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Great Plains Region

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Introduction

The Great Plains, here encompassing the States of Kansas, Montana, Nebraska, North Dakota, South Dakota, Oklahoma, Texas, and Wyoming (Fig. A5.1), is a diverse landscape consisting of a complex matrix of native, seminative, and non-native grasslands intermixed with riparian and prairie woodlands, shrublands, forests, and intensively cultivated agricultural lands. The composition and abundance of the native vegetation is strongly correlated with a north-south temperature gradient and an east-west precipitation gradient. Increasing pressure for intensive urban, agricultural, and energy development coupled with climate change is threatening maintenance of goods and services in the region. Because of the widespread and complex juxtaposition of privately owned lands with intensive agricultural use intermixed with native vegetation on public lands, invasive plants pose a unique challenge to both private and public land managers. Climate change is likely to enhance pathways for invasive species (see Chap. 4) which increases the risk of some species becoming locally adapted under a changing climate and then dispersed into adjacent lands dominated by native vegetation. Within this context, this regional assessment includes ten invasive plant species (or collections of species), along with examples of invasive animal, pests, and pathogens. Each species, or group of species, was selected for this assessment if the species is not covered extensively in other sections relating to the Great Plains or the species is managerially and ecologically significant. Pests and pathogens are included despite coverage elsewhere in this report since they are germane to the evaluation of invasive species in the Great Plains, especially given the 2016 Technical Report by Bergdahl and Hill (2016).

As a result of these selection criteria, Russian olive (*Elaeagnus angustifolia*), non-native perennial grass assemblages (*Agropyron*, *Bromus*, and *Poa* spp.), buffelgrass (*Pennisetum ciliare*), absinth wormwood (*Artemisia absinthium*), Johnsongrass (*Sorghum halepense*), tumble mustard (*Sisymbrium altissimum*), whitetop (*Lepidium appelianum* Al-Shehbaz), and field (Japanese) brome (*Bromus arvensis*, synonym Japanese brome (*B. japonicus*)) were chosen as examples of problematic invasive species on the Great Plains. Animal species chosen for inclusion are wild horse and burros (*Equus* spp.) and feral pigs (*Sus scrofa*). Invasive pests of trees in the Great Plains included here are emerald ash borer (*Agrilus planipennis*), Balsam woolly adelgid (*Adelges piceae*), European gypsy moth (*Lymantria dispar dispar*), pine wilt (the nematode *Bursaphelenchus xylophilus* which spread via pine sawyer beetles (*Monochamus* spp.)), Dutch elm disease (the fungus *Ophiostoma* spp.), and thousand cankers disease (the fungus *Geosmithia morbida* spread by the walnut twig beetle (*Pityophthorus juglandis*)). Descriptions of pests and pathogens are excerpted from Bergdahl and Hill (2016).

Exotic Perennials

The strong correlation of a north-south temperature gradient and an east-west precipitation gradient with the composition and abundance of plants in the Great Plains means that patterns in the prevalence and distribution of exotic grass species will largely depend upon the photosynthetic pathway of the constituent species. Grasses possessing the C3 photosynthetic pathway (cool-season grasses) are more common and productive in the northern Great Plains, while grasses possessing the C4 photosynthetic pathway (warm-season grasses) are more abundant in the southern Great Plains and eastern tallgrass prairie (Epstein et al. 1997; Terri and Stowe 1976). Where native cool- and warm-season grasses co-occur, they vary in their spatial distribution at the local level with warm-season grasses occupying warmer, open sites, while cool-season grasses tend to occur in cooler, more shaded sites (Barnes et al. 1983; Teeri 1979).

Northern C3-dominated native plant communities of the Great Plains face the threat of invasion by introduced cool-season perennial grasses, particularly smooth brome (*Bromus inermis* Leyss. ssp. *inermis*), Kentucky bluegrass (*Poa pratensis* L.), and crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.) (Christian and Wilson 1999; DeKeyser et al. 2013; Larson et al. 2001). Along those lines, a number of studies, primarily from researchers in Canada, document cases where these three species have escaped cultivation, invaded natural ecosystems, and adversely impacted native species diversity (Christian and Wilson 1999; Fink and Wilson 2011; Hansen 2007; Hansen and Wilson 2006; Henderson and Naeth 2005; Nernberg and Dale 1997; Otfinowski et al. 2007; Vaness and Wilson 2007). In fact, smooth brome was ranked as the eighth most serious invasive alien plant in Canada because of its impact on the abundance and diversity of native prairie species (Catling and Mitrow 2005). Although sparse, research in the United States attributes reductions in native plant diversity (Dillemuth et al. 2009; Frank and McNaugton 1992) and reduced habitat use by native ungulates (Trammell and Butler 1995) to smooth brome. Similar reductions in native plant species diversity have been reported for Kentucky bluegrass (Stohlgren et al. 1998) and crested wheatgrass (Fansler and Mangold 2011; Hulet et al. 2010). Large-scale conversions of native prairie to these exotic perennial grasses can be especially detrimental to prairie specialist butterflies (Swengel and Swengel 2015) and grassland songbirds (Ellis-Felege et al. 2013).

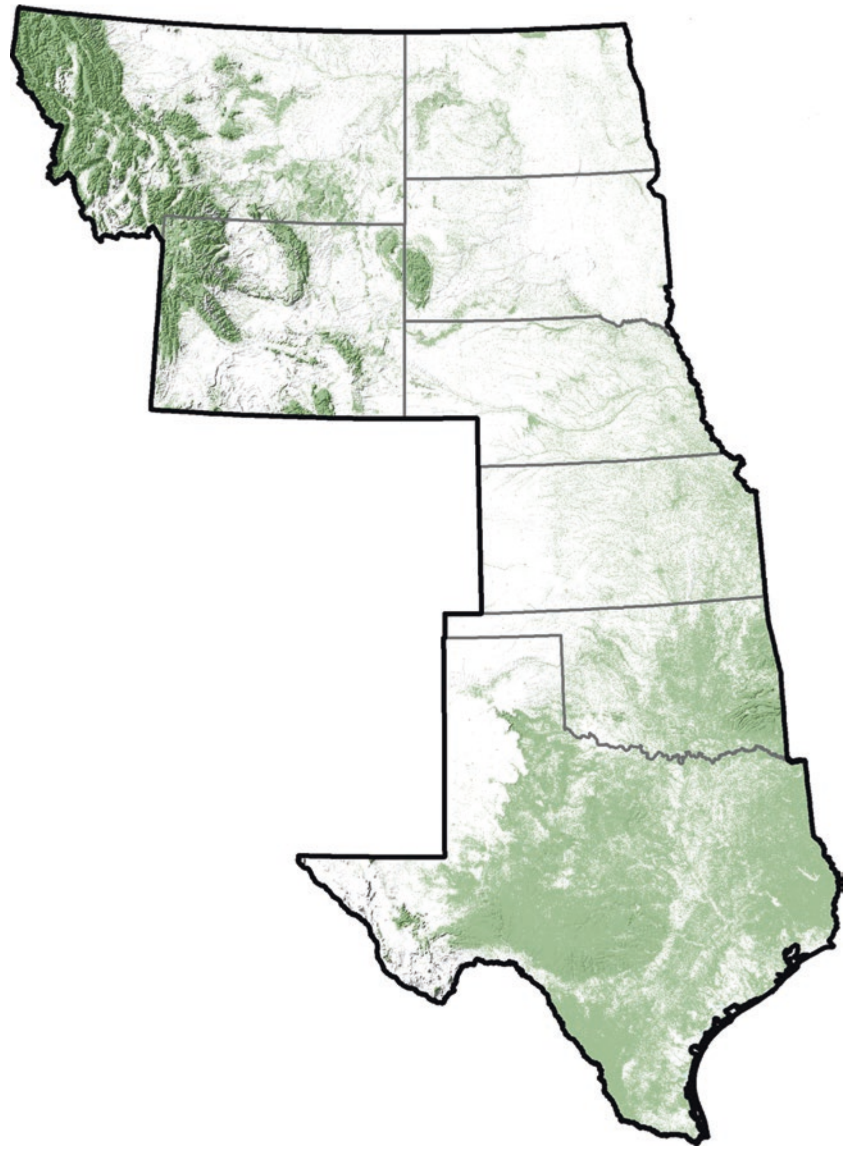

Fig. A5.1 The Great Plains region. (Figure courtesy of Daniel Ryerson and Andy Graves, USDA Forest Service Southwestern Region, Forest Health Protection)

The three cool-season species (smooth brome, crested wheatgrass, and Kentucky bluegrass) and the warm-season species (Johnsongrass) are generally not recognized as invasive, likely because of their perceived forage value. However, introduced exotic forage species are selected for traits that confer persistence under multiple stressors (drought, intensive defoliation, disease, etc.), and they likely often manifested in novel communities with a superior competitive advantage over native species, creating unique challenges in their management (Scasta et al. 2015). The collective evidence strongly indicates that these three non-native, perennial grasses have slowly and inexorably transformed relatively large tracts of non-forested ecosystems, and this transformation has largely gone unnoticed in the United States. The compositional balance of cool- and warm-season native and introduced grasses along the moisture and temperature gradient will undoubtedly be altered by climate change, likely in unknown ways.

Smooth Brome Smooth brome is native to Eurasia (Otfinowski et al. 2007) where it grows along roadsides, riverbanks, and borders of cultivated fields and in pastures (Kennedy 1899). Kennedy (1899) estimated that smooth brome was first introduced into the United States for pasture improvement in 1884 through the California Experiment Station. Initial seeding experiments showed that it was an aggressive rhizomatous species capable of rapidly displacing other plants (Kennedy 1899), a pattern confirmed by more recent experiments (Blankespoor and May 1996; Fink and Wilson 2011), including in areas where it is native (Liu et al. 2008). Once established, smooth brome is capable of higher production than adjacent native grasslands while reducing diversity through reductions in evenness (Fink and Wilson 2011; Otfinowski et al. 2007). Smooth brome readily outcompeted its native neighbors in northern mixed-grass prairie even under drought conditions (Nernberg and Dale 1997; Ulrich and Perkins 2014). Few efforts to control smooth brome have been completely effective (Bahm et al. 2011; Blankenspoor and Larson 1994; Bolwahn-Salesman and Thomsen 2011; Grilz and Romo 1995; Stacy et al. 2005; Willson and Stubbendieck 1996, 1997).

Crested Wheatgrass Crested wheatgrass is a cool-season bunchgrass native to a wide variety of grasslands in Central Europe, the Middle East, Central Asia, Siberia, China, and Mongolia where it is widely recognized as a valuable forage species (Rogler and Lorenz 1983). Crested wheatgrass is a complex of Eurasian species that were first introduced into North America (North Dakota) in 1898 (Dillman 1946). Crested wheatgrass has been widely planted throughout the northern Great Plains since the 1930s (Christian and Wilson 1999). It establishes quickly and is a successful competitor in many grassland ecosystems where it often outproduces and displaces native prairie species (Heidinga and Wilson 2002; Henderson and Naeth 2005). Grasslands dominated by crested wheatgrass contain few native species, especially forbs and grasses with growth forms similar to crested wheatgrass (Christian and Wilson 1999; Henderson and Naeth 2005). Christian and Wilson (1999) also reported that soils dominated by crested wheatgrass had less available nitrogen, total nitrogen, and less total carbon than soils under native prairie in Canada, potentially creating long-term ecosystem impacts. Crested wheatgrass is difficult to control primarily because of a large and persistent seed bank (Fansler and Mangold 2011; Hulet et al. 2010; Wilson and Pärtel 2003).

Kentucky Bluegrass Kentucky bluegrass is undoubtedly one of the most recognized and widespread *Poa* species in Europe, Asia, and North America (DeKeyser et al. 2015). Kentucky bluegrass is strongly rhizomatous, very productive, and highly palatable, making it a popular pasture grass in many ecosystems. Kentucky bluegrass was brought into North America by European traders, explorers, and missionaries in the mid- to late 1600s, largely because of its popularity as a forage grass (Schery 1965). Kentucky bluegrass has greatly expanded its range in North America over the last 100 years and is now a common species in many plant communities where is often considered an invasive species (Dekeyser et al. 2015; Toledo et al. 2014). Lower native species richness and declines in abundance of native warm-season grasses have been attributed to invasion by Kentucky bluegrass (Miles and Knops 2009). In a study on the National Wildlife Refuges in the Dakotas, Kentucky bluegrass accounted for 27–36% of the vegetation and was considered a contributing factor in the decline of the North American prairie (Grant et al. 2009). The current lack of regeneration of native green ash (*Fraxinus pennsylvanica*) woodlands in northern Great Plans grasslands has been attributed to the dense sod formed by the invasion of Kentucky bluegrass that greatly restricts establishment of green ash seedlings (Lesica 2009). The efficacy of using herbicides and fire to control Kentucky bluegrass and restore native species is generally highly variable (Bahm et al. 2011). When Kentucky bluegrass is successfully suppressed, the potential exists for the bare ground created by the reduction of Kentucky bluegrass to produce a secondary invasion by other exotic species (Adkins and Barnes 2013). At the same time, Kentucky bluegrass has also been classified as a major secondary invader following the successful suppression of leafy spurge (*Euphorbia esula*) using classical biological control (Butler and Wacker 2010).

Johnsongrass Johnsongrass (*Sorghum halepense*) is an exotic, perennial warm-season grass hybrid between *S. bicolor* (sorghum or millet) and *S. propinquum* (sorghum). Johnsongrass is a serious problem worldwide, especially in humid warm-temperate and subtropical regions (Follak and Essl 2012). Johnsongrass has long been recognized as aggressive invader of crop systems where heavy infestations can substantially reduce yields (Williams and Hayes 1984). No species of *Sorghum* are native to North America, and Johnsongrass is rapidly becoming a serious invader, adversely impacting the diversity of native prairies in the United States. The plant has several characteristics that are common to some of the most aggressive plant invaders, including a tall growth form and prolific seed production coupled with robust clonal growth through rhizomes. It also produces a defensive cyanogenic glycoside (dhurrin) (Abdul-Wahab and Rice 1967) and an allelopathic molecule (sorgoleone) that is exuded from root hairs (Czarnota et al. 2001). Collectively, these traits play a significant role in the ability of Johnsongrass to displace native species (Abdul-Wahab and Rice 1967; Follak and Essl 2012; Rout et al. 2013). Research is needed on the ecological impact of Johnsongrass in natural ecosystems and possible control strategies.

Buffelgrass Buffelgrass is native to India, Africa, and parts of Asia (Hauser 2008). It was introduced into Texas and Arizona in the 1930s and 1940s for soil stabilization and forage (Hauser 2008). Further establishment has occurred through seeds dispersed from Mexico. In Sonora, it is estimated that over 1000,000 ac of native desert and thornscrub vegetation have been converted to buffelgrass pasture (Burquez et al. 1998, 2002; Franklin et al. 2006). Within the Great Plains, buffelgrass occurs primarily in Texas, with outlying populations in Oklahoma (USDA, NRCS 2008). Within this limited distribution, buffelgrass occurs most often in desert thornscrub, mesquite-dominated shrublands, and cultivated buffelgrass pastures (Hamilton 1980; Hamilton and Scifres 1983; Mayeux and Hamilton 1983). Although the distribution in the Great Plains appears to be constrained by temperature, the ecological effects are significant. Buffelgrass alters plant communities and fire regimes and has been credited with creating "one of the most impressive ecosystem conversions happening in North America" (Nijhuis 2007) and is described as "one of the world's most notorious invaders" (Williams and Baruch 2000). The dramatic effects of buffelgrass on these communities are enhanced by a fire feedback cycle, since buffelgrass is a fire-adapted species (Burquez et al. 2002; Tellman 1997; Van Devender et al. 1997), enabling it to persist and spread following a fire. This is significant because the arid and warm sites that buffelgrass prefers often have extremely long fire intervals and support numerous succulent species that are not fire-adapted. Further, buffelgrass produces much greater fine fuel loads (often exceeding a threefold to fourfold increase in fine fuels (Esque et al. 2007) at the drier end of its invaded range) than native plants in these sites, thereby causing high mortality in native flora and fauna (Esque et al. 2007). Most of the effects of these fires driven by buffelgrass are documented from Sonoran Desert habitats located to the southwest of the Great Plains region. However, in the Chihuahuan Desert of western Texas, the endangered Chisos Mountains hedgehog cactus (*Echinocereus chisosensis*) is very vulnerable to mortality from increased fire frequency and effects from buffelgrass invasion (Hauser 2008). In addition, buffelgrass is competitive and invasive in southern mixed, shortgrass, and semi-desert grasslands of Texas and Oklahoma (Grace and Zouhar 2008; Rice et al. 2008). Since buffelgrass is a fire-adapted species, it is notably difficult to control using managed fire, but success may be enhanced with herbicide treatment or hand-pulling. Another factor increasing the difficulty of chemical control is that buffelgrass has been found to exhibit resistance to three of seven herbicides (Bovey et al. 1986), and older stands tend to tolerate herbicides better than small seedlings (Bovey et al. 1984). Recognizing the potential for very significant changes to southern ecosystems, the need for more research on control techniques has been noted (Hauser 2008).

Absinth Wormwood Absinth wormwood (*Artemisia absinthium*) is a coarse, erect herbaceous or semi-woody, clump-forming perennial that is native to parts of Europe and Asia (Maw et al. 1985; Selleck and Coupland 1961). Absinth wormwood was cultivated on a large scale in Europe for its reported hallucinogenic effects when consumed by humans (Maw et al. 1985) and its use as a folk remedy (see Makrini and Hassam 2016). Maw et al. (1985) further report that it was intentionally introduced into North America as a "medicinal and flavoring plant" in the early 1800s, but was banned in the United States in 1912. An online search in Web of Science using *Artemisia absinthium* in the title produced 170 articles with the vast majority of the papers reporting on the chemical compounds distilled or leached from the plant. Online information from Washington highlights the poisonous nature of the plant and cautions that no part of the plant should be consumed by humans or livestock (King County, WA 2017). Currently, absinth wormwood is naturalized in Canada and is listed as a noxious weed in only three States in the United States (Colorado, North Dakota, and Washington). The plant usually occurs in low densities (Selleck and Coupland 1961), but, because of its poisonous nature and its potential for expansion under climate change and a lack of research on its management, careful monitoring is needed.

Whitetop Three species of whitetop including globe-podded whitetop (*Cardaria pubescens*), lenspod whitetop (*C. chalepensis*), and heart-podded whitetop (*C. draba*) inhabit the Great Plains region. These species probably arrived in North America in the early 1900s (Zouhar 2004). Whitetop has an affinity for waste areas, roadsides, and degraded grasslands but is also attracted to moist environments such as irrigation ditches (Zouhar 2004).

Along the Bighorn River in Wyoming, globe-podded whitetop is often associated with Russian knapweed (*Rhaponticum repens*), Canada thistle (*Cirsium arvense*), and other non-native species co-occurring with saltcedar (*Tamarisk chinensis*), but it is generally rare in native shrublands (Zouhar 2004). Whitetop produces poor forage, crowds out desirable plants, and reduces animal diversity (USDA Forest Service 2014). Whitetop foliage contains glucosinolates, which are toxic to cattle and can impede germination and growth of other species (USDA Forest Service 2014).

Whitetop's extensive root system creates significant control difficulties, and it is a challenge to eradicate large populations once they are established (USDA Forest Service 2014). Treatment with herbicide can be effective, if it is performed during the correct life stage, but fire is not recommended as a solution for managing whitetop infestations. The extensive root system makes *Cardaria* spp. likely to survive even severe fire, but success has been noted using burners at close intervals (Rosenfels and Headley 1944). There have been significant economic and ecological effects of whitetop through reduced crop yields, cost of control, reductions in forage, and reduced quality of some agricultural products (Parsons and Cuthbertson 1992; Scurfield 1962). There is a lot of information available for whitetop control on croplands and heavily impacted lands (Chipping and Bossard 2000; McInnis et al. 2003; Sheley and Stivers 1999) but not for wildlands. Whitetop is considered a "moderate to serious" threat to native plant species in riparian and wetland settings and a “minor” threat in native grasslands (Zouhar 2004).

Russian Olive Russian olive (*Elaeagnus angustifolia*) is a tree or multi-stemmed shrub (5–12 m in height) that is native to Central and Western Asia. It is used worldwide as a nutritional agent or as a natural remedy for a range of illnesses. It was intentionally introduced in North America as a horticultural plant in the early 1900s, to be used for hedge rows and as a shade tree; by the 1940s, it was widely planted in windbreaks throughout the Great Plains (Katz and Shafroth 2003). Russian olive is currently found throughout the United States, where it has become the fourth most dominant woody plant in riparian areas through the Western United States (Friedman et al. 2005). Its rapid spread is potentially attributed to birds consuming the fruits (Edwards et al. 2014). The rapid dominance of Russian olive in riparian settings has generated considerable concern about its impact on natural communities and ecosystems (see review by Collette and Pither 2015). In their review, Collette and Pither (2015) described lower bird species richness and diversity in sites dominated by Russian olive, compared to noninfested sites. They presented evidence of enhanced nitrogen input into streams, likely related to the nitrogen-fixing ability of Russian olive, which could lead to eutrophication. At the same time, there is evidence that Russian olive provides a nesting habitat for the endangered southwestern willow flycatcher (*Empidonax traillii extimus*) and the threatened yellow-billed cuckoo (*Coccyzus americanus*), which creates potential conflicts in the management of the species. Because of its popularity as an ornamental, its ability to invade and dominate riparian areas, and the potential conflicts in its management (adversely affecting communities and ecosystems while providing habitat for endangered and threatened species), additional research is needed to address the ecological implications of the current and future range of Russian olive under climate change (Collette and Pither 2015; Katz and Shafroth 2003).

Exotic Annuals

Field Brome Although cheatgrass or downy brome (*Bromus tectorum*) is, without question, the most notorious and widely recognized annual exotic grass in North America, field brome has life history characteristics similar to cheatgrass (Baskin and Baskin 1981) and shows great potential for being just as invasive (Gasch et al. 2013; Haferkamp et al. 1997; Ogle et al. 2003). Field brome is a native Eurasian winter annual that has long been recognized as major weed of cropland systems worldwide (Sarani et al. 2016). While field brome is not as widely recognized as cheatgrass, it has greatly increased in abundance and distribution in Great Plains prairies (Haferkamp et al. 1997; Harmoney 2007; Ogle et al. 2003). This increase is often attributed to the removal of the interactive effects of fire and grazing, causing increases in litter, which favors the germination and establishment of field brome (Harmoney 2007; Whisenant 1990). It is difficult to assess the specific impacts of field brome on community and ecosystem properties because researchers sometimes lump cheatgrass and field brome together (Gasch et al. 2013; Ogle et al. 2003). Where the two species occur together, Gasch et al. (2013) have reported that annual brome-dominated sites had lower plant community diversity and carbon/nitrogen ratios, higher soil water infiltration rages, and altered soil microbial groups. Ogle et al. (2003) found that removal of both annual bromes resulted in more aboveground and belowground biomass at the end of the growing season. Studies specific to field brome found that, while removal of field brome increased production of associated perennial grasses, total production was reduced, at least for the duration of the study (Haferkam et al. 1997). Efforts to control field brome using fire and grazing suggest that while these treatments, used singly and in combination, may reduce field brome abundance, long-term control strategies are still elusive (Harmoney 2007; Whisenant 1990). Based on field studies conducted on cheatgrass (Blumenthal et al. 2016), long-term control of field brome under climate change may be difficult.

Tumble Mustard Tumble mustard probably came to North America in contaminated seed sources (Kostivkovsky and Young 2000) and is found throughout the continent. Westward expansion of the species was probably enhanced by inadvertent attachment to rail cars (Mitich 1983; Weber and Wittmann 1996). Though widespread, tumble mustard tends to occur most often on degraded sites with very low cover of native perennials and often co-occurs with other invasive annual species (Evans and Young 1970). In addition, tumble mustard is more common in rangeland and agricultural environments than in forested environments above the ponderosa pine (*Pinus ponderosa*) zone. Tumble mustard is a prolific seeder, and it is said that it can produce more than one million seeds per season (Clark and Fletcher 1923; Mitich 1983). Like other invasive annual species, tumble mustard germinates quickly after fire creating a fire-feedback cycle, thus enhancing its ability to regenerate. Tumble mustard is considered the second most invasive alien plant species in the Great Basin (Young and Evans 1972; Young et al. 1970), especially given its more effective seed dispersal mechanisms and earlier germination compared with native herbs (Allen and Knight 1984). With an affinity for degraded lands, tumble mustard is uncommon where there are high proportions of native perennial species and is an indicator of deteriorating land capability (Humphrey 1950). In addition, tumble mustard (and other annuals) can cause significant economic losses through reduction of forage for native and domestic ungulates (Pechanec and Stewart 1949). Other mustard species (*Brassicaceae* spp.) are frequently referenced as having unique tolerance to numerous herbicides, but, like many species, tumble mustard is most susceptible to herbicide application in the rosette stage.

Terrestrial Vertebrates

Historically, the grasslands of the Great Plains region have supported vast numbers of grazing ungulates, most notably the American bison (*Bison bison*). Perhaps it is not surprising then that the most significant animal invasive species affecting ecosystems of this region are ungulates, especially feral swine and feral horses and burros.

Feral swine (hereafter, pigs) in the region are largely restricted at present to the southern plains, particularly Texas and Oklahoma, where they are widespread, although their distribution is expanding continent-wide (Bevins et al. 2014; McClure et al. 2015). Pigs have been released or escaped continually since the arrival of the earliest European explorers in the sixteenth century (Mayer and Brisbin 1991). These animals are not simply grazers but are opportunistic generalists, feeding on plant material of all kinds (roots, stems, foliage, and seeds), fungi, invertebrates, reptiles, amphibians, small mammals, bird eggs, carrion, and refuse. Their rooting and wallowing have an impact on soil stability and chemistry, nutrient cycling, and microbe communities, as well as water quality. Plant community impacts include reduced species diversity, forb cover, leaf litter, and tree regeneration, as well as an increased prevalence of invasive plants (Timmons et al. 2012). In addition to direct impacts on native wildlife through depredation and habitat damage, pigs also compete with native wildlife for important foods (e.g., hard mast).

Feral horses in Western North America are descended from domestic horses of Eurasian and African origin, which were likewise introduced as early as the sixteenth century by European explorers. The number of horses apparently peaked around the mid-nineteenth century, declining thereafter; they were persecuted by grazing interests as competitors to cattle and sheep. Most wild horses and burros now occur on public lands administered by the Bureau of Land Management (BLM) or the USDA Forest Service, and they are protected and managed under provisions of the Wild and Free-Roaming Horses and Burros Act.

This protected status, however, has led to increased populations of wild horses and burros, at levels significantly above management objectives. To promote healthy conditions on the range, the BLM determines the Appropriate Management Level (AML), which is the number of wild horses and burros that can prosper in balance with other public land resources and uses. As of 2016, wild horses and burros exceed AML (which is 26,715) with an estimated population of 67,027, a 15% increase over the 2015 estimate (https://www.blm.gov/programs/wild-horse-and-burro/herd-management). This is consistent with the BLM's finding that wild horse and burro herds double in size about every 4 years. The disturbing trend in the growth of the herd size for these invasive equids has significant implications, given the reductions in rangeland health usually associated with their presence (Beever et al. 2008). Like most invasive species, the management or administration of feral horses and burros carries a significant economic burden; direct costs to the BLM alone topped $75 million in 2015. Depending on a variety of factors that include the abundance of horses in an area, overgrazing and trampling by equids can affect ecosystems through soil erosion and compaction, altered nutrient distribution, and altered plant species composition and abundance (Beever et al. 2008). These impacts, in turn, can affect the diversity and abundance of reptiles and mammals (Beever and Brussard 2004).

Invasive Pests of Trees in the Great Plains

Forests and "trees outside forests" (TOF) represent a relatively small portion of the land cover in the Great Plains. Nonetheless, they have long provided many ecosystem goods and services important to the well-being of humans living in this region (Droze 1977; McKay 1994). Agroforestry, a significant subset of TOFs throughout the Plains, has been used since the 1930s Dust Bowl days to protect soils, crops, livestock, and air and water quality. It is also used today to protect farmsteads, buildings, roads, and communities and to create habitats critical for wildlife, ranging from game species to pollinators (Schoeneberger et al. 2016). These "working trees" in the Great Plains are highly vulnerable to a number of factors (Joyce et al. 2018), including forest insect and disease pests (Bergdahl and Hill 2016; RMR FHP 2010). Exposure to environmental stresses, including the extreme shifts in temperature, moisture, and wind that are pervasive on the Plains, can exacerbate tree susceptibility to these pests (Ball 2016). Further, these severe and erratic weather-related events in the Plains are expected to increase in frequency and intensity in the coming years (Kunkel et al. 2013), further increasing tree vulnerability (Joyce et al. 2017).

While many of these forest pests are native to the region (RMR FHP 2010), there is a growing number of non-native pests threatening many of the key tree species occurring in this region. A few of the most potentially devastating invasive pests of Great Plains tree resources are presented in Table A5.1.

The potential ecological and economic losses related to non-native invasive tree pests have been estimated to be substantial (Lovett et al. 2016; Moser et al. 2009). Lovett et al. (2016) have noted that "non-native forest pests are the only disturbance agent that has effectively eliminated entire

Table A5.1 An overview of key invasive species in the Great Plains

Invasive pest of Great Plains trees	Overview	Current (2016) occurrence in the Great Plains[1/]								
		KS	MT	NE	ND	OK	SD	TX	WY	General comments
Emerald ash borer (EAB)	Highly invasive, exotic insect (*Agrilus planipennis* Fairmaire) introduced from China that attacks and kills all species of North American ash trees	**D**[2/] 2012	**N**	**D** 2016	**N**	**D** 2016	**N**	**D** 2016	**N**	Active monitoring/detection efforts occurring in most of **N** States.
Balsam woolly adelgid	Non-native, invasive insect (*Adelges piceae* Ratzeburg) impacting subalpine and grand fir	**ni**	**D** 2007	**ni**	**N**	**ni**	**ni**	**ni**	**ni**	Important in western Plains States, especially MT.
European gypsy moth (EGM)	*Lymantria dispar dispar* L.	**D***	**D***	**N**	**N**	**N**	**N**	**N**	**N**	Not yet established in any of the Plains States, all of which are suitable habitat for year-long survival of EGM
Pine wilt	Caused by the plant parasitic pinewood nematode (*Bursaphelenchus xylophilus*) via pine sawyer beetles in the genus *Monochamus*	**D** 1979	**N**	**D**	**N**	**D**	**D**	**N**	**N**	Mainly a threat to Scots, Austrian, and other non-native pines used extensively throughout the Plains
Dutch elm disease	Non-native, invasive wilt of elm species caused by the species of the fungus *Ophiostoma*, the most aggressive being *O. novo-ulmi*	**D**	**D**	**D**	**D** 1969	**D**	**D**	**D**	**D**	
Thousand cankers disease (TCD)	Invasive canker disease of black walnut caused by the walnut twig beetle (*Pityophthorus juglandis*) and its fungal associate *Geosmithia morbida*	**N**	**N**	**N**	**N**	**N**	**N**	**N**	**N**	Detected in every State (ID, UT, CO, and NM) bordering the western edges of the Great Plains States

[a]Based on information in the forest health reports up to 2016 from each of the Great Plains States (Montana [MT], Nebraska [NE], North Dakota [ND], Oklahoma [OK], South Dakota [SD], Texas [TX], and Wyoming [WY]) (RMR FHP 2010; Bergdahl and Hill 2016)
[b]**D = detected** (reported date), **N = not yet detected**, **ni = information not found**

tree species or genera from United State forests within decades." A good example of the level and cost of such an invasive tree pest is Dutch elm disease. This disease was responsible for the death and removal of most of the native elms (*Ulmus* spp.) throughout the United States over the past century and is still a disease of concern in the Great Plains (Dunnell and Bergdahl 2016). Thousand cankers disease, which causes widespread mortality of black walnut (*Juglans nigra* L.) and which was only noted in Colorado in 2001, is a major threat to the highly valued black walnut tree throughout the Great Plains (Tisserat and Cranshaw 2016).

Invasive forest pests are a particular concern in the Great Plains because the tree resources suitable to the environment in this region are limited; this greatly reduces tree diversity and, therefore, resilience to such attacks (Bergdahl and Hill 2016). Several of the main tree species long promoted and used in agricultural and community plantings have been removed from recommended planting lists in recent years either due to the high levels of mortality already occurring in the Plains (e.g., Scots (*Pinus sylvestris* L.) and Austrian (*Pinus nigra* L.) pines from pine wilt) or to the high levels of mortality being predicted to occur in the near future (e.g., ash (*Fraxinus* spp.) from the emerald ash borer and black walnut with thousand cankers disease).

Ash is one of the most prevalent species throughout the Plains. It is a significant component of riparian corridors, windbreaks, and community plantings (Rasmussen 2009). In 2008–2009, over four million ash trees were identified in urban settings with an additional 80 million identified in the rural areas just in the four northern Plains States (Schoeneberger et al. 2016). The emerald ash borer (EAB), a highly destructive pest of all North American ash trees, has already been detected in some easternmost areas of Kansas, Nebraska, Oklahoma, and Texas (USDA APHIS 2016a) and is expected to reach most of the Plains States within the next few years. The cost of treatment, removal, and replacement in response to EAB in the Plains could exceed $1 billion per State, along with the additional economic impacts from the loss of ecosystem services important to soil, water, and wildlife resources (Rasmussen 2009).

To better prepare and manage for the EAB and other invasive tree pests in the northern Plains States, State forestry agencies in Kansas, Nebraska, North Dakota, and South Dakota established the Great Plains Tree and Forest Invasives Initiative (GPI) in 2007 (Rasmussen 2009). This effort encompassed a comprehensive assessment of urban and agricultural tree resources across the four States, outreach and monitoring and detection programs, identification of marketing and utilization opportunities, and development of

State and regional planning strategies for EAB readiness and other invasive pests. Results from EAB parasitoid releases in the Northcentral region of the United States (Duan et al. 2017) indicate biocontrol may be a promising option for reducing EAB populations in the Great Plains. Further work is required to determine its success under the more extreme weather conditions and more fragmented ash occurrences experienced in this region.

Many other invasive insect and microbial pests of trees have the potential for significantly impacting Great Plains tree resources in the future. State forest plant health reports in the Plains also include the Asian longhorned beetle (*Anoplophora glabripennis* Motschulsky), which colonizes a wide range of hardwood hosts, and sirex woodwasp (*Sirex noctilio* F.), which has the potential to cause significant mortality of pines. Both are currently established within the United States. In addition, the Asian gypsy moth (*Lymantria dispar asiatica* Vinuskovkij), while not yet established in the United States, represents a major threat to all US tree resources because it can feed on over 100 botanical families (USDA APHIS 2016b). The realities of tree pest invasion in the Plains require more efforts like the GPI to be in place to manage the sustainability of Great Plains tree resources and the ecosystem services important to that region (Schoeneberger et al. 2016).

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Midwest Region

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Introduction

The Midwest region includes Minnesota, Iowa, Missouri, Wisconsin, Illinois, Michigan, Indiana, and Ohio (Fig. A6.1). Five States border the Great Lakes, in addition to numerous inland lakes and the Missouri and Mississippi River systems. Forty percent of all the water surface area in the continental United States is located within the Midwest. Abundance of water within the region influences trade (shipping ports, river traffic), recreation, agriculture, and ecology. All of these listed factors influence the distribution and impact of invasive species in both terrestrial and aquatic environments.

The diverse and ecologically complex forest ecosystems of the Midwest are dominated by northern and central hardwood forests, bordered by northern boreal forest to the north and prairie ecosystems to the south and west. Forests of the Midwest are productive and valuable, with forest-related businesses ranking in the top 10 for economic importance in every State. The oak-hickory (*Quercus*-*Carya*) forest type occupies the greatest proportion of the forested area (40%), followed by maple-beech-birch (*Acer*-*Fagus*-*Betula*) (15%) and aspen-birch (*Populus*-*Betula*) (14%). Conifer types, including 9% spruce-fir (*Picea*-*Abies*) and 6% pine (*Pinus*), are also important, particularly in the Lake States. Bottomland hardwoods rise to importance in this region, with 11% of the area comprising the elm-ash-cottonwood (*Ulmus*-*Fraxinus*-*Populus deltoides*) forest type.

The Midwest region also has many large cities and a very high presence of agriculture and industry. Human actions and their interactions with their environment exacerbate the movement and impacts of invasive species. Non-native

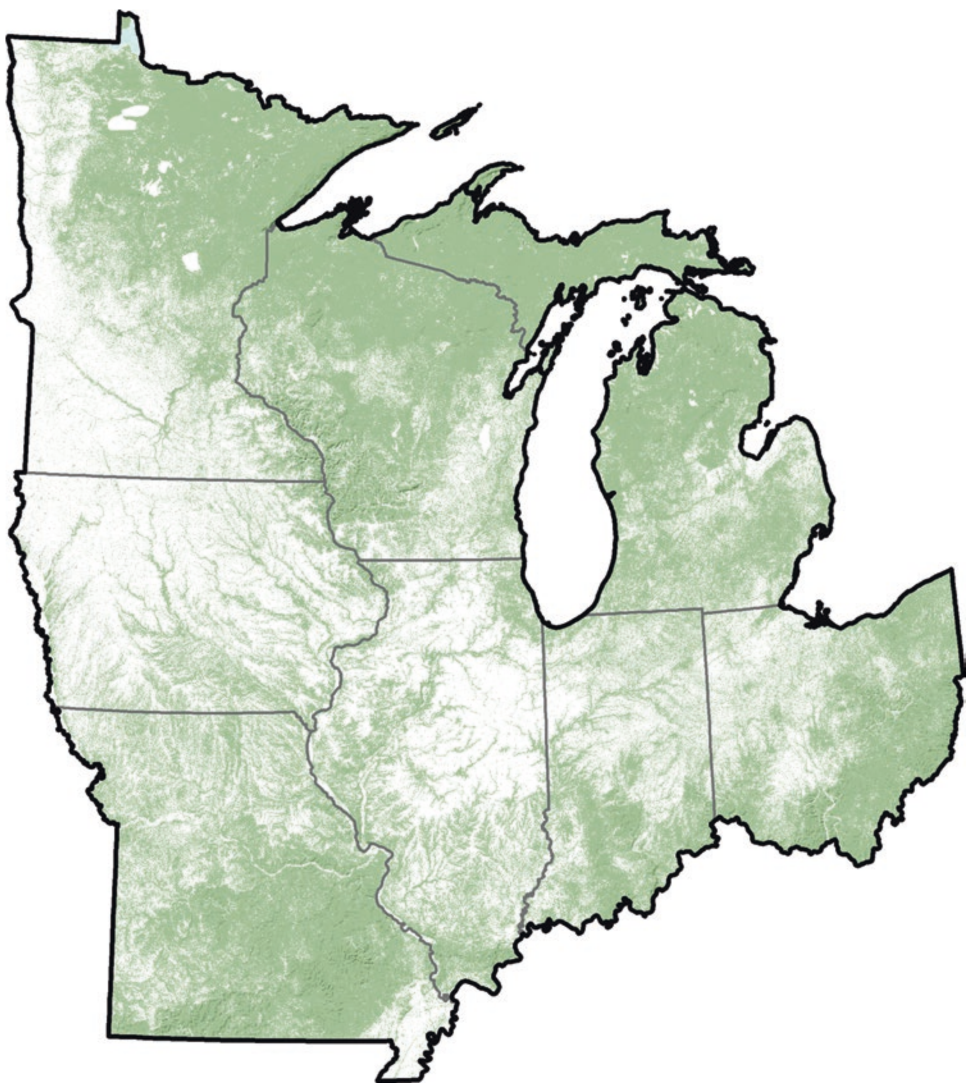

Fig. A6.1 The Midwest region. (Figure courtesy of Daniel Ryerson and Andy Graves, USDA Forest Service Southwestern Region, Forest Health Protection) invasive species have affected forests and aquatic systems since the time of European settlement, with landscape-level impacts extending into even the most remote areas of the region. We outline selected non-native species below, with focus on current distribution, significant impacts, and current management efforts.

Insect Pests of Trees

Many non-native insect pests occur in the region, and some have caused significant impacts on the region's forests. The focus in this summary is four species that have been of high interest or concern in recent years: gypsy moth (*Lymantria dispar*), hemlock woolly adelgid (HWA) (*Adelges tsugae*), emerald ash borer (EAB) (*Agrilus planipennis*), and Asian longhorned beetle (ALB) (*Anoplophora glabripennis*). Other non-native insects have had impacts that linger in our forests, including larch sawfly (*Pristiphora erichsonii*), larch casebearer (*Coleophora laricella*), Japanese beetle (*Popillia japonica*), birch leafminer (*Profenusa thomsoni*), European pine sawfly (*Neodiprion sertifer*), introduced pine sawfly (*Diprion similis*), and elongate hemlock scale (*Fiorinia externa*).

Gypsy moth caterpillars feed on hundreds of species of trees and shrubs, often causing severe defoliation and contributing to tree decline and mortality. The insect has been the focus of government-sponsored programs for more than 100 years. Currently, gypsy moth is established across Michigan and much of Wisconsin and in portions of Indiana, Illinois, Minnesota, and Ohio. A variety of biological control agents (i.e., parasitoids, predators, and entomopathogens) help regulate gypsy moth populations. In particular, the highly specific insect pathogen *Entomophaga maimaiga* has become widely established in the Midwest and may be contributing to the natural suppression of gypsy moth populations. Management of the insect at the Federal level consists of three distinct strategies (suppression, eradication, and slowing the spread), depending upon where the insect is found (USDA 2012a). Suppression is implemented to reduce adverse effects to trees caused by outbreaks of the insect. Gypsy moth populations in the region remained low between 2007 and 2016, with only Ohio and Wisconsin conducting modest State-led aerial suppression projects on about 44,000 ac (USDA 2017). Eradication is implemented to eliminate colonies of gypsy moth that are detected outside of the currently infested (regulated) area. Between 2007 and 2016, more than 17,000 ac in Indiana, Minnesota, Ohio, and Wisconsin were treated using eradication protocols (USDA 2017). The objective of the Slow the Spread (STS) program, which involves the collaboration of multiple jurisdictions and cooperators, is to slow the natural and short-range human-aided spread of the insect along the leading edge of the area generally infested by the insect. STS is a unique landscape-scale program across a 50-million-ac project area within 11 States from Minnesota to North Carolina. The design and implementation of STS is science-based with the overall strategy founded on research that indicated this was an optimal approach for minimizing spread. Since the start of the program, about 6 million ac have been treated in Iowa, Illinois, Indiana, Minnesota, Ohio, and Wisconsin, mostly employing the application of pheromone flakes to disrupt mating by gypsy moth adults (USDA 2017). Spread rates along the leading edge remained stable in the Midwest region in 2016, while rates across the entire STS project area were low (3.8 km/year).

The hemlock woolly adelgid (HWA) threatens the survival and sustainability of eastern hemlock (*Tsuga canadensis*). Hemlocks are considered a foundation species which define forest structure and control ecosystem dynamics (Havill et al. 2014). The insect, which causes tree decline and mortality, is now present in many eastern States and has recently been confirmed in the Midwest in 13 eastern counties of Ohio and 5 counties in Michigan. The National HWA Initiative, a landscape-scale effort, was established by the USDA Forest Service in 2003 to develop and implement tools to manage HWA and to reduce the adverse effects across the range of eastern hemlock. Current management of HWA in Ohio consists of enhanced survey and monitoring of HWA spread into uninfested areas, as well as the application of systemic insecticides to protect high-value trees in the near term, complemented with the release of biological control agents (predatory beetles) to manage HWA populations in the long term. The HWA predatory beetles *Laricobius nigrinus* and *L. osakensis* have been, and continue to be, released in the infested counties in Ohio. In summer 2015, infestations of HWA were detected in Ottawa and Muskegon Counties in western Michigan. Since then, HWA has also been detected in Allegan, Oceana, and Mason Counties. The State has quarantined the four infested counties and has initiated surveys to delimit the infested area and look for new infestations. Treatments relying heavily on systemic insecticides are being implemented in an attempt to contain local HWA populations. However, it is unlikely that HWA can be eliminated from Lower Michigan. This puts at greater risk more extensive hemlock stands in the Upper Peninsula of Michigan and northern Wisconsin.

Adults of the emerald ash borer (EAB) feed on leaves and larvae tunnel in the phloem. EAB is a significant tree killer that has decimated ash trees across much of the Midwest. Green, white, and black ash (*Fraxinus pennsylvanica*, *F. americana*, and *F. nigra*, respectively) are common and locally abundant. Pumpkin and blue ash (*F. profunda* and *F. quadrangulata*, respectively) are less common but locally important species. All are susceptible to EAB (Klooster et al. 2014). Tree losses from EAB are estimated to be in the hundreds of millions in the Midwest region. A few ash trees have survived in EAB-infested areas which suggests that there may be some resistance or tolerance in the population (e.g., Anulewicz et al. 2007; Knight et al. 2012; Rebek et al. 2008). First discovered in the Detroit metropolitan area in 2002, subsequent detections have occurred in Ohio (2003), Indiana (2004), Illinois (2006), Wisconsin (2008), Minnesota (2009), Iowa (2010), and Missouri (2008). Today, Federal and State quarantines exist in all or parts of every State in the Midwest region. Ash also is a common street and landscape tree in many Midwestern cities. The eventual cost of treatment, removal, and replacement of infested ash trees in communities is estimated to be as high as $10.7 billion over a 10-year period (Kovacs et al. 2010). Commerce and movement of infested nursery stock and wood products such as firewood are major contributors to the spread of the insect. The current management approach focuses on (1) containment of the insect; (2) regulating the movement of potentially infested materials to areas not infested with EAB; (3) survey and monitoring; (4) public outreach; (5) insecticide treatment to protect high value trees; and (6) management of the insect through the release and establishment of (currently) four biological control agents (parasitoids).

Native to China and Korea, the Asian longhorned beetle (ALB) is a wood borer that can penetrate deep into the wood. It poses a serious threat to the Midwest region's forests. At least 13 tree genera, and more than 100 different tree species, are known to be suitable hosts for ALB (USDA 2012b), although the insect mostly prefers maples (*Acer* spp.), poplars (*Populus* spp.), willows (*Salix* spp.), and elms (*Ulmus* spp.). The Midwest region's forests and urban landscapes include a large number of maples, poplars, and willow. The second confirmed detection of ALB in the United States occurred in the Midwest region, in the Chicago metropolitan area in 1998. An aggressive eradication effort was successful, eliminating the insect from that location by 2008. The next ALB detection in the Midwest region occurred in 2011 in Clermont County, OH, which is more rural compared to the Chicago metropolitan area. Current prevention and eradication protocols include (1) detection and monitoring for ALB via intensive surveys; (2) preventing movement of infested material with established quarantines; (3) public outreach and education; (4) removal and destruction of infested and high-risk host trees; and (5) the use of systemic insecticides. The goal is to eradicate the pest from the woodlots and natural forest stands in this Ohio infestation. ALB may spread faster in natural and managed forests than has been observed in urban and suburban environments (Dodds and Orwig 2011; Dodds et al. 2014). Current survey, monitoring, and control tactics developed for urban areas might need to be modified for rural lands.

Pathogens of Trees

Invasive pathogens have caused serious ecological and economic impacts to Midwestern forests. A few of the more significant current problems are highlighted below, in chronological order of recognition or introduction.

White pine blister rust, caused by the fungus *Cronartium ribicola*, was introduced during reforestation efforts in the early 1900s and is currently distributed throughout the range of eastern white pine (*Pinus strobus*). It causes mortality and top dieback, particularly on environmentally conducive sites. It is considered one of the most limiting factors in growing white pine in the region. The disease is managed by appropriate site selection, pathological pruning, and planting of putative resistant nursery stock (Geils et al. 2010).

Dutch elm disease (DED), caused by *Ophiostoma novo-ulmi* and *O. ulmi*, is a vascular wilt disease that has devastated native elms (*Ulmus americana*, *U. rubra*, and *U. thomasii*) across the region since the introduction of the fungi decades ago (*O. ulmi* in the 1930s and O. *novo-ulmi* in the 1970s). Successive waves of mortality can be attributed to ingrowth of susceptible elms and high populations of insect vectors of the DED fungi in affected areas. The vectors known to exist within the region include the native elm bark beetle (*Hylurgopinus rufipes*) and two non-native species, the smaller European elm bark beetle (*Scolytus multistriatus*) and the banded elm bark beetle (*Scolytus schevyrewi*). Management of the disease in urban settings is accomplished by sanitation to control the bark beetle vectors, chemical injections, and use of DED-tolerant cultivars. Operational trials are underway to evaluate the potential use of putative DED-tolerant elms in the restoration of riparian wild areas (Knight et al. 2017).

Oak wilt, caused by *Bretziella fagacearum* (syn. *Ceratocystis fagacearum*), is a devastating disease of red oak species (*Quercus* subsection Lobatae) that was first described in Wisconsin in 1942. It is considered by many experts to be non-native (Juzwik et al. 2008). The disease rapidly kills infected red oaks. It can also kill white oaks (*Quercus* subsection *Quercus*) in the Midwest, but tree death occurs over several to many years. Disease impact is generally more severe in landscapes with abundant red oaks compared to landscapes where white oaks are common. It is currently found in parts of all States in the region. The oak wilt range is expanding along the northern edge of its distribution. Oak wilt is now at epidemic levels in portions of affected States. Oak wilt is managed in urban and wildland environments by disrupting the overland and the belowground portions of the disease cycle to prevent the establishment of new infection centers and the expansion of existing centers. Current approaches to management on forest lands include preventing movement of diseased material, avoiding wounding during high-risk periods, and disruption of connected root systems (Juzwik et al. 2011).

Butternut canker (caused by *Ophiognomonia clavigignenti-juglandacearum*) was first reported on butternut (*Juglans cinerea*) in Wisconsin in 1967. Its origin is unknown, but it is believed to have been introduced to North America (Broders et al. 2014). It is now present throughout the natural range of butternut. The disease has killed up to 90% of the butternut trees in the region and may lead to extirpation of the species (Shultz 2003). Silvicultural approaches for butternut regeneration and selection of resistant trees have been proposed in an effort to promote survival of the species (LaBonte et al. 2015). There are no existing tools for management of the disease at this time.

Beech bark disease (BBD), caused by bark canker fungal species that colonize stylet wound damage of an exotic beech scale (*Cryptococcus fagisuga*), was first detected in the region (Michigan) in 2000 (O'Brien et al. 2001). Since that time, beech mortality has become widespread in parts of Michigan. The disease has also been confirmed in eastern Wisconsin and Ohio. As the disease moves through native forests, it kills a significant proportion of American beech *(Fagus grandifolia*), whose nuts are valuable as wildlife food. Mature beech trees can reach large size and are common in parts of Ohio, Michigan, and eastern Wisconsin. BBD is managed on the advancing front through salvage harvesting with retention of smooth-barked and unaffected trees and preventing the movement of infested materials (McCullough et al. 2005). An operational screening effort is underway to identify and propagate beech resistant to beech scale.

Diseases caused by *Phytophthora* species are an emerging concern throughout the region. White oak mortality in Ohio and Missouri has recently been attributed to *P. cinnamomi*, an exotic root-damaging pathogen (Balci et al. 2010). State and Federal plant regulatory agencies continue to monitor nursery stock for the introduction of *Phytophthora ramorum* which could affect the region's oak and ericaceous plants.

Invasive Plants of Terrestrial and Aquatic Systems

There are many non-native invasive terrestrial and aquatic plants distributed throughout the Midwest region. Many of these terrestrial plant species significantly affect the region's forest ecosystems, displacing native plant species and causing substantial damage. Several of the more important woodland species are highlighted below.

Garlic mustard (*Alliaria petiolata*) is a common invader in all Midwestern States (USDA, NRCS 2018). Brought from Europe as a food plant, this shade-tolerant species is now widely found in settings ranging from intact woodlands to disturbed areas (Kurtz and Hansen 2014). Garlic mustard is a biennial and forms large, nearly monospecific patches through heavy seed production, high seed germination rates, allelopathy, and disruption of mutualistic associations (Stinson et al. 2006). Biological control agents, including stem and root boring *Ceutorhynchus* spp. weevils (Becker et al. 2013), have been studied for nearly 20 years and are currently in the final stages of testing. A variety of tactics are employed to manage garlic mustard, including hand-pulling, removal of flowers before seed set, and herbicide application. Seeds are easily moved by animals, people, equipment, and vehicles, and new introductions are difficult to prevent. It can take years to manage large patches of garlic mustard even using multipronged management approaches.

Japanese barberry (*Berberis thunbergii*) was introduced as an ornamental. This species occurs in all Midwestern States but has a wide distribution in Ohio, Michigan, and Wisconsin (USDA NRCS 2018). It occurs in many habitats (closed canopy forests, open woodlands, wetlands, and fields), forming dense thickets and shading out other plants. It is very shade tolerant and grows under a wide variety of growing conditions. Thorns discourage some herbivores, but rabbits can feed on stems through the winter. Japanese barberry spreads through roots and branches that root when in contact with the soil. Birds and other animals eat the bright red berries and can disperse the seeds long distances. This species is typically managed by cutting, pulling, and herbicide use (Michigan DNR 2012).

Common buckthorn (*Rhamnus cathartica*) was also introduced as an ornamental shrub and is now prevalent in Minnesota, Wisconsin, and Michigan, occurring less frequently in the other Midwestern States (USDA, NRCS 2018). It grows as a shrub or small tree in habitats ranging from open fields to forests, forming dense thickets and crowding out native plants. This species has early leaf out and late leaf senescence and can have a longer growing season than other plants, in some cases by nearly as long as 2 months (Harrington et al. 1989). Common buckthorn is spread by birds that ingest fruit which ripens in the late summer. Control of this species can be difficult and can take years, because the thickets are difficult to work in and often resprout after cutting or pulling. Removal is generally followed by herbicide applications to cut stumps (NRCS 2007).

Exotic honeysuckles (*Lonicera* spp.) are common in forest, edges, wetlands, and disturbed areas, occurring in most counties of all Midwestern States (USDA NRCS 2018). Honeysuckles are shrubs, sometimes reaching 10–15 ft. in height, and produce flowers in spring and early summer that are attractive to bees. Fruits ripen in the fall and are dispersed by birds. Like with buckthorn, control is difficult, generally involving repeated efforts of cutting and stump treatments (Ohio State University Extension 2018).

The tree of heaven (*Ailanthus altissima*) is abundant in Ohio, Indiana, and Illinois and has spotty distributions in most other Midwestern States (USDA NRCS 2018). This fast-growing tree can approach 100 ft. in height and is found in many habitats, ranging from closed canopy forests to open fields and urban areas. Due to allelopathy, high seed production, and aggressive suckering, this species can completely dominate areas in which it grows and is difficult to control with cutting and herbicide stump treatments. Within the last 10–15 years, a soil-borne pathogen (*Verticillium nonalfalfae*) that causes vascular wilt and death in tree of heaven has been found in Ohio, Pennsylvania, and Virginia (Rebbeck et al. 2013). Further research is being conducted on this pathogen and its possible use as a biological control.

Reed canary grass (*Phalaris arundinacea*), phragmites (*Phragmites australis*), and purple loosestrife (*Lythrum salicaria*) are major invasive plants in wetland areas distributed throughout the entire region (USDA NRCS 2018). Biological control with beetles in the genus *Galerucella* has been a success in limiting purple loosestrife (Blossey et al. 2015), while reed canary grass and phragmites are generally managed with consecutive seasonal burns, mechanical removal, and herbicides (Michigan DEQ 2014).

Eurasian watermilfoil *(Myriophyllum spicatum*) is one of several invasive aquatic plants that is distributed widely throughout the region (USDA, NRCS 2018) and which can drastically alter the ecological processes and functioning of aquatic ecosystems. Other invasive aquatic plants in the Midwest include hydrilla (*Hydrilla verticillata*), starry stonewort (*Nitellopsis obtusa*), parrotfeather (*Myriophyllum aquaticum*), and curly-leaf pondweed (*Potamogeton crispus*). Management strategies include harvesting, rotovation, dredging, and aquatic herbicides (Mikulyuk and Nault 2009), but, as with aquatic animals, control of aquatic plants is costly and requires constant effort and investment. Eradication is all but impossible, so preventing new invasions is crucial to avoiding ecological and economic harm.

Invasive Animals of Terrestrial Systems

Invasive vertebrates and noninsect invertebrates threatening terrestrial ecosystems in the Midwest region include feral hogs (*Sus scrofa*) and invasive earthworms. Feral hogs damage native plants and crops and are problematic throughout Missouri, Indiana, Ohio, and Wisconsin. They are managed by trapping and removal, followed by improvement of the degraded habitat. Various species of invasive earthworms have been implicated in the degradation of native plant communities, especially throughout northern Minnesota and Wisconsin (Holdsworth et al. 2007). Best management practices have been developed and implemented to prevent further spread (e.g., Wisconsin Department of Natural Resources 2015).

Invasive Animals and Pathogens of Aquatic Systems

A variety of invasive aquatic animals are recognized as having important negative ecological and economic impacts in the Midwest region. These include fish such as sea lamprey (*Petromyzon marinus*), bighead carp (*Hypophthalmichthys nobilis*), and silver carp (*H. molitrix*); mollusks such as zebra mussel (*Dreissena polymorpha*) and quagga mussel (*D. bugensis*); crustaceans such as rusty crayfish (*Orconectes rusticus*) and spiny water flea (*Bythotrephes longimanus*); and pathogens such as viral hemorrhagic septicemia (VHS). These species and many other invasive aquatic animals in the region have disrupted native food webs and altered ecosystem functioning. In many cases, their impacts have reduced the value of ecosystem services and required the implementation of costly management activities to control invasive species and reduce their impacts. For example, sea lamprey, an invasive parasitic fish that feeds on the blood and body fluids of other fish, played a role in precipitous declines of Great Lakes fish stocks in the mid-twentieth century. Scientists discovered an effective lampricide (TFM, 3-trifluoromethyl-4-nitrophenol) in the late 1950s, and its application, along with several other management techniques, has been used to reduce sea lamprey populations. These control efforts are effective, but cost approximately $20 million each year.

In addition to sea lamprey, which invaded the Great Lakes from the North Atlantic Ocean through man-made canals, many other invasive aquatic animals have been introduced to the Great Lakes by the release of ballast water from transoceanic ships. Ship-borne species include zebra and quagga mussels, spiny and fishhook (*Cercopagis pengoi*) water fleas, round gobies (*Neogobius melanostomus*), and Eurasian ruffe (*Gymnocephalus cernua*). These, and some 50 other non-native aquatic species introduced to the Great Lakes by shipping, are estimated to reduce the value of ecosystem services from wildlife watching, commercial fishing, recreational fishing, and raw water usage by more than $100 million annually (Rothlisberger et al. 2012).

Invasive aquatic species that establish populations in the Great Lakes often spread to the rest of the Midwest and beyond. Zebra mussels, which invaded the Great Lakes in the 1980s, are a well-known biofouling organism. They quickly spread to rivers and inland lakes in the States surrounding the Great Lakes and, more recently, have become established in waterways in the Western United States.

Two invasive crayfish species that have serious impacts in the upper Midwest are native to the Southeast: the rusty crayfish and the red swamp crayfish (*Procambarus clarkii*). These species outcompete and hybridize with native crayfish and prey on native fish, crayfish, and gastropods.

Asian carps, including common carp (*Cyprinus carpio*), bighead carp, black carp (*Mylopharyngodon piceus*), grass carp (*Ctenopharyngodon idella*), and silver carp, are invasive fish that present significant concerns for the region. Asian carp species have had major impacts on native fish populations in the Mississippi River basin. Costly electric barriers to reduce the likelihood of Asian carp movement into the Great Lakes have been installed in the Chicago Ship and Sanitary Canal, a man-made hydrologic connection between the Great Lakes and the Mississippi River basin.

Other invasive fish of concern in the region include round goby and Eurasian ruffe, both of which are voracious benthivorous species with high reproductive rates. The piscivorous northern snakehead fish (*Channa argus*) has also been found in isolated locations in the Midwest region and threatens to become more widespread.

Pathogens that are not native to North America also cause harm to native fish species. Several of the diseases associated with these harmful non-native pathogens include viral hemorrhagic septicemia (VHS), salmonid whirling disease, and bacterial kidney disease. Cost-effective control methods are not yet available for most of the aquatic invasive animals in the Midwest region. Research into more effective and less expensive control methods is ongoing. Current management efforts emphasize spread prevention though campaigns to educate the public about the importance of not intentionally or inadvertently moving species among waterways and best practices for avoiding these movements. Direct intervention efforts such as inspecting and pressure washing recreational boats and trailers to remove invasive species propagules and laws requiring that no water be moved among waterways are also important prevention efforts.

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Northeast Region

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Introduction

The Northeast region is heavily forested with a high diversity of hardwood and conifer forest tree species. Northern hardwoods, including sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), yellow (*Betula alleghaniensis*) and paper birch (*B. papyrifera*), and aspen (*Populus tremuloides*) make up 44% of the forests, followed by the oak-hickory (*Quercus-Carya*) type (27%), pine (*Pinus*) types (white-red-jack pine (*P. strobus-P. resinosus-P. banksiana*), loblolly-shortleaf pine (*P. taeda-P. echinata*), and oak-pine) (12%), spruce-fir (*Picea-Abies*) type (11%), and bottomland types (elm/ash/cottonwood (*Ulmus/Fraxinus/Populus deltoides*) and oak/gum/cypress (*Quercus/Liquidambar/Taxodium*)) (5%). Topography, moisture gradient, and disturbance history highly influence where each forest type is found. The Northeast is also water rich, with over 10% of the total area covered by water. Aquatic ecosystems in the region include streams, swamps, lakes and ponds, rivers, and marine and estuarial habitats. In addition, New York has borders on two Great Lakes (Erie and Ontario), while Pennsylvania borders one (Erie).

The Northeast region comprises the New England and Mid-Atlantic States, including Maine, New Hampshire, Vermont, New York, Massachusetts, Connecticut, Delaware, Rhode Island, New Jersey, Maryland, Pennsylvania, and West Virginia (Fig. A7.1), and has a human population density greater than 330 people/mi^2. Many opportunities exist for human-mediated introductions of pests, including international shipping ports, a large urban/rural interface, highly industrialized areas, and high recreational use of forests. This region was colonized by Europeans earlier than most of the rest of the country, and coincidentally has the highest concentrations of invasive forest insects and pathogens in the country (Fig. A7.2). There are many