CHAPTER

3

Brush Management as a Rangeland Conservation Strategy: A Critical Evaluation

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INTRODUCTION

Rangelands support the majority of the world's livestock production (Safriel and Adeel 2005) and play an important role in human health and global carbon, water, and nitrogen cycles (Campbell and Stafford Smith 2000). Their extensive airsheds and watersheds provide habitat for game and nongame wildlife and myriad ecosystem goods and services important to rapidly growing settlements and cities that may be geographically distant. Rangelands thus have considerable, multidimensional conservation value. Stewardship of vegetation composition, cover, and production is the foundation of sustainable rangeland management, a key component of which is maintaining vegetation within a desirable mix of herbaceous and woody plants (WPs).

One of the most striking land cover changes on rangelands worldwide over the past 150 yr has been the proliferation of trees and shrubs at the expense of perennial grasses. In some cases, native WPs are increasing in stature and density within their historic geographic ranges; in other cases, nonnative WPs are becoming dominant. These shifts in the balance between woody and herbaceous vegetation represent a fundamental alteration of habitat for animals (microbes, invertebrates, and vertebrates) and hence a marked alteration of ecosystem trophic structure. In arid and semiarid regions, increases in the abundance of xerophytic shrubs at the expense of mesophytic grasses represent a type of desertification (e.g., Schlesinger et al. 1990; Havstad et al. 2006) often accompanied by reductions in primary production (Knapp et al. 2008a) and accelerated rates of wind and water erosion (Wainwright et al. 2000; Gillette and Pitchford 2004; Breshears et al. 2009). In semiarid and subhumid areas, encroachment of shrubs and trees into grasslands and savannas may have neutral to substantially positive effects on primary production, nutrient cycling, and accumulation of soil organic matter (Archer et al. 2001; Knapp et al. 2008a; Barger et al. 2011). While impacts of WP encroachment may vary among bioclimatic zones, there is one constant: grass-dominated ecosystems are transformed into shrublands, woodlands, or forest. As such, WP encroachment represents a threat to grassland, shrub-steppe, and savanna ecosystems and the plants and animals endemic to them, a threat on par with those posed by exurban and agricultural development (Sampson and Knopf 1994; Maestas et al. 2003).

Efforts to counteract the real and perceived threats of WP encroachment fall into the broad category of brush management. Brush management, defined by the Natural Resource Conservation Service (NRCS 2003) as the removal, reduction, or manipulation of nonherbaceous plants, has been an integral component of range management since its formal emergence in the 1940s. However, brush removal has historically been criticized, especially when large-scale programs have failed to consider the needs of diverse stakeholders and the impact on multiple goods and services during planning and implementation stages (e.g., Klebenow 1969; Belsky 1996).

Our goal here is to provide a contemporary, critical evaluation of "brush management" as a conservation tool. We begin with a brief review of potential drivers of WP encroachment. An understanding of these drivers will 1) shed light on the causes for the changes observed to date; 2) help us determine if management

Woody plant encroachment has been widespread in rangelands, including these desert grasslands in New Mexico. (Photo: Paolo D'Odorico)

TABLE 1. Potential causes for increases in woody plant (WP) abundance in rangelands. There is likely no single-factor explanation for this widespread phenomenon. Most likely, it reflects drivers that vary locally or regionally or from the interactions of multiple drivers. Changes in a given driver may be necessary to tip the balance between woody and herbaceous vegetation but may not be sufficient unless co-occurring with changes in other drivers. For detailed reviews and discussions, see Archer (1994), Archer et al. (1995), Van Auken (2000), Briggs et al. (2005), and Naito and Cairns (2011).

Driver	Mechanism	Potential vegetation response
Climate	Increased precipitation	Enhances WP establishment, growth, and density
	Decreased precipitation	Promote shifts from mesophytic grasses to xerophytic shrubs
	Shift from winter to summer precipitation	Favors WP over grasses, particularly on relatively deep, well-drained soils
Grazing	Utilization of grasses by livestock	Herbaceous production and species composition may shift to a community more susceptible to WP encroachment; livestock are effective agents of dispersal of some WP species; reductions in fine fuel mass and continuity (see "Fire")
	Seed dispersal	
Browsing	Reduced utilization of WPs by native herbivores	Elimination of browsers promotes WP recruitment and growth; WPs kept small in size by browsers more susceptible to fire
Fire	Reduced fire frequency, intensity, and extent	Increased WP recruitment and growth (see "Grazing")
Atmospheric CO_2	Increased atmospheric CO_2 concentrations	WPs with C_3 photosynthetic pathway may be favored over grasses with C_4 photosynthetic pathway
Nitrogen deposition	Increased N availability	Correlated with forest expansion into grassland

intervention is realistic; if so, 3) what approaches might be most effective; and 4) when, where, and under what conditions to apply them. We then discuss the ecological role of WPs in rangeland ecosystems and how human perspectives on WPs in rangelands influence management decisions and conservation objectives. The ecological impacts of WP proliferation are then reviewed with the aim of addressing the question, What are the environmental consequences of not managing WPs in rangelands? As it turns out, there are indeed consequences. Many of these have emerged relatively recently and hence are not yet reflected in current management guidelines. Advances in our understanding of the ecological consequences of WP proliferation in rangelands have paralleled changes in both perspectives on and approaches to brush management since the mid-1900s and have influenced how the NRCS has advised landowners. We therefore review the evolution of brush management in the spirit of putting current perspectives into their historical context. The basis for NRCS expectations underlying recommendations in the NRCS Brush Management Conservation Practice Standard matrix (hereafter described as "projected effects") is then evaluated on the basis of a pooling of expectations into five overarching areas: herbaceous cover, production, and diversity; livestock response; watershed function; wildlife response; and fuels management. Evaluations are then followed by recommendations, an itemization of knowledge gaps, and a series of overarching conclusions.

WHY HAS WP ABUNDANCE INCREASED ON RANGELANDS?

Understanding the drivers of tree/shrub encroachment can help identify when, where, how, and under what conditions management might most effectively prevent or reverse WP proliferation. Traditional explanations center around intensification of livestock grazing, changes in climate and fire regimes, the introduction of nonnative woody species, and declines in the abundance of browsing animals (Table 1). Historical increases in atmospheric nitrogen deposition and atmospheric carbon dioxide concentration are also potentially important drivers. Exploring this important question is beyond the scope of this discussion, but detailed reviews and discussion can be found in Archer (1994), Archer et al. (1995), Van Auken (2000), Briggs et al. (2005), and Naito and Cairns (2011). Likely all these factors have interacted to varying degrees, and the strength and nature of these interactions likely varies from one biogeographic location to another. Thus, local knowledge is important in developing WP management plans. In many respects, WP encroachment is a specific case of weed and invasive plant management, and the concepts and principles developed for those perspectives are widely applicable (Sheley et al. this volume).

It is important to note that once the process of WP encroachment is set in motion, grazing management per se may do little to prevent the conversion of grasslands and savannas to shrublands and woodlands (e.g., McClaran 2003; Browning et al. 2008). In fact, on sites with a long history of heavy grazing, removal of livestock may actually promote rather than deter WP encroachment (Smeins and Merrill 1988; Browning and Archer 2010). However, grazing management influences on WP encroachment are indirectly important in terms of how they affect the amount and continuity of fine fuels available for wildfire or prescribed burning (Fuhlendorf et al. 2008; Fuhlendorf et al. this volume). Because grazing management alone is generally not sufficient to curtail or reverse shrub encroachment, progressive brush management is a potentially important tool for grassland conservation.

Although WP encroachment has been formally documented and qualitatively observed in some areas, it should not be assumed that this transformation has been uniform or ubiquitous. Indeed, repeat ground photography in western North America documents areas where WPs have dominated landscapes since the 1800s (e.g., Humphrey 1987; Turner et al. 2003; Webb et al. 2007). Thus, many areas may have been historically comprised of mixtures of woody and herbaceous vegetation (e.g., shrub-steppe or shrub or tree savannas), and efforts to eradicate WPs from such sites may be misguided (McKell 1977) and sometimes detrimental to native plants and wildlife (e.g., Knick et al. 2003).

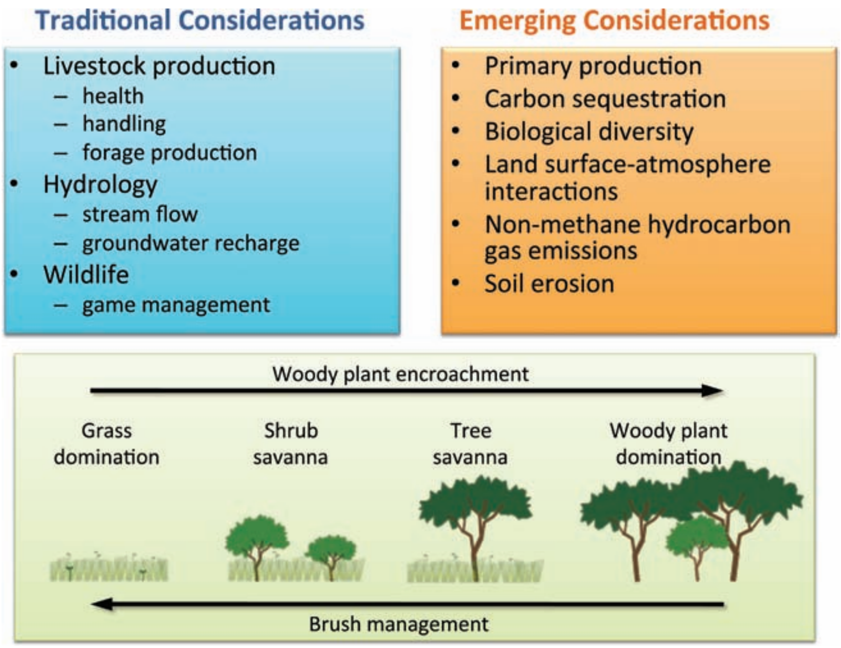
PERSPECTIVES ON WPs IN RANGELANDS

Brush management practices have historically focused on the goal of maximizing livestock production and promoting groundwater recharge and stream flow. Contemporary perspectives have been broadened to include impacts on biological diversity, ecosystem function (primary production and nutrient cycles), and land surface–atmosphere interactions (Appendix 2; Fig. 1). These broader perspectives are recognized to varying levels of specificity in NRCS Brush Management Conservation Practice Standards (code 314) and its projected effects. The current challenge lies with articulating these more explicitly in the CPPE worksheet, exposing landowners and the public to these perspectives, and articulating these perspectives in terms that can be quantified and objectively monitored and evaluated.

TABLE 2. Perspectives on woody plants (WPs) in rangelands. In areas subject to heavy livestock grazing, palatable species typically give way to less palatable, less preferred species, and in rangelands, these less palatable species are often shrubs. The fact that unpalatable shrubs dominate many grazed rangelands has led to the mistaken generalization that all WPs in rangelands are undesirable. WPs have been typically viewed as the problem on grazed rangelands, but in fact they are likely a consequence of past mismanagement. Brush management conducted in isolation of grazing management is therefore treating symptoms rather than addressing the root causes of the problem (excessive grazing and fire suppression). When assessing whether to invest in efforts to reduce WP cover or density, the points shown in the table should be considered. For further discussion, see McKell (1977), Archer and Smeins (1993), and Archer (2009).

• Palatable WPs may have been displaced along with palatable grasses and herbs (Lange and Willcocks 1980; Orodhu et al. 1990; Kay 1997).
• Shrubs may decrease grazing pressure on grasses and provide protection for heavily utilized herbaceous species.
• WPs may provide important habitat for a variety of vertebrate and invertebrate wildlife (nongame as well as game).
• Shrubs may provide an important and underappreciated source of nutrimental stability and reduce supplemental feed requirements during cold or dry periods (Le Houérou 1980; Coppock et al. 1986; Stuth and Kamau 1990; Styles and Skinner 1997).
• WPs may be the best adapted for the prevailing environmental conditions (Le Houérou 1994).
• Were it not for the “damn brush,” there might be little or no vegetative cover. It may not be realistic to expect brush management to enhance herbaceous production, especially where soils have extensively eroded.
• Will brush management stimulate herbaceous production and increase livestock carrying capacity sufficiently to offset treatment costs? If so, how much time will be required before a follow-up treatment? Will treating one problem perhaps create another (i.e., loss of valuable nontargeted species, invasion by weeds or exotic species, induced multiple-stemmed growth habit in shrubs, or replacement of nonsprouting species with sprouting species)?

FIGURE 1. Traditional and contemporary perspectives on woody plant (WP) encroachment (from Archer 2009).



THE COST OF DOING NOTHING

Changes in WP cover and density represent fundamental changes in vegetation composition and structure and animal (microbes, invertebrates, and vertebrates) habitat. These, in turn, can fundamentally alter ecosystem primary production, trophic structure, biological diversity, nutrient cycling, and land surface–atmosphere interactions (Fig. 2).

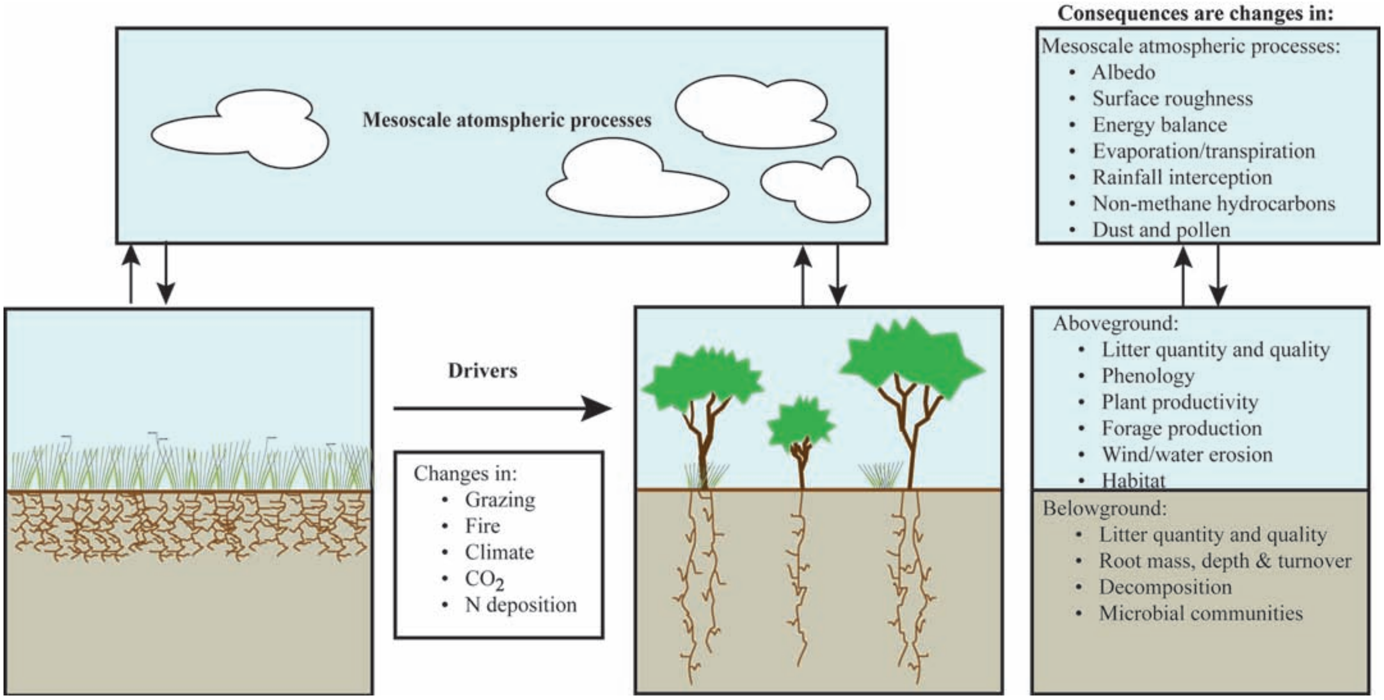
Herbaceous Cover and Production

The projected effects of brush management typically assume that herbaceous cover and production will increase following brush management. Implicit in this expectation is the assumption that WPs have a negative impact on ground cover. Does the literature support this perspective? The answer to this question is context dependent. Herbaceous cover and biomass typically decline as WP cover and basal area increase. However, the specific nature of the response ranges from an immediate linear or exponential decline to an initial stimulation, followed by a subsequent decline (Fig. 3; Table 3). The shape of these curves depends on the site and its grazing history, its climate, the physiology of the herbaceous vegetation (e.g., cool-season C_3 vs. warm-season C_4 grass), and the species of WP and its growth form (i.e., evergreen vs. deciduous), canopy architecture (i.e., single vs. multiple stemmed), size, density, and spatial arrangement (Jameson 1967; Mitchell and Battling 1991; Scholes and Archer 1997; Scholes 2003; Fuhlendorf et al. 2008; Teague et al. 2008a). When stocking rates are based on total area rather than grazable area, WP encroachment can intensify grazing pressure to further depress grass production unless stocking rates are adjusted to compensate for WP-induced losses of forage production. WP impacts on herbaceous plants must therefore be considered in the context of livestock management (Briske et al. this volume) and the ecological site(s) being managed.

It is important to note that relationships between WPs and grass are typically described at either the plant or the stand scale of spatial resolution. This can cause confusion and apparent contradictions. For example, velvet mesquite (*Prosopis velutina*) typically has neutral to positive effects on grasses at the plant scale but negative effects at the stand scale (McClaran and Angell 2007). Thus, care should be taken when generalizing results from a given study. Stand-scale assessments are generally most appropriate for pasture and landscape management.

How do declines in herbaceous cover and biomass that typically accompany WP encroachment impact overall ecosystem primary production? A recent comparison of sites around North America suggests aboveground primary production declines with WP encroachment in hot and cold deserts but that it increases dramatically as a function of annual rainfall in semiarid and subhumid regions (Knapp et al. 2008a). Recent estimates suggest that for every millimeter increase in mean annual precipitation above 330 mm, aboveground net primary production (ANPP) will increase by ~0.6 g $C \cdot m^{-2} \cdot yr^{-1}$ with shrub encroachment (Barger et al. 2011). Thus, losses of grass production can lead to a net decline in overall ecosystem production in arid areas, whereas increases in production attributable to WPs more than compensates for declines in herbaceous production in other bioclimatic zones.

FIGURE 2. Drivers of woody plant encroachment (see Text Box 1) and the potential consequences of ecosystem function and land surface–atmosphere interactions (from Archer 2009).



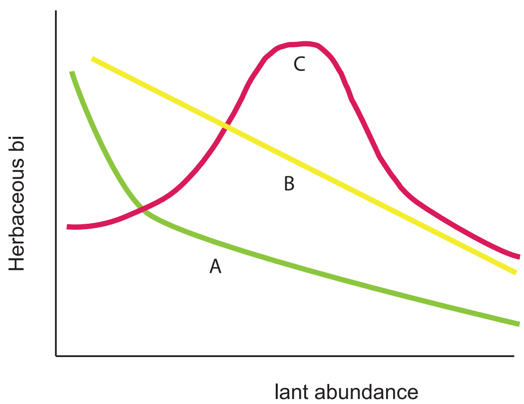
Soil Condition and Erosion

Projected effects generally assume that soil conditions and soil surface stability will be slightly to substantially improved by brush management and that soil erosion will be reduced by WP removal. Although not explicitly stated, these assumptions appear predicated on the expectation that WP proliferation adversely affects these parameters. Does WP encroachment lead to a deterioration of soil condition and site stability?

Changes in grass and WP abundance impact soils through alteration of above- and belowground productivity, quality of litter inputs, rooting depth and distribution, hydrology, microclimate, and energy balance (Fig. 2). The abundance of soil organic matter or, more precisely, soil organic carbon (SOC) is a good indicator of soil condition, as it integrates a variety of ecosystem processes that influence fertility, water-holding capacity, and site stability.

A substantial majority of the carbon in rangeland ecosystems resides in the SOC pool (Schlesinger 1997), but it is not yet clear how grazing, climate, and WP encroachment and "infilling" (shifts from relatively low to

FIGURE 3. Potential responses of herbaceous vegetation to increases in woody plant cover or basal area. See Table 1 and reviews by Jameson (1967), Mitchell and Battling (1991), Scholes and Archer (1997), Scholes (2003), Fuhlendorf et al. (2008), and Teague et al. (2008a).



relatively high WP cover or density) interact to affect gains and losses from these large carbon pools. Despite consistent increases in aboveground carbon storage with woody vegetation encroachment (Knapp et al. 2008a) and dryland afforestation (e.g., Nosetto et al. 2006), the trends in SOC are highly variable, ranging from substantial losses to large gains to

TABLE 3. Herbaceous response to shrub encroachment (US studies only).

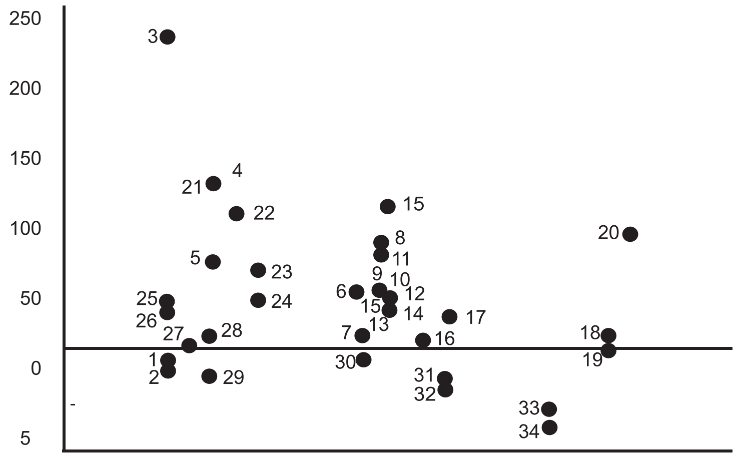
Dominant woody plant(s)		Herbaceous response[a]	Soils	MAP (mm)
Big sagebrush (*Artemisia tridentata*)		D^{1}	Aeolian sandy loams and loess (aridisols)	210
		A	Loam/sandy loam	225–405 (eight sites)
		D^{2}	Silt loam	406
		C	–	279
Broom snakeweed (*Gutierrezia sarothrae*)		A	Gravelly loam	323 (Vaughn) 328 (Roswell)
		B	Fine sandy loam	480
Creosotebush (*Larrea tridentata*)		B, D^{1}	Sandy loam	221–430 (4 sites)
		D^{1}	Gravelly/loamy	240
		A/B	Shallow sandy	
		C	–	255
		A	Gravelly	–
Honey mesquite (*Prosopis glandulosa*)		C	Clay loam	–
		A	Sandy loan	231
		B	Silt loam/clay loam	648
		B	Loamy sand	231
		B	Clay loams	665
		D^{2}	Shallow clay	
		C	Sandy loam	345
Huisache (*Acacia farnesiana*)		B (total); C (cool-season grass)	Clay	850
Juniper (*Juniperus* spp.)	**Redberry juniper (*J. pinchottii*)**	B	–	–
	Redberry juniper (*J. pinchottii*)	A (grazed); B (ungrazed)	Fine loam and clay	~525
	Western juniper (*J. occidentalis*)	B	Loams with variable rockiness	410
		D^{2}	Clay loam	430
		C	–	248
Pine (*Pinus* spp.)	**Ponderosa pine (*P. ponderosa*)**	A	–	–
	Ponderosa pine (*P. ponderosa*)	A	Limestone derived	560
	Ponderosa pine (*P. ponderosa*)	A	–	380–510 (31 sites, varies with elevation)
	P. taeda, P. echinata	A	–	–
	Longleaf pine (*P. palustris*)	B	Silty loam	~584
Pinyon–juniper (*Pinus edulis–Juniperus* spp.)		A	–	–
		D^{3}	–	345
Velvet mesquite (*Prosopis velutina*)		D^{2}	–	~330–432 (varies with elevation)
		C (one site); D^{2} (three sites)	–	~197–304 (varies with elevation)
		A	–	–

[a] As per Figure 3 (A, negative exponential decline; B, linear decline; C, initial positive response, followed by decline; and "D," "Other," including no change (D1), increase (D2), or decline in C4, increase in C3 (D3).

*Space-for-time substitution (sampling stands of different shrub abundance at one point in time).

	Scale (plant/ stand)	Study duration (yr)	Study location	Reference
	Plant	28 (1950–1978)	Southeastern ID	Anderson and Holte (1981)
	Stand	20	Northwestern NM	McDaniel et al. (2005)
	Stand	4 (1967–1970)	Southern ID	Hull and Klomp (1974)
	Stand	4 (1951–1954)	OR	Hyder and Sneva (1956)
	Stand	10 (1979–1989)	Vaughn and Roswell, NM	McDaniel et al. (1993)
	Stand	2 (1946–1947)	TX	Ueckert (1979)
	Stand	2* (1981–1982)	Chihuahua, Mexico, and southern AZ	Morton et al. (1990)
	Stand	14–17 (1984/1987–2001)	Southwest NM	Perkins et al. (2006)
	Stand	76 (1915–2001)	Central NM	Baez and Collins (2008)
	Stand	Model	South-central NM	Bestelmeyer et al. (2009)
	Stand	5 (1970–1975)	West TX	Dahl et al. (1978)
	Stand	105 (1858–1963)	South-central NM	Buffington and Herbel (1965)
	Stand	6 (1995–2001)	North-central TX	Teague et al. (2008a)
	Stand	45 (1935–1980)	Southern NM	Hennessy et al. (1983)
	Stand	1*1998	North-central TX	Hughes et al. (2006)
	Stand	46 (1957–2003)	Southern AZ	McClaran and Angell (2006)
	stand	2 (1978–1979)	South TX	Scifres et al. (1982)
	Stand	1*1995	Nolan County, TX	Johnson et al. (1999)
	Stand	3 (1984–1986)	Western TX	McPherson and Wright (1990)
	Stand	2* (2005–2006)	Northeastern CA	Coultrap et al. (2008)
	Stand	7 (1975–1982)	CA	Evans and Young (1985)
	Stand	13 (1991–2004)	Southeastern OR	Bates et al. (2005)
	Tree	1*	Northern AZ	Jameson (1967)
	Stand	1* (1988)	Northern AZ	Moore and Deiter (1992)
	Stand	1* (1984)	Rocky Mountain front range, CO and WY	Mitchell and Battling (1991)
	Stand	2* (1960–1961)	Eastern TX	Halls and Schuster (1965)
	Stand	11	Alexandria, LA	Grelen and Lohrey (1978)
	Stand	1*	Northern and central AZ	Jameson (1967)
	Stand	2 (1984–1985)	South-central NM	Pieper (1990)
	Stand	10 (nonsequential; 1954–1967)	Southern AZ	Cable (1971)
	Stand	5 (1945–1950)	Southern AZ	Parker and Martin (1952)
	Stand	—	Southeastern AZ	Upson et al. (1937)

FIGURE 4. Changes in soil organic pools following shrub encroachment in different precipitation zones (from Asner and Archer 2009). Numbers on symbols/legend entries indicate references listed in Appendix III.



no net change (Wessman et al. 2004; Asner and Archer 2009; Fig. 4).

Variation in SOC response to WP encroachment is perhaps not unexpected given the myriad factors that influence SOC pool and fluxes (Wheeler et al. 2007). These include growth characteristics of the WPs (e.g., evergreen or deciduous, N fixing or not, shallow or deep rooted, etc.), climate (mean annual rainfall and temperature), soil properties (e.g., texture, pH, carbonate content), initial conditions (e.g., amount, type, and distribution of SOC present at the time WP encroachment begins), and prior land management (e.g., whether WPs are establishing in native rangeland vs. abandoned cropland). In areas where shrub-induced increases in SOC have been documented, changes are typically restricted to the upper 10–20 cm of the soil profile, and accumulation appears to be a linear function of time since WP establishment (Boutton et al. 2009) with rates ranging from 8 g $C \cdot m^{-2} \cdot yr^{-1}$ to 30 g $C \cdot m^{-2} \cdot yr^{-1}$ (Wheeler et al. 2007). Some of the uncertainty in SOC response may reflect the fact that WP encroachment often occurs in areas with a history of livestock grazing, which itself has positive, neutral, and negative effects on SOC pools (Milchunas and Lauenroth 1993; Piñeiro et al. 2010; Briske et al. this volume). Where historical grazing and WP encroachment effects on SOC have been explicitly accounted for, it appears that losses of SOC associated with heavy grazing can be recovered subsequent to WP encroachment and that SOC in the shrub-dominated system can be substantially greater than that of the original grasslands (Archer et al. 2001; Hibbard et al. 2003).

SOC and N levels are typically highly correlated; hence, increases in SOC are typically accompanied by increases in soil N (Seastedt 1995; Wheeler et al. 2007). When the encroaching woody species is a nitrogen fixer, soil N levels can increase substantially (Geesing et al. 2000; Hughes et al. 2006), thus augmenting a key resource that, along with water, typically colimits rangeland productivity. The resultant increase in soil fertility and water-holding capacity likely drives the increase in herbaceous production that typically follows WP removal.

In arid regions, the loss of grass cover due to grazing is accompanied by loss and redistribution of soil resources from plant interspaces to areas beneath shrubs. Many studies have investigated this grass-erosion feedback, with the consensus that erosion by wind and water is capable of removing soil resources required for grass growth and propagation while creating semipermanent fertile islands beneath shrub canopies (see Okin et al. 2009). The net result is a dramatic increase in wind and erosion resulting from increased bare areas in shrublands compared to the grasslands they replaced. Aeolian sediment flux in mesquite-dominated shrublands in the Chihuahuan Desert are 10-fold greater than rates of wind erosion and dust emission from grasslands on similar soils (Gillette and Pitchford 2004). Flow and erosion plots in the Walnut Gulch Experimental Watershed in Arizona and the Jornada Long Term Ecological Research site in New Mexico have demonstrated significant differences in water erosion between grasslands and shrublands (Wainwright et al. 2000). For example, higher splash detachment rates (Parsons et al. 1991, 1994) and interrill erosion rates (Abrahams et al. 1988) are observed in shrublands compared to grasslands, and shrubland areas are more prone to develop rills, which are responsible for significant increases in overall erosion rates (Luk et al. 1993). Episodes of water erosion are often associated with decadal drought–interdrought cycles because depressed vegetation cover at the end of the drought makes the ecosystem vulnerable to increased erosion when rains return (McAuliffe et al. 2006). In hot desert systems where shrub encroachment has occurred, reestablishment of grass cover would help curtail erosion losses. However, the loss of topsoil to date, coupled with low and highly variable precipitation, make these among the most challenging environments in which to reestablish perennial grass cover once it has been lost (see the section "Herbaceous Vegetation and Native Communities").

Air Quality and Land Surface–Atmosphere Interactions

Brush management impacts on air quality are treated as "not applicable" or "neutral" with respect to particulate matter, ozone, and greenhouse gas production. However, these factors and others related to pollen production and land surface–atmosphere interactions may warrant more attention in the next generation of projected effects worksheets.

A synthesis of aeolian sediment transport studies spanning a grassland–forest continuum suggests 1) that among relatively undisturbed ecosystems, arid shrublands have inherently greater aeolian transport because of wake interference flow associated with intermediate levels of density and spacing of WPs and 2) that among disturbed ecosystems, the upper bound for aeolian transport decreases as a function of increasing amounts of WP cover because of the effects of the height and density of the canopy on airflow patterns and ground cover associated with WP cover (Breshears et al. 2009).

Pollen from WPs trigger nasal allergies and asthma (Chang 1993; Gutman and Bush 1993). Tree/shrub proliferation thus has the potential to influence the onset, duration, concentration, and total production of pollen allergens both locally and at great distances (Levetin 1998). However, the role of these allergens on human health is not well understood (Al-Frayh et al. 1999).

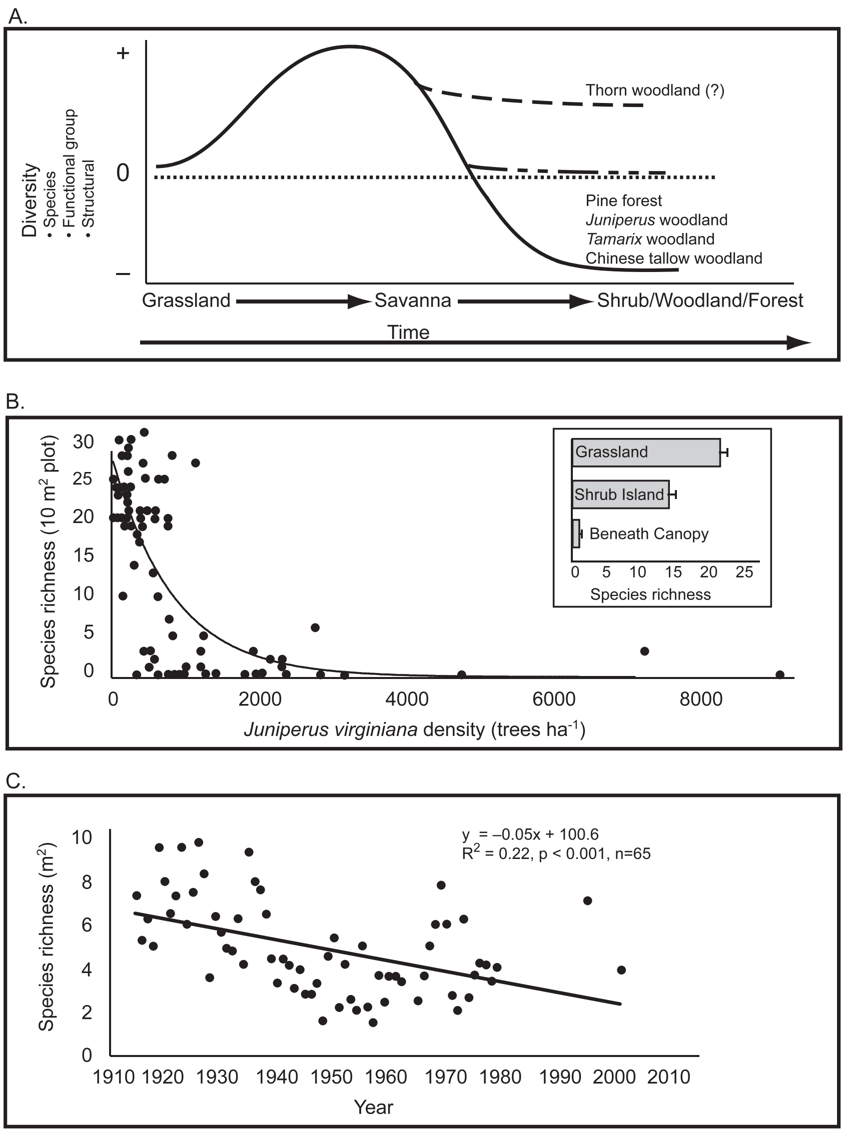
Climate and atmospheric chemistry are directly and indirectly influenced by land cover via biophysical and biogeochemical aspects of land surface–atmosphere interactions. Shifts from grass to WP domination have the potential to influence biophysical aspects of land–atmosphere interactions related to albedo, evapotranspiration, surface roughness, boundary layer conditions, and dust loading that affect cloud formation and rainfall (Figs. 1 and 2). Increases in C and N pools that occur when WPs proliferate in grasslands and savannas may be accompanied by increases in trace gas emissions (e.g., carbon dioxide, nitrous oxide, and methane; McCulley et al. 2004; Sponseller 2007; McLain et al. 2008) and nonmethane hydrocarbon emissions (Monson et al. 1991; Guenther et al. 1995; Klinger et al. 1998; Geron et al. 2006). Emissions of such compounds can influence atmospheric oxidizing capacity, heat retention capacity, greenhouse gas half-life, aerosol burdens, and radiative properties. As a result, air quality (Monson et al. 1991) and energy balance can be affected.

WP encroachment has been accompanied by increased dust production in arid regions (Gillette and Pitchford 2004). Dust can potentially influence weather and climate by scattering and absorbing sunlight and affecting cloud properties, though the overall effect of mineral dust in the atmosphere is likely

"Woody plant encroachment has been accompanied by increased dust production in arid regions"

Changes in vegetation have the potential to influence biophysical aspects of land–atmosphere interactions via effects on trace gas and hydrocarbon emissions, albedo, evapotranspiration, surface roughness, and dust loading. (Photo: Rich Reynolds)

FIGURE 5. (**A**) Conceptual model of landscape-scale changes in ecosystem biodiversity (species, growth form, or structural) that potentially accompany woody plant (WP) proliferation in grasslands and savannas (from Archer 2009). Plot-scale reductions in herbaceous species richness with increases in (**B**) juniper (*Juniperus virginiana*) density (10-m^2 plots) and within *Cornus drummondii* shrub islands and surrounding grasslands (m^2 plots; insert) in Tall Grass Prairie (Knapp et al. 2008b) and (**C**) with creosote bush encroachment in Desert Grasslands (Baez and Collins 2008). In **A**, diversity is predicted to increase during early stages of WP encroachment because of the mixture of woody and herbaceous floral/faunal elements. Maximum diversity might be expected in savanna-like configurations where woody and herbaceous plants co-occur. As WP abundance increases, loss of grassland components eventually occurs. In subtropical thorn woodland and dry forests with high WP species richness, a net increase in diversity may result. In other settings, there may be no net change in diversity, only a change in physiognomy. Where WPs form virtual monocultures with little or no understory (e.g., panels **B** and **C**), the loss of diversity may be profound. Regardless of the numerical changes in biodiversity, the existence of grassland and open savanna ecosystems and the plants and animals endemic to them are jeopardized with WP encroachment.



small compared to other human impacts (Intergovernmental Panel on Climate Change 2007). However, the mineral aerosols in dust originating from drylands are thought to play a major role in ocean fertilization and CO_2 uptake (Blain et al. 2007), terrestrial soil formation and nutrient cycling (Chadwick et al. 1999; Neff et al. 2008), and public health (e.g., Mohamed and El Bassouni 2007). Dust deposition also decreases albedo of alpine snowpack, thus accelerating melt and reducing snow-cover duration (Painter et al. 2007). In arid regions, erosion increases sediment delivery and changes flow conditions of rivers (Jepsen et al. 2003) and impacts water quality, riparian vegetation, aquatic fauna (Cowley 2006), and soil fertility and ecosystem processes (Valentin et al. 2005; Okin et al. 2006). Thus, the replacement of grasslands by shrublands in arid areas has potentially far-reaching ramifications.

Modified land cover can affect weather and climate (Bryant et al. 1990; Pielke et al. 1998). Changes in vegetation height and patchiness that occur when WPs replace grasses over large areas affect boundary layer conditions and aerodynamic roughness; changes in leaf area and rooting depth alter inputs of water vapor via transpiration; and changes in fractional ground cover, phenology, leaf habit (e.g., evergreen vs. deciduous), albedo, and soil temperature influence evaporation and latent and sensible heat exchange (Fig. 2; e.g., Graetz 1991; Bonan 2002). The extent to which these changes in structure influence meteorological conditions likely vary with annual rainfall (e.g., via leaf area changes accompanying shrub encroachment [Knapp et al. 2008a]), soil texture, shrub rooting depth, and proximity to water tables (Jobbaggy and Jackson 2004).

Effects of WP encroachment on mesoscale climate and local weather have not been assessed. However, evidence from tree-clearing studies suggest that decreases in WP cover can potentially influence evapotranspiration, the incidence of convective storms, and cloud formation (Jackson et al. 2007). Model simulations in tropical savannas indicate that clearing of woody vegetation could increase mean surface air temperatures and wind speeds, decrease precipitation and humidity, and increase the frequency of dry periods within the wet season (Hoffman and Jackson 2000). Thus, by extension, we might expect increases in WP abundance to have the reverse effect on local weather and climate.

Biodiversity

Effects of WP encroachment on biodiversity, whether quantified as the genetic diversity of populations, species richness, or the number of plant functional groups or animal guilds represented in an area, have not been widely quantified. At the landscape scale, colonization of grasslands by WPs initially represents new species additions and hence promotes biodiversity, and shrub modification of soil properties, vertical vegetation structure, and microclimate may subsequently promote the ingress and establishment of other plant and animal species (Fig. 5A). In its early stages, WP encroachment may have a multiplier effect on animal diversity by adding keystone structures and habitat heterogeneity (Tews et al. 2004b) and providing nesting, perching, and foraging sites and shelter against predators and extreme climatic conditions (Whitford 1997; Cooper and Whiting 2000; Valone and Sauter 2005; Blaum et al. 2007a). Indeed, numerous reptiles, birds, and mammals appear to prefer heterogeneous grass-dominated landscapes where scattered WPs provide up to 15% cover (Solbrig et al. 1996; Meik et al. 2002; Eccard et al. 2004; Bock et al. 2006; Thiele et al. 2008). In arid savanna rangelands, the diversity of small carnivores and their prey peaks at about 10–15% shrub cover (Blaum et al. 2007d). In the Chihuahuan Desert, shrub-invaded sites harbor four times the number of ant forager species found at a relatively pristine desert grassland site, suggesting that ant diversity is enhanced by shrub invasion and that several taxa benefit from it (Bestelmeyer 2005). The effects of WP encroachment vary among animal taxa and functional groups (e.g., Kazmaier et al. 2001), but as WP cover increases and habitat characteristics continue to shift, shrubland/woodland-adapted species are expected to become favored over grassland-adapted species.

Grassland-obligate plants and animals may be affected immediately and negatively by WP encroachment (Table 4). Even so, diversity may be maintained or enhanced if new species co-occur with the more broadly adapted original species and if the displacement of grassland-obligate species is more than offset by the arrival of new species (e.g., Sauer et al. 1999; Blaum et al. 2007b, 2007c). As WP cover and biomass continue to increase, the end result may be an overall gain in diversity, no net change in diversity, or a net loss in diversity (Fig. 5A). Qualitative observations suggest that tropical and subtropical grasslands may potentially experience net gains in diversity with WP encroachment because of large pools of tree and shrub species, large pools of herbaceous species capable of coexisting with WPs, and large pools of invertebrates

“Grassland-obligate plants and animals may be affected immediately and negatively by WP encroachment”

TABLE 4. Avifauna and woody plant (WP) encroachment. Grassland passerines are declining at a faster rate than any other bird group in North America (Peterjohn and Sauer 1999). Woody plant encroachment associated with livestock grazing is among the contributing factors (Bakker 2003; Brennan and Kuvlesky 2005).

Vegetation change	Effects on grassland avifauna	Citation(s)
Thresholds of WP cover and height exceeded	Reduction in suitable habitat	Lloyd et al. (1998); Grant et al. (2004); Gottschalk et al. (2007)
Proximity to woodlands	Decreased food abundance; increased predation and brood parasitism	Johnson (2001); Bakker et al. (2002); Fletcher and Koford (2002); Thiele et al. (2008)
Grasslands converted to shrublands	Increased overall avian species richness but declines in ground-nesting passerine and gallinaceous species	Whitford (1997); Pidgeon et al. (2001); Rosenstock and van Riper (2001); Fuhlendorf et al. (2002)
Juniper encroachment into sagebrush–steppe communities	Eliminates sagebrush-obligate species habitat	Connelly et al. (2000); Miller et al. (2000); Crawford et al. (2004); Reinkensmeyer et al. (2007)

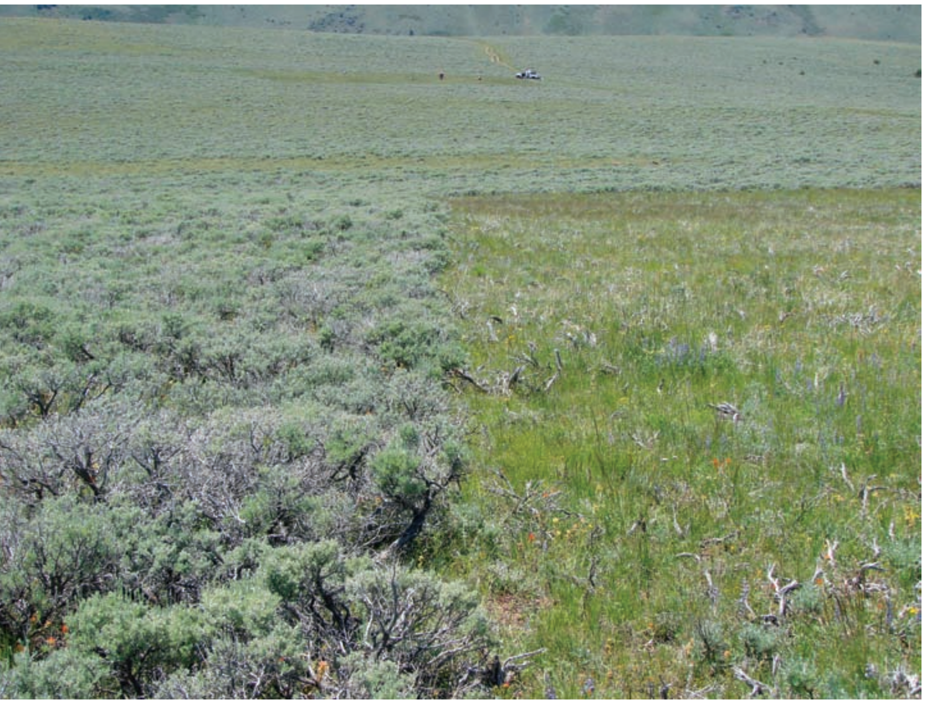


FIGURE 6. Undisturbed (left) and mowed (right) mountain big sagebrush (*Artemisia tridentata vaseyana*)–bunchgrass plant community in southeastern Oregon. (Photo: K. W. Davies)

and passerine bird species. In other cases, the number of encroaching woody species may be very small and their traits detrimental to the persistence of other plant species. WP encroachment may then result in virtual monocultures of vegetation (Figs. 5B and 5C) with concomitant impacts on faunal diversity.

Changes in aboveground biological diversity with WP proliferation may also be accompanied by changes in the diversity of microbial communities belowground. Shifts from bacterial to fungal populations may accompany shifts from herbaceous to woody domination (e.g., Imberger and Chiu 2001; Purohit et al. 2002), enabling the microbial biomass to effectively deal with lower litter quality and thus maintain or even increase soil respiration and mineralization. Aanderud et al. (2008) found differences in gram-positive bacteria, Actinobacteria, and fungi communities in soils below and between shrubs. Thus, changes in microbial communities would be expected to accompany changes in composition and abundance of shrubs.

Parasitic nematodes and nematodes feeding on bacteria and fungi in the immediate vicinity of plant roots are indicator taxa for changes in belowground microbial communities. The maximum depth of occurrence of these organisms increased from 2.1 m in grasslands to 4.0 m in areas where WPs have replaced grasses, but the composition of the nematode food web at this depth was markedly reduced from five trophic groups to two (Jackson et al. 2002). Invaded woody sites also had lower species richness in soils due primarily to the loss of root feeding species.

The conceptual model in Figure 5A is based on numerical assessments of species, functional group, and structural diversity. However, from the perspective of physiognomic diversity, WP encroachment is transformative. Grasslands become shrub or tree savannas, and shrub and tree savannas become shrublands or woodlands. Thus, even in cases where numerical diversity may be maintained or enriched by WP encroachment, there is a loss of grassland and savanna ecosystems and the plants and animals endemic to them. Thus, while brush management has historically been advocated from the perspective of potential benefits for livestock production and hydrology, it should also be considered from the perspective of maintaining the existence of grassland and savanna ecosystems.

BRUSH MANAGEMENT: A BRIEF HISTORY

WP encroachment has long been of concern to rangeland managers (Leopold 1924). Thus, there is a long history of devising management tools for reducing WP cover. The basis for concerns over WP proliferation was historically centered around the adverse effects of shrubs on forage production (Fig. 6) and livestock safety (e.g., WPs as cover for predators), health (e.g., as habitat for insect and arthropod pests and parasites such as ticks and horn flies [Teel et al. 1997]), and handling (difficulty in gathering and moving animals with increasing WP stature/cover/density). This traditional focus on rangeland value for livestock production was also the impetus for other management practices, such as efforts to eliminate competitors (e.g., certain predators, herbivores, and insects) viewed as directly or indirectly reducing ranch profits. In some cases, these wildlife may have played an important role in keeping WPs in check, and their systematic elimination may have opened the door for WPs to increase in abundance (e.g., prairie dogs; Weltzin et al. 1997).

The 1940s and 1950s

During the post–World War II era, heavy equipment and chemicals were readily available and were used on a broad scale. Our understanding of ecosystem processes and ecosystem goods and services was in its infancy during this period, and few environmental regulations were in place. Applied research in range science focused on the development and application of herbicides and mechanical techniques (Scifres 1980; Bovey 2001), often with the goal of eradicating shrubs. Brush management during this period was typically applied indiscriminately.

The 1960s and 1970s

Efforts aimed at widespread eradication in the 1940s and 1950s gave way to efforts aimed at selective control and containment in the 1960s and 1970s. By this time, it was clear that there were no "silver bullets" for brush management. Unlike many herbicide products available today that provide a long treatment life (15–50 yr; McDaniel et al. 2005; Perkins et al. 2006; Combs 2007), treatments in the past were relatively short lived (Jacoby et al. 1990a). Following chemical spraying, shrub cover often returned to pretreatment levels (or higher) within 5–15 yr. The necessity of retreating landscapes at relatively high frequencies made brush management nonsustainable and difficult to justify when cost often exceeded revenues generated from subsequent livestock production.

Basic and applied research from the 1940s to the 1970s led to the realization that brush management practices:

- were treating symptoms (the shrubs) rather than root causes of land cover change (e.g., disruption of historic grazing and fire regimes);
- must be conducted in concert with progressive livestock grazing management;
- when applied in an indiscriminant manner without careful planning, can
    - be detrimental to wildlife populations,
    - lead to homogenization and loss of biological diversity,
    - increase risks for catastrophic soil erosion and weed invasion, and
    - be too costly for a ranching enterprise, and
- can be short lived, with shrubs reestablishing dominance in 5–10 yr.

Collectively, these realizations led to the development of integrated brush management systems (IBMS) in the 1980s (Scifres et al. 1985; Brock 1986; Hamilton et al. 2004).

1980s to Present: IBMS

IBMS are long-term planning processes that move away from a purely livestock production perspective and toward management of rangelands for multiple uses and values. The IBMS planning process begins by identifying management goals and objectives for a specific site and the surrounding management unit. These might include increasing forage production; maintaining or promoting suitable wildlife habitat; augmenting stream flow or groundwater recharge; controlling pests, pathogens, or invasive species; maintaining scenic value; reducing wildfire risk; or preserving grassland and savanna ecosystems. Specific objectives are refined on a comprehensive inventory of ecosystem components (plants, animals, and soils), projecting the responses of those components to brush treatment alternatives, and considering the effects of treatment alternatives on management goals on other sites (Hanselka et al. 1996). Brush management techniques (herbicidal, mechanical, and prescribed burning) differ with respect to environmental impacts, implementation costs, efficacies, and treatment longevities. Thus, the IBMS approach advocates consideration of the type and timing of a given brush management technology and makes explicit allowances for consideration of the type and timing of follow-up treatments. This, in turn, requires knowledge of how woody and herbaceous plants will respond and how climate, soils, topography, and livestock and wildlife management might mediate plant responses. The IBMS approach is therefore inherently interdisciplinary and dependent on the active collaboration of a diverse group of management professionals.

Examples of the IBMS approach abound (Teague et al. 1997; Grant et al. 1999; Paynter

"IBMS are long-term planning processes that move away from a purely livestock production perspective and toward management of rangelands for multiple uses and values."

“The type, timing, and sequencing of brush management are the keys to long-term success”

and Flanagan 2004; Ansley and Castellano 2006; Ansley and Wiedemann 2008). Nearly every State Cooperative Extension Service in the western region offers a long list of guides and publications that address the IBMS approach and give specific recommendations for management of important WPs. There are expert system tools (e.g., PESTMAN 2009) available to assist land managers in selecting appropriate brush management practices and techniques (Hanselka et al. 1996). There are also special adaptations of the IBMS approach for small landholdings (McGinty and Ueckert 2001). NRCS guidelines should promote the IBMS approach and explicitly utilize it in the conservation planning process.

BRUSH MANAGEMENT: CURRENT PERSPECTIVES

> Brush management efforts must be viewed as an integral part of the overall system for wise, efficient use and conservation of grasslands. Available brush management and conservation methods are complex tools, the effectiveness of which depends primarily upon the resource manager's understanding of their proper application approached with consideration of all potential uses of rangeland. (Scifres et al. 1983, p. 11)

This statement, made over 25 yr ago, is still relevant today. This rationale is echoed in recent textbooks addressing brush management and ecosystem restoration (Valentine 1989; Whisenant 1999; Bovey 2001; Hamilton et al. 2004). The NRCS (2006) currently recognizes six reasons for undertaking brush management:

- Restore natural plant community balance
- Create the desired plant community
- Restore desired vegetative cover to protect soils, control erosion, reduce sediment, improve water quality, and enhance stream flow
- Maintain or enhance wildlife habitat, including that associated with threatened and endangered species
- Improve forage accessibility, quality, and quantity for livestock
- Protect life and property from wildfire hazards

A Texas survey found that the two primary goals of landowners investing in brush management were to increase forage production and to conserve water (Kreuter et al. 2005). Other reasons included improvement of aesthetic values, benefit the next generation, improve wildlife habitat, and improve real estate value.

Short- and Long-Range Planning Is Essential

A well-thought-out, comprehensive resource management plan reflecting short-, medium-, and long-term goals and objectives should be in place before attempting brush management.

In areas where shrubs are dense, herbicides or mechanical treatments, such as this crawler tractor equipped with push blades and discs, may be initially required to open up areas to promote herbaceous production and enable the subsequent use of prescribed fire as a management tool. (Photo: Tim Fulbright)

Evaluating multiple scenarios is useful for explicitly assessing the advantages and disadvantages of several alternatives. Given that the type, timing, and sequencing of brush management are the keys to long-term success, management plans should identify long-term objectives and work to ensure that resources and commitments are in place at the right time. Programs should include explicit information on what constitutes success and, if feasible, should address alternatives to primary objectives. Objectives will vary with the specific needs of the landowner, the community, and context of the action. Well-defined short- and long-term objectives are critical to determining when, where, why, how, and under what conditions brush management should be undertaken.

Considerable momentum is now building for landscape-scale IBMS projects. Such projects usually require close consultation, coordination, and cooperation among multiagencies and associated stakeholders. Managing rangeland brush and weeds requires adapting management methods to match species physiological and morphological traits and ecological site conditions (Fig. 7). The decision-making process associated with landscape-scale IBMS projects can be restricted by numerous factors, including equipment availability, financial constraints, available manpower, time needed to complete the task, environmental regulations, and agency mission and policy.

The Need for Monitoring

Monitoring is intrinsically linked to the IBMS process (i.e., treatment monitoring, control, revegetation (natural or planted), and pre- and posttreatment monitoring; Scifres et al. 1983; Scifres 1987; DiTomaso 2000). Assessing baseline (pretreatment) conditions is essential to determine the effects of brush management efforts. Monitoring of weather and seasonal growth of the plants targeted for manipulation should begin well in advance (3–6 mo) of planned treatment. Posttreatment monitoring should be conducted at least annually to evaluate responses.

Field variables to be measured for baseline inventories and posttreatment monitoring should be selected on the basis of the objectives of the brush management project. The following represent initial field variables that

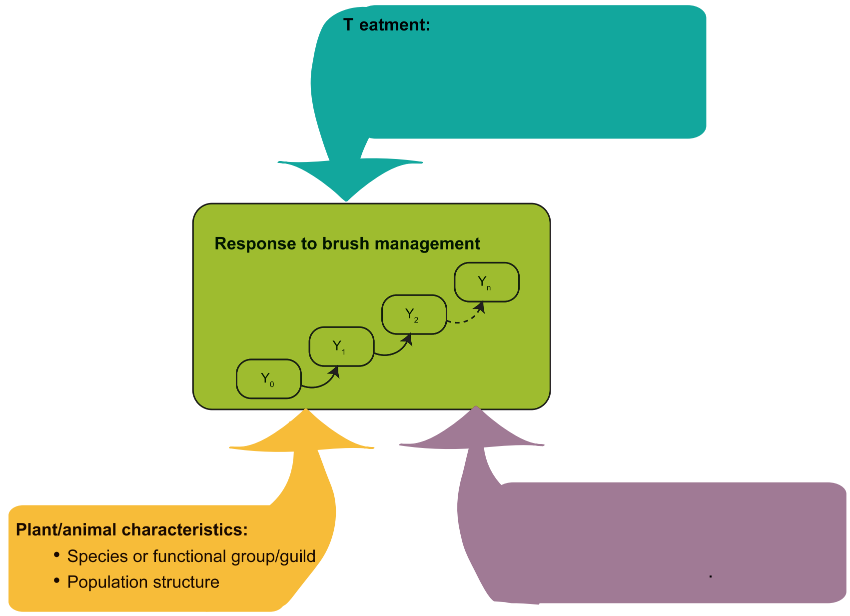


FIGURE 7. Ecosystem response to brush management varies with time since treatment (Y_1, Y_2, etc.) and is determined by a variety of interacting factors.

should be assessed in baseline inventories and to evaluate posttreatment responses:

- Shrub age (or size as a rough proxy), class distribution (baseline only), height, and stem density and diameter
- Plant composition and frequency of occurrence
- Ground cover of grass, forb (total and by species), litter, and bare ground
- Biomass (seasonal or peak live standing crop; preferably by species)
- Specific metrics will depend on the goals and objectives of the brush management project.

Treatment Options

Understanding the ecology of WPs and herbaceous plants and how they interact with each other on a particular site is crucial in determining the IBMS strategy. Relevant stand characteristics include plant community composition, plant phenology, plant density, plant size (stem diameter; canopy area, volume, and height), canopy architecture, and patterns of biomass allocation to leaves and stems aboveground and roots belowground. Each of these can influence the effectiveness and longevity of a given brush management practice. Realization of brush management objectives often benefits from spatially explicit prescriptions that take into account

TABLE 5. Factors influencing the effectiveness and longevity of a given brush management practice. Cookbook or "one-size-fits-all" approaches for brush management seldom succeed. A given project should be tailored to individual goals, objectives, and circumstances, and these, in turn, will be mediated by the items shown in the table.

• Site accessibility and terrain: If the site is difficult to reach or traverse (e.g., sandy soils, uneven topography), less labor-intensive methods (e.g., aerial spraying) might be more effective.
• Stand characteristics: Extent, age, biomass, and plant density will be factors in selecting the most cost-effective methods.
• Proximity to endangered species: The presence of a federally listed species or environmentally sensitive sites may preclude some types of management methods or limit the season of application.
• Presence of desirable plant species or other important resources (e.g., archaeological sites): Locally targeted methods (e.g., individual plant treatments) may be warranted to protect other resources in the area.
• Extent of area to be treated: Suitability and efficiency of a given treatment method may vary by the size of the area targeted for treatment.

the topoedaphic heterogeneity of landscapes (i.e., uplands, side slopes, riparian, valley bottomland, etc.; Taylor and McDaniel 2004; Table 5).

BRUSH MANAGEMENT AS A CONSERVATION TOOL: A CRITICAL ASSESSMENT

The projected effects of brush management on ecosystems and ecosystem processes boils down to its effects on: 1) herbaceous vegetation and native plant communities; 2) livestock; 3) watershed properties related to erosion, soil condition, water quality, and water quantity; 4) wildlife; 5) air quality; and 6) human dimension considerations. The following sections summarize the scientific literature addressing the first five of these and the question, Are the outcomes expected from brush management being realized? For treatment of economics and the human dimensions area, see Tanaka et al. (this volume).

Our assessment began with a series of literature searches using the Web of Knowledge[SM]. Search strings included "brush management" and terms referring to specific brush management techniques. Search results were filtered to include only studies conducted in the United States on rangelands and only those quantifying responses to brush management. Studies quantifying herbaceous responses dominate the brush management literature in the United States and comprised 48.7% of those in the sample (Fig. 8). Treatment efficacy and shrub regeneration studies accounted for another 28.7%. Studies documenting water (4.1%) and soil (2.0%) responses were the least common.

Herbaceous Vegetation and Native Communities

The general expectations associated with brush management are that it will mitigate soil erosion, improve soil condition, enhance water quantity and quality (via improvement in infiltration and reductions in runoff, which interact to reduce sedimentation), and improve livestock production. Each of these expectations is based on the assumption that herbaceous ground cover will increase following brush management. How good is this assumption?

Herbaceous Response. The majority (>80%) of studies in our literature sampling reported positive herbaceous responses following brush management (Appendix I). Herbaceous plant growth increases an average of 3- to 5-fold for brush management conducted on productive range sites, including sites with Wyoming big sagebrush (*Artemisia tridentata*; Hyder and Sneva 1956; McDaniel et al. 2005) and broom snakeweed (*Gutierrezia sarothrae*; McDaniel et al. 1993). Management of other woody species, including mesquite (*Prosopis glandulosa*) and creosote bush, can result in substantial forage increases on productive sites with adequate rainfall (Ethridge et al. 1984; Perkins et al. 2006; Combs 2007). In semidesert grasslands at the Santa Rita Experimental Range in Arizona, herbage yields following velvet mesquite removal increased (Parker and Martin 1952; Paulsen 1975; Cable 1976) or remain unchanged in zones <1100 m in elevation and when velvet mesquite cover was <25% (McClaran and Angell 2006). These patterns are consistent with field studies in southern New Mexico (Warren et al. 1996; Drewa and Havstad 2001).

No consistent relationship between posttreatment changes in herbaceous production and annual rainfall were found;

however, a survey of data across a range of management contexts suggests an upper limit for the herbaceous production responses that might be expected for a given rainfall zone (Fig. 9A). Herbaceous response corrected for annual rainfall varies with time since brush treatment (Fig. 9B). The median first-year response is 0 and highly variable, with half the treated sites responding positively and half negatively. By year 2, the median response is slightly positive but also highly variable. After year 2, the response becomes more consistent and peaks in year 5. The response then drops off in years 7 and 8, being slightly but consistently positive.

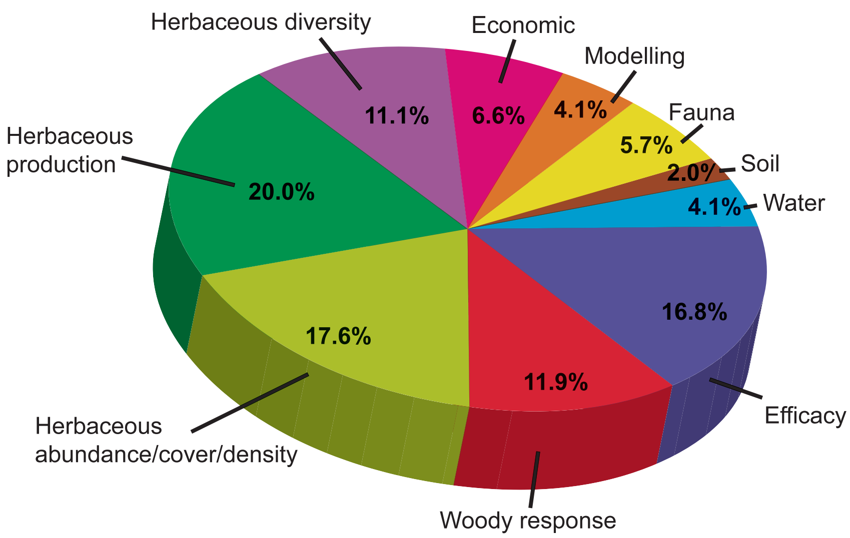
The longevity of brush management treatments varies widely by type of treatment applied, shrub species, effectiveness of the initial treatment, composition of the herbaceous vegetation, and soil properties (Fig. 10). Variations in the Figure 10 conceptual model have been illustrated for velvet mesquite (Cable 1976), honey mesquite (Heitschmidt et al. 1986; Ansley et al. 2004b), big sagebrush (McDaniel et al. 2005), and creosote bush (Gibbens et al. 1987; Morton and Melgoza 1991; Perkins et al. 2006). The change in foliage cover and herbaceous response to brush management ranges from 5 to 20 yr for velvet mesquite (Cable 1976), from 10 to 25 yr for honey mesquite (Dahl et al. 1978; Jacoby et al. 1991, 1990a, 1990b; Combs 2007), >25 yr for sagebrush (McDaniel et al. 2005), and >40 yr for creosote bush (Perkins et al. 2006). The general curve shape of the overstory–understory relationship for these shrub species is similar, but average grass yield associated with overstory cover is scaled quite differently: from 2- to 3-fold greater for mesquite relative to big sagebrush and creosote bush, respectively. Mesquite management typically provides a greater forage response, but it is of shorter duration than for big sagebrush and creosote bush removal. Accordingly, timing of investments to re-treat communities dominated by these shrubs would be on the order of about 4–12 yr for mesquite, 20–30 yr for big sagebrush, and >30 yr for creosote bush (Torell and McDaniel 1986; Torell et al. 2005a).

Although studies have or currently are being conducted across different ecological sites in the western United States, adequate data to statistically estimate the relationships in Figure 10 as a function of rainfall, soils, site productivity, and so on are not generally available. Instead, qualitative assessments by experienced range scientists and economists are currently the norm for projecting forage response to brush management (Fig. 11).

Soil Condition. Some of the projected effects of brush management on soil are associated with the assumption that soil organic matter depletion will occur with shrub encroachment. However, as reviewed earlier (see the section "Soil Condition and Erosion"), this is not a robust assumption. It may be true in certain cases, most likely those in arid areas where disruption of grass cover by grazing has accelerated wind and water erosion. But even in those instances, soil resources may undergo not a net change in abundance but, rather, a change from a homogenous to a heterogeneous distribution wherein they are concentrated within shrub islands (Schlesinger and Pilmanis 1998). Thus, were it not for shrubs, soil resources may have been lost from the site because of grazing rather than being spatially rearranged. Brush management, by reducing shrub cover on fertile shrub islands, may put

> "Robust generalizations regarding brush management effects on soil carbon are currently not possible."

FIGURE 8. Proportion of brush management studies quantifying various categories of treatment effects. Published papers resulting from Web of Knowledge search strings that included "brush management" and terms referring to specific brush management techniques were distilled to a database of 333 articles that reported quantitative responses. These were then classified into the categories shown. Articles reporting data for multiple metrics were tallied in multiple categories. Thus, the graph reflects the information reported in the literature but not on a per-paper basis.



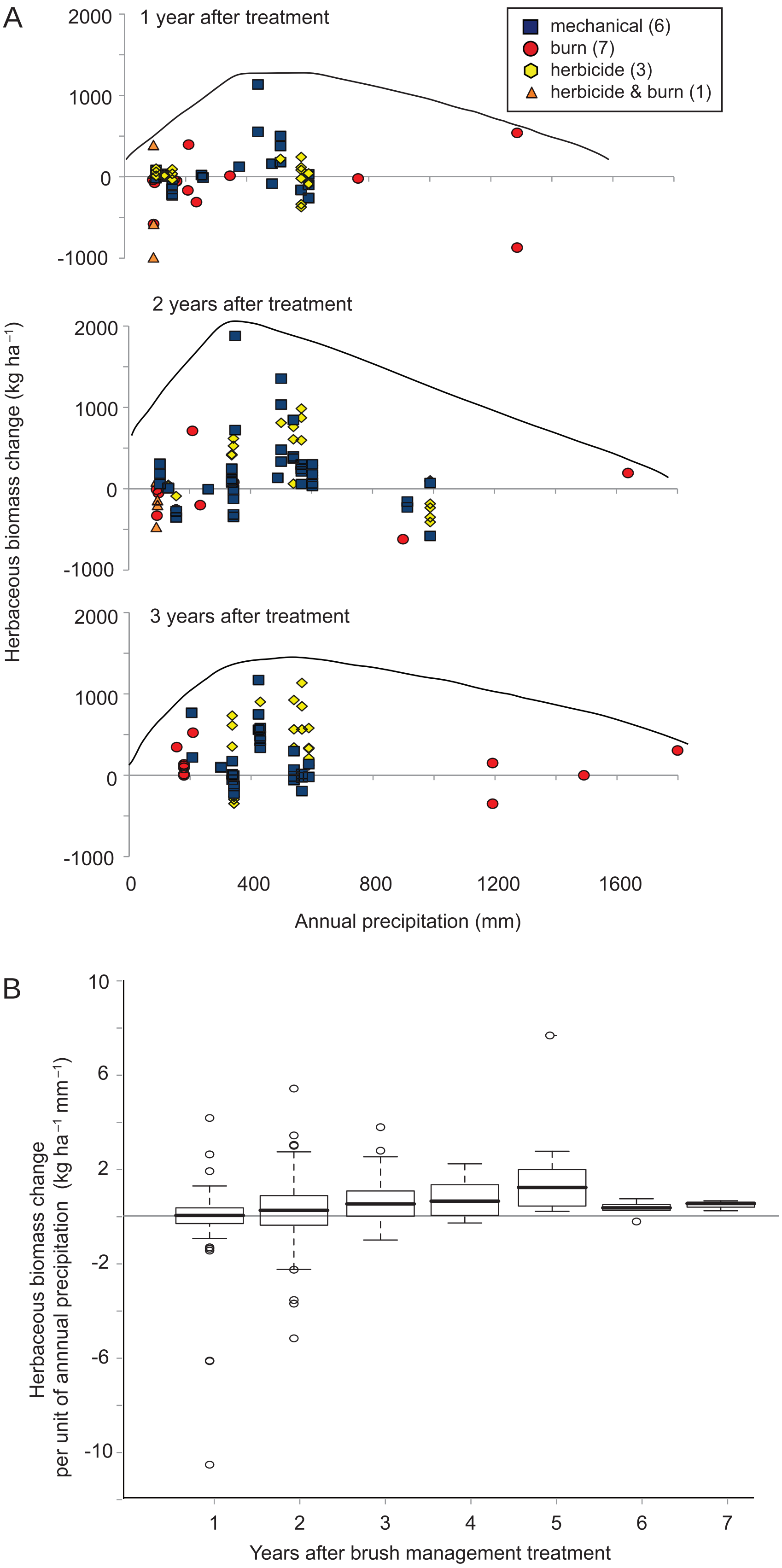


FIGURE 9. (**A**) Changes in herbaceous biomass production ($kg \cdot ha^{-1}$) 1, 2, and 3 yr after brush management as a function of current years' annual precipitation (PPT, mm). Multiple observations for a given PPT value reflect multiple sites or different brush management applications. PPT was determined from nearby weather stations if not reported. The number of studies pertaining to a given brush treatment are listed parenthetically in the key. (**B**) Change in herbaceous biomass per millimeter of annual precipitation received after brush management. Responses are from 13 studies representing brush management with fire, herbicides, and mechanical treatments. Tukey box plots show inner-quartile range (IQR; rectangle) and the median (bold line). Whiskers indicate the maximum and minimum values or the values within 1.5*IQR of the third and first quartiles, respectively. Values beyond 1.5*IQR of the first and third quartiles are considered statistical outliers and are indicated with open circles. *N* = 13, 13, 11, 8, 5, 3, and 2 for years 1 through 7, respectively. The number of studies pertaining to a given brush treatment are listed parenthetically in the figure legend. Citations used to generate the data points are given in Appendix III.

these sites at risk for net losses of soil nutrients unless ground cover is quickly established. Alternatively, nutrient losses from shrub islands following brush management may help reinstate the homogeneous distribution of resources by disrupting the processes that lead to the concentration of nutrients in and around shrub canopies (e.g., Davies et al. 2009a; Ravi et al. 2009). In the latter scenario, the likelihood of getting grasses reestablished within intershrub zones may improve. Site-specific factors may dictate which of these competing scenarios is most likely on a given landscape.

WP encroachment can have a moderate to strong positive impact on SOC and N pools on many sites (Fig. 4). This shrub-induced improvement in SOC and N may be an important factor underlying the extent to which herbaceous vegetation production increases following brush management (Fig. 9). The degree to which shrubs might increase soil resources beneath their canopies is a function of how long the shrubs have occupied the site (Wheeler et al. 2007; Throop and Archer 2008). Thus, stand age at the time of brush management will have an important bearing on soil conditions. Removal of individual shrubs causes depletion

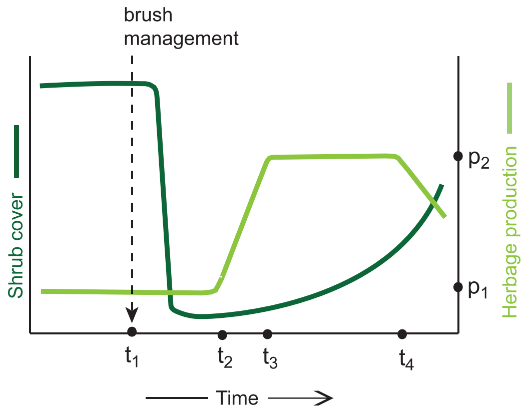


FIGURE 10. Generalized conceptual model of herbaceous response to brush management. The lag time in response (t_1–t_2), the magnitude of (p_1–p_2) and time (t_1–t_3) to peak herbaceous response, the duration of peak elevated production response (t_3–t_4), and the time frame over which herbaceous productions decline as shrubs reestablish (t_4 onward) vary with numerous factors. Knowledge of the relationships depicted in this conceptual model for a given ecological site will help determine the type, timing, and appropriate sequencing of brush management practices in an Integrated Brush Management Systems (IBMS) approach.

of the associated resource pool and the availability of nutrients over the 10–15 yr following treatment, the extent depending on whether shrubs regenerate (Klemmedson and Tiedemann 1986; Tiedemann and Klemmedson 1986, 2004). Losses of SOC and N accumulating in soils associated with mature shrubs killed by herbicide ranged from 67% to 106% at 0–5-cm soil depths and from 78% to 93% at 5–10-cm soil depths over a 40-yr period (McClaran et al. 2008). Data from these individual plant perspectives suggest that brush management will cause a decline rather than an increase in SOC and N pools in hot, semidesert rangelands but that shrub regeneration can arrest or reverse such declines (Hughes et al. 2006). These findings contrast with those of Teague et al. (1999), who compared SOC and N on sites 4–22 yr after root plowing against untreated controls in the southern Great Plains to test the hypothesis that removal of honey mesquite would result in steady decline in SOC because of a loss of mesquite inputs and reductions in shading (and therefore higher soil temperatures and higher oxidation rates). Overall, they found no significant differences between treated and control sites. Similarly, Hughes et al. (2006) found that while aboveground C and N pools increased markedly with mesquite stand development following brush management (more so on sandy sites than shallow, clayey sites), near-surface SOC and N pools were unaffected. Thus, as with WP encroachment (Fig. 4), robust generalizations regarding brush management effects on soil condition are currently not possible.

“Timing of investments to re-treat communities is about 4–12 yr for mesquite, 20–30 yr for big sagebrush, and >30 yr for creosote bush”

FIGURE 11. Projected changes in livestock stocking rates for aerial spraying and mechanical plus grass seeding treatment of a *Prosopis*–mixed brush system in southern Texas relative to untreated controls (J. Conner et al., unpublished data, 1998).

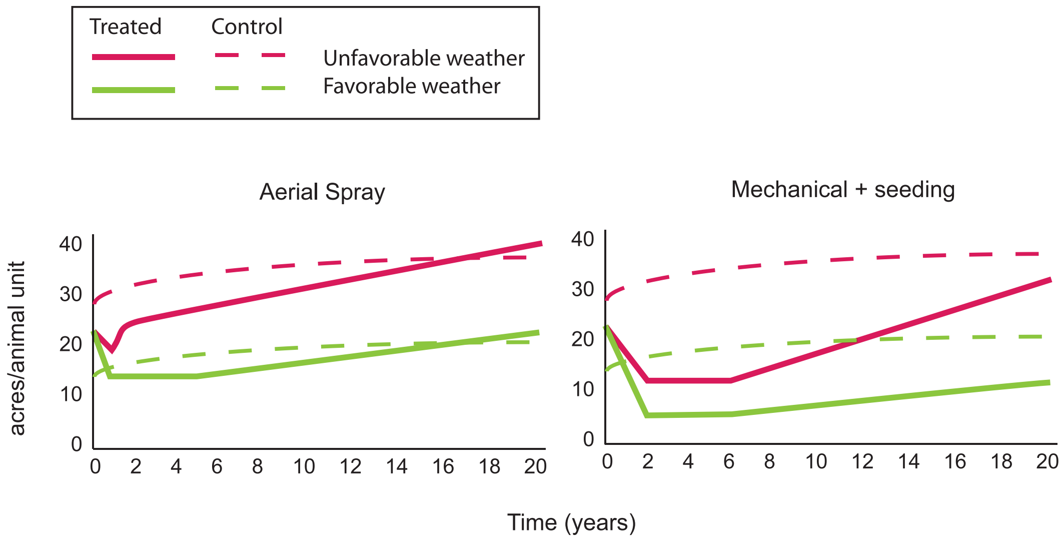


FIGURE 12. (**A**) Mojave Desert scrub near Las Vegas, Nevada, USA (foreground) and area invaded by the nonnative annual grass red brome (*Bromus rubra*; background) following a fire that carried from desert floor upslope into pinyon–juniper woodlands. This exotic grass has instigated a positive disturbance (fire) feedback that reduces ecosystem carbon storage, threatens the biodiversity, and constitutes a new ignition source for fire in upper-elevation woodlands and forests. Photo: T. E. Huxman. (**B**) Nonnative annual grass (medusahead, *Taeniatherum caput-medusae*) invaded area (foreground) transitioning into a Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) plant community in southeastern Oregon. The presence of this exotic annual grass increases the probability that the sagebrush plant community above it will burn. (Photo: K. W. Davies)

Non-native species invading or purposely seeded following brush management (see Fig. 12 and the section "Biodiversity and Nonnative Species") may significantly reduce ecosystem C accumulating with WP encroachment. Indeed, estimates of aboveground C loss with conversion of Great Basin shrublands and woodlands to annual grasslands are on the order of 8 Tg C, with estimates of 50 Tg C release to the atmosphere over the next several decades (Bradley et al. 2006). In cold desert sagebrush steppe ecosystems, this level of C release with annual grass invasion could completely offset any increases in C with woody encroachment that has occurred over the past century. However, the story may be quite different in southwestern rangelands where highly productive, deeply rooted perennial grasses introduced from Africa are expanding and sequester substantially more C than annual grasses (e.g., Williams and Baruch 2000; Franklin et al. 2006).

Specific brush management techniques will likely differ in their impact on litter decomposition, depending on the type of disturbance they cause, treatment efficacy, and the extent to which they co-occur with other land use practices, such as livestock grazing. Brush management treatments that minimally disturb soils (e.g., herbicide applications and prescribed burning) may be most advisable for managers wishing to minimize short-term SOC losses. In contrast, brush management techniques that cause extensive disturbance to the soil surface, such as chaining, root plowing, and grubbing, may increase decomposition rates due to surface soil disturbances. These practices likely superimpose a variety of new short- and long-term direct and indirect effects on decomposition processes via their dramatic alteration of surface roughness, water infiltration and runoff, vegetation cover, and ANPP and by initiating large, synchronous inputs of leaf, stem, and coarse woody debris onto the soil surface with widely varying degrees of contact and incorporation into the soil. Such treatments ostensibly increase exposure to direct sunlight and UV radiation and may promote soil movement via wind and water, particularly during the immediate posttreatment period when vegetation is reestablishing.

Vegetation responses to brush management have been widely described, but very little is known of its effects on soils and nutrient cycling (Fig. 8). There have been few attempts to model brush management effects on ecosystems (but see Carlson and Thurow 1996; Grant et al. 1999), and the future development of such models would likely benefit from field studies elucidating how various brush management practices might impact C and N cycling. Lessons learned from studies of temperate forest clear-cutting and tropical deforestation would be instructive starting points, but it is likely that the shrublands, savannas and woodlands of drylands would have novel behaviors. For example, predicting brush management effects on litter decomposition in semidesert grasslands will require information on shrub–grass interactions and herbaceous biomass influences on soil movement at a decadal time scale (Throop and Archer 2007).

Biodiversity Response. Biodiversity responses can be assessed at the species (e.g., genetic variation in populations), the organismal (species richness), the structural (vegetation strata and physiognomy), and the functional (plant functional groups and animal guilds) levels. Studies at the organismal level are typically restricted to a select class of organisms (e.g., perennial herbaceous plants or small mammals) without regard for other classes (annual plants, shrubs, reptiles, avifauna, large mammals, microbes, etc.). To further complicate things, diversity varies with scale (e.g., alpha, beta, and gamma diversity) and topoedaphic heterogeneity. Objectives aimed at preserving, restoring, and monitoring biodiversity should thus be phrased to specifically articulate the facets of biological diversity being addressed.

From a Web of Knowledge search generating 333 studies quantifying responses to brush management (Fig. 8), 39 articles reporting herbaceous plant diversity emerged; of these, 13 were conducted on rangelands and were amenable to comparative analysis. From the 90 data points emerging from these studies, it appears that brush management treatments typically have neutral (30% of data points exhibiting <10% change) to positive (60% of data points exhibiting >10% increase) effects on grass/forb diversity (Fig. 13). Cases where brush management had negative effects on herbaceous diversity (10% of data points exhibiting >10% decline) were typically associated with herbicide treatments, ostensibly reflecting adverse impacts on forbs. The few long-term data available suggest that posttreatment stimulation of herbaceous diversity is relatively short lived (<15 yr).

In the subtropical southern Great Plains characterized by a diverse flora of encroaching WPs, WP communities developing after brush management have lower shrub diversity and higher densities of less desirable browse species than the previously existing community (Fulbright and Beasom 1987; Ruthven et al. 1993). In systems where shrubs aggressively regenerate vegetatively, use of low-intensity fire and herbicides can promote a savanna physiognomy (e.g., Ansley et al. 1997, 2003) and ostensibly promote diversity.

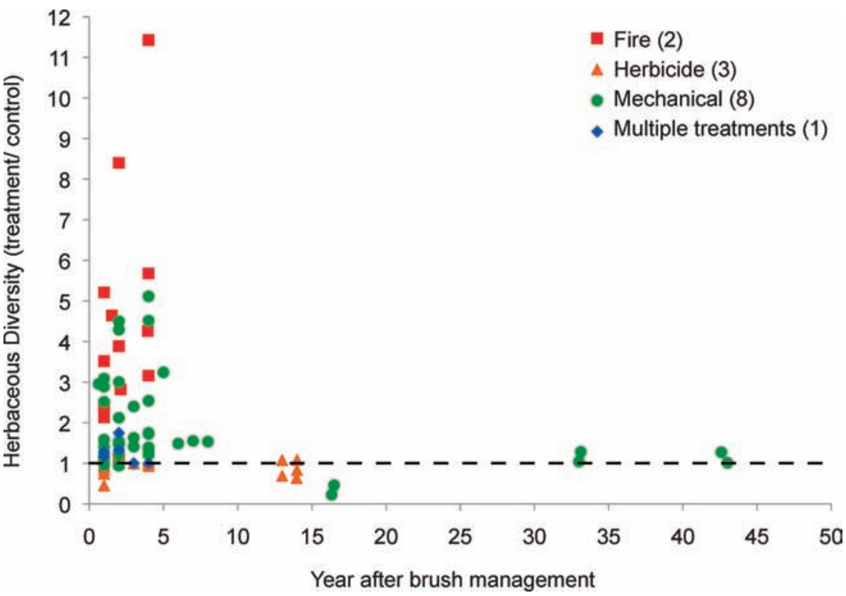


FIGURE 13. Changes in herbaceous vegetation diversity following brush management. Results (90 data points) show ratios of values (richness and various indices) reported for treated and control areas. Ratios near 1.0 indicate that diversity metrics on treated and control areas were similar, values <1.0 indicate that brush management decreased diversity, and values >1.0 indicate increases in diversity. The number of studies pertaining to a given type of brush treatment are listed parenthetically in the key. See Appendix III for citations.

> “Brush management has the potential to create conditions favorable for the establishment and growth of weeds and invasive nonnative species.”

Faunal diversity response to brush management varies with the organisms of interest (see the section “Wildlife”). For example, although Jones et al. (2000) reported that relative total abundance and species richness of herpetofauna was similar among a variety of treatments, amphibians were most abundant in untreated and herbicide-only sites, lizards were most abundant on untreated sites, and snakes were most abundant on sites receiving herbicide and fire. Rodent and avian relative frequency, richness, and diversity have been observed to be unaffected by brush management (Nolte and Fulbright 1997; Peterson 1997).

The biodiversity response to brush management may be strongly influenced by the pattern of treatment application (see Bestelmeyer et al. this volume). A “wall-to-wall” application may yield one result, whereas applying a treatment or combination of treatments in “strips” may have a quite different outcome by creating more habitat edge and creating patches of grassland habitat interspersed with shrubland habitat (e.g., Scifres et al. 1988). For example, diversity of native perennial grasses may be promoted by a mixture of open areas interspersed with cover of mature shrubs (Tiedemann and Klemmedson 2004). Effects of brush management on biological diversity are poorly understood and need to be investigated at larger scales across longer time periods.

Biodiversity and Nonnative Species. Brush management has the potential to create conditions favorable for the establishment and growth of weeds and invasive nonnative species (Young et al. 1985; Belsky 1996; Bates et al. 2007) that can have adverse affects on biodiversity. Brush management is therefore often conducted in conjunction with seeding operations intended to accelerate establishment of ground cover and a forage base (see Hardegree et al. this volume). In many cases, the grasses used for seeding are nonnative perennials (Cox and Ruyle 1986; Ibarra-Flores et al. 1995; Martin et al. 1995; Christian and Wilson 1999). Seeds from such species may be more readily available, and their establishment success rates may be higher than that of natives (Eisworth et al. 2009). When seeding of nonnative grasses is successful, the result is often a persistent, long-lived near-monoculture of nonnative vegetation. While this may be valued for livestock production and ground cover and may make the site more resistant to invasion by undesirable exotic annual grasses (Davies et al. 2010) by virtue of their superior competitive ability (Eissenstat and Caldwell 1987), these plants may represent threats to the biodiversity of native plants and animals (McClaran and Anable 1992; Williams and Baruch 2000; Schussman et al. 2006). Their unintended spread into areas beyond where they were planted may make it difficult to achieve conservation goals on other lands. Thus, there are clear trade-offs that should be explicitly considered and evaluated.

A Tool to Promote Landscape Heterogeneity and Biodiversity? Disturbances associated with fire and herbivory (granivory, grazing, browsing, burrowing, trampling, and dung/urine deposition) interact with climate variability and extremes to generate patchiness across the landscape and contribute to the maintenance or enhancement of biological diversity. It is now recognized that such disturbances should be explicitly included in ecosystem management and conservation plans (Pickett et al. 1997; Fuhlendorf and Engle 2001).

In the IBMS approach, brush management techniques can be targeted for certain portions of a landscape and distributed across landscapes in both time and space such that mosaics of vegetation structures, patch sizes, shapes, and age states are created. This, in turn, would promote the co-occurrence of suites of insect, reptile, mammalian, and avian species with diverse habitat requirements (Jones et al. 2000). The logistics of planning and applying spatially heterogeneous brush management practices at appropriate scales is facilitated by advances in geomatics (e.g., global positioning satellites, geographic information systems, and remote sensing imagery) and landscape ecology that allow habitat and population data to be readily linked over large areas. Thus, a low-diversity shrubland or woodland developing on a grassland site can be transformed to a diverse patchwork of grassland–savanna–shrubland communities using a spatial placement of landscape treatments that promotes biological diversity at multiple scales (Scifres et al. 1988; Fulbright 1996).

Rocky Mountain elk in a sagebrush community where encroaching western juniper trees have been cut to preserve habitat for sagebrush associated wildlife. (Photo: K. W. Davies)

Livestock Response

Livestock grazing contributes significantly to the economy and social fabric of most rural communities. Brush management is a tool used to restore native ecosystems that have the capacity to provide a steady source of forage for livestock while facilitating other uses and resource values (NRCS 2006; US Department of the Interior, Bureau of Land Management 2007). The decision of whether to apply brush management for the betterment of domestic livestock is influenced by numerous factors, including the extent to which declines in carrying capacity (Olson 1999), animal performance (Ralphs et al. 2000), animal loss from poisoning (Williams 1978; Panter et al. 2007), animal handling (Hanselka and Falconer 1994), and animal health (Teel et al. 1998) will be impacted. Even when grazing has contributed to shrub increases, simply removing livestock or reducing their numbers is unlikely to remedy a brush encroachment problem (Browning et al. 2008). Passive treatments may help, but in many situations aggressive intervention is necessary (Olson 1999). Livestock can be used as part of the vegetation treatment program, especially when goats and sheep are used to apply browsing pressure on unwanted shrubs and weeds (Riggs and Urness 1989; Frost and Launchbaugh 2003).

Studies quantifying forage response to reductions in brush cover are relatively numerous (Fig. 8), but few have quantified direct commodity (livestock) benefits. Potential changes in livestock carrying capacity for contrasting brush management × precipitation scenarios are illustrated in Figure 11. These projections illustrate that a range in livestock returns should be anticipated because of differences in forage response during favorable, normal, and unfavorable rainfall conditions that may occur over a 20-yr horizon. In this example, using cattle prices equal to the average of the past 20 yr, current operating costs, and current costs of brush management practices, the returns on aerial spraying and mechanical practices may be relatively high when environmental conditions support high levels of herbaceous production. However, returns are greatly reduced when conditions for plant growth are poor. Subjective projections such as these are based on the best available information, and actual results are known to vary widely, depending on the specific situation. As research continues, more accurate and reliable projections can be developed.

Increases in available forage following brush management do not necessarily warrant an increase in livestock numbers. In some cases, justification for brush management may be to maintain stocking rates nearer its true capacity (i.e., recognition that current stocking rates cannot be sustained). For example, big sagebrush management on public land helps avoid potential conflict and lawsuits with grazing permittees and environmentalists because positive steps are taken to reduce grazing pressure without forcing major herd reductions (Torell et al. 2005a). Similarly, forestalling the need for controversial grazing reductions was a primary benefit of the 11-yr (1962–1972) Vale Rangeland Rehabilitation Program initiated in eastern Oregon (Bartlett et al. 1988). In the case of big sagebrush, brush management is not always acceptable because of its adverse impact on habitat for sagebrush-obligate wildlife species (Rhodes et al. 2010).

Reductions in brush and weeds potentially benefit livestock operators by increasing grazable land area (McDaniel et al. 1978). However, returns based solely on gains in animal performance are not always economically justified, especially when public assistance is not available (McBryde et al. 1984; Torell et al. 2005b). Lee et al. (2001) found that costs for brush management projects in the Edwards Plateau area of Texas exceeded livestock returns by 7–31%. Similarly, Torell et al. (2005b) found that a 30% cost-share agreement was required to justify big sagebrush management in New Mexico when the added forage from the brush management practice was valued at an intermediate level of $7/AUM (in 2003 dollars). A range improvement practice that increases forage during critical or limiting seasons can be economically feasible (Evans and Workman 1994).

While other resources (soil, water, wildlife, etc.) may benefit from IBMS, the economics of brush management practices continue to be evaluated on the basis of the amount of forage and meat products gained by implementing the practice (Tanaka and Workman 1988; Watts and Wamboldt 1996; Lee et al. 2001). The economic component of the holistic decision support system PESTMAN (2009) is driven by the anticipated forage response to brush management. Yet, as noted over 30

At advanced stages of shrub encroachment, brush management can improve biological diversity while potentially benefitting livestock production by increasing grazable land area. (Photo: K. W. Davies)

FIGURE 14. Overland flow is an important mechanism of runoff generation for many semiarid landscapes. For woodlands such as the pinon–juniper stand in the Jemez Mountains, New Mexico, pictured here, much of the runoff and erosion is generated from the intercanopy spaces. (Photo: Bradford Wilcox)

yr ago by Smith and Martin (1972), most range improvements show a negative benefit/cost ratio (costs exceed benefits) when based only on the value of the added forage for livestock production. This is a consistent and continuing conclusion that increased returns from improved animal performance and production are often too low for brush management to be economically justified (McBryde et al. 1984; Lee et al. 2001; Torell et al. 2005a). Landowners recognize this, and many if not most recognize other benefits to conducting brush management beyond livestock production. Additionally, most landowners conducting a brush management project do so under cost-share arrangements with state and federal agencies. When the value of ecosystem goods and services beyond those associated with livestock production are taken into account, a more favorable picture of brush management begins to emerge (see Tanaka et al. this volume).

Watershed Function

The NRCS makes a number of assumptions related to the hydrological consequences of brush removal. These assumptions fall into three broad categories: 1) horizontal fluxes—the removal of WPs will reduce overland flow (surface runoff) and erosion, primarily by improving infiltration rates and increasing ground cover; 2) vertical fluxes—the removal of WPs will reduce the evapotranspiration (ET) and thus increase groundwater recharge; and 3) landscape effects—as a result of assumptions 1 and 2, the removal of WPs will reduce gully erosion and increase stream flow. We review the validity of these assumptions below on the basis of relevant literature. Our review is organized by the primary geographic regions in the United States where information is available: the Southwest, the Northwest, and the southern Great Plains.

Horizontal Fluxes—Surface Runoff and Erosion. The expectation that surface runoff and erosion are higher from woodlands or shrublands than from grasslands is implicit in the assumption that reductions in WP cover will reduce overland flow and water erosion (Fig. 14). In some cases, this is true, but in many cases, it is not. It is most likely in the xeric climates that support creosote bush shrublands and piñon–juniper (*Pinus* spp.–*Juniperus* spp.) and juniper woodlands. The influence of woody species encroachment on surface runoff and erosion depends on the impacts of encroachment on herbaceous vegetation and subsequently bare ground. Surface runoff and erosion increase when WP

FIGURE 15. Reductions in herbaceous ground cover resulting from grazing and woody plant encroachment can increase connectivity between bare patches and lead to higher runoff and erosion (Davenport et al. 1998). (**A**) pinyon–juniper woodland in the Jemez Mountains, New Mexico, and (**B**) creosote bush shrubland in southern New Mexico. (Photo: Bradford Wilcox)

encroachment decreases herbaceous vegetation and increases bare ground; however, if WP encroachment does not decrease herbaceous vegetation and increase bare ground, then surface runoff and erosion would not increase. Brush management does not always reverse the impacts of WP encroachment on surface runoff and erosion. In some cases, depending on the woodland type and the method of shrub management, surface runoff and erosion may actually increase.

Southwest. There is clear evidence that as desert grasslands transition to creosote bush, juniper, or mesquite shrublands or woodlands, there is more bare ground and better-connected interspaces, resulting in lower net infiltration, more surface runoff, and higher erosion (Fig. 15) (Parsons et al. 1996; Schlesinger et al. 2000; Mueller et al. 2008; Wainwright et al. 2000). However, the reverse has not been demonstrated. In other words, brush management on creosote bush shrublands does not necessarily curtail surface runoff and erosion (Tromble et al. 1974; Tromble 1978, 1980; Wood et al. 1991).

There has been relatively little work evaluating the hydrological implications of managing mesquite in the Southwest. Long-term watershed studies at the Santa Rita Experimental Range suggest that runoff and sediment yields may decline with mesquite removal (Lane and Kidwell 2003). The results are, however, equivocal because no pretreatment monitoring took place.

Surface runoff is a relatively small portion of the water budget in piñon–juniper woodlands (Gifford 1975), primarily because of internal storage within the hillslopes (Reid et al. 1999; Wilcox et al. 2003a). Surface runoff is higher when snowmelt occurs (Baker 1984; Wilcox 1994). Infiltration rates are higher under tree canopies than in the interspaces spaces (Reid et al. 1999), even though the hydraulic conductivity of canopy and intercanopy soils is similar (Wilcox et al. 2003b), likely because of the buildup of duff under that canopy. In these woodlands, small-plot infiltration studies indicate that shrub management has little effect or even a negative effect on infiltration rates (Gifford et al. 1970; Blackburn and Skau 1974; Roundy et al. 1978). Runoff and erosion are the highest following chaining and windrowing. If debris is left in place, there is little difference in surface runoff between treated and untreated locations (Gifford 1973). Watershed-scale experiments in Arizona indicate little effect of brush management on surface runoff (Clary et al. 1974; Collings and Myrick 1966). Although runoff may be relatively small in general, it may be much higher on woodlands occurring on slopes >10% (Wilcox et al. 1996a, 1996b). In these landscapes, cutting trees and leaving the slash in place has proven to dramatically decrease erosion rates, primarily because of increases in herbaceous cover (Hastings et al. 2003).

Northwest. Major shrublands of the northwestern United States are those dominated by sagebrush or western juniper

(*Juniperus occidentalis*). Erosion on sagebrush rangelands is generally very low (Coppinger et al. 1991). Effects of brush management on horizontal water fluxes in sagebrush landscapes are equivocal. Increasing surface runoff and erosion has been documented in some cases, and declines have been observed in others (Blackburn 1983; Brown et al. 1985). Mechanical treatments that disturb soil can increase runoff and erosion. For example, plowing reduced infiltration and increased runoff and erosion up to 12 yr (Gifford 1982). Soil erodibility was initially ~2-fold greater in burned compared to unburned sagebrush communities, but was comparable 1 yr postfire, and infiltration and runoff was comparable on burned and unburned hillslopes in the first and second years postfire (Pierson et al. 2001). However, in another study, burning had little effect on runoff but resulted in a large increase in erosion (Pierson et al. 2008). Balliette et al. (1986) found little change in infiltration, runoff, or erosion following herbicide treatment. In contrast, Blackburn and Skau (1974) found that plowing and reseeding of big sagebrush increased infiltration rates and lowered surface runoff. Effects can also vary with season. At the small catchment scale (2–4 ha), summer runoff and erosion declined by 75% and 80%, respectively, following conversion of sagebrush to introduced grasses, whereas snowmelt runoff increased 12% (Lusby 1979). The contrasting results from this population of studies likely reflect differences in responses of the herbaceous understory to sagebrush clearing and differences in disturbance impacts associated with various brush management techniques.

Western juniper has been aggressively encroaching into sagebrush communities across the intermountain West (Miller et al. 2005). Subsequent to its establishment, western juniper excludes other vegetation and increases bare ground (Miller et al. 2000). Although Belsky (1996) found little compelling evidence that surface runoff and erosion were higher following western juniper encroachment, other work suggests that runoff and erosion can be significantly accelerated and that brush management can significantly mitigate these effects (Buckhouse and Mattison 1980; Gaither and Buckhouse 1983). Indeed, Pierson et al. (2007) found that 10 yr after juniper removal, treated hillslopes had significantly more vegetation cover, higher infiltration rates, and 15-fold less erosion than nontreated sites.

Southern Great Plains. In the southern Great Plains, the major shrublands of concern are those dominated by mesquite or juniper. Most of the research related to WPs, and water has been conducted in relation to Ashe juniper (*Juniperus ashei*) in the Edwards Plateau with some additional work completed on mesquite woodlands in the Rolling Plains and the South Texas Plains.

Extensive woodlands dominated by Ashe juniper occur on the Edwards Plateau of central Texas. As with other juniper woodland types, there is a widely held perception that encroachment by this WP has promoted surface runoff and erosion. However, there is little evidence in support of this assumption. Infiltration rates within this woodland type are relatively high, and erosion is low unless the area is heavily grazed (Hester et al. 1997; Wilcox et al. 2007, 2008b; Taucer et al. 2008).

For mesquite shrublands in the Rolling Plains of northern Texas, small-plot rainfall simulations indicate that shrub management may improve infiltration capacity and reduce erosion as a result of increased herbaceous cover (Bedunah 1982; Brock et al. 1982). Larger-scale plot and catchment studies, however, suggest the honey mesquite management would not significantly alter surface runoff and erosion (Carlson et al. 1990; Wilcox et al. 2006). Weltz and Blackburn (1995) reached a similar conclusion for mesquite–mixed shrub rangelands in the Rio Grande Plains.

Vertical Fluxes, ET, and Groundwater Recharge. WPs have the potential to alter the fluxes of water moving in a vertical direction, ET, and recharge by virtue of the fact that deep root systems allow WPs access to water not available to more shallow-rooted vegetation. The ability of WPs to access deep water is, however, modulated by soil depth, texture, and the underlying geological structure, the latter also being a key determinant of whether groundwater recharge events will affect stream base flow. In principle, in locations where WPs are accessing deeper water, there is the potential

> "There are few examples demonstrating that brush management enhances ground water recharge or streamflow."

"Historical stream flow records in the Edwards Plateau indicate that base flows have actually increased substantially since 1960 in spite of the fact that WPs have increased markedly since that time."

to use vegetation management to enhance recharge. In practice, however, there are few examples of where this has been demonstrated and then only at relatively small scales.

Southwest. There is little difference in ET between creosote bush shrublands and desert grasslands (Small and Kurc 2003; Kurc and Small 2004). Recent work suggests that removal of shrubs could increase groundwater recharge but not in amounts that would appreciably affect water supplies (Sandvig and Phillips 2006).

Recharge rates in most piñon–juniper woodlands are very small, and it is unlikely that brush management would lead to higher recharge (Newman et al. 1997; Sandvig and Phillips 2006). However, decreasing piñon–juniper cover by chaining increased soil moisture in the upper 60–90 cm of the soil profile, with only minor differences at greater depths (Gifford and Shaw 1973). We are not aware of any work comparing ET between piñon–juniper woodlands and comparable grassland areas.

Northwest. Removal of sagebrush can increase soil water content and presumably recharge (Sturges 1993; Seyfried and Wilcox 2006). Sturges (1993) suggested that reductions of sagebrush cover can increase water yield if sagebrush roots are not confined to the same volume of soil as grass roots. Along these lines, Darrouzet-Nardi et al. (2006) found that sagebrush in herbaceous meadows in the Sierra Nevada Mountains was in fact accessing deeper water than the herbaceous vegetation. Sagebrush management decreases water withdrawal from the upper 1 m of soil for 2 yr posttreatment (Sonder and Alley 1961; Cook and Lewis 1963; Tabler 1968; Shown et al. 1972; Sturges 1977). However, over longer periods of time, water depletion to 0.9-m soil depth can be greater where sagebrush is removed compared to where it is not because of increases in herbaceous vegetation production (Sturges 1993). The replacement of sagebrush by nonnative annual grasses and forbs (e.g., Fig. 12) can alter the timing of ET and patterns of soil moisture storage. For example, Prater and De Lucia (2006) found that early spring ET rates were higher from areas converted to cheatgrass (*Bromus tectorum*), an exotic annual grass, than for native sagebrush.

Large scale brush management programs focusing on mesquite rangelands in Texas have not resulted in increased streamflow, in spite of public perceptions to the contrary. (Photo: Bradford Wilcox)

Interception by western juniper canopies can reduce the amount of precipitation reaching the ground by 20% at the edge of the canopy, 50% halfway between the canopy edge and the trunk, and 70% at the trunk (Young et al. 1984). Stem flow is low, and thus the moisture captured in western juniper canopies is lost through evaporation (Miller et al. 2005). Cutting western juniper increases soil water throughout the growing season in at least the first 2 yr posttreatment (Bates et al. 2000). We are not aware of longer-term studies evaluating the influence of western juniper management on soil moisture.

Southern Great Plains. Ashe juniper intercepts 40–50% of rainfall (Fig. 16; Hester et al. 1997; Owens et al. 2006). Transpiration from an Ashe juniper community should be greater than that from an herbaceous community because evergreen Ashe juniper canopies can transpire much of the year in the subtropical portions of their range, and plants can access water to deep depths. Mature Ashe juniper trees transpire as much as 150 $L \cdot d^{-1}$, the equivalent of about 400 $mm \cdot yr^{-1}$ (Owens and Ansley 1997). Dugas et al. (1998), using the Bowen ratio/energy balance method, compared ET between intact and cleared Ashe juniper stands. For the 2-yr period following treatment, the difference in ET was about 40 $mm \cdot yr^{-1}$, but the treatment effects on ET disappeared in the third year, by which time ET was similar in treated and untreated areas.

For honey mesquite shrublands in the southern Texas plains, water balance studies suggest that conversion of mesquite to grasslands will increase recharge 15–20 $mm \cdot yr^{-1}$ (Weltz and Blackburn 1995; Moore et al. 2008). In the Rolling Plains of Texas, honey mesquite utilizes both deep and shallow soil water (Ansley et al. 1990, 1992a, 1992b), with individual plants using 30–200 $L \cdot d^{-1}$ and plants in open savanna settings using more water per tree than plants in dense stands (Ansley et al. 1991, 1998). At the stand scale, ET was comparable on cleared and uncleared honey mesquite rangelands (Dugas and Mayeux 1991); hence, the potential for increasing soil recharge or water yield by reducing mesquite cover in these systems is low (Carlson et al. 1990). Honey mesquite stands in the southern Great Plains can occur on fine, montmorillonitic clay soils with high shrink–swell potential. When dry, these soils develop extensive fissures that allow rapid and deep-percolation of rainfall. Mesquite removal on these soils reduced ET and increased soil moisture by about 80 $mm \cdot yr^{-1}$ (Richardson et al. 1979).

Landscape Effects. Streamflow. Brush management is commonly presumed to increase stream flow because of assumed increases in the base flow derived from increases in groundwater recharge. This has not been widely demonstrated except at the small-watershed scale, where stream flows are generated from winter precipitation (Huxman et al. 2005). A very prominent example of enhancement of stream flow subsequent to brush management is from chaparral shrublands characterized by winter rainfall (Rowe 1948; Ingebo 1972; Davis and Pase 1977; Hibbert 1983).

Increases in stream flow of ~150% were demonstrated on a 147-ha watershed following herbicide treatment in northern Arizona. Stream flow occurred mainly as a result of winter precipitation (Baker 1984). A larger-scale watershed treatment, however, failed to generate additional stream flow (Collings and Myrick 1966). Annual water yield initially increased 20% on the herbicide-treated sagebrush sites (Sturges 1994), then returned to pretreatment levels within 11 yr as sagebrush density increased. Small-watershed studies in western Colorado indicate that runoff from summer thunderstorms was reduced following conversion of sagebrush to grass (Lusby 1979). A paired watershed study in central Oregon indicated that late season spring flow may increase as a result of juniper management (Deboodt et al. 2009).

In the Edwards Plateau of Texas, Huang et al. (2006) found that spring flow increased by about 45 $mm \cdot yr^{-1}$ following Ashe juniper removal. Studies of juniper removal on small catchments where no springs were present found surface runoff was about 20% (13 $mm \cdot yr^{-1}$) lower following root plowing, which was attributed to increased surface roughness that enhanced shallow surface storage (Richardson et al. 1979). In another study, Dugas et al. (1998) found that when juniper cover was removed by hand cutting, the treatment had little influence on surface runoff from 4- and 6-ha small catchments. Similarly, Wilcox et al. (2005) found no change in runoff following juniper removal. Paradoxically, historical stream flow records in the Edwards Plateau indicate that base flows have actually increased substantially since 1960 in spite of the fact that WPs have increased markedly since that time (Wilcox and Huang 2010). The higher base flows were attributed

FIGURE 16. Rainfall simulation is a valuable tool for understanding how woody plants alter hydrological properties on rangelands. Large-scale rainfall simulation experiments, like the one conducted here on Ashe juniper rangelands, have the advantage of being able to apply known amounts of water above the tree canopies and enable quantification of canopy interception and water and erosion dynamics at the hillslope scale. (Photo: Bradford Wilcox)

"Projects that remove saltcedar and Russian olive with the intention of reducing ET and increasing flow in streams have produced mixed results, with most studies failing to demonstrate significant long-term changes."

FIGURE 17. Brush management is commonly applied with hopes of improving stream flow and groundwater recharge. However, studies indicating that brush management may not be achieving desired outcomes with respect to water yield are accumulating. Estimates of the economic benefits of shrub control based solely on water salvage are therefore questionable. However, it may be desirable to manage cover of nonnative shrubs, such as the tamarisk shown here, to enhance wildlife habitat, biological diversity, and soil health (Shafroth et al. 2005, 2010). (Photo: Charles Hart)

to the fact that ground cover has improved across the Edwards Plateau because of livestock destocking in the region. In the Rolling Plains of Texas, small-watershed and landscape-scale evaluations within the plains found little evidence that honey mesquite removal had an appreciable affect on stream flow (Wilcox et al. 2006, 2008a).

Early studies suggested that transpirational water loss from WPs such as saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) was substantially higher than that of native riparian vegetation. Expansion of these nonnative species along riparian corridors in the western United States was thus presumed to reduce river flows and groundwater supplies, and their removal was expected to promote stream flow and groundwater recharge (Fig. 17). However, recent studies indicate that saltcedar and Russian olive transpiration is on par with that of native species (Owens and Moore 2007), and projects that remove saltcedar and Russian olive with the intention of reducing ET and increasing flow in streams have produced mixed results, with most studies failing to demonstrate significant long-term changes (Shafroth et al. 2010).

Sediment Delivery. There are few studies of brush management effects on sediment yield at the catchment or watershed scale. Hastings et al. (2003) found that cutting trees and spreading slash in piñon–juniper woodlands in New Mexico significantly reduced erosion from 1-ha catchments. Lusby (1979) found that shrub management reduced erosion by 80% on two 4-ha sagebrush watersheds. Such studies suggest that brush management may help curtail erosion, but additional studies and studies at larger scales are needed before broad generalizations can be made with confidence.

Wildlife Response

NRCS goals of brush management for wildlife include 1) maintaining or enhancing habitat—including threatened and endangered species, with enhancements encompassing (a) slight to substantial improvement in cover, usable space, and habitat fragmentation; (b) improvement of imbalances among and within populations; and (c) neutral effects on endangered species—and 2) improving food accessibility, quality, and quantity. The challenge in meeting these goals lies with the fact that wildlife species and functional groups vary widely in their habitat requirements (Krausman et al. this volume). In addition, and as reviewed earlier, the response of vegetation and other habitat components to brush management varies, depending on a variety of factors. Impacts—positive, neutral, or negative—of brush management on wildlife therefore depend on a variety of factors (Fig. 7). Stating that brush management maintains or enhances wildlife habitat, consequently, is an oversimplification. Goals of brush management should be stated with the interacting factors that influence impacts on specific wildlife species taken into account.

Habitat is species specific, and habitat for one species may not serve as habitat for another species or group of species (Hall et al. 1997; Krausman 2002). Clearing a large tract of sagebrush to create grassland, for example, may improve habitat for grassland birds (Reinkensmeyer et al. 2007) but destroy habitat for sagebrush obligates (Klebenow 1969; Martin 1970; Green and Flinders 1980). A fundamental concept in wildlife management is that wildlife species vary in their response to disturbance. Northern bobwhites (*Colinus virginianus*), for example, are frequently considered "early ecological succession stage" species, whereas white-tailed deer (*Odocoileus virginianus*) are considered "midsuccession species" and grizzly bears (*Ursus arctos*) "climax" species (Bolen and Robinson 2002). This implies that bobwhites, for example, should respond positively to disturbance, whereas climax wildlife species may be negatively impacted by human-imposed disturbances, such as brush management.

Brush management may affect sexes of the same wildlife species differently (Leslie et al. 1996; Stewart et al. 2003). For example, male and female white-tailed deer selected different herbicide and fire treatments in Oklahoma (Leslie et al. 1996). Anticipated conservation benefits should be stated on the basis of the species, functional group (e.g., grassland birds, woodland birds, large mammals, small mammals, etc.) or the gender that they will benefit; broad generalizations that all wildlife will be benefited by brush management should be avoided.

Brush management may affect the same species differently, depending on seasonal use patterns of the habitat being treated. For example, thinning dense big sagebrush stands can benefit sage-grouse (*Centrocercus urophasianus*) during brood rearing (Dahlgren et al. 2006) but decrease its value as winter habitat for sage-grouse and other wildlife species (Davies et al. 2009b). Mechanical brush clearing during active nesting can destroy eggs and kill nestlings. The magnitude of the impact of brush clearing during active nesting on North American bird populations is unknown.

Wildlife species response to brush management can also vary by the species of brush. Sagebrush-obligate wildlife species are negatively impacted by reductions of sagebrush abundance (Klebenow 1969; Martin 1970; Green and Flinders 1980). However, sagebrush-obligate wildlife species benefit from control of western juniper encroaching into sagebrush communities (Miller et al. 2000; Reinkensmeyer et al. 2007).

Density and canopy cover of brush before treatment and amount of brush removed strongly influence wildlife responses to brush management. Clearing some brush in a landscape with a 100% canopy cover of WPs, for example, may benefit wildlife such as white-tailed deer (Fig. 18), whereas clearing brush in a landscape with only 25% canopy cover may be detrimental (Fulbright and Ortega-Santos 2006). In areas where the two species overlap, reducing WP canopy cover to < 50% favors mule deer (*Odocoileus hemionus*) over white-tailed deer (Wiggers and Beasom 1986; Ockenfels et al. 1991; Avey et al. 2003). Northern bobwhites use habitat patches where woody cover is ≥ 30%; therefore, reducing woody canopy cover in landscapes that marginally provide sufficient woody cover may be detrimental to bobwhites (Kopp et al. 1998; Ransom et al. 2008).

> "Anticipated impacts should consider game and non-game species; and should be tailored to specific species or functional groups."

Climate and Soils Mediate Outcomes.

Variation in precipitation and soil fertility may override effects of brush management on wildlife species abundance and richness in certain cases. Nutrition, productivity, and distribution of white-tailed deer, for example, may be more strongly related to variation in precipitation than to alterations in vegetation resulting from brush management. Seventeen years after root plowing, treated sites in the eastern Rio Grande Plains of Texas were dominated by huisache (*Acacia farnesiana* [L.] Willd.; Ruthven et al. 1994). Browse species important to white-tailed deer were either absent from the huisache communities that replaced the original honey mesquite-mixed brush communities or present in greatly reduced numbers compared to the mesquite–mixed brush community. Nutritional condition and population status of white-tailed deer, however, were similar in untreated and root plowed sites. Changes in body condition, reproduction, and diet were associated with variation in precipitation rather than with plant community differences. Similarly, patch burning and grazing had little effect on white-tailed deer distribution in southern Texas because drought limited vegetation response to the treatments (Meek et al. 2008). Lack of a difference in the use of aerated and aerated and burned patches by white-tailed deer has also been attributed to lack of precipitation, which constrained forb response to the treatments (Rogers et al. 2004).

FIGURE 18. Wildlife response to brush management is species and situation specific. Mule deer and white-tailed deer respond differently to changes in shrub cover, and the white-tailed deer depicted in this photo may respond positively when shrub cover is high but may be adversely affected by brush management imposed when shrub cover is <25%. (Photo: Tim Fulbright)

Brush management effects on wildlife food and cover vary with soil productivity (Fulbright et al. 2008). Root plowing may result in long-term loss of WPs that are important as browse for white-tailed deer on upland soils, whereas in ephemeral drainages, root-plowed sites supported brush communities similar in species composition and diversity to sites that had not been disturbed (Fulbright and Beasom 1987; Nolte et al. 1994). Ephemeral drainages receive runoff from uplands and tend to have more productive soils (Wu and Archer 2005). A possible explanation for the lack of reduction in species diversity in ephemeral drainages is that growing conditions are more favorable for the reestablishment of diverse WP species following root plowing than in upland sites.

Vegetation dynamics following brush management on fertile soils in mesic environments may follow directional change toward climax as predicted by traditional models of ecological succession. In arid or semiarid environments, however, vegetation change following disturbance may be nondirectional (Briske et al. 2005). Disturbance by brush management may push vegetation across a threshold to a different plant community than existed before treatment and one that is relatively stable. This new plant community may or may not provide better-quality habitat for specific wildlife species than the plant community that existed before brush management. For example, exotic annual grasses can rapidly increase and dominate plant communities after brush management in the intermountain West (Stewart and Hull 1949; Evans and Young 1985; Young and Allen 1997). Nonnative annual grass invasion in sagebrush communities decreases their habitat value for sagebrush-obligate and facultative wildlife (Davies and Svejcar 2008). Buffelgrass may increase following root plowing or disking in South Texas (Gonzalez and Dodd 1979; Johnson and Fulbright 2008) with adverse effects on bobwhite populations (Flanders et al. 2006). Thus, the potential for undesirable shifts in plant communities following brush management must be carefully considered before implementing treatments (see the sections "Biodiversity and Nonnative Species" and "A Tool to Promote Landscape Heterogeneity and Biodiversity").

Scale and Pattern. Effects of brush management on wildlife may vary dramatically, depending on scale of application (see also Bestelmeyer et al. this volume). Many grassland-adapted species may respond in a positive fashion to broad-scale conversion of woodland to grassland (Fitzgerald and Tanner 1992; Smythe and Haukos 2010). Conversely, these large-scale conversions reduce northern bobwhite and Texas tortoise populations (Kazmaier et al. 2001; Ransom et al. 2008). Extensive brush removal (>60% of the landscape) reduces landscape use by white-tailed deer (Rollins et al. 1988; Reynolds et al. 1992). Large areas of untreated brush provide habitat for many nongame bird species, and brush management efforts should be limited in scope in areas where conservation of this wildlife group is a priority (Fulbright and Guthery 1996).

Range management has traditionally promoted vegetation uniformity rather than heterogeneity (Fuhlendorf and Engle 2004). Promoting uniformity, deemed prudent for increasing livestock production, included practices such as clearing WPs completely from the landscape, planting monotypic stands of grasses, and taking steps to promote livestock grazing distribution. Wildlife needs were relegated to lesser importance in this traditional management approach. Wildlife response to amount and interspersion of brush patches varies among species. Many wildlife species reach maximum diversity or density in heterogeneous landscapes such as those containing a mosaic of brush and interspersed tracts dominated by herbaceous vegetation (Roth 1976; Tews et al. 2004a; see also the sections "Biodiversity," "Biodiversity Response," and "Biodiversity and Nonnative Species"). Diversity and richness of birds is greatest in plant communities with structural heterogeneity (Reinkensmeyer et al. 2007). For example, providing a mosaic of plant communities including closed-canopy oak forest and open pastures derived from forest increased breeding nongame birds richness in Oklahoma (Schulz et al. 1992). Brush management is commonly done in strips or other patterns to create mosaics of WP communities interspersed with communities dominated by herbaceous plants to benefit wildlife (Fulbright and Ortega-Santos 2006). Brush sculpting is another approach to brush management (Fulbright 1997; McGinty and Ueckert 2001). Brush sculpting refers to selective removal of brush to accomplish multiple-use objectives, such as habitat improvement for wildlife and increased forage for livestock (Ansley et al. 2003). Anticipated effects of brush management should take into account the extent to which habitat heterogeneity is important for wildlife species (Fulbright 1996; Kie et al. 2002; Tews et al. 2004a).

Patch size is also an important consideration when creating vegetation mosaics (Bestelmeyer et al. this volume). Selection of patch size depends on management objectives and the wildlife species or functional group being managed. Mosaics may be created to either maximize wildlife species diversity or optimize habitat for a particular species. Edge and interior species are more prone to be affected by patch size than are generalist species (Bender et al. 1998). A mosaic consisting of patches that are too small essentially functions as edge and does not provide habitat for interior species. For woodland-adapted birds, patch size and shape are important because nest parasitism and nest predation may increase with increasing edge, although this relationship has been questioned in recent literature (Patton

"Vegetation mosaics may be created to either maximize wildlife species diversity or optimize habitat for a particular species."

1994; Lahti 2001). Patches that are large with relatively little perimeter support fewer edge species.

Patch size and configuration requirements vary among wildlife species. Grassland birds, for example, require patches >50 ha (Helzer and Jelinski 1999). Ratio of patch perimeter to area is also important; bird species richness is greatest in patches with larger interiors that are free from edge effects. For grassland birds, landscape composition may interact with patch size in that larger core areas may be more important in landscapes with a mixture of grassland and woodland than in treeless landscapes (Winter et al. 2006).

Although the idea of creating patchy mosaics through brush management has been discussed in the literature, using brush management to achieve an "optimum" size and configuration of patches has received little attention (Fulbright 1996). Part of the reason for the lack of attention to the concept of optimal patch size/configuration may be that many of the game species that are often the focus of research on brush management effects are edge associates that show little response to variation in patch size. Northern bobwhites, for example, appear to be adapted to an almost infinite set of patch configurations; therefore, an "optimum" arrangement may not exist (Guthery 1999).

Brush management may increase connectivity and reduce habitat fragmentation for grassland-adapted species; conversely, brush management may fragment habitat of shrubland or woodland adapted species if the cleared areas limit wildlife movement between tracts of woody vegetation. Patches of habitat for a wildlife species should be linked by corridors that facilitate movements among habitat patches (Bennett 2003). Ensuring that connectivity exists among habitat patches should be a priority when vegetation is manipulated.

Improving Food. Brush management may improve food accessibility, quality, and quantity for some wildlife species or functional groups (e.g., grazers) but reduce it for others (e.g., browsers). A review of publications in the *Journal of Range Management*, *Rangeland Ecology & Management*, and *Ecology* and articles emerging from a search of BIOONE, JSTOR, Science Direct, and Springer using the search strings "brush management," "brush management wildlife," "herbicides birds," "brush control deer," "brush control prairie chicken," and "brush control sage grouse" yielded 50 articles addressing 59 cases of effects of brush management treatments or combinations of treatments (e.g., fire and herbicides) on wildlife food plants. Effects on food plants ranged from positive (53%) to neutral (32%) to negative (16%). In most cases, negative responses occurred where brush management reduced mistletoe (a parasitic plant on honey mesquite that is eaten by

Mechanically clearing juniper in strips provides edge and brush piles for wildlife, forage for livestock and opportunities for future use of prescribed fire as a management tool. (Photo: Kirk McDaniel)

Plant communities with mixtures of herbaceous- and shrub-dominated patches provide excellent habitat for a diversity of game and non-game species. (Photo: Tim Fulbright)

deer), reduced browse plants preferred by white-tailed deer, or increased thorns or secondary compounds in browse regrowth. In the review, we considered treatment effects to be neutral when they resulted in only temporary (<3 yr) increases in forb seeds or insects. Chemical, mechanical, and pyric brush management methods vary in their impact on woody and herbaceous food for wildlife. Chemical treatments, for example, tend to cause a temporary reduction in forbs, whereas fire may stimulate growth and abundance of early successional forbs that benefit many species of animals (e.g., Fig. 13) (Beasom and Scifres 1977; Bozzo et al. 1992a). Fire may top kill WPs, encouraging production of palatable sprouts (Schindler et al. 2004b). Anticipated benefits of brush management to wildlife should be predicated on the brush management approach to be used and the wildlife species potentially affected.

Endangered Species. Brush management potentially reduces habitat for endangered species that depend on WP communities, such as ocelots (*Felis pardalis*), which need woodland with >97% canopy cover, or pygmy rabbits (*Brachylagus idahoensis*), which forage primarily on big sagebrush (Green and Flinders 1980; Harveson et al. 2004). Conversely, brush management potentially could improve habitat for grassland-adapted species, such as the Attwater's greater prairie chicken (*Tympanuchus cupido attwateri*). Documentation of the effects of brush management on habitat of species listed as endangered in the United States is lacking, however.

Herbicide Toxicity. Herbicides used in rangeland brush management are usually not used in concentrations harmful to wildlife and dissipate from the ecosystem following the growing season they are applied (Scifres 1977; Freemark and Boutin 1995; Guynn et

Wyoming big sagebrush mowed in strips creates a mosaic of treated and untreated sagebrush habitat to increase diversity and maintain critical habitat for sagebrush-obligate wildlife. (Photo: K. W. Davies)

al. 2004). Herbicides are generally not acutely toxic to soil organisms (Freemark and Boutin 1995). Certain aspects of herbicide toxicity to wildlife, such as the role of surfactants and inert ingredients, and possible synergistic effects of multiple chemicals applied simultaneously are unknown (Guynn et al. 2004). Herbicides may negatively affect insects directly or indirectly, but little is known of the effects of rangeland herbicides on these organisms. A better understanding is needed since native rangelands may serve as a reservoir of pollinator and predator insects important to crop production in nearby cultivated areas (Freemark and Boutin 1995). In addition, invertebrates are a critically important food resource for many grassland bird species (O'Leske et al. 1997). Research on herbicide effects on reptiles and amphibians is also lacking (Freemark and Boutin 1995; Guynn et al. 2004).

Although rangeland herbicides are generally not highly toxic to wildlife, acute effects of the herbicide 2,4-D have been documented. The herbicide is toxic to cutthroat trout (*Salmon clarkia*) (Woodward 1982). Spraying 2,4-D dramatically reduced pocket gopher (*Thamomys talpoidis*) populations in Colorado (Keith et al. 1959).

Predators. Anticipated benefits of brush management stated by NRCS focus largely on forage production and habitat structure for herbivores; however, brush management also alters predator habitat and may change behavioral responses of prey. Ungulates, for example, may use cleared patches within woodland or shrubland because of enhanced visual detection of predators (Bozzo et al. 1992b). Florida panthers (*Felis concolor coryi*) are attracted to recent prescribed burns where prey species such as white-tailed deer congregate (Dees et al. 2001). Landscape-level reduction of brush may remove perching structures important for raptors and increase susceptibility to nest predators. Prickly pear (*Opuntia* spp.) control, for example, has the potential to reduce nest sites and increase nest susceptibility to predators for bird species that prefer nesting

in prickly pear. Treating prickly pear with herbicides, however, did not reduce nesting success of bobwhites in central Texas (Hernandez et al. 2003). Prey population densities may also change in response to brush management. Effects of mechanical brush management on the mortality of small mammals and immobile wildlife species at the time of treatment are unknown. Habitat changes following treatment may have unintended consequences, such as favoring increased prey densities. For example, cotton rat (*Sigmodon hispidus*) densities were six times greater on root-plowed rangeland in Texas than in untreated rangeland (Guthery et al. 1979). Rodent populations are strongly cyclical. Flushes in rodent abundance may be followed by increases in predator abundance; but subsequent abrupt declines in rodent populations may cause the now-abundant predators to shift to a prey base of livestock or ungulates such as white-tailed deer.

Brush management may also affect visual cues used by predators to locate prey. Logged areas in the boreal forests of Canada have less debris on the forest floor than uncut stands. Efficiency of predation by martens (*Martes americana*) is greater in uncut timber stands because coarse woody debris act as sensory cues and enhance hunting success (Andruskiw et al. 2008). Brush management may likewise affect structure and amounts of woody debris in shrubland habitats, potentially affecting predator efficiency. Herbicide application may have little influence on habitat use by coyotes (*Canis latrans*) and bobcats (*Felis rufus*) possibly because standing woody material remains after treatment and herbaceous community structure is not drastically altered (Bradley and Fagre 1988).

Treatment Longevity. Brush management initially reduces shrub canopy cover, but over time, stem and foliage cover returns. In Texas, the estimated duration of treatments range from 10 yr to 20 yr for root plowing and from 3 yr to 9 yr for roller chopping (Fulbright and Taylor 2001; Schindler and Fulbright 2003). Potential benefits of brush management for wildlife, therefore, are transient. Brush management, for example, may benefit a wildlife species initially, but as the WP community reestablishes (e.g., Fig. 10), benefits may be lost. The temporary nature of treatments and the need for follow-up treatments must therefore be explicitly considered in statements of anticipated benefits (see the previous sections “Integrated Brush Management Systems” and “Treatment Options”).

Single applications of mechanical brush management with no follow-up treatments may adversely impact wildlife habitat. For example, density of WPs palatable to white-tailed deer may be lower in WP communities that reestablish following root plowing than in undisturbed communities (Fulbright and Beasom 1987). Density of woody legumes such as honey mesquite and huisache may be greater on root plowed areas than on untreated areas >17 yr posttreatment (Fulbright and Beasom 1987; Ruthven et al. 1994). WPs that regenerate following roller chopping may have longer and more numerous spines than undisturbed plants, which could reduce bite rate of browsers (Schindler and Fulbright 2003; Schindler et al. 2004a).

Measuring Habitat Improvement. The statement of anticipated benefits of brush management to wildlife is based on the assumption that improvements in food, cover, space, imbalance among populations, and fragmentation are evidence of habitat improvement. Vegetation characteristics are commonly linked with habitat quality in the wildlife literature (Guthery 1997; Hall et al. 1997; but see Johnson 2007). However, increases in a specific habitat characteristic do not constitute improvement if that characteristic is not limiting to wildlife (Guthery 1997). For example, rangeland disking may increase abundance of seed-producing forbs. However, seeds may not be limiting to northern bobwhites (Guthery 1997). In this case, the assumption that increasing food (e.g., seed-producing forbs) resulted in habitat improvement may not be valid. Further, numerous confounding factors exist in natural ecosystems, and an increase in food and cover alone may not result in habitat improvement if some other factor, such as nesting cover, is limiting.

Brush management is assumed to have improved wildlife habitat quality in an area if it results in greater food abundance, better interspersion of plant communities, and habitat requirements, less fragmentation, or better cover characteristics. An underlying assumption is that population density in an area increases with increasing habitat quality (Guthery 1997). However, increased densities following brush management does not necessarily indicate sustained improvement in habitat. Treated areas may provide resources needed by an organism only during part of the year, and untreated areas may be needed to meet needs during other times of the year. White-tailed deer, for example, do not exhibit preference for a particular level of woody canopy cover during winter, but during summer, deer densities increase with increasing WP cover, with areas >80% canopy cover receiving greatest use (Steuter and Wright 1980). Improvements in habitat quality should be expressed in terms of increased survival and reproduction in addition to increased population densities and availability of key habitat components (Van Horne 1983; Hall et al. 1997; Crawford et al. 2004). For northern bobwhites, evidence that their abundance increases with habitat quality variables such as food supplies and interspersion is limited and equivocal (Guthery 1997). Instead, abundance of bobwhites is proportional to the amount of usable space (habitat for which a species is fully adapted), and only practices that increase the abundance of usable space are likely to improve bobwhite numbers (Guthery 1997; Guthery et al. 2005). The usable space concept has also been applied to white-tailed deer management (Hiller et al. 2009).

Demographic characteristics of wildlife populations and usable space are more difficult and time consuming to quantify than habitat characteristics such as food production. As a result, comparisons of survival

Brush management may improve food accessibility, quality, and quantity for some wildlife species or functional groups but reduce it for others. (Photo: Tim Fulbright)

and reproduction of wildlife on sites with and without brush management are limited (Appendix II). Consequently, restricting statements of anticipated benefits to treatments and species for which increased reproduction, survival, and density or increases in usable space resulting from brush management have been documented is impractical. A better approach would be to acknowledge that while brush management may improve various habitat properties, its impact on habitat quality for many species is unclear.

Fuels Management

Brush management is increasingly being applied in shrubland and woodland settings to reduce fire risk or create fuel breaks (Keeley 2002; Davies et al. 2009b); however, little information is available to evaluate its effectiveness. In forest systems, mechanical brush management alters fuel characteristics and influences fire behavior (Kane et al. 2009); however, current fire models have not yet been parameterized to represent these modified behaviors. Although the impact of brush management on fire characteristics and spread are unclear, fire suppression efforts can be facilitated simply by reducing fuel height (Keeley 2002). However, while brush management can effectively reduce the mass and continuity of canopy fuels, it may promote production and continuity of fine surface fuels (e.g., grasses) and thus promote fire risk (Keeley 2002; Perchemlides et al. 2008; Huffman et al. 2009).

RECOMMENDATIONS

- Care is needed when using words and phrases such as "vigor," "health," "biodiversity," "encouraging growth," and "suitable" when projecting the effects of brush management. These terms are vague or ill-defined and often value laden and should be replaced with words and phrases that refer to specific and tractable metrics to define more specific and measurable conservation outcomes.
- Integrated Brush Management Systems have proven effective in WP management and are likely to yield the greatest conservation benefits. Brush management is a long-term commitment. Adaptive management, coordination with grazing management, a plan and funding for follow-up restoration and brush treatments, and periodic monitoring are essential. Emphasize flexibility and objectivity.
- Customize brush management prescriptions according to the stakeholder's vision and management objectives and the inherent capability or limitations of the ecological site. This perspective on human dimensions should be incorporated into the list of purposes in practice code 314: "Work closely and cooperatively with clientele to apply brush management practices that meet both land and personal conservation objectives."
- Evaluate and define when, where, how, and under what circumstances brush management should be undertaken and what specific outcomes are to be attained. Recommendations should be thoroughly vetted and justified. Do not assume that brush management is needed simply because shrubs are present.
- Tailor statements of potential hydrological benefits of brush management to specific bioclimatic zones.
- Anticipated effects of brush management should take into account the extent to which habitat heterogeneity is important for wildlife species. Do not assume that brush management will result in improvement of habitat for a wildlife species or functional group. Tailor statements of anticipated benefits of brush management to specific habitat variables or characteristics, such as food production, and to specific wildlife species or functional groups. State which wildlife species or functional groups may be negatively impacted by brush management under specific sets of circumstances.
- Develop and maintain a relational database to evaluate brush management treatments. Important information may include treatment approaches and longevities; location and spatial pattern(s) of treatment in relation to soils and topography; pre- and posttreatment soil, plant, livestock, and wildlife responses; environmental conditions; and predicted trade-offs and outcomes based on published literature (Table 6). This database should be updated as new information becomes available and used to communicate anticipated benefits for specific locations and regions.

TABLE 6. Example of a matrix approach to communicating anticipated benefits of brush management for wildlife. A similar matrix could be developed for plants, soils, and so on.

Brush management approach	Scale	Climate	Existing woody canopy cover (%)	Wildlife species or group	Anticipated impact[1]
Mechanical	Landscape	Humid	60–100	Grassland obligates	+
				Woodland obligates	–
				Edge-associated species	–
				Habitat generalists	0
			25–59	Grassland obligates	+
				Woodland obligates	–
				Edge-associated species	–
				Habitat generalists	0
			<25	Grassland obligates	+
				Woodland obligates	–
				Edge-associated species	–
				Habitat generalists	0
Chemical, fire	Mosaic, patch	Subhumid, semiarid, arid	<25	Habitat generalists	

[1]+, positive; 0, neutral; –, negative.

- Seeding of nonnative plants following brush management should be avoided, but if considered, it should be explicitly justified.
- Articulate and critically evaluate positive and negative trade-offs in brush management impacts on various ecosystem goods and services. For example, gains in livestock production and herbaceous diversity accruing from brush management may be at the expense of ecosystem carbon sequestration.
- Develop a mechanism to integrate conservation planning on individual properties into and consistent with local/regional conservation plans. Specific goals and objectives from brush management may vary by ownership and agency, but by pooling expertise and financial resources, there will be better opportunities for treating and restoring larger areas.

KNOWLEDGE GAPS

- The extent to which pre–brush treatment management conditions drive posttreatment responses is largely unknown, as are the effects of follow-up treatments.
- Projected effects of brush management mention numerous variables related to air quality as "not applicable." However, available information, albeit scant, suggests that changes from grass to WP dominance can significantly increase emissions of trace gases and volatile organic carbon compounds and the production of dust, aerosols and allergens. The extent to which brush management might reverse these is unknown, as are the implications for human health, tropospheric chemistry, and land surface–atmosphere interactions.
- ANPP can be dramatically enhanced by shrub encroachment (Knapp et al. 2008a; Barger et al. 2011), but the effects of brush management on ANPP are largely unknown. Plant production responses to brush management have focused on the herbaceous vegetation, and there is scant data on WP ANPP during the postmanagement period. Thus, we are ill equipped to evaluate brush management from a carbon-accounting perspective.
- The belowground organic carbon pool (roots+soil) typically dwarfs the aboveground pool in rangeland ecosystems. Robust generalizations as to how WP encroachment (Fig. 4) and brush management affect this large belowground

pool are not yet possible. Studies that have quantified soil responses to brush management are few (Fig. 8) and have relied on comparing random samples from a treated site(s) to a nearby, untreated site. Given the extensive edaphic heterogeneity on shrub-encroached rangelands (e.g., Bai et al. 2008; Liu et al. 2010), such coarse comparisons are probably not too reliable. Studies quantifying soil resources in a spatially explicit fashion before and following brush management are sorely needed, as are studies quantifying the response of shrub roots to brush management. Decreases in plant and SOC pools that may occur following brush management could have important but as yet poorly understood implications for ecosystem carbon management.

- Quantification of trade-offs between livestock production, hydrology, erosion, carbon sequestration, biodiversity, and so on and approaches for weighting them is a current challenge that must be addressed if we are to advance our ability to comprehensively evaluate the conservation value of brush management. Brush management has the potential to modify the provisioning of numerous ecosystem services at both local and regional scales. Attempts must be made to monitor and value these nontraditional nonmarket services.
- Many of the potential benefits of brush management depend on the extent to which herbaceous production and ground cover can be reestablished and the duration of the herbaceous response. General models of WP effects on herbaceous vegetation (Fig. 3) need to be better quantified to determine when it might be most effective to implement brush management, and conceptual models of posttreatment herbaceous vegetation response to brush management (Fig. 10) need to be made operational to obtain quantitative ecological (Fig. 9) and socioeconomic (Fig. 11) assessments of

Woody plant encroachment represents a threat to grassland, shrub-steppe, and savanna ecosystems and the plants and animals endemic to them. (Photo: Tim Fulbright)

brush management. Simulation modeling has been underutilized (Fig. 8). Given the advent of inexpensive, user-friendly software for personal computers, this tool can now be readily used to integrate existing information for assessment, scenario development, and forecasting (e.g., Grant et al. 1999; Fuhlendorf et al. 2008).

- The major knowledge gap related to brush management and water is our limited understanding of landscape-level implications. With the exception of a few studies (e.g., Collings and Myrick 1966; Wilcox et al. 2008a), there has been little documentation of the large-scale impacts of brush management on water and erosion processes. As a result, there is considerable uncertainty concerning the efficacy of extrapolating from fine-scale studies to the landscape level (Wilcox and Huang 2010).
- Biodiversity responses to shrub encroachment are poorly documented, and responses to brush management have focused largely on herbaceous vegetation. Responses of various faunal groups, including soil biota, are few and scattered. The implications of changes in biodiversity for ecosystem function have been the topic of much discussion in the research community but remain poorly understood.
- Brush management effects on wildlife have focused mainly on game species, particularly white-tailed deer, northern bobwhites, and sage-grouse. Nongame species, including predators, passerines, small mammals, and reptiles, have been largely neglected. Habitat requirements of many nongame species are not well understood, making it challenging to even speculate about effects of brush management. These gaps must be filled for statements of anticipated benefits to be made for specific species or functional groups.
- The extent to which brush management-induced changes in habitat attributes translate into improvements in carrying capacity and animal birth rates, longevity, nutritional status and body mass are largely unknown.
- Further research that addresses the interrelationship between brush management and fire behavior is needed to provide robust conclusions on its effectiveness for reducing fire risk and spread. Trade-offs between reducing WP canopy mass and continuity and promoting fine fuel production needs further study among different WP communities.
- A framework for conceptualizing how climate change, invasions of nonnative species, and increases in atmospheric CO_2 and nitrogen deposition might influence future grass–woody states and ecosystem responses to brush management is needed.

CONCLUSIONS

Successful long-term management programs (typically >5 yr) usually involve an integrated

Rangeland conservation goes beyond traditional concerns of livestock production to include potential effects on a variety of ecosystem services. The research community is challenged with measuring and monitoring these varied impacts; and the management community with creating or maintaining woody-herbaceous mixtures in arrangements that satisfy competing objectives. (Photo: Tim Fulbright) brush management systems and restoration approach that includes a suite of mechanical, fire, biological, and chemical methods. A combination of methods customized for local ecological site conditions is particularly important when the primary objective is to achieve long-term native plant stability that supports conservation and resource function.

Assessing revegetation potential is a critical first step before proceeding with brush management. Brush management and revegetation costs are high, and careful selection of areas with a high potential for reestablishment is necessary for long-term, sustainable brush management. In many situations, herbaceous vegetation on treated areas will recover naturally after brush management without revegetation. In other situations, planting or seeding of grasses or forbs may be necessary. Sites with particularly dense brush cover, poor hydrologic integrity, or related conditions may have limited revegetation potential. An in-field evaluation and soil survey should always be used to evaluate soil and other factors that will ultimately influence replacement of the vegetation community. With these caveats in mind, our synthesis suggests the following conclusions regarding the conservation value of brush management:

- Conservation of grasslands and savannas as ecosystem types and the plants and animals endemic to them should be a high priority (Fig. 19). Loss of grassland-obligate organisms occurs with shrub encroachment, even if overall numerical biological diversity is enhanced or unaffected. Brush management programs are essential to maintain grassland, steppe, and savanna ecosystems and the biodiversity and services they provide. Progressive brush management protocols will be required to achieve this conservation goal in many instances.
- Herbaceous cover, production, and diversity are typically enhanced by brush management. However, exceptions occur, and the possibility for deleterious outcomes should always be anticipated and considered when planning. Furthermore, treatment longevity will vary, so plans for follow-up are required.

FIGURE 19. Brush encroachment threatens habitat for grassland-obligate species such as this savanna sparrow. Brush management may be required to generate and maintain shrub cover amounts and patterns within acceptable limits for such species. (Photo: Tim Fulbright)

- Returns arising from improved livestock performance and production are important, but benefits beyond livestock production are being increasingly recognized. When the value of ecosystem goods and services beyond those associated with livestock production are taken into account, a more favorable picture of brush management begins to emerge.
- Although frequently justified on the basis of benefits to water quality and quantity, brush management does not necessarily produce the hydrological benefits that are commonly attributed to it. In most cases, these perceived benefits are exaggerated and have not been documented, and there is little or no evidence that brush management is a viable strategy for increasing ground water recharge or stream flows at meaningful scales. Outcomes depend on the vegetation type and geological setting. In some cases, depending on the vegetation community and the method of shrub management, surface runoff and erosion may actually increase. Local/regional knowledge should therefore guide brush management prescriptions with respect to hydrological impacts. In settings where winter precipitation predominates or where WPs are accessing deep stores of water, there is the potential to use vegetation management to enhance groundwater recharge and stream flow. However,

A burned (left) and untreated (right) mountain big sagebrush plant community on the Hart Mountain National Wildlife Refuge in southeastern Oregon. (Photo: K. W. Davies)

projections for how this translates to watershed- and regional-scale hydrology is based more on speculation than data.

- Statements that brush management maintains or enhances wildlife habitat are oversimplifications. Habitat requirements of many nongame species are poorly understood, making it challenging to even speculate about effects of brush management on these organisms. Clearer definitions of what constitutes a benefit of brush management to wildlife are needed, and these should be tailored to species or functional groups. Statements should focus on the habitat characteristics or attributes that are anticipated to be improved.

Technology and the tools available for brush management are dynamic and ever changing. Keeping educated and up to date on new developments is paramount. There are knowledge gaps in brush management, but there always will be, and it is important that managers strive to use the best available information. In some instances, practices applied and approaches followed to manage a particular WP species may

not be known. Thus, it is recognized that land managers are often placed in situations where they must exercise flexibility, responsibility, and their best professional judgment when developing a planning strategy and carrying out an action program.

Brush management presents a series of dilemmas and challenges as a response to WP encroachment. The recognition that WP proliferation can substantially promote ecosystem primary production and carbon stocks may trigger new land use drivers as industries seek opportunities to acquire and accumulate carbon credits to offset CO_2 emissions. WP proliferation in grasslands and savannas may therefore shift from being an economic liability in the context of livestock production to a source of income in a carbon sequestration context. Policy and management issues related to grazing land conservation thus extend well beyond the traditional concerns of livestock production and game management (wildlife valued for sport hunting) to include potential effects on hydrology, carbon sequestration, biological diversity, atmospheric chemistry, and the climate system. The research community is challenged with quantifying and monitoring these varied impacts and the management community with devising approaches for creating or maintaining woody–herbaceous mixtures in arrangements that satisfy competing conservation objectives.

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APPENDIX I. Herbaceous Response to Brush Management

A search of articles with the key word "brush control" in the *Journal of Range Management* and *Rangeland Ecology & Management* at http://www.uair.arizona.edu/search?page_set = 51 yielded 1277 separate articles. Among these, about 80% (1021 articles) either assessed target plant mortality (764 articles) or described changes in herbaceous (grass and broadleaf) plant abundance (e.g., cover, biomass, and frequency; 257 articles). Of the 257 articles reporting on herbaceous responses, 216 (84%) characterized the response as positive, 21 (8%) reported no change, and 20 (8%) report a negative response to brush management.

In another, more directed search, we sampled published accounts of how brush management influences herbaceous vegetation. Web of Knowledge searches resulted in 532 unique references, 36 of which were field studies conducted on rangelands in the United States and 22 of which measured the response of herbaceous or grass production.

Among these 22 studies, herbicide was the most frequently assessed brush management technique (15 studies, or 68%). As with our initial, broader survey, most of these (18 studies, or 82%) reported increases in herbaceous production. The majority of experiments were conducted over short periods of time, with only eight studies (36%) lasting more than 5 yr and only five (23%) lasting longer than 10 yr.

APPENDIX II. Brush Management and Wildlife Habitat Quality

- Peer-reviewed publications were surveyed to determine the proportion of studies that measured effects of brush management on wildlife density.
- A total of 97 publications emerged in this compilation, which included articles in the *Journal of Range Management*, *Rangeland Ecology & Management*, and *Ecology*, along with those emerging using the search strings "brush management," "brush management wildlife," "herbicides birds," "brush control deer," "brush control prairie chicken," "brush control sage grouse," "fire sagebrush," "sage grouse prescribed fire," and "prescribed fire" in the search engines BIOONE, JSTOR, Science Direct, and Springer.
- Only 45% of these articles reported some measure of organism abundance in response to brush management.
- Only about 5% reported the demographic information that Van Horne (1983) and Hall et al. (1997) suggest as necessary to assess habitat quality.

APPENDIX III. Citations for Data Points in Figures 4, 9, and 13

Data points in Figure 4 are from the following: 1–3 = Schlesinger and Pilmanis (1998); 4–5 = Asner et al. (2003); 6, 8–12, 17 = Geesing et al. (2000); 7 = Hughes et al. (2006); 13–15 = Boutton et al. (1998); 16 = Tilman et al. (2000); 18–19 = Mordelet et al. (1993); 20 = San Jose et al. (1998); 21–24 = Wheeler et al. (2007); and 25–34 = Jackson et al. (2002).

Data points in Figure 9 are from Ansley et al. (2006), Bedunah and Sosebee (1984), Clary (1971), Griffith et al. (1985), McDaniel et al. (1982), Morton et al. (1990; mechanical treatments), Augustine and Milchunas (2009), Bates et al. (2005, 2009), Cable (1967), Engle et al. 1993, 1998), Teague et al. (2008b; prescribed fire), Bedunah and Sosebee (1984), McDaniel et al. (1982), Morton et al. (1990; herbicides), and Engle et al. (1993; multiple treatments).

Data points in Figure 13 are from Baeza and Vallejo (2008), Davies et al. (2007), Edwards et al. (2007), Maccherini et al. (2007), Maron and Jefferies (2001), Nolte and Fulbright (1997), Nolte et al. (1994), Olson and Whitson (2002), Page et al. (2000), Ponzio et al. (2006), Ruthven et al. (1993), Ruthven and Krakauer (2004), and Sheley et al. (2006).