FIRE AND INVASIVE EXOTIC PLANT SPECIES IN EASTERN OAK COMMUNITIES: AN ASSESSMENT OF CURRENT KNOWLEDGE

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Abstract.—Successful regeneration of oak-dominated communities in the Eastern United States historically requires disturbance such as fire, making them vulnerable to invasion by exotic plants. Little is currently known about the effects of fire on invasive plant species and the effects of invasive plant species on fire regimes of this region. Seventeen common eastern invaders were evaluated for their response to fire and potential to change current fire regimes. Twelve species are potentially controllable with repeated growing-season burns (decreasers); five may increase in abundance in response to fire (increasers). Most of the woody decreasers are also potential resisters of fire at maturity. The presence of a seedbank or an outside seed source (evaders) for all but one species and a positive germination response to post-fire conditions (e.g., higher soil temperature, nitrogen availability, and light) make it less likely that most eastern plant invaders can be controlled by fire alone. Shifts in fire regime in eastern oak communities are undocumented but may occur due to changes in community flammability after an invasion. Current fire models are inadequate for predicting fire behavior in these oak communities due to a lack of information on eastern native and exotic plant species fuels. Consequently, fire behavior predictions are best made on a site-by-site basis, especially for sites with multiple invaders composed of increasers and decreasers as well as fire promoters and inhibitors.

INTRODUCTION

The dependence of oak ecosystems on disturbance for regeneration (Loftis 1983; Cook 1998) can result in a vulnerability to invasion by exotic plant species, successful establishment of which often is linked to disturbance (Anderson 1999; Lonsdale 1999; Debinski and Holt 2000; Knapp and Canham 2000; Mack et al. 2000; Buckley et al. 2002). Disturbances that increase understory light (e.g., fire, harvesting, herbicide treatments or combinations of these) are effective in increasing oak regeneration (Dolan and Parker 2004; Miller et al. 2004; Rebbeck et al. 2004). If disturbance were the only important factor determining invasion success, oak systems requiring greater disturbance levels for maintenance would be the most likely to be invaded by exotic plant species. Factors other than disturbance, such as high resource availability (Richardson et al. 1994; Burke and Grime 1996; Higgins et al. 1999; Lonsdale 1999; Stohlgren et al. 1999; Davis et al. 2000; Pysek et al. 2002; Thomson and Leishman 2005) and historic agricultural use (Dupouey et al. 2002, Ramovs and Roberts 2003) also can increase a community's vulnerability to invasion.

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For this paper, the Eastern United States includes the Midwest, Great Lakes Region, Ozarks, Appalachian Mountains, Mid-Atlantic, New England, and Southeast. Within these regions are at least five community types with a dominant oak component that may be affected by fire and invasive exotic plant species: 1) mesic forests (mixed mesophytic), 2) dry forests (oak-hickory), 3) oak woodlands/savannas, 4) oak glades/barrens, and 5) oak shrublands (scrub) (Jones et al. 1984; Andreas 1989; Haney and Apfelbaum 1990; Grossman et al. 1998). Definitions for woodland/savanna and glade/barren overlap in the literature, but are separated here because glades/barrens have a more patchy distribution of trees than woodlands/savannas due to less available water or nutrients. Seventeen exotic plant species that are potential invaders of at least one or all of these five oak communities are evaluated for their response to and effects on fire (Table 1).

Fire and Exotic Plant Invasion History

Before 1850, most fires in the eastern region were surface fires set by native Americans. Fire intervals ranged from 1 to 17 years, with shorter intervals in glades/barrens and woodlands/savannas than in forests. There was an increase in frequency between 1850 and 1930 due to European conversion of wilderness areas to farms. Between ~1850 and 1930 (and up to the 1950's in the

Table 1.—Seventeen exotic invasive species organized by habit type. Nomenclature in the table and text follows Gleason and Cronquist (1993) and the Integrated Taxonomic Information System (ITIS; 2005) http://www.itis.usda.gov/servlet/.

Herbaceous Plants
Canada thistle (*Cirsium arvense* (L.) Scop.)
Cogongrass (*Imperata cylindrica* (L.) Beauv.)
Garlic mustard (*Alliaria petiolata* (M. Bieb))
Japanese stiltgrass (*Microstegium vimineum* (Trin.) A. Camus)
Leafy spurge (*Euphorbia esula* L.)
Spotted knapweed (*Centaurea biebersteinii DC*.)
Vines
Japanese honeysuckle (*Lonicera japonica* Thunb.)
Kudzu (*Pueraria montana* (Lour.) Merr. var. *lobata* (Willd.) Maesen & Almeida)
Oriental bittersweet (*Celastrus orbiculatus* Thunb.)
Shrubs
Autumn olive (*Elaeagnus umbellata* Thunb.)
(Russian olive – *E. angustifolia* was used as an example of EHOC value in Table 2 that may correlate with autumn olive)
Bush honeysuckle (*Lonicera* L. spp.)
Data in the text was presented specifically for Morrows bush honeysuckle (*L. morrowii* A. Gray), Amur honeysuckle (*L. maackii* (Rupr.)), and Maxim and Bell's bush honeysuckle (*L*. x *bella* Zabel).
Common buckthorn (*Rhamnus cathartica* L.)
Japanese barberry (*Berberis thunbergii* DC.)
Common barberry (*B. vulgaris* L.) was used as an example of EHOC value in Table 2 that may correlate with Japanese barberry.
Multiflora rose (*Rosa multiflora* Thunb.)
Privet (*Ligustrum* L.spp.)
Chinese privet (*L. sinense*) and common privet (*L. vulgare* L.) had specific research data described in the text.
Trees
Norway maple (*Acer platanoides* L.)
Tree of heaven (*Ailanthus altissima* (Mill.) Swingle)

Ozarks) catastrophic wildfires increased after logging. Before this time, fire-rotation periods generally were less than 10 years, though this varied by community type and region. After 1930-50, fire suppression increased rotation periods to as long as 6,000 years for the Eastern United States (Dey 2002). Despite these changes in fire frequency and rotation, most fires in the East were and remain human caused and often are correlated with drought (Dey 2002; Muzika et al. 2005).

Recorded invasions of exotic plants and other exotic species since 1800 have increased at an accelerated rate, presumably due to increased intercontinental mobility (Liebhold et al. 1995). Similar increases in exotic plant species at a more local scale are evident from herbarium data (Huebner 2003). Although the correlation is circumstantial and weaker for the exotics, there is a noticeable increase in all species collected after the intensive logging and burning phase (1879-1920) in

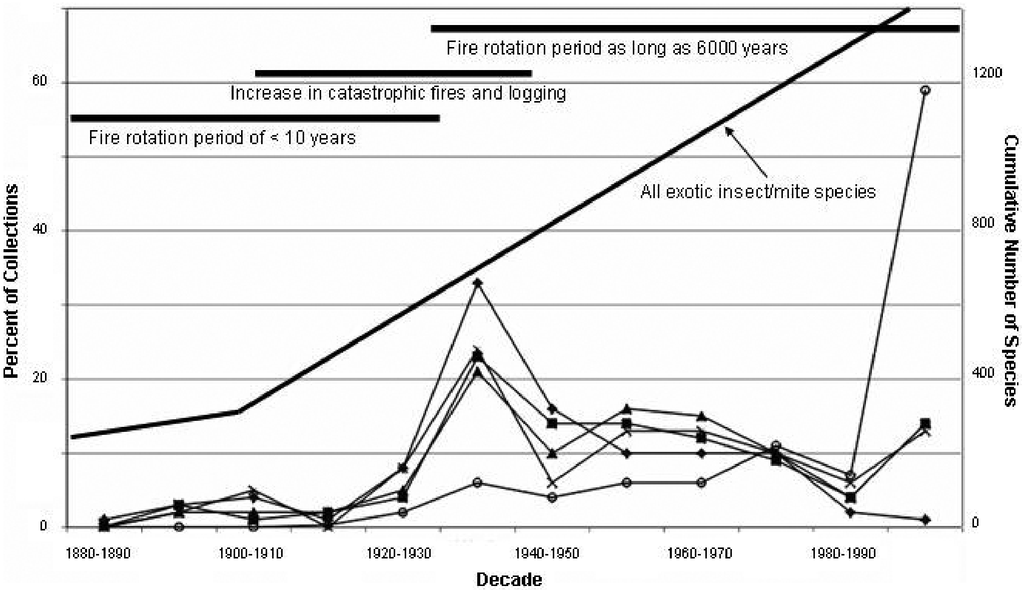

Figure 1.—Time trend in percentage of the total number of species in herbarium collections from West Virginia for nine exotics (o), pteridophytes/gymnosperms (diamond), dicots (square), Asteraceae members (triangle), *Solidago canadensis* (x) (Huebner 2003) overlaid by total number of exotic insect and mite species recorded in the US (Liebhold et al. 1995) and the estimated fire rotation periods in the eastern hardwood forests (Dey 2002). The nine species include three bush honeysuckles (amur, morrow, and tatarica), garlic mustard, Japanese honeysuckle, Japanese stiltgrass, multiflora rose, oriental bittersweet, and tree of heaven.

West Virginia (Clarkson 1964). Thus, while fires in the Eastern United States have decreased since the 1950's, exotic invasive plants have increased since the 1800's, the tenuous connection being human intervention (Fig. 1).

Native Oak Communities' Response to Fire

In general, oak species may benefit from fire because: 1) intense fires followed by low-intensity fires open areas for encroachment by oak species and/or 2) low-intensity fires in forests open the subcanopy or canopy and reduce competition from shade-tolerant species (Lorimer et al. 1994; Miller et al. 2004). Hom (2003) proposed that shrub oak barrens, oak woodlands, and oak forests require fire-return intervals of 15 to 25, 20 to 30, and 100 to 200 years, respectively, for community maintenance. These intervals are inversely proportional to each community's likeliness to burn (Streng and Harcombe 1982).

Spring burns are often more successful at reducing non-oak woody vegetation with epigeal germination (cotyledons are above ground) and slow root-system development that contrast with oak species which generally have hypogeal germination (cotyledons are below ground) and relatively rapid root growth. Fall and winter burns have less effect on woody species, which have translocated much of their energy into below-ground storage (VanLear and Brose 2002; Richburg et al. 2001). Repeated spring and fall burns may increase prairie forb cover in oak woodlands/savannas and oak glades/barrens by reducing competition of grass species, a possible management concern for threatened forbs, e.g., wild lupine (*Lupinus perennis* L.) (Pauly 1997).

Flammability and Fuels of Invasive Plant Species

Flammability has four components: 1) combustibility, 2) ignitability, 3) consumability, and 4) sustainability all of which are influenced, on an individual plant basis, by plant moisture content, percent make-up of carbon, presence of volatile compounds, leaf thickness, and overall surface area-to-volume ratio (Behm et al. 2004). A few comparisons have been made on combustibility of eastern native and exotic plants using effective heat of combustion (EHOC; quantified with a cone calorimeter) (Dibble et al. 2003b). Compared to western fuels' combustibility (14.5 to 21.6 MJ/kg), fuels of both native and nonnative species in the Eastern United States tend to have lower EHOC values (10.9 to 16.0 MJ/kg). These studies also showed that nonnative species had lower EHOC values (10.0 to 14.2 MJ/kg) for combustibility than native species (11.7 to 16.0 MJ/kg; Table 2). The available EHOC data weakly suggest that eastern invasive plant species may tend to reduce fire occurrence; flammability data, using all components, on more species are needed (White et al. 2002). The ability to predict fire behavior in oak communities also requires knowledge of the flammability of the fuel bed. Surface fuels beds are composed of a range of categories, including litter, downed wood, shrubs, grasses, and forbs and are described by their average depth and percent cover and weighted averages of the 1) loading (lbs/ft^2), 2) surface-area-to-volume ratio, 3) heat content (BTU/lb), and 4) moisture content of each sub-class of fuel. Fuel sub-classes are determined by the general fuel category and the size of particles within a category and whether the material is dead or alive (Richburg et al. 2004). Community flammability is determined by multiple, interacting species, and, consequently, is more difficult to measure than flammability of individual species.

Species and community flammability, as well as current weather conditions, are all needed to predict fire behavior at a site. Models like BEHAVE's NEWMDL program (allowing for custom fuel models) may improve predictive capabilities by incorporating data from eastern forests (Richburg et al. 2004). However, it is essential that the fuel bed characteristics of each community type be well-defined. For instance, in woodlands/savannas, the surface-to-volume ratio of fine fuels has a disproportionately large effect on fire spread, whereas in a closed hardwood stand, both the surface-to-volume ratio and the fuelbed depth of fine fuels have a disproportionately large effect on the spread rate of fire (Ducey 2003). In a study with paired invaded and noninvaded mixed hardwood and hardwood stands, the invaded stands had less nonwoody litter, more 100-hr fuels, less duff depth, more graminoid and shrub cover, and less basal area than the noninvaded stands (Dibble et al. 2003a). Thus, while invasive plants may have low individual flammability, they may change a community's flammability such that fire can burn and spread rapidly due to the presence of graminoids, or burn slower and more intensely in the presence of 100-hr fuels.

Exotic Plant Invaders' Response to Fire

Invasive plant species responding to fire can be classified as: 1) evaders -- species with long-lived propagules stored in the soil (or a reliable outside seed source), 2) endurers -- resprouters, or 3) resisters -- species that survive low-intensity fire due to certain adaptive characteristics. These can be classified secondarily as increasers -- likely to increase in abundance after a burn, maintainers -- likely to maintain their current population size after a burn, or decreasers -- likely to decrease in abundance after a burn (Harrod and Reichard 2001). Each of the 17 species evaluated has been categorized on the basis of the available literature (Table 2).

Several nonnative woody species, including autumn olive (Szafoni 1991a), common buckthorn (Solecki 1997; Richburg et al. 2004); Chinese privet (Faulkner et al. 1989); Japanese barberry (Richburg et al. 2004); Japanese honeysuckle (Barden and Matthews 1980; Solecki 1997); kudzu (Radar et al. 1999); Morrow's and Bell's bush honeysuckle (Nyboer 1992; Luken and Shea 2000; Solecki 1997; Richburg et al. 2004); multiflora rose (Szafoni 1991b; Solecki 1997); Norway maple (Simpfendorfer 1989; Maissurow, 1941, based on similarities with sugar maple); and two herbaceous plants, garlic mustard (a biennial) (Nuzzo 1996; Schwartz and Heim 1996) and spotted knapweed (a perennial), can be reduced in community importance after repeated (annually consecutive for at least 2 to 5 years) growing-season (spring to early summer) fires. The data for spotted knapweed, which can resprout after a burn like all of the above species (Sheley et al. 1998), are unpublished[2]. Japanese stiltgrass, an annual, can be expected to decrease in cover after a burn, given no seed source, though there is no direct evidence for this and resprouting is possible (Tu 2000; Gibson et al. 2002). Though all 12 of these species are technically

[2]Zouhar, K. 2001. Centaurea maculosa. In: Fire Effects Information System. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. Available: http://www.fs.fed.us/database/feis/. Accession date: October 20, 2005.

Table 2.—EHOC values, colonizer type, response to fire, and possible effects on fire regimes documented for each exotic invasive plant species. Species are judged based on their response to repeated spring fires. na = no data; ? indicates no literature is available to support the possible categorization.

Species	EHOC (MJ/kg)	Colonizer type after fire	Population response to fire	Promote or inhibit fire
		Herbs		
Canada thistle	na	Endurer	Increaser Evader	Promote?
Cogongrass	na	Endurer	Increaser Evader (off-site seed only)	Promote?
Garlic mustard	na	Evader	Increaser	Neither? Decreaser
Japanese stiltgrass	11.14	Evader	Increaser Decreaser	Promote?
Leafy spurge	High (no value)	Endurer Evader (but hot fires kill seed)	Increaser	Promote?
Spotted knapweed	na	Evader	Increaser Decreaser	Inhibit?
		Vines		
Japanese honeysuckle	14.34	Evader (but poor seed production) Resister	Decreaser Maintainer	Promote? (as ladder fuel) Inhibitor?
Kudzu	na	Evader Resister (older individuals)	Maintainer Decreaser (younger plants)	Inhibit? Promote? (as ladder fuel)
Oriental bittersweet	11.44	Endurer Evader (off-site seed only)	Increaser	Inhibit? Promote? (as ladder fuel)
		Shrubs		
Autumn olive	na (13.42) (Russian olive)	Evader (off-site seed, possible seedbank)	Increaser Decreaser	Neither?
Bush honeysuckle	na	Evader (off-site seed only)	Increaser Decreaser	Neither?
Common buckthorn	na	Evader (off-site seed, possible seedbank) Resister (older individuals)	Increaser Decreaser Maintainer	Neither?

continued

Table 2.—continued.

Species	EHOC (MJ/kg)	Colonizer type after fire	Population response to fire	Promote or inhibit fire
Japanese barberry	na (14.04)	Evader	Increaser	Neither?
	(common barberry)	(off-site seed source, possible seedbank)	Decreaser	
Multiflora rose	12.30	Evader	Increaser	Neither?
				Decreaser
Privet	na	Evader	Increaser	Neither?
		(off-site seed only)	Decreaser	
		Trees		
Norway maple	11.11	Evader	Increaser	Inhibit?
		(off-site seed; seedbank possible but unlikely)	Maintainer	
		Resister	Decreaser	
		(older individuals)		
Tree of heaven	na	Endurer	Increaser	Neither?
		Evader		
		(off-site seed only)		

endurers, they may be considered decreasers because of the evidence that their underground reserves and ability to sprout may be insufficient to withstand repeated growing-season burns. However, this evidence is not limited to oak-dominated communities; some studies were conducted in pine plantations (Japanese honeysuckle), prairies (spotted knapweed) and pastures (multiflora rose). Also, in the case of garlic mustard, there is evidence of increased garlic mustard abundance after repeated growing season burns in oak systems (Nuzzo et al. 1996; Luken and Shea 2000). The absence of a seedbank or outside seed source of all these species would ensure their status as decreasers or maintainers (Table 2).

Conversely, fires may be ineffective against large, mature shrubs of common buckthorn (Solecki 1997) and against large, woody individuals of any of these species. Similarly, kudzu can maintain a high water content due to its deep taproot so it may not experience topkill during a growing season burn (Winberry and Jones 1973). Tissue moisture content of Japanese honeysuckle (Slezak 1976) and Norway maple (Horvitz et al. 1998) may also be resistant to some fires. Thus, depending on age and size, autumn olive, common buckthorn, Japanese barberry, Japanese honeysuckle, kudzu, the bush honeysuckles, multiflora rose, privet, and Norway maple may be classified as resisters and maintainers.

Several species with deep and/or extensive underground root reserves, including oriental bittersweet (Dreyer et al. 1987), tree of heaven (Lepart and Debussche 1991), Canada thistle (Adams et al. 1987; Solecki 1997), leafy spurge (Cole 1991), and cogongrass (King and Grace 2000; Grace et al. 2001) may respond positively to repeated growing-season burns by increasing in abundance due to an ability to root sucker or sprout, keeping in mind that the research is not directly related to fire response (oriental bittersweet and tree of heaven) or took place in a pine plantation (cogongrass) or prairie/grassland (leafy spurge and Canada thistle). These species can be considered potential endurers and increasers after a fire (Table 2). Although evidence is stronger that Canada thistle responds positively to fire, some studies show it decreasing in abundance after summer or fall burns (Kirsch and Kruse 1973; Hogenbirk and Wein 1991). Seedlings of cogongrass respond positively to a burn and this grass is likely to respond to fire like other perennial warm-season grasses. However, its growth response to multiple fires in different seasons is unclear (King and Grace 2000; Grace et al. 2001).

Seed germination of kudzu often is promoted after a fire due to scarification (Takahashi and Kikuchi 1986; Susko et al. 2001). Garlic mustard (Hintz 1996; Byers and Quinn 1998), Japanese stiltgrass (Barden 1987; Anderson and Schwegman 1991; Anderson et al. 2000; Glasgow and Matlack 2005), multiflora rose (Szafoni 1991b; Kaye et al. 1995; Leck and Leck 1998; Luginbuhl et al. 1999; Glasgow and Matlack 2005), spotted knapweed (Davis et al. 1993), Canada thistle (Kellman 1970; Turner et al. 1997), and leafy spurge (Wolters et al. 1994) all have seedbanks and may respond positively to fire (or, rather, the resulting removal of litter and increases in soil surface temperature and nutrients) with respect to on-site seed germination. Thus, these species are potential <u>evaders</u> and <u>increasers</u> (Table 2).

Seed banking for tree of heaven (Kowarik 1995, but see Kostel-Hughes and Young 1998), oriental bittersweet (Van Clef and Stiles 2001; Ellsworth et al. 2004), bush honeysuckles (*Lonicera maackii* (Rupr.) Maxim.; Ingold and Craycraft 1983), privet (*Ligustrum sinense* Lour. and *L. vulgare* L.; Panetta 2000; Shelton and Cain 2002), cogongrass (Grace et al. 2001), and garlic mustard (Baskin and Baskin 1992) is minimal and most establishment by seed after a fire is by off-site dispersal. It is uncertain whether Japanese barberry, autumn olive (Katz and Shafroth 2003), common buckthorn, and Norway maple (Hong and Ellis 1990) have seedbanks. Tree of heaven seed is dispersed as far 200 m by wind (Graves 1990; Kota 2005); Norway maple also is wind dispersed but distances are not known. Oriental bittersweet, autumn olive, and multiflora rose have been dispersed as far as 500 m by starlings (LaFleur and Rubega 2005), while Japanese barberry seeds have been dispersed at least 80 m by birds (Silander and Klepeis 1999). Bush honeysuckle and common buckthorn also are dispersed by birds but distances are not published. If outside seed sources exist, all of these species would be potential <u>evaders</u> and <u>increasers</u>. There is little evidence of Japanese honeysuckle having a seedbank and this species is a comparatively less prolific seed producer due to pollinator limitations (Larson et al. 2002) and low fruit viability (Haywood 1994). Thus, establishment by seed of Japanese honeysuckle after a fire may be unlikely.

A species' population response to fire may differ from its individual response. For instance, a patchy forest burn likely only removes portions of a large garlic mustard population. Because the population growth of this species is density dependent (Meekins and McCarthy 2002), a large population of garlic mustard likely will show a positive growth response after a patchy burn due to a decrease in intraspecific competition.

Potential to Change Current Fire Regimes

There are no specific data showing shifting fire regimes in the Eastern United States for the 17 species evaluated. Leafy spurge may alter fire intensity because of the high oil content in its leaves (Davis 1990). However, fires with this species present may become hot enough to kill leafy spurge (and native species) seedbank seeds (Wolters et al. 1994). This could halt leafy spurge regeneration if sprouting were unlikely (Brooks et al. 2004). Canada thistle, Japanese stiltgrass, and cogongrass can build up fine, continuous fuels such that a rapidly moving, even spreading fire is likely should a burn occur (Barden 1987; Hogenbirk and Wein 1995; Rice 2004). Such species may cause similar changes in fire regime as was found with the burn and regeneration cycle of cheat grass (*Bromus tectorum* L.; D'Antonio and Vitousek 1992; Brooks et al. 2004). The mesic habitats that Japanese stiltgrass tends to invade and the low flammability of this grass (compared to western species) may make such a regime change unlikely.

If a community is invaded by vines, such as oriental bittersweet, the vine may act as a ladder fuel that could result in a rapidly spreading canopy burn under ideal climatic conditions. Spotted knapweed tends to increase the patchiness of the grassland communities it invades, making it more difficult for such communities to carry a fire (Sheley et al. 1998). Also, if shrubs are invading a community that was not previously dominated by shrubs, there may be a subsequent decrease in fine fuels and a reduction in the community's ability to carry a fire, though any fire that does occur may be more intense. If the community is a savanna or barren, the latter may result in the loss of that community type over time. However, such resource-limited communities may not be invaded as easily. Japanese honeysuckle (Slezak 1976) and Norway maple (Horvitz et al. 1998) may promote an increase in shade-tolerant species by shading out intolerant species, which, in turn, may reduce the likelihood of a burn. These species would be similar to the Brazilian pepper tree (*Schinus terebinthifolius* Raddi.), which retains high moisture in its leaves and litter and has reduced the fire frequency in once pyric pine rocklands (Gordon 1998).

CONCLUSION

If an invaded site is being managed for oak regeneration, the repeated burning required to control certain invasive species, e.g., bush honeysuckle and privet, may have a detrimental effect on oak, e.g., northern red oak (*Quercus rubra* L.), and other native species in the more mesic communities. However, such burns may be required to maintain certain xeric forests, woodland/savanna and glade/barren communities, though repeated burns might promote the invasion of other nonnative species, e.g., leafy spurge and Canada thistle.

We still lack adequate data with which to discern reliable patterns. The comparatively well-studied species, garlic mustard and Canada thistle, have variable responses to fire. As additional research is conducted on these 17 and other species, it is likely that the patterns discussed here will become more complex and variable. Such variability probably will result from differences in site conditions (community flammability, species flammability, resource availability, land-use history) before and after a burn (which are often unknown or difficult to measure) as well as each invasive plant's population age structure and size just before a burn (both of which also are often unknown and difficult to measure). Moreover, many of these species occur together and joint effects on how a site will burn may differ from individual effects. Management of communities with multispecies invasions in which multiple increasers and decreasers as well as promoters and inhibiters might be present may be of such a complex nature that a site-specific approach always is required.

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