



Chapter 7:

Fire and Nonnative Invasive Plants in the Central Bioregion

Introduction

The Central bioregion is a vast area, stretching from Canada to Mexico and from the eastern forests to the Rocky Mountains, dominated by grasslands and shrublands, but inclusive of riparian and other forests. This bioregion has been impacted by many human-induced changes, particularly relating to agricultural practices, over the past 150 years. Also changed are fire regimes, first by native peoples who used fire for a variety of purposes and then by European settlers, who directly and indirectly contributed to a great reduction in the frequency of fire on the landscape. Perhaps of even greater importance has been the introduction of nonnative plant species, which have come to impact every community type to some degree.

Nonnative plants have a wide array of impacts on native ecosystems and populations in the Central bioregion, and these impacts continue to mount and evolve. Many long-time invaders, such as smooth brome (*Bromus inermis*), and leafy spurge (*Euphorbia esula*), have already spread to large areas, and their ranges may still be expanding. Others, such as tamarisk or saltcedar (*Tamarix* spp.) and buffelgrass (*Pennisetum ciliare*), are rapidly spreading at the present time, while still others have likely not yet shown their full

potential for expansion. In this volume, as well as in this chapter, our emphasis is on the interaction of nonnatives with fire, how it affects them and how they affect it.

The ecosystems of the Central bioregion have been shaped by fire, including fires associated with natural ignitions and those deliberately set by humans. Both grasslands and shrublands in this bioregion experienced frequent and widespread fires during their evolution (Stewart 2002). Prescribed fire is now widely used to manage some areas for their natural characteristics. Thus, while changed in character, both by conditions that now limit wildfire occurrence and spread and by prescribed burning, the Central bioregion remains one with a high fire frequency (Wade and others 2000).

Fire interactions with nonnative plants can have important impacts. In some cases, fire can be a means of reducing impacts of nonnative species (chapter 4). In other cases, fire may facilitate the establishment and spread of nonnatives (chapter 2). Some nonnative species can radically change the fire regime itself (chapter 3). Because of the widespread use of prescribed fire in this bioregion, it is important to know how nonnative species interact with fire and whether there are means whereby these interactions can be controlled.

Geographic Context and Chapter Organization

Grasslands characterize most of the Central bioregion, much of which has varying amounts of shrubland and woodland components. Two major subdivisions occur within the Central bioregion: the mesic tallgrass prairie subregion in the east and the drier Great Plains subregion in the west. A wide variety of conditions and species exist within each of these two major subregions. To capture some of this variability, these subregions are further divided into formations based on dominant life forms and north-south gradients. Within each of these formations, some nonnatives are conspicuously associated with riparian zones, which are typically forested even if surrounded by grassland. The boundaries of vegetation communities in the Central bioregion are not usually strictly delimited; rather, many types occur scattered throughout the eastern deciduous biome and western and southwestern forests. The central grasslands intergrade from open expanses into arid grasslands and savannas in the west and southwest and into grassland inclusions in the east.

A great range of climate, topography, soil conditions, and historic land use practices occur within the Central bioregion. In the east, most of the pre-Columbian tallgrass prairies have been converted to agricultural fields, some of which have been abandoned and allowed to succeed to various states that include a variety of nonnative and potentially invasive species. In the west, grasslands have also been used for crop production but are used more often for livestock grazing, which means that much larger expanses of grassland exist in at least a quasi-native state in the Great Plains subregion. The embedded wetland and riparian habitats also vary in their condition. In areas where wetlands were small and isolated, there was a tendency for them to be eliminated. In the more extensive coastal wetlands, the communities generally retain more of their native character and are often comparatively pristine.

We would like to be able to predict individual species' interactions with fire within ecoregional boundaries. What is most predictive is information about the biology of individual species. Consistent botanical traits lead to relatively consistent outcomes. Less reliable is our ability to predict how a species will respond in the presence of other species. Competitive ability varies with environment, and complex interactions with the varying native flora and fauna ensure that outcomes will be somewhat conditional. The inadequacy of available information and the complexity of particular situations make it imperative that we keep in mind the possibility that particular situations may deviate from general guidelines.

This chapter presents individual species and their relation to fire within the mesic tallgrass prairie and Great Plains subregions, which match up approximately with the tallgrass prairie in the east (covered by Wade and others 2000) and the plains grasslands in the west (covered by Paysen and others 2000). The treatment is further divided by formations where individual species seem to be most problematic. At this level, we follow the system presented by Risser and others (1981). Using this approach, we recognize two formations in the mesic tallgrass prairie subregion, (1) the northern and central tallgrass prairie and (2) the southern tallgrass prairie. Within the Great Plains subregion, we recognize (1) the northern mixedgrass prairie, (2) the southern mixedgrass prairie, and (3) the shortgrass steppe (fig. 7-1). Finally, while not uniform throughout, we recognize a riparian formation as distinct in character. These classifications recognize the strong role that climate plays in nonnative species distributions and impacts. The reader is reminded that these classifications are approximations and that nonnative invasive species' ranges are not always precisely known and may be expanding. Within-formation variation, which can also be of great importance, will be discussed for each species within this framework when evidence of such variation is available.

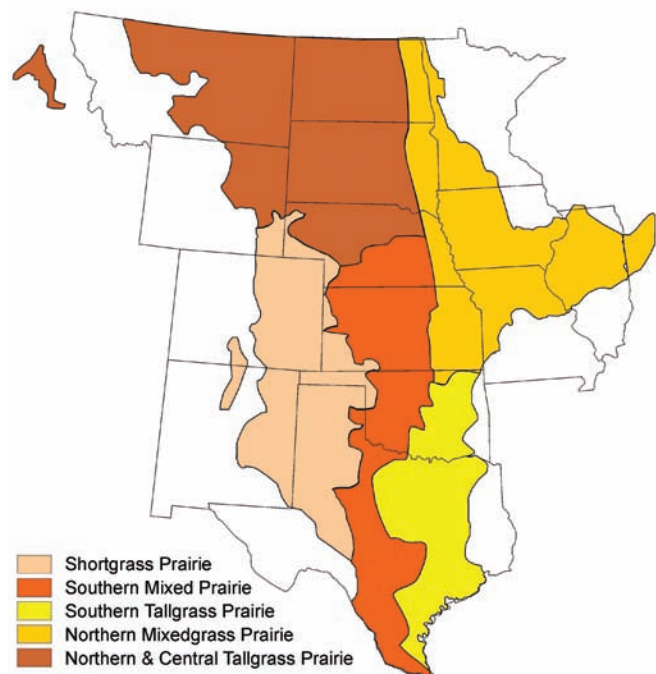


Figure 7-1—Approximate distribution of major vegetation formations in the Central bioregion. Riparian areas are not shown.

Fire Regimes

Grassland and shrubland fire regimes have many similarities. Presettlement fires were frequent and the natural or reference fire regime was dominated by growing-season fires, while the anthropogenic fire regime during the past few millennia seems to be heavily slanted toward dormant season burning (Stewart 2002). Most of these plant communities have now been either cultivated or grazed, and many common features follow from these major influences. To provide useful information, the challenge is to present a picture that is general enough to supply broad guidance to burning programs and decision making. At the same time, we must recognize that there is a great deal of complexity hidden by general descriptions. Our purpose is to focus on some of the major nonnative plants in this bioregion and exemplify the kinds of information known and unknown relating to their interactions with fire. In this vein, in the Effects of Fire on Flora volume (Brown and Smith 2000) figure 1-2 (page 7) provides a useful depiction of fire regime types, including a representation of the average conditions in the Central bioregion.

Wildfires in grasslands are generally warm-season fires that consume most of the aboveground herbaceous growth; that is, they are “stand-replacement” fires as defined by Brown (2000). Presettlement fire regimes in the mesic subregion (northern, central, and southern tallgrass prairies) are characterized as stand-replacement fires with average return intervals of 10 years or less (Wade and others 2000). The Great Plains subregion contains communities with fire regimes that fall into the categories of either stand-replacing or mixed-severity fires (due to the presence of shrubs) and have fire-return intervals of 35 years or less (Paysen and others 2000). Fire severity patterns depend largely on the continuity and abundance of fuels, which in turn is influenced by rainfall and local factors such as grazing. Fire exclusion has a profound impact on the current fire regime. While many managed areas are subject to routine prescribed burning, the majority of the landscape is now managed for fire exclusion. Also of great importance is grazing. Where livestock grazing is intensive, the availability of fuels can be greatly reduced, strongly impacting the likelihood and character of fire.

The Conservation Context

Temperate grasslands, which predominate in the Central bioregion, include some of the most threatened ecosystems in the world (Ricketts and others 1999). In North America, their widespread use for agriculture and livestock grazing, in addition to the effects of urbanization and other human activities, have led to dramatic alterations in their extent and condition.

Nonnative plants constitute a major additional threat to conservation, rehabilitation, and restoration of temperate grasslands (Smith and Knapp 1999; Stohlgren and others 1999a; Westbrooks 1998). Studies indicate that the success and consequences of an invasion depend on many factors, including fire. Because of the historic importance of fire in this bioregion, the interactions between invasives and fire is likely to be of critical importance in these systems.

A variety of interactions between invading species and fire is possible. In some cases, fire may act as an environmental filter that eliminates or reduces nonnative invaders. In other cases, fire-adapted invaders will be quite impervious to burning, and fire may facilitate the establishment and spread of certain nonnative plants. Alternatively, fire exclusion may provide a window of opportunity for the establishment of certain nonnatives that may not be easily displaced once they have established. Species that invade and become dominant may drastically change the fire regime and, through that change, have detrimental effects on the native community. Both the natural characteristics of a landscape and anthropogenic modifications can be expected to influence the interactions between invaders and native communities. Also, interactions between fire and invasives can be complicated by additional factors such as grazing and other disturbances (Collins and others 1995, 1998; Stohlgren and others 1999b). More information is needed in order to manage natural and seminatural systems using fire in ways that are of greatest utility to conservation, particularly in the face of nonnative invasions.

Overview of Nonnative Plants that Impact or Threaten the Central Bioregion

Many nonnative plants occur in the Central bioregion (table 7-1). Some species have very broad distributions and have demonstrated invasiveness in many, but not all, ecosystems within their ranges. These species include cheatgrass (*Bromus tectorum*), smooth brome, tamarisk, leafy spurge, spotted knapweed (*Centaurea biebersteinii*), and Canada thistle (*Cirsium arvense*). Other species are of increasing concern, such as Caucasian bluestem (*Bothriochloa bladhii*), itchgrass (*Rottboellia cochinchinensis*), and guineagrass (*Urochloa maxima*). Still others (such as Angleton bluestem (*Dichanthium aristatum*)) are restricted to a limited portion of the range of grassland types; and the status of some nonnatives is not well documented.

The degree of concern associated with these nonnatives varies depending on climate and other factors. Of greater importance in the Northern and Central States are leafy spurge, cheatgrass, smooth brome, spotted knapweed, Canada thistle, musk thistle (*Carduus nutans*), sweetclovers (*Melilotus* spp.), Dalmatian

Table 7-1—Nonnative plants of major concern in the central United States (from Grace and others 2001). Scientific nomenclature is from the ITIS Database (<http://www.itis.usda.gov/>).

Grasses	
Angleton bluestem	<i>Dichanthium aristatum</i> (Poir.) C.E. Hubbard
Bahia grass	<i>Paspalum notatum</i> Fluegge
Bermudagrass	<i>Cynodon dactylon</i> (L.) Pers.
Buffelgrass	<i>Pennisetum ciliare</i> var. <i>ciliare</i> (L.) Link
Caucasian bluestem	<i>Bothriochloa bladhii</i> (Retz.) S.T. Blake
Cheatgrass	<i>Bromus tectorum</i> L.
Cogongrass	<i>Imperata cylindrica</i> (L.) Beauv.
Crested wheatgrass	<i>Agropyron cristatum</i> (L.) Gaertn.
Fountain grass	<i>Pennisetum glaucum</i> (L.) R. Br.
Giant reed	<i>Arundo donax</i> L.
Giant sugarcane plumegrass	<i>Saccharum giganteum</i> (Walt.) Pers.
Guineagrass	<i>Urochloa maxima</i> (Jacq.) R. Webster
Itchgrass	<i>Rottboellia cochinchinensis</i> (Lour.) W.D. Clayton
Japanese brome	<i>Bromus japonicus</i> Thunb. ex Murr.
Johnson grass	<i>Sorghum halepense</i> (L.) Pers.
Kentucky bluegrass	<i>Poa pratensis</i> L.
Kleberg bluestem	<i>Dichanthium annulatum</i> (Forsk.) Stapf
Lehmann's lovegrass	<i>Eragrostis lehmanniana</i> Nees
Orchard grass	<i>Dactylis glomerata</i> L.
Quackgrass	<i>Elymus repens</i> (L.) Gould
Ryegrass spp.	<i>Lolium</i> spp. L.
Smooth brome	<i>Bromus inermis</i> Leyss.
Vaseygrass	<i>Paspalum urvillei</i> Steud.
Yellow bluestem	<i>Bothriochloa ischaemum</i> (L.) Keng
Forbs	
Brazilian vervain	<i>Verbena brasiliensis</i> Vell.
Canada thistle	<i>Cirsium arvense</i> (L.) Scop.
Common mullein	<i>Verbascum thapsus</i> L.
Crown vetch	<i>Coronilla varia</i> L.
Dalmatian toadflax	<i>Linaria dalmatica</i> (L.) P. Mill.
Diffuse knapweed	<i>Centaurea diffusa</i> Lam.
Garlic mustard	<i>Alliaria petiolata</i> (Bieb.) Cavara & Grande
Japanese climbing fern	<i>Lygodium japonicum</i> (Thunb. ex Murr.) Sw.
Kochia	<i>Kochia prostrata</i> (L.) Schrad.
Leafy spurge	<i>Euphorbia esula</i> L.
Missouri bladderpod	<i>Lesquerella filiformis</i> Rollins
Musk thistle	<i>Carduus nutans</i> L.
Oxeye daisy	<i>Leucanthemum vulgare</i> Lam.
Purple loosestrife	<i>Lythrum salicaria</i> L.
Red-horned poppy	<i>Glaucium corniculatum</i> (L.) J.H. Rudolph
Russian knapweed	<i>Acroptilon repens</i> (L.) DC.
Scotch thistle	<i>Onopordum acanthium</i> L.
Sericea lespedeza	<i>Lespedeza cuneata</i> (Dum.-Cours.) G. Don
Spotted knapweed	<i>Centaurea biebersteinii</i> DC.
Squarrose knapweed	<i>Centaurea triumfettii</i> All.
White Sweetclover	<i>Melilotus alba</i> Medikus
Whitetop	<i>Cardaria draba</i> (L.) Desv.
Yellow sweetclover	<i>Melilotus officinalis</i> (L.) Lam.
Yellow starthistle	<i>Centaurea solstitialis</i> L.
Yellow toadflax	<i>Linaria vulgaris</i> P. Mill.
Woody Species	
Chinese privet	<i>Ligustrum sinense</i> Lour.
Chinese tallow	<i>Triadica sebifera</i> (L.) Small
Common buckthorn	<i>Rhamnus cathartica</i> L.
Japanese honeysuckle	<i>Lonicera japonica</i> Thunb.
Macartney rose	<i>Rosa bracteata</i> J.C. Wendl.
Multiflora rose	<i>Rosa multiflora</i> Thunb. ex Murr.
Russian-olive	<i>Elaeagnus angustifolia</i> L.
Tamarisk	<i>Tamarix</i> spp. L.

toadflax (*Linaria dalmatica*), and Kentucky bluegrass (*Poa pratensis*). In the southern portion of the Central bioregion, buffelgrass, guineagrass, Bermudagrass (*Cynodon dactylon*), sericea lespedeza (*Lespedeza cuneata*), Johnson grass (*Sorghum halepense*), Chinese tallow (*Triadica sebifera*), Macartney rose (*Rosa bracteata*), Caucasian bluestem, and several other escaped pasture grasses are the most often mentioned species. Riparian and wetland habitats in many parts of the Central bioregion have been invaded by tamarisk, Russian-olive (*Elaeagnus angustifolia*), and purple loosestrife (*Lythrum salicaria*). In southern wetlands, nonnative invaders include giant reed (*Arundo donax*).

The species covered in this chapter represent only a fraction of the nonnatives currently present in temperate grasslands. In a review of invasive species in the USGS Central Region (Burkett and others 2000), resource management agencies throughout the Central States were polled to obtain their views on which nonnative invaders (plants and animals) were of greatest concern. Some 69 plant taxa were reported to be of concern, which gives an indication of the magnitude of the problem. A preliminary assessment of the flora of the southern tallgrass prairie formation (Allain and others, unpublished report, 2004) has identified over 150 nonnative species in this type of prairie alone. Also, of the 304 grass species in Texas coastal prairies and marshes, 85 (26 percent) are introduced (Hatch, S. and others 1999). Our current knowledge about the full suite of introduced species in the Central bioregion and how they relate to fire is incomplete. While certain generalizations may apply to nonnative invaders, experience tells us that species-specific and site-specific effects are often highly important and difficult to anticipate without a great deal more information than is currently available.

Plant-Fire Interactions

The conceptual model presented by Grace and others (2001) represents a framework for summarizing and evaluating how an invading nonnative may interact with the native community and the fire regime. In this framework, the major categories of influences are the native community characteristics, the fire regime, the growth conditions for both nonnative and native species, and the influences that disturbances, human impacts, and landscape characteristics have had in the past and will have in the future. According to this model, several major relationships will determine whether an invasive species can successfully establish and persist in a habitat in the absence of fire. These include whether an invading species can survive and spread when fires occur and the degree to which the invader may ultimately alter that system.

The process of invasion can be broken into four stages: (1) establishment, typically from seed, though long-distance dispersal of vegetative propagules may occur in some cases; (2) survival and reproduction (that is, persistence); (3) density increase within a site, which includes spread within a site, either through seed or vegetatively, as well as increases in abundance in surrounding areas; and (4) dominance, which implies not only the establishment of substantial abundance but also the suppression of other species or other types of ecological harm.

Several questions about how an invasive species relates to fire need to be addressed in order to predict the species relationship to fire in a particular setting:

1. Does fire enhance establishment by the nonnative?
2. Does fire result in the mortality of the nonnative?
3. Are burned plants able to regrow following fire and, if so, how rapidly do they recover?
4. How important is competition with native species to the response by a nonnative to fire? and
5. What effects does a nonnative species have on the characteristics of the fire regime?

The answers to these questions, though approximate, are used to frame the discussion of each of the invasive species discussed below and can be used to classify them into different functional types as described by Grace and others (2001).

The goal of this chapter is not to produce an exhaustive treatment of all major nonnatives in the Central bioregion, which can be found in Grace and others (2001). Rather, we summarize available information relating to nonnatives of greatest concern with regard to interactions with fire in these ecosystems. Following an introduction to the subregion and formations, three main topics will be discussed for species or groups of species within each formation in that subregion: (1) the role of fire or fire exclusion in promoting nonnative species invasions, (2) the effects of nonnative invasives on fuels and fire regimes, and (3) the use of fire to control nonnative invasives. Table 7-2 lists the species, organized by the formations in which they pose the greatest problem, that will be discussed in detail in this chapter.

The Mesic Tallgrass Prairie Subregion

This subregion is represented by tallgrass prairie ecosystems that extend, though not continuously, from Canada to Mexico (Risser and others 1981). Both characteristic grasses (for example, prairie cordgrass (*Spartina pectinata*) in the north and brownseed paspalum (*Paspalum plicatulum*) in the south) and

Table 7-2—Approximate threat potential of several major nonnative species in the Central bioregion within seven broad vegetation formations. L= low threat, H = high threat, P = potentially high threat, N= not invasive

Species		Formations					
		Northern & Central tallgrass prairie	Southern tallgrass prairie	Northern mixedgrass prairie	Southern mixedgrass prairie	Shortgrass steppe	Riparian
<i>Arundo donax</i>	Giant reed	N	N	N	N	N	H
<i>Bothriochloa ischaemum</i>	Yellow bluestem	N	H	N	H	P	N
<i>Bothriochloa bladhii</i>	Caucasian bluestem	H	H	H	P	P	N
<i>Bromus inermis</i>	Smooth brome	H	N	P	N	N	N
<i>Bromus japonicus</i>	Japanese brome	N	N	H	N	H	N
<i>Bromus tectorum</i>	Cheatgrass	N	N	H	P	H	N
<i>Centaurea</i> spp.	Knapweeds	L	N	H	H	H	N
<i>Cirsium arvense</i>	Canada thistle	H	L	H	L	H	N
<i>Coronilla varia</i>	Crown vetch	H	N	L	N	N	N
<i>Elaeagnus angustifolia</i>	Russian-olive	N	N	H	L	H	H
<i>Euphorbia esula</i>	Leafy spurge	H	N	H	N	N	N
<i>Imperata cylindrica</i>	Cogongrass	N	H	N	N	N	N
<i>Lespedeza cuneata</i>	Sericea lespedeza	H	H	L	N	L	N
<i>Melilotus alba</i>	White sweetclover	H	N	H	N	L	N
<i>Melilotus officinalis</i>	Yellow sweetclover	H	N	H	N	L	N
<i>Poa pratensis</i>	Kentucky bluegrass	H	N	H	N	N	N
<i>Pennisetum ciliare</i> var. <i>ciliare</i>	Buffelgrass	N	L	N	H	H	N
<i>Rhamnus cathartica</i>	Common buckthorn	H	N	N	N	N	H
<i>Rosa bracteata</i>	Macartney rose	N	H	N	N	N	N
<i>Sorghum halepense</i>	Johnson grass	N	H	N	N	N	N
<i>Tamarix</i> spp.	Saltcedar, tamarisk	N	H	N	H	H	H
<i>Triadica sebifera</i>	Chinese tallow	N	H	N	N	N	H
<i>Urochloa maxima</i>	Guineagrass	N	N	N	H	N	N

forbs (for example, Canadian milk vetch (*Astragalus canadensis*) and prairie cinquefoil (*Potentilla arguta*) in the north, and Texas coneflower (*Rudbeckia texana*) and tropical puff (*Neptunia pubescens*) in the south) are regionally restricted to the northern versus southern lobes of the tallgrass prairie (Kucera 1992; Smeins and others 1992).

Historically, Native Americans were probably the primary source of ignition for fires in the mesic prairie subregion during normal weather conditions. Evidence regarding fire frequencies from this subregion is most reliably taken from scars embedded in trees. Such data indicate that prior to 1870, when European influences had clearly come to dominate, fires occurred at roughly 5-year return intervals, while for the period after that, return intervals of around 20 years are more characteristic (Wade and others 2000). Because of the strong influence of human-caused fires on the fire regime, predominant fire season has varied depending on the practices of the peoples living in the area more than on the influences of lightning strikes and dry frontal passages. Fuel loads in mesic grassland are characteristically high, in the range of 5 to 10 mt/ha. Such fuels characteristically produce flame lengths of 10 to 13 feet (3 to 4 m) and typically precluded the

establishment and persistence of woody plants in areas where moisture conditions were adequate for forest development. The exclusion of fire leads to rapid conversion of many areas of mesic prairie to woodland (Wade and others 2000). Species of oak (*Quercus* spp.) and juniper (*Juniperus* spp.) are among the native woody plant groups whose ranges have expanded in recent times.

Successional responses to fire among prairie natives are influenced by season and frequency of burning. Prescribed burning in tallgrass prairie has been most commonly applied in spring or fall, with late spring or fall fires having a tendency to favor warm season grasses compared to early spring fires. Generally, late spring burns are recommended to foster the density of warm-season (C_4) grasses and prevent the establishment of nonnatives (Willson and Stubbendieck 2000). However, it is possible that while such a strategy may be successful against cool-season nonnatives (for example, Kentucky bluegrass) it may be less successful in cases of warm-season nonnatives (for example, Johnson grass).

Native prairie species respond differentially to fire frequency. Generally, big bluestem (*Andropogon gerardii*) shows no response to time since fire, while

little bluestem (*Schizachyrium scoparium*), indian-grass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*) all decrease with time since fire. Gibson (1988) found that perennial forbs and cool-season grasses increased with time since fire, while annual forbs and warm-season grasses decreased. Annual or frequent burning tends to reduce herbaceous plant diversity in tallgrass prairie (Wade and others 2000).

The management of wildlife species may also influence burn times in order to avoid direct mortality of individuals, particularly during nesting season. Integration of livestock grazing and fire management is constrained by the needs of the animals and by the effects of grazing on fuels. Results from various studies suggest that for the conservation of native diversity, a variety of burn times should be used (Howe 1995). However, when nonnatives are involved, many such recommendations must be reconsidered to avoid promotion of nonnative dominance.

Northern and Central Tallgrass Prairie Formation

In the northern and central tallgrass prairie, from Canada to Oklahoma, the topography is mostly gently rolling plains. Some areas are nearly flat, while other areas have high rounded hills. Elevation ranges from 300 to 2,000 feet (90 to 600 m). Summers are usually hot and winters cold, with the frost-free season ranging from 120 days in the northern portion to 235 days in the central portion. Average annual precipitation ranges from 20 to 40 inches (510 to 1,020 mm), and falls primarily during the growing season (Bailey 1995).

Most of the area is cultivated, and little of the native vegetation remains. Where it does occur, native vegetation in this formation is predominantly prairie dominated by bluestems (big bluestem, little bluestem, sand bluestem (*Andropogon gerardii* var. *paucipilus*)) and variously codominated by switchgrass, indian-grass, prairie sandreed (*Calamovilfa longifolia*), needle-and-thread grass (*Hesperostipa comata*), and hairy grama (*Bouteloua hirsuta*) (Küchler 1964). Many flowering forbs are also present. Most of these prairie species are classified as warm-season plants. Woody vegetation is uncharacteristic in native prairie, except within the eastern ecotone where the prairie grades into oak-hickory (*Quercus* spp.-*Carya* spp.) forest, and on floodplains and moist hillsides (Garrison and others 1977; Küchler 1964). Additionally, in places where fire is excluded and grazing is limited, deciduous forest is encroaching on the prairies (Bailey 1995). Where conditions are more mesic or otherwise favor tree and shrub growth, savannas (for example, oak savanna) are common. In these savannas, the vegetation between trees comprises species typical of tallgrass prairie (Nuzzo 1986).

The effects of long-term, frequent fires on central tallgrass prairie plant communities and the relationship between nonnative species and fire in these communities was examined in a 15-year study at Konza Prairie in northeastern Kansas (Smith and Knapp 1999). Long-term annual spring burning resulted in 80 percent to 100 percent reductions in number and abundance of nonnative plant species compared with infrequently burned plots. Nonnative species were absent from sites that had been burned 26 of 27 years, and nonnative species richness steadily increased as the number of times a site was burned decreased. The highest nonnative species richness occurred on sites burned fewer than 6 times over the 27-year period. Thus the cumulative effects of fire seem to be important in controlling invasion by nonnative species in tallgrass prairie. This effect may be due more to the increased productivity of dominant native C₄ grasses under a regime of frequent fire rather than to direct negative impacts of fire on nonnative species (Smith and Knapp 1999). Some species such as Caucasian bluestem appear to be less easily controlled by frequent fire (USDA, NRCS 2006).

The effects of frequent fire on plant community composition in tallgrass prairie varies with burn season (Howe 1995; Towne and Kemp 2003), other land management practices (for example, grazing by livestock and bison), climatic variation, topographic position, and the impact of management practices on soil, moisture availability, fire patchiness, and propagule pressure (Coppedge and others 1998; Hartnett and others 1996; Trager and others 2004; Vinton and others 1993). For example, while annual burning appears to reduce invasibility of areas studied at Konza Prairie, there is evidence that nonnative species richness increases 2-fold in annually burned sites that are grazed when compared to similar ungrazed sites (Smith and Knapp 1999). Similarly, the percent of species that were nonnative in bison wallows on Konza Prairie was higher than in surrounding grazed prairie sites whether sites were burned annually or on a 4-year rotation. In both wallow and prairie sites, most nonnative species (smooth brome, common pepperweed (*Lepidium densiflorum*), Kentucky bluegrass, and prostrate knotweed (*Polygonum aviculare*)) had higher cover on annually burned sites than on those burned every 4 years, although differences were not always significant (Trager and others 2004).

When there is a source of nonnative propagules in the area surrounding an annually burned site, the site may be more susceptible to establishment and spread of these species in the postfire environment (Grace and others 2001). Further studies at Konza Prairie by Smith and Knapp (2001) suggest that the size and composition of local species pools surrounding a target community is as important as the effects of

fire in determining invasibility at this tallgrass prairie site. So, while fire is an important management tool for promoting dominance of native C₄ grasses and reducing nonnative species invasions, frequent fire may not sufficiently limit nonnative species invasions if the local nonnative species pool is relatively large. This is an important consideration as prairie ecosystems become increasingly fragmented and nonnative species establish in surrounding areas (Smith and Knapp 2001).

Important nonnative species in the tallgrass prairie formation include several introduced perennial pasture grasses such as smooth brome, Kentucky bluegrass and Caucasian bluestem; perennial forbs such as Canada thistle, crown vetch (*Coronilla varia*), sericea lespedeza, biennial sweetclovers, and leafy spurge; and the shrub common buckthorn (*Rhamnus cathartica*). Information on fire relationships for these species follows, although Caucasian bluestem is covered in the section on southern tallgrass prairie and leafy spurge is covered in the section on northern mixedgrass prairie.

Smooth brome—Smooth brome is a widespread nonnative grass that has been widely planted to increase forage or to reduce erosion following wildfires; it readily escapes into other habitats, particularly in the northern portion of its range (Sather 1987a, TNC review). It generally invades after disturbance and is a common invader of prairie habitat throughout the Great Plains (Howard 1996, FEIS review). Smooth brome is a cool-season, perennial grass that begins its growth in early spring and continues growth late into fall. It is a prolific seed producer (figure 7-2), setting



Figure 7-2—Smooth brome in flower. This widely distributed species, characteristic of the northern and central tallgrass prairie subregions, is a major invader of disturbed native prairie and restoration sites. (Photograph by Mike Haddock, with permission, (see www.lib.ksu.edu/wildflower).)

seed mid-summer through fall. Studies indicate that it has a high level of drought resistance while not being particularly flood tolerant (Dibbern 1947). Most smooth brome cultivars are rhizomatous, though some northern cultivars have bunchgrass morphology. Because of the magnitude of morphological variation among cultivars, Howard (1996) suggests that postfire recovery may differ among cultivars, although such differences have not been documented in the literature.

Exclusion of fire from tallgrass prairie sites may promote smooth brome invasion, and smooth brome can be adversely affected by burning in some cases (Blankespoor and Larson 1994; Kirsch and Kruse 1973). According to Masters and Vogel (1989), smooth brome is usually found in tallgrass prairie in areas with a history of overgrazing and/or fire exclusion. A comparison of burned and unburned prairie remnants in northwestern Illinois indicates that fire exclusion favors nonnatives such as smooth brome and Kentucky bluegrass over native prairie species such as little bluestem and porcupinegrass (*Hesperostipa spartea*), while periodic spring burning over a 20-year period increased mean native species richness and reduced nonnative species richness (Bowles and others 2003).

Smooth brome invasion may sometimes be enhanced by fire, and it appears to suffer little mortality when burned, responding to fire by sprouting from rhizomes and possibly by tillering. Fire in early spring or fall may promote smooth brome by removing litter from sod-bound plants (Howard 1996). Over a 15-year period, annual spring burning did not reduce this species in tallgrass prairie in northeastern Kansas (Smith and Knapp 1999). Smooth brome's phenological stage at the time of burning may be more important in determining its response to fire than the date of burning. Burns during the spring growth period may reduce smooth brome vigor, and several researchers have reported reductions in smooth brome after late-spring fires (Howard 1996). Grilz and Romo (1994, 1995) found no significant effect of fall or spring burning on smooth brome stem density in tallgrass prairie in Saskatchewan. The authors speculate that a single dormant-season burn in this C₃-dominated fescue prairie is not expected to reduce smooth brome, and may actually increase smooth brome density if native species are suppressed (Grilz and Romo 1994).

Willson and Stubbendieck (2000) have proposed a model for managing smooth brome using prescribed fire in northern tallgrass prairie. One fundamental aspect of their proposed approach

relates to the plant's phenology. Tiller emergence in smooth brome occurs twice: once after flowering in early summer and again in fall (Lamp 1952). Fall tillers overwinter and then elongate in early May in the central Great Plains and Midwestern States. Willson (1991) found a 50 percent reduction in the density of tillers following prescribed burning in eastern Nebraska in early May, presumably because of reduced carbohydrate reserves during the time of tiller elongation. Kirsch and Kruse (1973) and Old (1969) also reported that burns in April and May have negative effects on smooth brome. Additional studies by Willson and Stubbendieck (1996, 1997) in Minnesota and Nebraska have examined burning during (1) tiller emergence, (2) tiller elongation, (3) heading (initiation of morphological changes associated with sexual reproduction), and (4) flowering. Burning during any of these periods resulted in reductions in plant growth.

A second aspect of control of smooth brome relates to native competitors. Field observations indicate that burning when smooth brome tillers are elongating in the spring shifts the competitive balance to favor warm-season tallgrass species such as big bluestem. In the absence of native warm-season grasses, the adverse effects of fire on smooth brome are not sufficient to control smooth brome (Willson and Stubbendieck 2000). Fire and competition in combination may inhibit this species, though in the absence of native warm-season competitors, even fire precisely timed to have maximum impact on smooth brome will not be sufficient to lead to sustained population reductions (Willson and Stubbendieck 2000). Other examples of this pattern have been observed involving Kentucky bluegrass, quackgrass, and possibly also crested wheatgrass and Canada thistle.

There is no indication that smooth brome alters the fire regime of systems it invades. In the absence of specific information, we might hazard a guess that this results partly from the fact that the grasslands that smooth brome invades are typically high in herbaceous cover and naturally prone to high fire frequencies. Another factor that might contribute is its perennial nature and the fact that its aboveground tissues do not dry and become highly flammable early in the summer. Since the literature suggests that smooth brome can be burned almost any time of year, it is unclear how important the difference in fuel drying may be.

Kentucky bluegrass—Kentucky bluegrass, like smooth brome, is a perennial, cool season, rhizomatous invader of native grasslands that is widely planted, in this case primarily for lawns, as well as for pastures and erosion control. Kentucky bluegrass is a significant invader in more mesic sites in the upper Great Plains as well as in eastern prairies and it is considered a major problem for tallgrass and mixedgrass prairies (Hensel 1923; Sather 1987b, TNC review; Stohlgren and

others 1998). Due to its strongly rhizomatous nature, Kentucky bluegrass is capable of rapid vegetative spread (Etter 1951), and populations are persistent.

Exclusion of fire from tallgrass prairie sites seems to favor nonnative, cool season grasses such as Kentucky bluegrass (Benson 2001; Bowles and others 2003; Smith and Knapp 1999), and many studies have shown that burning has negative effects on Kentucky bluegrass production in tallgrass prairie. Fire impacts on Kentucky bluegrass are most pronounced when burning takes place during tiller elongation in the spring (for example, Archibold and others 2003; Curtis and Partch 1948; Ehrenreich 1959) and when fires are repeated at relatively frequent intervals. Kentucky bluegrass cover tends to be lower on annually burned prairie sites than on sites burned at 4-year or longer intervals (Collins and others 1995; Hartnett and others 1996; Vinton and others 1993). Kentucky bluegrass is not always adversely affected by burning. Kirsch and Kruse (1973) found it to be unaffected by a May burn in mixedgrass prairie in North Dakota.

The effect of fire on Kentucky bluegrass is strongly influenced by available soil moisture, which may either enhance or nullify any detrimental impact of fire (Anderson 1965; Blankespoor and Bich 1991; Zedler and Loucks 1969). Late-spring burning on a tallgrass prairie remnant in South Dakota resulted in a significant ($P < 0.01$) reduction in Kentucky bluegrass biomass on plots with low water content and an insignificant reduction on plots with high water content. Kentucky bluegrass biomass increased on unburned plots regardless of soil moisture conditions (Blankespoor and Bich 1991).

Because Kentucky bluegrass is a cool-season grass that elongates early in the growing season (an attribute that helps to make it a desirable forage species), prescribed burning designed to shift the competitive balance to native grasses is used widely for controlling this species. In addition to the importance of timing and frequency of fire and available moisture, native competitors make a critical difference in the impact of burning on Kentucky bluegrass (Schacht and Stubbendieck 1985). Where Kentucky bluegrass grows with warm-season native grasses (as would be typical in tallgrass prairie), repeated spring burning offers a substantial opportunity for shifting the competitive balance toward native species (Owensby and Smith 1973; Towne and Owensby 1984). Indeed, native warm-season grasses dominated and Kentucky bluegrass occurred less frequently in annually-burned, mesic, tallgrass prairie sites at the Konza Preserve in Kansas, compared to infrequently burned (20-year burn cycle) or unburned sites (Benson 2001; Smith and Knapp 1999). In more arid regions of the western Great Plains, where native cool-season species are more common, only a narrow window of opportunity exists

for burning while Kentucky bluegrass is elongating but before native species (for example, western wheatgrass (*Pascopyrum smithii*) and needlegrasses (*Achnatherum* and *Hesperostipa* spp.)) elongate (Sather 1987b). The use of early spring burning becomes somewhat more complex when both smooth brome and Kentucky bluegrass occur together because the optimum time for controlling each of these species with spring burning does not appear to be the same. At present, there are insufficient examples for drawing firm conclusions about whether Kentucky bluegrass can be eliminated from tallgrass prairie with fire.

Canada thistle—Canada thistle is a perennial forb that spreads by seed and by creeping, horizontal roots. It occurs throughout much of Canada and the United States in nearly every upland herbaceous community within its range, particularly prairie communities and riparian habitats. In the Central bioregion Canada thistle threatens northern mixedgrass prairie and shortgrass steppe formations, in addition to northern and central tallgrass prairie. It is an early successional species, establishing and developing best in open, moist, disturbed areas. Canada thistle often grows in large clonal patches, and individual clones may reach 115 feet (35 m) in diameter, interfering with native species (Zouhar 2001d, FEIS review).

Canada thistle is adapted to establish on exposed bare soil on recently burned sites and to survive fire. Several examples in the literature indicate Canada thistle establishment from wind-deposited seed, anywhere from 2 to 9 years after fire, on sites where it was absent from the prefire plant community and adjacent unburned areas (Zouhar 2001d). Additionally, Canada thistle may dominate the soil seed bank where it occurs, as was observed on mixedgrass prairie sites in North Dakota (Travnicek and others 2005). Its extensive root system allows Canada thistle to survive major disturbances and fires of varying severity by producing new shoots from adventitious buds on roots. The response of Canada thistle to fire is variable, however, depending on vegetation and site characteristics, as well as frequency, severity, and season of burning (Zouhar 2001d).

Results from studies on prairie and riparian sites demonstrate Canada thistle's variable response to fire. There were no significant differences ($P < 0.05$) in Canada thistle cover after spring burning in the prairie pothole region of Iowa (Messinger 1974). Prescribed burning in spring either reduced or did not change canopy cover of Canada thistle on bluestem prairie sites in Minnesota. Results varied among sites that differed in plant community composition and in time and frequency of burning (Olson 1975). In a prairie site at Pipestone National Monument, Minnesota, 5 years of annual spring burning in mid- to late April, with fires of low to moderate severity, reduced the frequency

of Canada thistle over time until it was absent after the fifth year (Becker 1989). Similarly, observations in tallgrass prairie sites in South Dakota indicate that late spring prescribed burning (when native species are still dormant) on a 4 to 5 year rotation (as per the historic fire regime) encourages the growth of native plants and discourages the growth of Canada, bull, and musk thistles. Livestock use must be carefully timed following burning, since grazing early in the growing season can potentially negate beneficial effects of prescribed fire (Dailey, personal communication, 2001). In a study conducted on a mesic prairie site in Colorado, plots that were burned frequently (5 times over 7 years) had lower density of Canada thistle than did an area that was burned only twice during the same period. Results were inconclusive, however, since the final season of the study saw increased spread of Canada thistle from the surrounding area, probably due to clonal growth from existing plants (Morghan and others 2000). On a common reed marsh in Manitoba, Canada, thistle response to burning varied with burn season. Above-ground biomass, stem density, and seedling density were unchanged on spring burns but increased on both summer and fall burns (Thompson and Shay 1989). A Canada thistle clone in a mid-boreal wetland site in northeastern Alberta was not noticeably changed when burned in the spring with a propane torch to simulate both "light" and "deep" burns (Hogenbirk and Wein 1991). The authors concluded that there exists a moderate to high probability that Canada thistle and other Eurasian xerophytic species will dominate these wet meadows in the short term after fire, and that they will continue to dominate small areas for longer periods (Hogenbirk and Wein 1995).

Canada thistle may change the fuel characteristics of invaded sites with its abundant aboveground biomass. Hogenbirk and Wein (1995) suggest that in boreal wet-meadows, Canada thistle has the potential to increase fire frequency and perhaps severity as a result of its abundant and readily ignited litter. No additional information on the potential for Canada thistle to alter fuel characteristics or fire regimes in the Central bioregion is available in the literature.

Season of burn is an important consideration for determining species composition and cover in the postfire tallgrass prairie community (Howe 1994a, b). According to Hutchison (1992a) prescribed burning is a "preferred treatment" for control of Canada thistle, and late spring burns effectively discourage this species, whereas early spring burns can increase sprouting and reproduction. During the first 3 years of control efforts, he recommends that burns be conducted annually. At a mixedgrass prairie site at Theodore Roosevelt National Park, North Dakota, fall burning appeared to increase Canada thistle stem density because it emerged more rapidly on burned sites than on unburned

sites. By the second postfire growing season, however, Canada thistle stem densities were similar on burned and unburned sites. Grass foliar cover was unaffected by burning, while forb cover increased to 4 percent in the burned area compared to a 1 percent increase in the unburned area. The number and variety of species in the soil seed bank were not affected by prescribed burning, and Canada thistle and Kentucky bluegrass accounted for over 80 percent of germinants from the soil seed bank (Travnicek and others 2005).

Crown vetch—Crown vetch is a perennial forb that has been widely planted in the Northern United States, primarily for erosion control but also for pasture, mine reclamation, and ornamental ground cover. It tolerates a broad range of conditions and has been reported to escape from cultivation in every state except Alaska, California, North Dakota, and Louisiana. In the Central bioregion, it occurs primarily in northern and central tallgrass and northern mixedgrass prairies, growing best in areas with more than 18 inches (460 mm) annual precipitation. A review by Tu (2003, TNC review) indicates that crown vetch infestations commonly begin along roads and rights-of-way and spread into adjacent natural areas such as grasslands and dunes in Missouri, Minnesota, Illinois, and Iowa. It sometimes forms monotypic stands and can exclude native plant species by covering and shading them. Crown vetch spreads by abundant seed production, a multi-branched root system, and strong rhizomes. Observations suggest that crown vetch seed can remain viable in the soil for many years, although maximum longevity is not known (Tu 2003).

Little research is available regarding the invasiveness, impacts, longevity, and control of crown vetch in any plant community, particularly in relation to fire. One may speculate that as a rhizomatous, perennial legume it is likely to occur in the initial postfire community by sprouting from rhizomes or establishing from seed in the soil seed bank. Crown vetch withstands some grazing pressure and sprouts when aboveground tissue is removed by pulling, cutting, or mowing. Observations by managers indicate both sprouting and stimulation of germination following burning (Tu 2003).

Some managers use fire as part of their management program for crown vetch. Solecki (1997, review) suggests that burning in late spring controls crown vetch seedlings but that mature plants are only top-killed. A manager in central Minnesota uses a 3-year burning cycle with the objective of exhausting the crown vetch seed bank by stimulating germination. Burning kills crown vetch seedlings and slows spread of mature plants. Where crown vetch dominates, burning alone is not likely to control it. In large infestations with sufficient fuel, fire can reduce crown vetch on the periphery of stands but leave infestation interiors

unaffected. Frequent (annual) prescribed fire in late spring may control sparse infestations of crown vetch in ecosystems dominated by species adapted to such fire regimes (Tu 2003).

Crown vetch infestations may alter fuel characteristics on invaded sites, especially where it occurs in single-species stands. Managers report that dense infestations of crown vetch do not carry fire well (Tu 2003). More information is needed on the fire ecology of this species.

Sericea lespedeza—*Sericea lespedeza* is a warm-season, perennial, erect, multi-stemmed forb that grows 2 to 7 feet (0.5 to 2 m) tall. It spreads by seed and can sprout following damage to aboveground tissues. *Sericea lespedeza* has been widely planted in the United States and has escaped cultivation in many areas. It occurs from southern New England west to southern Wisconsin, Iowa and Nebraska, and south to Texas and Florida. Within this range it may be invasive on open sites such as prairie, grassland, and pasture, especially near areas where it was previously planted. It grows best in areas with more than 30 inches (760 mm) of annual precipitation, although its deep, well-developed root system allows *sericea lespedeza* to grow on relatively dry or infertile sites (Munger 2004, FEIS review).

No published information specifically describes adaptations of *sericea lespedeza* to fire. Perennating buds on *sericea lespedeza* are located 0.5 to 3 inches (1 to 8 cm) below the soil surface; therefore, individual plants are likely to survive the heat of most surface fires despite damage or destruction of aboveground tissue. *Sericea lespedeza* is also known to sprout in response to mechanical damage of aboveground tissue. It may develop a seed bank and thereby establish from on-site seed sources after fire. *Sericea lespedeza* seedlings may be favored in postfire environments (Munger 2004). Conversely, frequent fire may exclude *sericea lespedeza* from plant communities adapted to such fire regimes. For example, on three prairie remnants in eastern Arkansas, *sericea lespedeza* was absent from two sites that were annually burned for over 60 years and occurred on one site where fire was excluded for 17 years prior to study (Irving and others 1980). Although inconclusive, these results suggest that frequent fire in prairie communities may exclude *sericea lespedeza*. Conversely, Griffith (1996) implicates annual burning among reasons for spread of *sericea lespedeza* based on field observations.

Some studies suggest that fire effects on *sericea lespedeza* populations depend on a number of factors including fire conditions, climatic conditions before and after fire, plant community composition, and interactions with other disturbances or management practices. In a study designed to evaluate interactive effects of fire and grazing in an area primarily

characterized as tallgrass prairie (dominated by little bluestem, big bluestem, and indiangrass) in central Oklahoma (Fuhlendorf and Engle 2004), sericea lespedeza occurred throughout the study area. Treatments included (1) burning of spatially distinct patches within a treatment unit (one sixth of each unit burned each spring (March to April) and one sixth burned each summer (July to October), representing a 3-year return interval for each patch), and free access by moderately stocked cattle; and (2) no burning with free access by moderately stocked cattle. Sericea lespedeza occurred in all treatment units and increased overall during the study. Cover and rate of increase of sericea lespedeza were significantly greater in the unburned units during the 4 years of the study. Sericea lespedeza cover fluctuated at low values in the units with burned patches. These results suggest that the use of prescribed fire in conjunction with grazing in a shifting mosaic pattern may help reduce the importance and spread of sericea lespedeza, while grazing in the absence of burning leads to continued increases in sericea lespedeza cover (Fuhlendorf and Engle 2004).

Few studies document the results of prescribed fire used to control sericea lespedeza populations, although it is suggested that spring fires may promote seed germination (Segelquist 1971) and sprouting (Stevens 2002, TNC review). Under these circumstances, a useful strategy might be to promote spring growth of native species with a prescribed burn and follow up with control measures that focus on eradication of sericea lespedeza sprouts and seedlings. Sericea lespedeza seedlings may be susceptible to mortality from spring burning (Cooperative Quail Study Association 1961), although it is unclear at what age newly established plants can survive fire damage by sprouting. Prescribed fire may be useful for reducing density of sericea lespedeza seed banks. Evidence from laboratory studies on heat tolerance of sericea lespedeza seed suggests that seeds exposed to fire may sustain lethal damage (Munger 2004).

No information is available on the effects of sericea lespedeza invasion on fuel or fire regime characteristics in invaded communities. More research is needed to understand its response to fire in tallgrass prairie.

Biennial sweetclovers—White sweetclover (*Melilotus alba*) and yellow sweetclover (*Melilotus officinalis*) are widely planted annual or, more typically, biennial leguminous forbs that have escaped cultivation and are established in disturbed areas, along roadsides, and in many grasslands and prairies (Turkington and others 1978). Sweetclover sometimes occurs in dense patches that exclude native vegetation (for example, Heitlinger 1975; Randa and Yunger 2001). Sweetclover reproduces by seed, with peak germination in early spring. Seeds are hard upon dispersal and may remain viable in the soil for many years and possibly

decades. Biennial sweetclovers produce a vegetative shoot in the first season of growth that dies back in fall. Most root development occurs in late summer, and during this time crown buds form just below or at the soil surface. The taproot and crown buds overwinter; the following spring and early summer, one or more flowering shoots emerge from the buds and rapidly elongate (Turkington and others 1978).

Literature reviews indicate that burning enhances sweetclover establishment in grasslands (for example, Heitlinger 1975; Turkington and others 1978; Uchytel 1992a), and evidence suggests that burning helps to break seed dormancy and enhance postfire germination (Kline 1986). The effect of fire on sweetclover populations depends on timing and frequency. Sweetclover plants are killed when burned during the growing period (Heitlinger 1975), though they survive if burned before shoots begin to elongate (Eckardt 1987, TNC review; Kline 1986). A fire on a tallgrass prairie site in Minnesota in early May resulted in an increased number of first-year sweetclover plants and a decreased number of second-year plants, compared to an unburned area. Fire on the same site in early July reduced the number of both first- and second-year sweetclover plants (Heitlinger 1975). Infrequent burning, especially in combination with grazing (Hulbert 1986, review), actually promotes spread of these species (Kline 1986; Randa and Yunger 2001).

Prescribed fire is often used as a part of management plans for controlling sweetclover. Eckardt (1987) presents information from managers who use prescribed fire as part of their management plan for prairie sites where sweetclover occurs. Based on work in Minnesota, Heitlinger (1975) suggests three possible strategies for using prescribed fire to reduce soil seed reserves and prevent production of additional seed: (1) burning annually in early May, or when second year shoots are clearly visible; (2) burning every other year in early July, before second-year plants ripen seed; and (3) burning annually in early September near the beginning of the “critical growth period” when roots are growing rapidly. Any surviving second-year plants must be hand-pulled or clipped at the base before they produce seed (Heitlinger 1975). In Wisconsin, a combination of an April burn followed the next year by a May burn was more successful in reducing white sweetclover than other burning combinations. Heavily infested prairie stands where this burning combination was conducted twice, separated by 2 years without burning, became almost completely free of white sweetclover (Kline 1986). Problems with this method may arise if the burn is either too early or patchy, leaving viable seeds or undamaged second-year shoots (Cole 1991). These relatively aggressive fire prescriptions designed to control sweetclover will have variable impacts on desirable native species that

must be considered before implementing management plans. In arid regions, there may be insufficient fuels for frequent burning as well. Combinations of mowing or herbicides with prescribed burning are also suggested for sweetclover control (Cole 1991; Eckhardt 1987).

No information is available regarding the potential for sweetclover to alter fuel characteristics or fire regimes in tallgrass prairie.

Common buckthorn—Common buckthorn is a dioecious, large shrub or small tree that attains heights up to 20 feet (6 m). Although it has the ability to produce vegetative sprouts, it reproduces primarily by seed. While it can successfully establish in undisturbed sites, there is evidence that excess shading reduces seedling growth (Converse 1984a, TNC review).

Common buckthorn invades oak savanna, pastures, fens, and prairies. Savanna and prairie communities where fire has been excluded may be especially vulnerable to common buckthorn invasion (for example, Apfelbaum and Haney 1990; Packard 1988¹). Boudreau and Willson (1992) report that where it has invaded oak savanna habitat at Pipestone National Monument in Minnesota, native midstory and understory species have been virtually eliminated. Godwin (1936) reported an example in Europe for a related buckthorn species, glossy buckthorn (*Frangula alnus*), where seedlings established and developed into a continuous shrub thicket in 20 years in a mixed sedge marsh. The most detailed description of stand development in common buckthorn yet presented is by Archibold and others (1997). In this study, initial establishment of a few individuals on a site near Saskatoon, Saskatchewan, was followed by a localized increase in seedling recruitment once the initial trees became reproductive.

Common buckthorn often forms dense, even-aged thickets in both wetlands and woodland understories (for example, Archibold and others 1997; Packard 1988). These stands may reduce or alter understory herbaceous density and composition, thus altering fuel characteristics on invaded sites. This topic deserves further study.

Common buckthorn reproduces from seed. Seeds are dispersed by birds and mammals, and require scarification for germination. Seedling recruitment is most successful where there is ample light and exposed soil (Converse 1984a). These qualities imply that common buckthorn may be adapted to seedling establishment in the postfire environment, although research on this topic is unavailable.

Common buckthorn also sprouts vigorously following removal of or damage to topgrowth (Heidorn 1991),

and a limited amount of information suggests that large common buckthorn plants are able to survive fire but that small individuals and seedlings are killed. Boudreau and Willson (1992) report that spring burning in an oak savanna in Minnesota killed common buckthorn seedlings, which were unable to sprout after fire. However, mature trees were only top-killed and survived by sprouting. Converse (1984a) suggests that prescribed fire treatments do not control *Rhamnus* species, in part because species in this genus suppress herbaceous fuel beneath the canopy.

Repeated fires may facilitate development of the herbaceous layer and lead to eventual control of common buckthorn, with the caveat that adequate fuels must exist for control to be obtained through burning. Heidorn (1991) suggests that regular prescribed fire kills buckthorn (*R. cathartica*, *Frangula alnus*, and *R. davurica*) seedlings and larger stems in fire-adapted upland and wetland sites, and that annual or biennial burns may be required for 5 to 6 years or more to reduce the population. Prescribed spring burns conducted annually for 5 years from mid- to late April (when cool-season grasses had initiated growth) at Pipestone National Monument resulted in no appreciable change in common buckthorn cover (Becker 1989). In a dry oak forest dominated by hybrid black oak (*Quercus velutina* x *Q. ellipsoidalis*) and heavily infested with bush honeysuckle (*Lonicera x bella*) and common buckthorn, prescribed burns in 2 consecutive years followed by a year with no burn resulted in a decrease in cover of buckthorn in each of the burn years, followed by a slight increase in the no-burn year. Most individuals sprouted from the base, but sprouts were not vigorous (Kline and McClintock 1994).

Southern Tallgrass Prairie Formation

The southern tallgrass or coastal prairie formation occurs in east-central Texas and along the Gulf coastal plain in Texas and Louisiana. It is composed of gently rolling to flat plains, with elevations ranging from sea level to 1,300 feet (400 m). Most of the Coastal Plain streams and rivers are sluggish, with numerous wetland areas along the coast. Winters are warmer and there is more precipitation than in the more temperate northern and central prairies. Average annual precipitation ranges from 35 to 55 inches (890 to 1,410 mm) from west to east along the coast. The length of the frost-free season varies from 300 days to an almost entirely frost-free climate along the coast (Bailey 1995; Garrison and others 1977).

¹ Packard (1988) provides a particularly poignant quote, “An especially sad landscape features forlorn, aristocratic old oaks in an unbroken sea of buckthorn – the understory kept so dark by the dense, alien shrubbery that not one young oak, not one spring trillium, not one native grass can be found”

Vegetation in the southern tallgrass formation consists of savannas, prairies and coastal wetlands. Post oak and blackjack oak dominate the cross timbers habitat that abuts the prairie from northeastern Texas to southeastern Kansas, and oak and hickory are common on the eastern ecotone near the forest region. At higher elevations, little bluestem and indiangrass dominate. Switchgrass and eastern gamagrass (*Tripsacum dactyloides*) become dominant in low areas. In areas where prairie and salt marsh intergrade (salty prairie), gulf cordgrass becomes the dominant graminoid. Big bluestem is present in moist, sandy soils but is less common than in the northern and central tallgrass prairies. Brownseed paspalum is an important and conspicuous part of the vegetation of the upper coastal prairie north of the San Antonio River, while meadow dropseed (*Sporobolus asper* var. *asper*) replaces it in the lower coastal prairie (Smeins and others 1992).

The coastal prairie has been largely converted to agricultural fields and urban areas and is severely infested with invasive species. Only about 1 percent of the original coastal prairie remains in a seminatural condition (USGS 2000). Research in coastal prairie at the Brazoria National Wildlife Refuge, Angleton, Texas, provides some preliminary observations about the relationship between fire and invasive species in this landscape. Factors such as previous soil disturbance and soil salinity influence both variety and abundance of native and nonnative species. For example, nonnative invasive species such as Chinese tallow and Macartney rose are well established in abandoned rice fields and previously overgrazed areas, respectively. Areas with high-salinity soils are generally free of invasive species, with the exception of tamarisk. Several other invasive species occur on roadsides and in disturbed areas of the refuge but have not yet invaded native communities. This suggests that these species may be inhibited by the competitive effects of native dominant species, and that practices that support native species' dominance may discourage further invasions. For example, since fire has been reintroduced to this landscape, native species seed production has increased and recolonization of former agricultural fields by native species has been observed. Further research is needed to evaluate the threat potential of nonnative species and the role that fire plays (Grace and others, unpublished report, 2005).

The following discussion provides information on the biology and fire ecology of some important nonnative invasive species in the southern tallgrass formation. Common nonnative invasive woody species include Chinese tallow, tamarisk, and Macartney rose. Several introduced pasture grasses have become invasive in southern tallgrass ecosystems and mesquite savannas of southern Texas. Invasive grass species in these

ecosystems include yellow and Caucasian bluestems, guineagrass, and Johnson grass. Cogongrass (*Imperata cylindrica*) is just beginning to invade areas of southern tallgrass prairie, and itchgrass is also potentially invasive in southern tallgrass prairie ecosystems. In addition to the species covered in this section, species such as Canada thistle, sericea lespedeza, and buffelgrass are also invasive in southern tallgrass prairies, but are covered in other sections of this report.

Chinese tallow—Chinese tallow is an early successional tree (Bruce and others 1995) that can reach heights as great as 60 feet (18 m) and diameters in excess of 3 feet (1 m). The maximum life span is not known, but after a few decades trees typically become senescent. This species has exceptional growth rates and precocious sexual reproduction, producing seed within only a few years of establishment. Chinese tallow is also capable of vigorous sprouting from the roots following top-kill or damage, either at the base of the tree or at distances up to 16 feet (5 m) from the trunk. North American populations of Chinese tallow may be susceptible to herbivory, but the predators that control them in their native and other habitats do not occur in southern tallgrass prairie (Rogers and Siemann 2002, 2003, 2004, 2005; Siemann and Rogers 2003a, b).

Seeds of Chinese tallow are dispersed primarily by birds or via surface water flow. Seed germination typically takes place during fall through spring. Cameron and others (2000) indicate that seeds can survive up to 7 years when stored in refrigerated conditions. Chinese tallow appears to develop at least a short-lived seed bank (Renne and Spira 2001), though it is doubtful the seeds last more than a few years in the field. Preliminary analyses indicate that in the field, tallow seeds germinate within their first year without a light requirement if moisture is available (Billock, personal communication, 2003). There is also evidence that germination is stimulated by fluctuating temperatures (Donahue and others 2004, Nijjer and others 2002), but there is no indication that germination and establishment are particularly promoted by fire; information is lacking on this point.

The distribution of Chinese tallow extends from South Carolina in the east to southern Texas, and it has recently established in areas of southern California. Studies indicate that it has a broad tolerance of soil and moisture conditions (Butterfield and others 2004), although it can be limited by excess salinity (Barrilleaux and Grace 2000). Chinese tallow is likely to be a problem invader only in the southernmost parts of the Central bioregion. Chinese tallow invades a variety of habitats, but the principal grasslands it occupies are coastal marsh and coastal prairie. Here it establishes in abandoned rice fields, riparian zones and fence lines, and then spreads into the prairie (fig. 7-3). It appears



Figure 7-3—Chinese tallow is a fast growing tree that invades both disturbed and undisturbed prairie (as well as other habitats) in the southern tallgrass prairie formation. (Photo by Jim Grace.)

to be able to establish in undisturbed sod at some sites (J. Grace, personal observations, Brazoria National Wildlife Refuge, Texas, 1997-2002), though competition from native vegetation can inhibit persistence (Siemann and Rogers 2003a). It is likely that this species poses the greatest single threat to the conservation and restoration of the southern tallgrass prairie due to its capacity for rapid invasion (Bruce and others 1995) and its dramatic impacts (Grace 1998).

Studies indicate that Chinese tallow is vulnerable to fire when fuels are adequate (Grace 1998). Seedlings, in particular, suffer substantial mortality when burned within their first year. Because of its well-developed capacity for regrowth, it has been hypothesized that Chinese tallow is able to survive burning once it reaches a sufficient size. The critical size would depend in part on available fuel and related fire intensity/severity. Therefore, if burning can be conducted at sufficient frequencies and fuel conditions are sufficient for complete burns to be maintained, fire can be used to keep trees below the critical size and maintain the pyrogenicity of the system. Several experiments (Grace and others, unpublished report, 2005) indicate that the relationship between fire effect and tree size is actually complex and depends on season of burn as well as other factors. Available evidence indicates that burns have long-term detrimental effects on even the largest Chinese tallow trees. Thus, for isolated trees surrounded by a sufficient layer of herbaceous fuels, even the largest individuals can be top-killed by repeated fires (J. Grace, personal observations, Brazoria National Wildlife Refuge, Texas, 1997-2002).

What makes Chinese tallow invasions more problematic than invasions by other woody species is the change in fuel properties of tallow-invaded communities. When trees grow in closed stands, herbaceous fuel

biomass and continuity are rapidly reduced, making the system less flammable. Currently, only widespread chemical applications are capable of returning these ecosystems to a condition where grass cover is sufficient to carry fire.

Substantial effort is underway to determine whether a prescribed burning regime can be used to prevent Chinese tallow from invading native prairie (Grace and others, unpublished report, 2005). Many factors influence successful management, including (1) growth conditions for tallow, (2) growth conditions for fuel species, (3) site history, (4) site management, and (5) seed sources and site conditions for tallow recruitment. Results are preliminary and in need of verification. Nonetheless, there are several tentative findings from this work:

1. Fire results in substantial mortality of tallow trees only when they are small (less than 1 foot tall).
2. Top-killing of Chinese tallow can be accomplished for trees of any size as long as there is sufficient fuel around the tree to expose the base of the tree to prolonged lethal temperatures. Tallow cannot be managed successfully using fire alone when fuel conditions are insufficient.
3. Resprouting rates are high (greater than 50 percent) for established tallow trees, and regrowth can be rapid (within weeks following a growing-season fire).
4. Equal rates of mortality or top-kill can be obtained using either growing- or dormant-season burns. However, regrowth rates are greater following dormant-season burns. Thus overall, growing-season burns are more effective in controlling tallow, though dormant-season burns can be of some use.
5. The presence of individual trees does not suppress herbaceous growth sufficiently to render fire ineffective for management. However, on sites previously under cultivation, fuel load and continuity are often poor and individual trees are not likely to be controlled by fire.
6. Dense seed sources can overwhelm even mature prairie. Only small gaps in grass cover are needed for tallow to establish successfully. Thus, unmanaged prairie is highly vulnerable to complete replacement by tallow when seed sources are within bird-dispersal range.
7. It appears that when fuel conditions are sufficient to obtain complete burns, growing-season burns approximately every 3 years can keep tallow from invading and taking over prairie. However, single burns that are not repeated can actually increase the vulnerability of prairie to tallow invasion (Hartley and others 2007).

Chinese tallow has ability to establish and grow rapidly, reducing fine fuels in the understory. Frequent burning and vigilance will be needed to prevent further conversion of great tracts of southern mesic grasslands to tallow forests. Many areas have already succumbed and now require substantial rehabilitation to recover any sort of native character.

Tamarisk—Tamarisk refers to a complex of taxa that are of disputed affinity, of which “saltcedar” typically refers to *Tamarix ramosissima* or *T. chinensis*. In practice and in this report, these entities are generally combined for ecological and management discussions. Tamarisk is a long-lived (50 to 100 or more years), deciduous shrub or tree that attains heights of up to 26 feet (8 m), develops a deep taproot, and can spread by adventitious roots (Zouhar 2003c, FEIS review).

Tamarisk is most widespread and invasive along riparian areas of the Southwestern United States (chapter 8). As a facultative phreatophyte that is tolerant of high salinity (Zouhar 2003c), tamarisk is also well adapted to invade coastal prairie sites with saline soils, and is spreading along the Gulf of Mexico into the southern tallgrass prairie (Grace and others 2001). In the coastal prairie, sites occupied by tamarisk are generally upland and part of the fire-dependent ecosystem (J. Grace, personal observations, San Bernard National Wildlife Refuge, Texas, 1999-2000).

Reproductive strategies of tamarisk (evident from research in other bioregions) may allow it to establish and spread after fire in southern tallgrass and coastal prairie. As of this writing, no information is available in the literature regarding the response of tamarisk to fire in these ecosystems. However, it appears to be able to withstand a great deal of damage without incurring death (J. Grace, personal observation, San Bernard National Wildlife Refuge, Texas, summer 2001). Additionally, research in other ecosystems indicates that tamarisk is highly tolerant of burning and usually sprouts vigorously after fire (see section on “Riparian Formation” for more details page 137).

While changes in fuel and fire regime characteristics are sometimes attributed to tamarisk invasion in southwestern riparian areas (chapter 8), the effects of tamarisk invasion are difficult to disentangle from the many impacts of modern development that have enhanced its spread (Glenn and Nagler 2005; Zouhar 2003c). Fine fuels are much less abundant beneath tamarisk canopies in southern tallgrass prairies than in uninvaded prairie sites (J. Grace, personal observations, San Bernard National Wildlife Refuge, Texas, summer 2001), suggesting a lower likelihood for an invaded community to carry surface fire than an uninvaded grassland. Researchers in other areas note tamarisk plants with many stems and high rates of stem mortality, resulting in dense accumulations of dead, dry branches that catch leaf litter above the

ground surface and enhance the crowns’ flammability (Busch 1995; Racher and others 2002). Attempts to burn tamarisk in southern tallgrass prairie suggest that it is little affected by prescribed fire because its foliar moisture reduces flammability, thereby reducing fire continuity (J. Grace, personal observations, San Bernard National Wildlife Refuge, Texas, summer 2001).

The use of fire to control tamarisk has been investigated in riparian areas in shortgrass prairie and southern mixedgrass prairie in the Central bioregion (see the “Riparian Formation” section for details page 137). Tamarisk appears to be relatively unaffected by fire or competition. This is not to say that fire has no effect on tamarisk; rather, fire alone does not appear adequate as a control measure.

Macartney rose—This nonnative is an evergreen, perennial, thorny vine or shrub (Vines 1960), introduced from China primarily for erosion control and as a natural fence for pasture. Widely planted in the southern states, it has escaped in various habitats from Texas to Virginia. Macartney rose is a widespread invader in southern Texas and has been estimated to represent a problem on over a quarter million hectares (Scifres and Hamilton 1993). It is unpalatable and increases under grazing, forming dense, impenetrable mounds up to 20 feet (6 m) in height. It is an especially troublesome species in pastures as well as in prairies that have been formerly overgrazed and that are now being rehabilitated.

Existing evidence does not indicate that either fire or fire exclusion promotes the establishment and spread of Macartney rose (Scifres and Hamilton 1993). Rather, it seems that Macartney rose is very well adapted to surviving fire once it is established. Scifres and Hamilton (1993) suggest that Macartney rose may possess volatile oils, and we have observed that it burns readily (Grace and Allain, personal observations, San Bernard National Wildlife Refuge, Texas, 2001-2003). Fires generally top-kill the plants regardless of the season of burn. However, regrowth following winter or summer burns is rapid, and complete recovery may occur as early as the following season (Scifres and Hamilton 1993). Because of low mortality and rapid regrowth following burning, repeated burning is required to prevent Macartney rose from spreading further. We have observed substantial growth of Macartney rose following repeated growing-season burns that were followed by periods of extreme drought.

Macartney rose is not likely to be controlled using fire once it has established. Probably the best approach to controlling this species is through preventive management, such as by avoiding overgrazing and through early detection and eradication. Combined use of burning and herbicide treatments can kill individual plants, though the herbicide treatment is the critical element.



Figure 7-4—(A) Caucasian bluestem (in foreground), with inflorescence highlighted against the sky, invading tallgrass prairie. This species is a recent addition to the list of species of concern to mixedgrass and tallgrass prairie in the central and southern part of the central grasslands. (Photo by Mike Haddock, with permission, (see www.lib.ksu.edu/wildflower).) (B) A near monoculture of yellow bluestem at Barton Creek Preserve, Texas. (Photo by John M. Randall, The Nature Conservancy.)

Perennial grasses—Several nonnative perennial grasses have been introduced to increase forage value of rangelands or pastures in the Central bioregion. Among these, cogongrass, yellow bluestem (*Bothriochloa ischaemum*), Caucasian bluestem (fig. 7-4), guineagrass, and Johnson grass may pose some threat to southern tallgrass prairies. All of these species are at least partially adapted to fire, and most are capable of remaining in late-successional grasslands. Several of the invasive grasses found in the coastal tallgrass prairie are actually a greater threat in other formations, particularly the southern mixedgrass prairie. Here we shall treat those of greatest concern to coastal prairie, cogongrass and Johnson grass. Others will be discussed in the “Southern Mixedgrass Prairie Formation” section.

Information on the relationships of nonnative perennial grasses to fire and fire regimes in southern tallgrass prairie is limited. Cogongrass is the most studied with regard to fire ecology and the impacts of invasion on native fire regimes (Howard 2005b, FEIS review). Cogongrass is invasive in coastal systems from Florida west to Louisiana, with an outlying population in eastern Texas (fig. 7-5). It is not yet known to have invaded coastal prairie, although given its westward spread along the Gulf Coast, it is expected to establish in this ecosystem (Grace and others 2001). In parts of the eastern Gulf of Mexico, cogongrass is noted for



Figure 7-5—Cogongrass in eastern Texas. This grass is invasive along the coast of the Gulf of Mexico from Florida to Louisiana, and there is concern regarding its westward spread that it will become invasive in southern tallgrass prairie. (Photo by Jim Grace.)

the formation of monocultures once established. However, its small seeds and need for outcrossing appear to slow, though not stop, its rate of spread (King and Grace 2000). See Howard (2005b) and chapter 6 for more information on this species.

Johnson grass survives fire by sprouting from rhizomes that occur 8 inches (20 cm) or more below ground. As a seed banking species that establishes in open, disturbed sites, it is likely that Johnson grass is capable of postfire seedling establishment, although this is not documented. In some areas, Johnson grass may produce more persistent dry-matter biomass than associated native species, thereby altering fuel characteristics of invaded sites. Fire may be a useful tool for controlling Johnson grass when used in conjunction with postfire treatments to control rhizome sprouts (Howard 2004b, FEIS review). More information is needed to understand the relationship of Johnson grass to fire and fire regimes in southern tallgrass ecosystems.

The Great Plains Subregion

The Great Plains subregion represents the area of the central states that lies between the mesic tallgrass prairie subregion and the Rocky Mountains. It is generally characterized by mid- and shortgrass species (Coupland 1992).

This subregion coincides with the plains grassland and Texas savanna ecosystems described by Garrison and others (1977). The plains grassland ecosystem occurs on a broad area of high land that slopes gradually eastward and down from an altitude of 5,500 feet (1,700 m) near the eastern foothills of the Rocky Mountains to an altitude of 1,500 feet (460 m) in the Central States, where it merges into the prairie ecosystem. The prairie in the Great Plains is characterized by periodic droughts, with mean annual precipitation ranging from 10 inches (250 mm) in the north to more than 25 inches (630 mm) in the south (Garrison and others 1997). Droughts are less frequent and severe along the eastern edge, where annual precipitation approaches 30 inches (770 mm) from Oklahoma to Nebraska, and about 20 inches (510 mm) in North Dakota. The frost-free season ranges from less than 100 days in the north to more than 200 days in Oklahoma and Texas. Winters are cold and dry, and summers are warm to hot (Bailey 1995; Garrison and others 1977). In the Texas savanna, elevation ranges from sea level to 3,600 feet (1,100 m) on the Edwards Plateau. Precipitation ranges from 20 to 30 inches (510 to 770 mm) annually, and the frost-free season ranges from about 250 to well over 300 days (Garrison and others 1977). Three formations are described for this subregion, based on differences in climate, soils, and vegetation: (1) northern mixed-grass prairie, (2) shortgrass steppe, and (3) southern mixedgrass prairie.

Presettlement fire frequencies in the Great Plains varied, with estimates ranging from 4 to 20 years (Paysen and others 2000). Before Euro-American settlement, fires likely burned over very large areas in this subregion, and dry lightning storms perhaps played a greater role than in the mesic subregion. In areas where fuel might have been strongly limited by precipitation, fire regime characteristics would have varied over time. When cattle grazing became widespread, fuel loads were probably greatly reduced and there was an accompanying reduction in fire frequency. Generally, fuel loads in the Great Plains are substantially lower (about 1 t/ha) than in the mesic subregion. For this reason, continuity of fuels and the presence of flammable shrubs are even more critical to sustained ignition. Fire intensities associated with the herbaceous component in the Great Plains would generally be on the low side, but the presence of flammable shrubs could increase fire severity. More detail about historic changes in fire regime can be found in Paysen and others (2000).

Wide temperature ranges are found in the Northern Great Plains, though a somewhat reliable moisture supply makes the native grasses more hardy. Caution must be exercised to avoid overgrazing, particularly at times of peak evapotranspiration. In the Southern Great Plains, moisture tends to be predominantly available during mid growing season (though this can be variable), and fires during drought lead to conditions deleterious to vegetation recovery. Kucera (1981) offered an interesting theory about the relative differences in successional development following fire in semi-arid versus mesic grasslands, in which the development of thatch plays a central role. Thatch development can occur rapidly in mesic grasslands, leading to reduced production for grazers. In contrast, in semi-arid grasslands thatch is slow to develop and helps conserve moisture in the system. Thus, to maintain productivity, the optimum fire interval would probably be longer in more arid grasslands, particularly in shortgrass steppe, than in mesic grasslands.

The interaction of nonnative species with the fire regime in the Central bioregion is perhaps more dramatic and widespread in the Great Plains than in the mesic tallgrass subregion. Across the central grasslands, vegetation types high in native plant diversity have been heavily invaded by nonnative plant species (Stohlgren and others 1998, 1999a, b). In general, both native and nonnative species richness and cover are positively correlated with soil fertility and soil water holding capacity (Stohlgren and others 1999a). In shortgrass steppe and northern mixedgrass prairie, significantly more nonnative species occur in riparian zones than in adjacent upland sites (Stohlgren and others 1998). While grazing is thought to play a role in the spread of invasive species, a study in temperate and

montane grasslands found nonnative species richness and cover were no different in long-term grazed areas than in long-term ungrazed areas (Stohlgren and others 1999b). We might expect that cessation of grazing would increase fuel loads and fire frequency.

Northern Mixedgrass Prairie Formation

The northern mixedgrass prairie occurs from eastern Wyoming and parts of Nebraska north through Montana, South and North Dakota, and into Canada. Vegetation in this formation is dominated by herbaceous species such as western wheatgrass, blue grama, needle-and thread grass, Idaho fescue, rough fescue, green needlegrass, big bluestem and buffalo grass (Küchler 1964). It is characteristic of mixedgrass prairies that they contain elements of both tallgrass and shortgrass formations. In most cases, the major invaders of the shortgrass systems can probably also impact mixedgrass prairies. This is expected because tolerances are asymmetric. Drought tolerant species grow quite well with abundant moisture supplies while mesic species do not tolerate drought in most cases. We can also expect invaders that are very competitive in arid lands to be somewhat less competitive in more mesic areas compared to the highly productive, mesic natives. For these reasons, variations in competitiveness and threat level for invaders will likely vary depending on moisture supply. Latitudinal and elevational influences of a similar nature are expected. It should be noted that in this subregion, many factors can limit the application of prescribed burning (Larson, personal communication, 2006). Spring is often wet, and dominant cool-season grasses green up too quickly to carry fire in many cases. Also, by the time that cool-season grasses cure sufficiently to permit burning, dry conditions bring restrictions on burning. An additional difficulty is that both natives and nonnatives are frequently cool-season grasses, preventing the ability to time burning to selectively impact the nonnatives. Despite these restrictions, fire is a common feature of the landscape, and complex effects are frequently observed, often depending on community type (Larson and others 2001).

Cheatgrass—Cheatgrass occurs throughout the United States and southern Canada, though it is less common in the Southern States. It is highly invasive and occupies large areas of shrubland in the Interior West bioregion (chapter 8). It is also a significant and increasing component of grasslands along the eastern front range of the Rocky Mountains (Carpenter and Murray 1999, TNC review). In the Central bioregion, cheatgrass is problematic in northern mixedgrass prairie and even more troublesome in shortgrass steppe formations.

Cheatgrass is an annual that is able to complete its life cycle in the spring before the dry summer weather begins. The phenology of this species is particularly important to both its desirability as an early season forage grass and to its ability to compete with native species, survive fire, and alter fire regimes. It is able to germinate from fall through spring whenever moisture conditions are suitable, and growth proceeds throughout the winter irrespective of cold temperatures. Rapid early growth allows it to set seed before most other species. Once it begins to senesce, its persistent litter produces a highly flammable fuel (Zouhar 2003a, FEIS review).

Studies of cheatgrass in the Great Basin indicate that it is well adapted to fire and thrives under a regime of relatively frequent fires (3- to 10-year return intervals) in these ecosystems (Zouhar 2003a). Little published information is available regarding the relationship of cheatgrass with fire in the Central bioregion. Cheatgrass is favored by occasional burning at study sites within shortgrass steppe and mixedgrass prairies (Grace and others 2001). However, Smith and Knapp (1999) provide evidence that cheatgrass and other nonnative species are less frequent on annually burned tallgrass prairie sites at Konza Prairie, Kansas than they are on less frequently burned and unburned sites.

Because it dries 4 to 6 weeks earlier than native perennial species and also retains its dead leaf tissues for up to 2 months longer in fall, cheatgrass infestations may increase opportunities for fire. Fire has been shown to reduce densities of cheatgrass the following year; however, the plants that establish generally have especially high seed production (perhaps due to increased soil resources and reduced intraspecific competition). Because of this, cheatgrass recovery after fire is rapid (Young and Evans 1978).

In mesic grasslands, cheatgrass does not appear to be an especially successful competitor against fire-adapted native perennial grasses, and it does not appear to pose as great a threat to native communities. In semiarid and arid grasslands and shrublands, the tolerance of cheatgrass to dry conditions, its ability to fill in the spaces between shrubs and other grasses, and its retention of dead leaf tissue gives it the potential to change fuel characteristics and ultimately alter fire regimes. This property probably does not have as dramatic an effect in the Central bioregion as in more arid regions, where cheatgrass can completely alter fire regimes (chapter 8).

Japanese brome—Japanese brome (*Bromus japonicus*) is a cool-season annual grass that has been widely planted for rangeland improvement and is highly palatable to deer and bison. It is a prolific seed producer, and germination appears to be enhanced by a layer of

litter that helps to retain moisture (Howard 1994, FEIS review). Most successful seed germination occurs in fall when sufficient rainfall occurs (Whisenant 1990b). Because of dormancy mechanisms, fall germinants are typically from the previous year's seed crop (Baskin and Baskin 1981).

In the Western United States, Japanese brome occurs in prairie, sagebrush (*Artemisia* spp.) steppe, piñon pine (*Pinus edulis*, *P. monophylla*) woodland, and arid grasslands. It is most common on disturbed sites and grazed areas (Nagel 1995; Smith and Knapp 1999; Stohlgren and others 1999b), but is also reported to invade native communities. It has been found to invade shortgrass prairie in eastern Wyoming (Fisser and others 1989), mixedgrass prairie in southwestern South Dakota (Cincotta and others 1989), tallgrass prairie in central Oklahoma (Ahshapaneck 1962) and Kansas (Smith and Knapp 1999), mesquite (*Prosopis* spp.) savanna in Texas (Heitschmidt and others 1988), and riparian zones in south-central Oklahoma (Petranka and Holland 1980). This species is considered a threat to native diversity in rangelands and prairie because of its ability to out-compete many native perennials for water and nutrients. Disturbance appears to enhance Japanese brome populations. For example, mowing in a Kansas tallgrass prairie in August resulted in a major increase in its abundance (Gibson 1989). The absence of disturbance may lead to the disappearance of Japanese brome from a site when woody plants invade (Fitch and Kettle 1983), though it appears to persist even into late successional conditions when woody plants do not dominate (Fitch and Kettle 1983; Hulbert 1955).

The current view of the effects of fire on Japanese brome population dynamics is based on the need for this species to have sufficient moisture for successful establishment from seed and the role of plant litter in retaining soil moisture (Whisenant 1989). Fire not only kills the majority of Japanese brome plants and much of the seed retained by the plant, but also removes the litter layer. Thus, populations of Japanese brome may be substantially reduced following fire (for example, Ewing and others 2005; Whisenant and Uresk 1990). The rate at which they recover depends on precipitation; when there is ample fall precipitation, litter is not required for successful establishment and populations can rebound immediately (Whisenant 1990b). When moisture is less available, a litter layer is typically required before population recovery occurs. An example consistent with this hypothesis comes from Anderson (1965), who observed that annually burned, native prairie in the Flint Hills of Kansas remained free of Japanese brome while grazed prairie that was not burned was invaded by both this species and Kentucky bluegrass. Similarly, after 17 years of periodic burning (about once every 3 years,

on average) and resting from grazing at Willa Cather Prairie in Nebraska, relative abundance of Japanese brome and cheatgrass was reduced from about 4.5 percent to about 0.03 percent (Nagel 1995). A similar pattern was observed in tallgrass prairie in Kansas, where Japanese brome occurred at higher frequency on unburned sites (25 percent) than on annually burned sites (14 percent) (Smith and Knapp 1999).

Moisture availability may be more important to Japanese brome population dynamics than the effects of fire. For example, Japanese brome density increased dramatically on both burned (from 12,141 to 194,253 plants per acre 1 year after fire) and control plots (from 6,295 to 482,036 plants per acre during the same period) in the Hill Country of Texas following unusually high winter rainfall (Wimmer and others 2001). In circumstances where Japanese brome can overcome moisture limitation, recovery from fire permits it to develop substantial dominance.

Knapweeds—Several species of knapweeds occur in temperate grassland ecosystems, although they tend to be more problematic in the Western United States than in the Central bioregion. A partial list of knapweeds that are important in semi-arid regions of the Central bioregion includes spotted knapweed, diffuse knapweed (*Centaurea diffusa*), Russian knapweed (*Acroptilon repens*), and yellow starthistle (*C. solstitialis*). Except for spotted knapweed, very little information is available on the ecology and impacts of these species in the Central bioregion.

Spotted knapweed—Spotted knapweed is a short-lived perennial that has spread throughout most of the United States except for the Southeastern States. The highest concentrations are in the Interior West bioregion (chapter 8) and western Canada, though it is also of major importance in the upper Great Plains and North Central States. Spotted knapweed reproduces from seed and forms a persistent seed bank. Seeds are dispersed by gravity and by human activities, and they typically germinate in fall or early spring (Schirman 1981; Spears and others 1980). Established plants are reported to have an average lifespan of 3 to 5 years and a maximum lifespan of at least 9 years (Boggs and Story 1987). As plants age, they may develop short lateral shoots and multiple rosettes (Watson and Renney 1974, review).

Fires create the type of disturbance that may promote establishment and spread of knapweeds by creating areas of bare soil and increasing the amount of sunlight that reaches the ground surface. Spotted knapweed seedlings may emerge from the seed bank or establish on bare ground from off-site seed sources following fire, and mature spotted knapweed can survive on-site and sprout from root crowns following fire (Zouhar 2001c, FEIS review). It is generally

presumed that a single, low-severity prescribed fire will not control spotted knapweed, and may lead to increased population density and spread of this species. This is supported by research and observations in the Interior West bioregion (chapter 8; Sutherland, personal communication, 2006; Zouhar 2001c).

Research and observations from the Central bioregion, however, suggest that prescribed fire can be used to control spotted knapweed under some conditions. In a greenhouse study using seeds from a spotted knapweed-infested site in Michigan and temperatures simulating controlled burns, germination of spotted knapweed seeds was significantly ($P < 0.05$) reduced when exposed to 392 °F (200 °C) for 120 seconds or more, and when exposed to 752 °F (400 °C) for 30 seconds or more (Abella and MacDonald 2000). Additionally, results from a greenhouse experiment suggest that burning shortly after seed germination (mid-April to early May) (using fuel loads near the average biomass for temperate grasslands, and intended to simulate both low- and high-severity fires) may reduce spotted knapweed seedling establishment. Burning significantly reduced seedling establishment, regardless of whether the burn occurred prior to seed germination, during the cotyledon stage (1 week), or after seedlings had initiated primary leaves (2 weeks). Timing, rather than fuel load, had a greater effect on seedling establishment, with the greatest reduction occurring when burning occurred 2 weeks after seeding (MacDonald and others 2001). These results suggest that spring burns (timed to kill recently emerged knapweed seedlings) in combination with measures to remove surviving knapweed plants between burns could result in reduced density of spotted knapweed in plant communities dominated by warm-season grasses and forbs that are stimulated by burning during this season (Howe 1995; MacDonald and others 2001).

Emery and Gross (2005) compared the effects of spring, summer, and fall burns at two frequencies (annually from 2000 to 2003 and biennially, 2001 and 2003) on survival, growth, and reproduction of spotted knapweed populations in prairie remnants in southwestern Michigan. Annual summer burning was the only treatment that significantly reduced overall population growth rates. Results further indicated that attempts to control spotted knapweed should focus on combinations of treatments that target both reproductive and non-reproductive adults.

On prairie sites in Michigan with low to moderate spotted knapweed density and sufficient fine fuels to carry a fire, annual spring prescribed burning under severe conditions (when humidity and dead fine fuel moisture are as low as possible) serves to reduce spotted knapweed populations and increase the competitiveness of the native prairie vegetation. On some sites, 3 years of this regimen reduced spotted knapweed to

the point where it could be controlled by hand-pulling individuals and increasing the fire-return interval to 3 to 5 years (McGowan-Stinski, personal communication, 2001).

Prescribed burning of spotted knapweed can be difficult, especially if no fine grass fuels are present, because fire does not easily carry through spotted knapweed stems (Xanthopoulos 1988). In dense infestations (>60 rosettes/m²) in Michigan, broadcast burning is ineffective due to lack of adequate fuel to carry the fire. In this case, spotted knapweed plants can be killed with repeated spot-burning (using a propane torch) of individuals and sprouts 3 to 4 times during the growing season until root reserves are depleted. Seedlings emerge in the time between burning treatments and are killed with subsequent burning, thus depleting the seed bank. This treatment does not seem to harm existing prairie natives; however, newly germinating natives may be at risk, and seeding after the last burn treatment may help them to recover (McGowan-Stinski, personal communication, 2001).

Spotted knapweed infestations can alter fuel characteristics and may reduce flammability in invaded communities (Xanthopoulos 1988), although spotted knapweed fuel loading varies between sites (Xanthopoulos 1986). Existing grass fuel models work poorly for spotted knapweed unless associated grasses exceed 40 to 50 percent cover (Xanthopoulos 1988), so a specific fuel model was developed for this species. In order for a stand of spotted knapweed to carry fire, both modeling and field tests indicate that burns should be conducted in early spring prior to grass and forb growth (because of the high moisture content of grasses and forbs in the spring). Moreover, the sparse foliage and discontinuous nature of spotted knapweed biomass suggest that, under low wind speeds and flat or declining slopes, fires will likely fail to spread (Xanthopoulos 1986).

Diffuse knapweed—Diffuse knapweed functions as a biennial or short-lived, semelparous perennial. It has a well-developed taproot but is not capable of vegetative spread, relying solely on seed for population growth (Zouhar 2001b, FEIS review). Most germination takes place in spring, and seedlings develop into rosettes that persist for 2 to several years before flowering and dying (Watson and Renney 1974). Reproductive success depends on plant size, and the proportion of diffuse knapweed plants that survive to flower and produce seed each year is largely a function of competition for moisture and increases with available growing space (Powell 1990). Diffuse knapweed currently occurs largely in the Western United States and Southwestern Canada. It is not widespread in the Great Plains or tallgrass prairie region, although it does occur in Iowa, Missouri, Minnesota, Indiana, Michigan, Kentucky, and Tennessee. There are no published studies on the relationship of diffuse knapweed to fire; however, its

capacity to withstand repeated mowing (Roche and Roche 1999, review; Watson and Renney 1974) and its possession of a taproot suggest that it may be able to tolerate moderate-severity fire.

Russian knapweed—Russian knapweed is a perennial herb that occurs throughout the Western and Central United States, including the Great Plains, but is largely absent from eastern and southeastern areas (Watson 1980, review). This species is used by bighorn sheep and white-tailed deer, although it is not palatable to livestock and causes a neurological disease in horses (Zouhar 2001a, FEIS review). Botanical descriptions indicate that Russian knapweed has an extensive root system, is capable of rapid vegetative spread by root buds, and typically forms dense, often monotypic colonies that may persist more than 75 years (Watson 1980). Russian knapweed is also capable of establishing from seed; however, seed dispersal over any distance larger than the height of the plant requires a dispersal agent.

No information regarding fire adaptations of Russian knapweed is available in the literature, although its deep, extensive root system is likely to survive even severe fire. The tolerance of Russian knapweed seeds to heating is unknown. Russian knapweed probably sprouts from root buds after fire and may establish from on-site seed or from seed brought on site by animals or vehicles (Zouhar 2001a). Researchers burned plots of Russian knapweed in Wyoming after first mowing them to a height of 3 to 5 inches (8 to 12 cm). Observations following these treatments suggest that Russian knapweed plants were injured to a depth of 1 to 2 inches (2.5 to 4 cm) below the soil surface, and that lateral roots at the 3- to 6-inch (7.5 to 15 cm) depth did not appear to be injured. Russian knapweed seedheads were also burned but the seed “appeared to be viable”; however, Russian knapweed seedlings were not observed after burning (Bottoms, personal communication, 2002).

Yellow starthistle—Yellow starthistle, an invasive winter annual, is most problematic and best studied in California (chapter 9) and Interior West (chapter 8) ecosystems. Infestations of yellow starthistle in the eastern two-thirds of the United States are sporadic and localized, where populations apparently fail to establish and persist on a year-to-year basis, possibly because of unfavorable growing conditions (Great Plains Flora Association 1986; Maddox and others 1985). More research is needed to understand the role of yellow starthistle invasion in ecosystems of the Central bioregion. For information on the use of fire to control yellow starthistle and the potential for yellow starthistle to impact fire regimes, see chapter 9, or the FEIS review for this species (Zouhar 2002a).

Leafy spurge—Leafy spurge is a perennial forb that quickly develops an extensive root system reaching depths of 15 to 30 feet (4.5 to 9 m) (Simonin 2000, FEIS review). This species has a prolific capacity for vegetative spread and regrowth, which is facilitated by the production of a basal crown of tissue just beneath the soil surface (Biesboer and Eckardt 1987, TNC review). This basal crown persists for many years, and develops a large number of buds that produce aboveground shoots as well as new roots. In addition, leafy spurge produces rhizomes and root buds that are capable of forming independent plants, which is likely to occur when disturbed (Selleck and others 1962). A notable characteristic of leafy spurge is its capacity to recover from damage by sprouting from rhizomes and root buds as early as 7 to 10 days after germination. Another important characteristic is prolific seed production in explosive capsules that can disperse seed up to 15 feet (4.5 m). Seeds may remain viable in the soil for up to 8 years, though it has also been reported that most seeds germinate or perish within 2 years of production (Selleck and others 1962). Based on this information, it appears that leafy spurge relies on its bud bank as much as its seed bank for regeneration.

The highest concentrations of leafy spurge in the United States are currently in Oregon, Idaho, Montana, Wyoming, Colorado, North Dakota, South Dakota, Nebraska, Minnesota, and Wisconsin. Leafy spurge occupies a broad range of ecological conditions from xeric to subhumid, though its most rapid growth is in more mesic sites. In particular, it invades disturbed areas with exposed mineral soil (Belcher and Wilson 1989; Wilson 1989), but it can also establish and spread in pastures and ungrazed native grasslands (Selleck and others 1962). Forbs and grasses may be completely displaced where leafy spurge invades (Biesboer and Eckardt 1987) and forms dense patches (fig. 7-6). In addition to adverse effects on native diversity and restoration efforts (Belcher and Wilson 1989; Butler and Cogan 2004), it has been found to be toxic to cattle and humans, though goats and sheep are able to consume it without adverse effects (Landgraf and others 1984; Stoneberg 1989). In balance, leafy spurge perhaps poses the greatest problems in mixedgrass prairies, though it is also a problem in semiarid and mesic conditions.

One of the more frequently cited examples of adverse effects of fire on leafy spurge is attributed to Dix (1960). In this study, he compared three pairs of unburned and previously burned sites in North Dakota native grasslands to evaluate the effects of fire on community composition. Leafy spurge was found in only one of the six sites examined, and this site happened to be one of the unburned grasslands. Dix drew no conclu-



Figure 7-6—Meadow infested with leafy spurge in Rist Canyon, west of Fort Collins, Colorado. (Photo by William M. Ciesla, Forest Health Management International, Bugwood.org.)

sions about the effects of fire on leafy spurge from this study, and given the absence of information about its distribution prior to the burns, we feel that no firm conclusions can be drawn from this example.

Other studies have generally found prescribed fire to have limited effects on leafy spurge. Prosser and others (1999) examined the effects of fall burning on leafy spurge in North Dakota and found no effects on stem densities the next year compared to unburned controls. Wolters and others (1994) conducted studies of the effects of spring and fall burns on leafy spurge in mixedgrass prairie at the Little Missouri National Grassland in North Dakota. They found that both spring and fall burns reduced seed germination but increased stem density through vigorous postfire sprouting. Similarly, Masters (1994) found that stem densities increased with late spring burning in mixedgrass prairies in Nebraska. Fellows and Newton (1999) found that burning in late spring or early fall produced immediate reduction of leafy spurge densities, but these effects were not evident the following growing season. In grasslands at the Tewaukon National

Wildlife Refuge in North Dakota, leafy spurge canopy cover was not changed 2 months after fire, but canopy cover was greater in burned versus unburned plots 1 year after fire (Olson 1975). These studies suggest that leafy spurge is especially well adapted to sprouting following fire, and that burning alone is ineffective in controlling leafy spurge.

While there is evidence that spurge can be impacted by competition with commercial grass cultivars (Lym and Tober 1997), there is no evidence that fire can be used to tip the competitive balance toward native species to a sufficient degree to negatively impact leafy spurge.

Prescribed burning in conjunction with herbicide treatments may be more effective than either treatment alone for reducing leafy spurge (Masters and others 2001; Nelson and Lym 2003; Winter 1992; Wolters and others 1994). For example, Wolters and others (1994) found that herbicides with or without burning were most effective in reducing leafy spurge stem density, while spring burning, with or without fall herbicide application, was the most effective treatment for reducing leafy spurge seed germination. They concluded that a fall application of picloram plus 2,4-D followed by spring burning would most effectively reduce both germination and stem density (Wolters and others 1994). Prescribed fire was used to enhance chemical control efforts at Bluestem Prairie in Minnesota. According to Winter (1992), prescribed fire effectively removed plant litter in the most heavily infested leafy spurge areas, thereby increasing visibility of leafy spurge plants, especially small shoots, and enabling more chemical to reach leafy spurge foliage and roots (Winter 1992).

Success in the control of leafy spurge will likely involve biocontrol agents (flea beetle species), though it is not clear at this point the degree of success that can be expected (Larson and Grace 2004). Preliminary evidence suggests that the use of herbicide treatments can interfere with biocontrol success (Larson and others 2007) because the initial decline in spurge density following herbicide treatment can result in a decline in biocontrol agents. Fellows and Newton (1999) examined the effects of prescribed burning on the flea beetle *Aphthona nigriscutis*, a potential control agent, at the Arrowwood National Wildlife Refuge in North Dakota. They found that burning prior to beetle release enhanced establishment of most beetles. The insects did not persist on all plots, but on plots where they did persist, spring and fall burns 2 years later did not affect their population size.

Leafy spurge has a tremendous capacity to sprout following top-kill or damage to aboveground tissues. Even extensive control programs that attempt to eliminate leafy spurge using a combination of mechanical removal, biological control, chemical treatments, and

prescribed fire have found that this species must be treated repeatedly for many years to achieve satisfactory reductions.

Southern Mixedgrass Prairie Formation

The southern mixedgrass prairie occurs in eastern Kansas and Oklahoma, south through central Texas and into Mexico. This formation is dominated by little bluestem, side-oats grama, and blue grama. Shrubs become more common in the southernmost part of the formation, in the area described by Garrison and others (1977) as the Texas savanna ecosystem. Here the vegetation is characterized by dense to very open synusia of deciduous and evergreen low trees and shrubs. The grass varies from short to medium tall, and the herbaceous vegetation varies from dense to open (Garrison and others 1977). Common species include little bluestem and seacoast bluestem (Küchler 1964). Mesquite is the most widespread woody plant. Other woody species include *Acacia* species, oaks, juniper, and ceniza along the Rio Grande valley and bluffs (Garrison and others 1977; Küchler 1964). At present, there is a relatively small list of important nonnative invaders in southern mixedgrass prairie (table 7-2); these are primarily introduced grasses and trees such as tamarisk (covered in Riparian section).

Yellow and Caucasian Bluestems—Yellow and Caucasian bluestems, perennial, C₄ bunchgrasses, are among several species of Old World bluestems that have been seeded throughout much of the southern Great Plains for stabilizing marginal cropland or increasing forage production on rangelands (Berg 1993; Marietta and Britton 1989; McCoy and others 1992; Phillips and Coleman 1995). Spring burning and nitrogen fertilization are common management practices to improve productivity of Old World bluestem pastures (for example, Berg 1993; Phillips and Coleman 1995).

Spring burning of uniform stands of yellow bluestem in established pastures on former cropland in northwestern Oklahoma significantly reduced ($P < 0.01$) herbage yields by 6 to 30 percent per year. Soils were calcareous and noncalcareous loams (Berg 1993). Winter burning (February) of artificial grasslands (chaparral sites seeded to Lehmann lovegrass (*Eragrostis lehmanniana*) and yellow bluestem) in central Arizona resulted in a 16 percent increase in bluestem production following fire and rootplowing (Pase 1971). In central Texas, there is some indication that yellow bluestem has a high tolerance of burning and a wide tolerance of soil conditions (Gabbard 2003), though many details are lacking at present.

Caucasian bluestem appears to be well adapted to frequent fire. Research in India indicates that annual and biennial burning in either winter or summer resulted in increased abundance of Caucasian bluestem

over a 2-year period (Gupta and Trivedi 2001). According to an NRCS Plant Fact Sheet, however, Caucasian bluestem “does not respond as strongly to fire as do our native species,” although it is unclear what plant community or native species are being compared (USDA, NRCS 2006). Furthermore, at Konza Prairie, when compared to native C₄ grass species such as big bluestem, Caucasian bluestem had greater plant biomass, lower pools of soil N, lower rates of decay and carbon cycling, and higher foliar and root tissue C:N ratio in response to spring burning. Additionally, areas dominated by Caucasian bluestem had significantly lower plant species diversity. The authors conclude that the threat of invasion by nonnative C₄ species such as Caucasian bluestem raises a dilemma for using prescribed fire to promote native C₄ grasses in these prairies (Reed and others 2005).

Guineagrass—Guineagrass, an important invader of the western portion of coastal prairie, has short, creeping rhizomes and produces wind-dispersed seed, although seeds are often nonviable. It often forms dense stands in open pastures and disturbed areas and may displace natives on fertile soils. It is drought tolerant but will not survive long periods of desiccation. According to a review by the Invasive Species Specialist Group (2006), guineagrass survives grass fires (quick-moving fires that do not harm the rhizomes), regenerates rapidly from rhizomes, and may dominate the postfire plant community. More information is needed on this species’ relationship to fire in the Central bioregion.

Shortgrass Steppe Formation

The shortgrass steppe occurs from eastern Colorado and New Mexico into western Kansas and the panhandles of Oklahoma and Texas (fig. 7-1). This formation is characterized by tablelands and rolling lands, with numerous inclusions of eroded hills, buttes, and valleys, abutting the Rocky Mountains to the west. Presettlement fire regimes were variable in this ecosystem depending on ignition sources and fuel loads, with fire frequencies estimated at 4 to 20 years (Paysen and others 2000). The shortgrass steppe formation is dominated by blue grama (*Bouteloua gracilis*) and buffalo grass (*Buchloe dactyloides*) (Küchler 1964). According to Lauenroth and Milchunas (1992), about 60 percent of the shortgrass steppe remains in native vegetation and is used for grazing, while the remaining 40 percent is largely used for cropland. The shortgrass steppe is still considered to be largely comprised of native species and not yet significantly impacted by nonnative invasive species (Kotaniemi and others 1998). The list of invaders of major importance is short at present (table 7-2), with primary concern surrounding buffelgrass in the southern range and tamarisk in riparian zones.

Buffelgrass—Buffelgrass is a drought-tolerant, perennial bunchgrass, ranging in height from 4 to 60 inches (10 to 150 cm) at maturity, and reproducing both vegetatively and by wind- and animal-dispersed seed (Tu 2002a, TNC review). It is well established in disturbed and intact desert shrub communities in the Southwestern United States (chapter 8). It has been widely planted and established as a pasture grass in northern Mexico and western and southern Texas. It thrives on sandy soils where annual precipitation ranges between 8 and 47 inches (200 to 1,200 mm) (Hanselka 1988; Ibarra-F. and others 1995), and tends to form dense swards that exclude native vegetation (Tu 2002a).

Buffelgrass is well adapted to survive fire. Cool-season prescribed fires are sometimes used in southern Texas to maintain and rejuvenate buffelgrass pastures by removing litter and controlling woody invaders (Hamilton and Scifres 1982; Mayeux and Hamilton 1983; Scifres and Hamilton 1993). Research in India indicates that annual and biennial burning in either winter or summer increased abundance of buffelgrass over a 2-year period (Gupta and Trivedi 2001). A study in the west-central portion of the South Texas Plains produced similar results, where burning in February resulted in greater buffelgrass production (kg/ha) by mid-spring. However, buffelgrass production was lower on burned than unburned plots as soil water was depleted during summer (Hamilton and Scifres 1982). Prescribed winter (December or February) burning of a buffelgrass pasture in the lower Rio Grande Valley of southern Texas similarly resulted in small, temporary increases in buffelgrass standing crop (kg/ha), while burning combined with broadleaf herbicides to control associated invasive forbs increased buffelgrass standing crop as much as 3-fold (Mayeux and Hamilton 1983). Summer prescribed burning of buffelgrass in Sonora, Mexico, produced a mixed response, depending on plant phenology at the time of burning and possibly related to soil water availability. Burning in summer, after the accumulation of 2 inches (50 mm) of precipitation and between the second leaf stage and early culm elongation in buffelgrass, appeared to stimulate buffelgrass growth for 3 to 4 years. Conversely, burning later in the growing season, at the peak of buffelgrass live biomass production, reduced buffelgrass production by almost 50 percent for 4 years (Martin-Rivera and others 1999).

Where buffelgrass invades desert shrublands in the Southwestern United States and Mexico, it appears to increase fire frequency and spread (chapter 8) by introducing a novel life form (in other words, fuel type) in invaded communities (D'Antonio 2000). No information is available regarding the impacts of buffelgrass on fuel characteristics and fire regimes in invaded

communities in shortgrass steppe, but one might speculate that it would have less of an impact on fuel characteristics in a community already dominated by grasses.

Prescribed burning is not likely to be an effective control method for buffelgrass, especially when conducted in winter or other times when soil water availability is high (Tu 2002a), but might be more effective under drier conditions. However, no reports were found in the literature discussing application of prescribed burning in summer as a potential control method for buffelgrass in shortgrass steppe ecosystems.

Riparian Formation

The dominant variable influencing the distribution of species in the central grasslands is the availability of water. For species associated with riparian zones, this constraint is lifted, at least part of the time. Characteristics of riparian zones can vary substantially, as can the degree of constancy of water supply and edaphic influences. Although riparian zones in the Central United States are a small part of the total area, they are ecologically important and distinct from the surrounding communities in their species composition. The vegetation of the riparian zones, for example, is often of special importance as wildlife habitat (Smith 2000). As a group, plants of the riparian zones are distinctly different from plants of upland habitats, so the invaders of riparian areas are treated separately here.

Riparian zones are often dominated by trees, contrasting with the surrounding landscape (Shelford 1963), and range from narrow fringes of vegetation to broad floodplains kilometers wide. Little is known about fire regimes in riparian communities, though it is suggested that they differ from the surrounding communities in fire regimes and adaptations. For some riparian zones, fire frequency is quite low (Wright and Bailey 1982), though in other cases fires can be frequent. Periodic flooding, along with forest canopy development, can reduce herbaceous fuels and make fire less likely to spread into riparian vegetation.

Riparian zones are particularly prone to invasion by nonnative invaders (Kotanen and others 1998), even when phylogenetically independent comparisons are made. Because of their rich alluvial soils, floodplains along large rivers have often historically been converted to cropland and consequently support propagules of many nonnative invasive species. On the Rio Grande Delta, for example, restoration of native woody species is often impeded by a number of invasive grasses including bermudagrass, guinea-grass, Johnson grass, buffelgrass, Kleberg bluestem (*Dichanthium annulatum*), Angleton bluestem, and

Natal grass (*Rhynchelytrum repens*). Although research from these areas is unavailable, there is concern that these grasses not only prevent establishment of native trees and shrubs but also increase wildfire risk (for example, Twedt and Best 2004). Many riparian invaders are widespread, capable of transforming ecosystems through their complete domination, and difficult to control. Here we discuss two species in some depth, tamarisk and Russian-olive, and mention a third of lesser importance in the Central bioregion, giant reed.

Tamarisk—In the Central bioregion, tamarisk is most common in riparian areas in southern mixedgrass prairie and shortgrass steppe ecosystems in eastern New Mexico and western Texas, and it is spreading along the coastal plain of the Gulf of Mexico (see “Southern Tallgrass Prairie Formation” section page 125).

Tamarisk establishment and spread appear to be facilitated by fire, although experimental evidence is limited. Tamarisk produces abundant, wind-dispersed seed throughout the growing season that can germinate and establish on bare, saturated soil (Zouhar 2003c). Additionally, burned tamarisk trees that are not killed may show an enhanced flowering response (Stevens 1989). Tamarisk is also capable of vigorous and abundant sprouting from roots and root crown following fire or other disturbance that kills or injures aboveground portions of the plant (Zouhar 2003c). Even when completely top-killed, it may produce sprouts 7 to 10 feet (2 to 3 m) tall in the next year. For example, growth of surviving tamarisk plants after a July wildfire at Lake Meredith National Recreation Area, Texas, exceeded 6 feet (1.8 m) at the end of that growing season (Fox and others 1999).

Postfire response of tamarisk depends on season of burning, fire frequency, fire severity, and postfire plant community composition. Burning during the peak of summer appears to have the strongest adverse effect on tamarisk, presumably due to ensuing water stress (Grace and others 2001). Tamarisk mortality exceeded 60 percent 12 months following a July wildfire at Lake Meredith National Recreation Area, Texas (Fox and others 1999). In cases where annual burning can be achieved for several years, it is suggested that tamarisk can be controlled with fire alone (Duncan 1994). Similarly, control of tamarisk using prescribed burning has been reported by land managers when several years of fire treatments were applied in succession with high fuel loads (Racher and Mitchell 1999).

Several factors seem to influence flammability of tamarisk:

- High water and salt content make tamarisk difficult to burn under some conditions (Busch 1995; Racher and others 2002).
- Flammability of tamarisk increases with the build-up of dead and senescent woody material

within the plant, such that old, dense stands of tamarisk can be highly flammable (Hohlt and others 2002; Racher and others 2002).

- Tamarisk in dense stands that have not burned in 25 to 30 years exhibit extreme fire behavior and crowning due to the closed canopy, regardless of time of year. They can have flame lengths exceeding 140 feet (43 m), resulting in consumption of a majority of the woody material.
- Stands reburned within 5 to 6 years show much less intense fire behavior, carrying fire only if there is adequate fine fuel load and continuity. Few trees torch during burning, though some trees are top-killed (Hohlt and others 2002; Racher and others 2001).

Tamarisk stands in the Pecos Valley, New Mexico, can burn “hot,” with erratic fire behavior and numerous firebrands transported over 500 feet (150 m) downwind, regardless of whether the stand has burned before (Racher and others 2001, 2002).

Use of fire alone to control tamarisk is generally ineffective. In some areas, prescribed fire can be used to manage tamarisk by eliminating the closed canopy, slowing the rate of invasion, and allowing desirable vegetation to respond, thereby increasing diversity in monotypic tamarisk stands. Burning these communities under controlled conditions can also reduce the potential for costly wildfires that must be suppressed to avoid property loss (Racher and others 2001). Typically control of tamarisk requires some combination of mechanical and chemical control methods, sometimes integrating prescribed fire (Zouhar 2003c). Tamarisk plants that are stressed following defoliation by biocontrol beetles may also be more susceptible to mortality following fire (Brooks personal communication, 2007b). See the tamarisk review in FEIS for more details on the ecology and management of tamarisk and related species.

Much current research on prescribed burning of tamarisk has been conducted in the Pecos Valley, New Mexico, and in the Texas panhandle. Abstracts summarizing the status of this research are available in Texas Tech University’s *Research Highlights*. Research includes studies of fire behavior and spotting potential of fires in tamarisk-dominated communities (Racher and others 2001; Racher and others 2002); effects of season of burning on response of tamarisk to prescribed fire (Bryan and others 2001; Hohlt and others 2002; Racher and Mitchell 1999); response of herbaceous species to prescribed fires in tamarisk-dominated communities (Bryan and others 2001); effectiveness of using herbicides in conjunction with prescribed burning on tamarisk control (Racher and Mitchell 1999); and tamarisk response to wildfire followed by mechanical and chemical control efforts (Fox and others 2000; Fox, R. and others 2001).

Russian-olive—Russian-olive is a deciduous shrub or small tree that has been cultivated for a variety of purposes both in its native range and in North America. It is widely distributed from the west coast to the Dakotas, Nebraska, Kansas, Oklahoma, and Texas (Knopf and Olson 1984). Russian-olive readily establishes and spreads on disturbed floodplains, stream banks, and in some situations, marshy areas and wet grasslands (Olson and Knopf 1986). Once established, it appears to have a substantial capacity to withstand periodic drought and out compete native woody species, and is considered the successional climax species in many cases (Howe and Knopf 1991). Russian-olive was introduced to many of the Great Plains States by the early 1900s and it was planted extensively in windbreaks in the 1930s and 1940s in association with government programs. It has persisted and spread in many areas in the Central bioregion, especially in the understory of riparian communities in northern mixedgrass prairies in the western part of the Great Plains (Zouhar 2005b, FEIS review). It is also invasive in riparian areas in much of the Interior West bioregion (chapter 8).

Transplanted Russian-olive seedlings typically produce fruit about 3 to 5 years after transplanting; however, its average reproductive age varies with latitude (Borell 1971). On the Marias and Yellowstone rivers in eastern Montana, the average age of first reproduction for Russian-olive is around 10 years (Lesica and Miles 2001). Seeds are typically dispersed by birds and small mammals, and probably also by water and ice. Longevity of Russian-olive seed under field conditions is unknown, though stored seeds retain viability for about 3 years. Mature Russian-olive seed is typically dormant and may require scarification for germination. Russian-olive may be able to exploit suitable germination conditions over a relatively long time compared to native taxa with which it commonly associates (Shafroth and others 1995). It can establish on disturbed sites in full sun, in shade, or in intact ground cover, though it grows best in full sun. Shade tolerance, reproduction, growth, and recruitment rates may vary with latitude, making Russian-olive less invasive at the upper and lower latitudinal limits of its North American range (Borell 1971; Lesica and Miles 2001; Zouhar 2005b). Russian-olive can produce adventitious roots on buried stems (Brock 1998), and Bovey (1965) suggested that Russian-olive spreads by “underground rootstalks”; however, there is no evidence in the literature that indicates that Russian-olive spreads by asexual reproduction under field conditions, except following injury or top-kill.

No information in the literature specifically addresses Russian-olive regeneration or establishment from seed after fire, although several researchers and managers report that Russian-olive sprouts from the trunk, root crown, and/or roots after top-kill or damage

(Zouhar 2005b), and some report sprouting from roots and root crown following fire (Brock 1998; Caplan, personal communication, 2005; Winter, personal communication, 2005). Mixed species stands along the Rio Grande often become monospecific stands of Russian-olive following fire due to vigorous Russian-olive root sprouting (Caplan 2002). Managers should be alert to the possibility of Russian-olive establishment and spread in the postfire environment when it occurs on-site or in the vicinity of burn areas.

The growth habit (in other words, fuel arrangement) of Russian-olive varies among plant communities and site conditions. It may have a single trunk or multiple stems branching at ground level. On many sites, Russian-olive grows in dense thickets with close spacing, sometimes with scattered mature cottonwood in the canopy. Stands may be so dense that other riparian species are excluded entirely. The lower 6.5 feet (2 m) of vegetation may contain a tangle of dense, often dead, branches with little live foliage (Zouhar 2005b). Katz and Shafroth (2003) present data describing density and cover of several established Russian-olive populations in western North America.

Dense growth of Russian-olive may be more fire-prone than the native communities that it invades, although this has not been studied or reported in the literature and deserves further investigation. Additionally, there is little quantitative information on prehistoric frequency, seasonality, severity and spatial extent of fire in North American riparian ecosystems, where Russian-olive is commonly invasive. Many interrelated factors, such as urban and agricultural development, groundwater pumping, dams, and flood suppression, contribute to altered disturbance regimes in these ecosystems.

There is little information available on the use of fire to control Russian-olive. Observational evidence indicates that Russian-olive is top-killed by prescribed fire in tallgrass prairie (Winter, personal communication, 2005) and by wildfire in riparian communities on the Rio Grande River (Caplan, personal communication, 2005). Fire in tallgrass prairie generally does not top-kill trees greater than 2-inch (5 cm) DBH, so Russian-olive is maintained at “brush height” with regular burning in tallgrass prairie sites in Minnesota (Rice and Randall 2004; Winter, personal communication, 2005). According to Deiter (2000, review), stump burning may successfully control sprouting in Russian-olive, but it is time-consuming and therefore likely to be costly. Prescribed fire is probably most effective as a control method for Russian-olive when combined with other control techniques. The species’ occurrence in riparian zones may make the use of fire problematic in cases where adverse impacts on fire-intolerant natives may be greater in magnitude. This kind of information is needed to determine the role fire may play in management of Russian-olive.

Giant reed—Giant reed is an escaped ornamental grass that occurs in riparian and wetland areas throughout the Southern United States, from Virginia to California. Giant reed is clump-forming and can grow up to 20 feet (6 m) tall. In deep water, shoots of this species can be semi-floating. Giant reed spreads locally by rhizomes and is reported to flower only once every few years (McWilliams 2004, FEIS review), though in southern Louisiana it appears that some plants flower every year (J. Grace, personal observation, Brazoria National Wildlife Refuge, summer 2002). This species establishes and spreads in riverbanks, marshes, and floodplains and has been observed to replace common reed (*Phragmites communis*) on some sites. The above-ground material of giant reed is quite flammable once leaves have dried. It is capable of surviving fire because of extensive storage organs belowground (McWilliams 2004). At present, there appears to be no information on its effects on fire regimes or succession, nor information about whether fire aids in its establishment.

Conclusions

In general, native species of the central grasslands are adapted to frequent fires. Many native species are promoted by fires in appropriate seasons, and many nonnatives decline. According to Solecki (1997), fire and competition from native prairie plants eliminate most nonnative invaders in prairie ecosystems. However, a fraction of nonnative invaders can tolerate, avoid, or

alter the fire regime and pose major threats to central grassland ecosystems. Management of habitats that contain these species or are exposed to them will benefit from considering the interaction of these invaders with fire.

The interaction of fire and nonnatives is an important topic that we are only beginning to address, and additional work is needed. The literature describing how particular nonnatives react to burning and alter fuels and fire regimes is satisfying in its details for only a few species. The few examples where precise applications of burning can be used to favor native species over nonnatives provide encouragement that there is considerable potential for such uses of fire. On the other hand, the degree to which such potentials exist or whether they can be successfully accommodated in burn programs is not known at present.

Several areas are in need of research. One emerging topic of particular interest is how fire impacts biocontrol. Biocontrol agents are being used increasingly as a primary means of controlling nonnative plants, a trend that is likely to continue. How this will relate to the use of fire is largely unknown. Preliminary results for select cases, such as for fleabeetles that feed on leafy spurge, indicate that biocontrol agents can persist in the presence of periodic fire. It seems unlikely that this will universally be the case. Detailed studies will be required for each situation where biocontrol is used in combination with burning.