching systems (53, 109). Tropical deforestation has been a regional syndrome largely driven by nutrient-poor soils, underlying a social and political demand for cattle production (150). Climatological impacts of deforestation have been heavily studied, suggesting overall decreases in continental-scale precipitation but possible increases in rainfall at the landscape-to-regional levels.

As the human population grows and land scarcity increases, intensification and shifts in traditional animal production are occurring globally. Traditional extensive pastoral systems are declining in some arid regions, and integrated pastoral farm management is disappearing in tropical highlands. These are being replaced by more concentrated grazing systems that can lead to greater degradation of pasture land (151). In the northeastern United States, northwestern Europe, and densely populated areas of Asia, animal production has become mechanized and dependent on external fertilizer and feed inputs. Industrial meat production is growing rapidly. From 1991 to 1993, it provided 37% of global meat production and 43% by 1996 (152). In these systems, animal waste exceeds the absorptive capacity of the land and pollutes the surrounding environment (153). Excess nitrogen and phosphorus from livestock is a substantial source of nonpoint pollution in the United States, causing eutrophication of freshwater and marine ecosystems, toxic algal blooms, and fish kills (154). Excess nitrogen in the environment can have many negative impacts on human health (155). Other biogeochemical impacts of intensification are only starting to be studied and quantified.

Knowledge Gaps and Research Needs

We contend that a combination of three regional syndromes—desertification, woody encroachment, and deforestation—represents a major component of global environmental change promoted and mediated by managed grazing activities. There is a rich literature on the ways that cropland expansion and intensification have altered both terrestrial and aquatic ecosystems (156, 157). There are also clear effects of urban and suburban land use on regional ecological dynamics, and the role of urbanization on climate is now being recognized at the global scale (158). Managed grazing systems cover more of the Earth's surface than any other form of land use, yet pastoral operations are spatially diffuse and natural looking (except in the case of deforestation). Nonetheless, managed grazing has resulted in typological responses of ecosystems during the past few centuries if not beforehand. These responses can be organized by climate-edaphic conditions on a regional basis, and in doing so, the global footprint of grazing systems appears to be quite large. This synthesis helps stitch together this globally relevant land use into a single framework for further study and perspective.

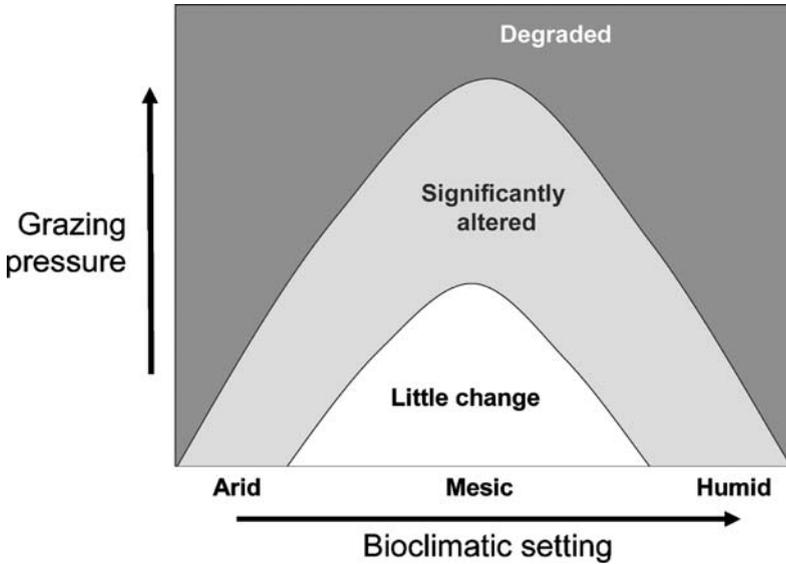


Figure 11 The grazing intensity at which ecosystems become degraded or are significantly altered changes with bioclimatic setting, with greater impacts and lower sustainability in very dry and very humid ecosystems.

As the result of the emergent themes from our synthesis, we hypothesize that ecosystem responses to grazing vary along a bioclimatic gradient from arid to mesic to humid environments (Figure 11). The literature suggests that mesic environments, such as in temperate grasslands, can endure the highest grazing intensities (19). At some point, however, changes in the structure, biogeochemistry, and hydrology of these systems occur, and the degree of such changes will be mediated by management practices. Arid environments are predisposed to changes in ecosystem structure, biogeochemistry, and hydrology in ways that truly degrade the land, even at relatively low grazing pressures. Humid forest ecosystems (especially tropical) are immediately and significantly altered by grazing systems, and degradation of pastureland often occurs under poor edaphic conditions.

These hypotheses are well supported by the scientific literature, but our overall understanding of ecosystem responses to managed grazing remains somewhat diffuse and fragmented for several reasons. The extent and intensity of managed grazing operations are very poorly known at regional, continental, or global scales. The GIS maps presented and synthesized here are state of the art, yet we recognize the generality and inaccuracy of them, especially in regions, such as the Indian subcontinent, the U.S. southwest, South America, and northern Mexico. Remote sensing technologies cannot identify grazing lands without going to very

high-spatial and -biogeophysical resolution (37, 44, 80). Such approaches are impractical at the global scale, but a strategic global sampling would be tractable if we knew how to stratify such a sampling, using process-level knowledge of grazing systems and the resulting regional syndromes.

Our lack of process-based knowledge not only impedes our ability to make the appropriate observations for analysis and monitoring, but it also limits our predictive capabilities. We are currently unable to forecast the onset of desertification, woody encroachment, or even deforestation because we lack the approaches to understand the interaction between ecological, climatological, and socioeconomic factors. Recent synthetic work has made some progress in this regarding desertification (159) and deforestation (6), but the basic observations of when, where, and under what conditions these regional syndromes occur are still lacking. Even remote sensing studies of tropical deforestation produce wide-ranging estimates of the amount of pastureland emplaced annually (5, 12). In sum, the observations are limited by technological barriers, whereas the scientific understanding of grazing systems and global change are limited by insufficient observations. Research and progress are needed on both fronts to better understand the role of managed grazing in the global environment.

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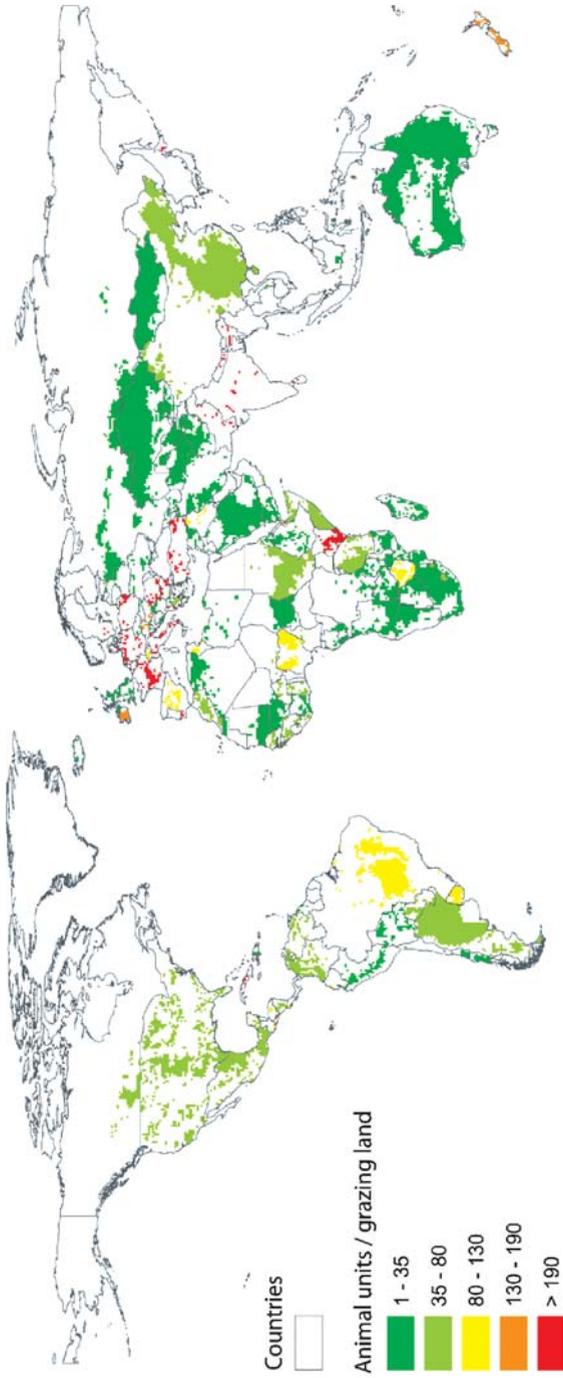


Figure 1 The present global distribution of grazing extent and intensity (stocking rates), derived by combining References 1 and 2.

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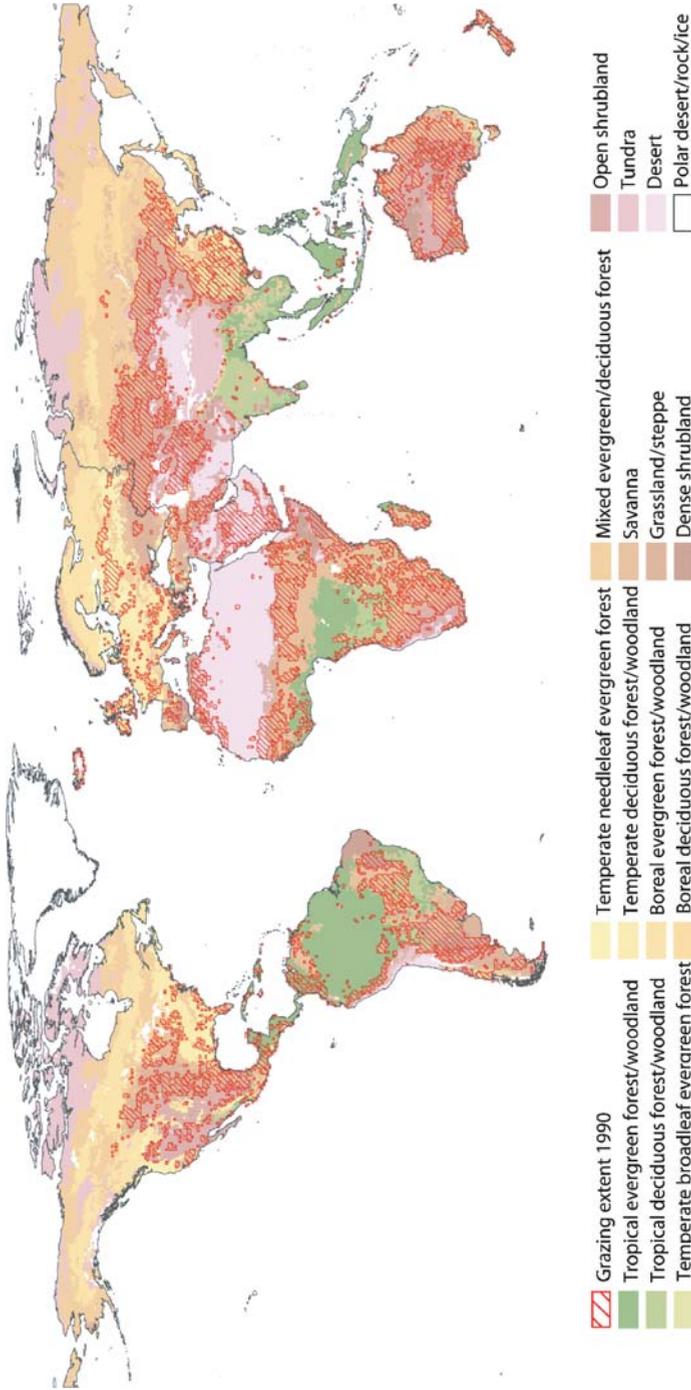


Figure 3 Global distribution of managed grazing systems overlaid on the global distribution of biomes (1, 160).

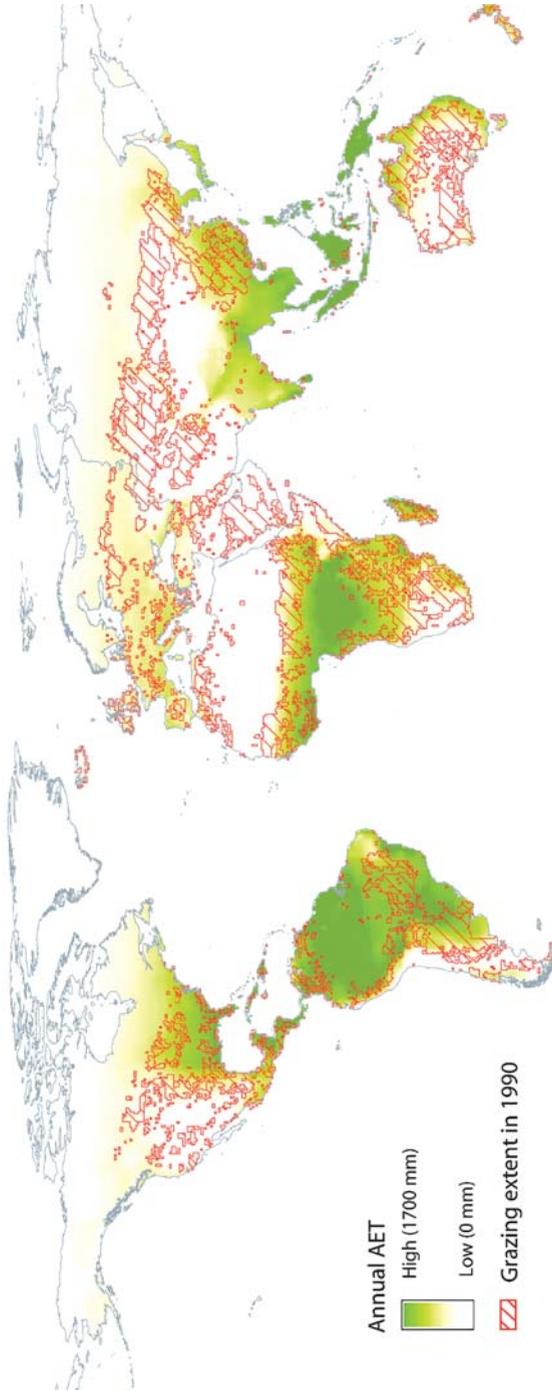


Figure 4a Global distribution of managed grazing systems and actual evapotranspiration (AET), a metric of bioclimatic stress. Derived by combining References 1 and 8.

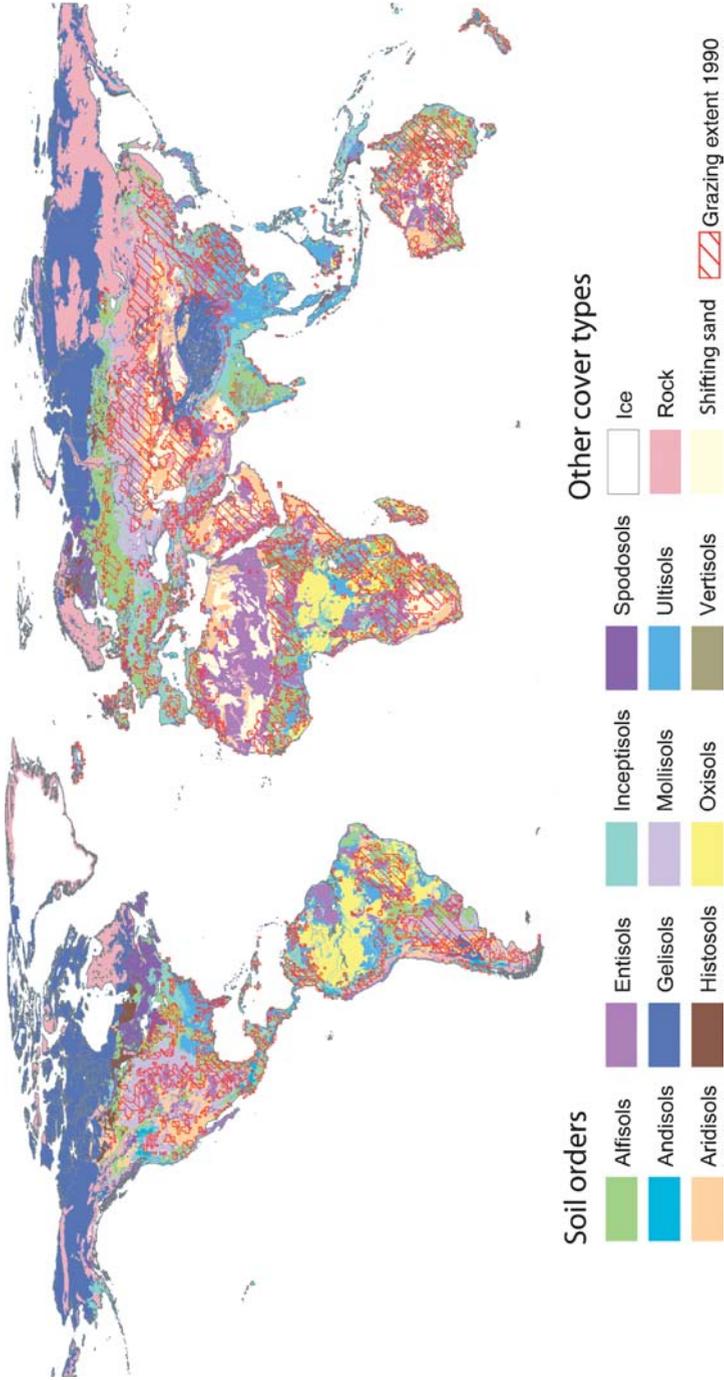


Figure 4c Global distribution of managed grazing systems and soil taxonomic order. Derived by combining References 1 and 10.

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