# Nonnative Plant Response to Silvicultural Treatments: A Model Based on Disturbance, Propagule Pressure, and Competitive Abilities

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ABSTRACT

Invasion by nonnative plants can result in substantial adverse effects on the functions of native forest ecosystems, including nutrient cycling and fire regimes. Thus, forest managers need to be aware of the potential impacts of management activities, including silvicultural treatments, on nonnative vegetation. To aid in that effort, we created a conceptual model of potential responses of nonnative plants to silvicultural treatments, used the model to make a general set of predictions, and tested our predictions against observed responses published in the scientific literature. Of a total of 42 studies that addressed the effects of silvicultural treatments on nonnative plants, 90% found a posttreatment increase in at least one nonnative plant species. All of the studies that assessed the effect of disturbance intensity on nonnative plants found that invasion success increased with increasing disturbance intensity or number of management entries. As predicted by the model, there was substantial among-species and among-site variation in nonnative plant responses to silvicultural treatments; variation in responses were probably driven by local conditions including propagule pressure, condition of the forest community, or characteristics of the local flora (native and nonnative species). Because species- and location-specific changes in nonnative plants are dependent on local conditions, local knowledge is important for predicting invasion potential. In addition, monitoring is essential for early detection of postharvest invasions and/or expansions of nonnative plants.

**Keywords:** competition, disturbance, propagule pressure, exotics, invasives

In 2007, more than 1.3 million ac of federal lands were mechanically treated to "help save the lives of firefighters and citizens and to reduce the risk of catastrophic fire" (US Department of the Interior [USDOI] and US Department of Agriculture [USDA] 2008). This has been a threefold increase in hazardous fuels thinning treatments since the National Fire Plan was initiated in 2001. With increased interest both in treating wildland fuels to mitigate fire hazard and in using biofuels to replace petroleum, this number could increase to more than 3 million ac/year by 2030 (US Department of Energy [USDOE] and USDA 2005, US Forest Service 2008). The types of mechanical treatments that are being implemented to reduce fuels have the potential to increase abundance and richness of nonnative plants. Disturbance associated with treatments can result in increased levels of light, water, nutrients, and exposed mineral soil and in decreased plant competition (Covington et al. 1997, Battles et al. 2001, Gundale et al. 2005). These factors promote colonization, establishment, and expansion of native and nonnative ruderal species (Hobbs and Huenneke 1992, Bailey et al. 1998). Because of the potential threat that nonnative plant species pose to native species, ecosystems, and fire regimes (Vitousek et al. 1996, Mack et al.2000, Zouhar et al. 2008), managers need to be aware of the impact that silvicultural treatments can have on nonnative vegetation, alter their harvesting techniques to minimize negative nonnative impacts, and have weed control strategies in place to deal with increases in nonnative plant populations after harvest.

To aid in that effort, we created a conceptual model of the impacts of disturbance on nonnative plants and used the model to make predictions about the responses of nonnative plants to silvicultural treatments. We then tested our predictions by analyzing published literature on the responses of nonnative plants to silvicultural treatments in North America north of Mexico. We included all types of mechanical harvest activities in our literature review, rather than limiting our review to thinning treatments, to capture the widest disturbance gradient.

## The Model

Invasions by nonnative plants are regulated by the interactions of ecosystem properties (such as disturbance regimes and resource availability), nonnative propagule pressure (the number and frequency of nonnative plant seeds and parts), and species-specific characteristics of local native and nonnative plants (which determine competitive interactions) (Lonsdale 1999). Disturbance can increase available resources directly by altering the form or availability of nutrients (such as a nutrient pulse after fire) or indirectly by reducing competition by injuring or destroying existing vegetation (Hobbs and Huenneke 1992, Davis et al. 2000). If viable nonnative plant propagules are present after disturbance, either from onsite or off-site seed sources, they can germinate and compete with existing vegetation for available resources (Rouget and Richardson 2003, Lockwood et al. 2005). Ultimately, it is species-specific plant traits that determine the outcome of native and nonnative resource competition (Vilà and Weiner 2004).

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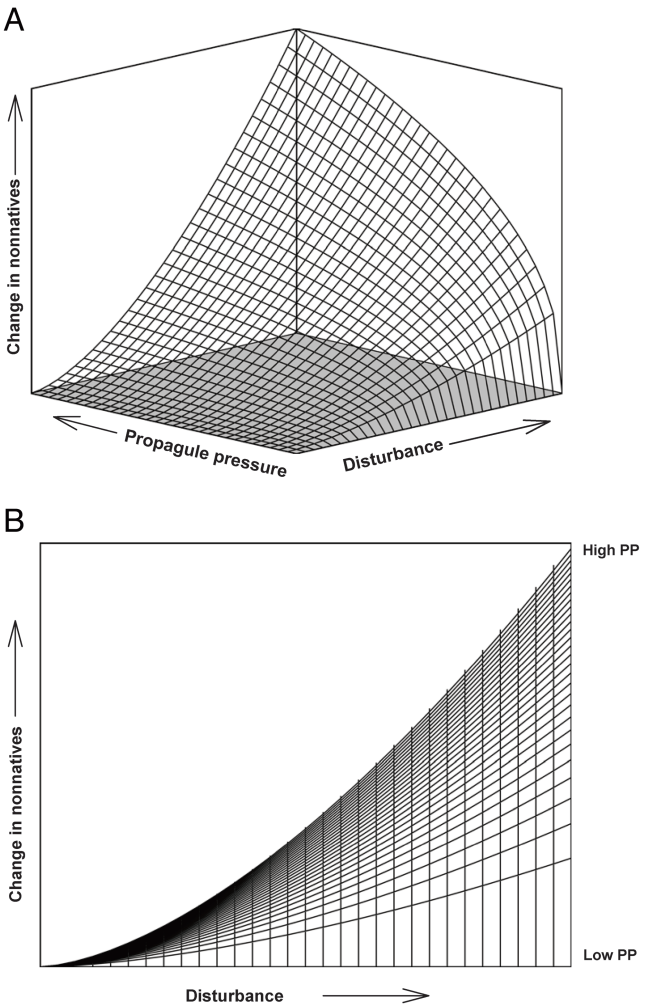


**Figure 1. Conceptual model of nonnative plant response to disturbance. (a) Hypothetical three-dimensional model of the impact of propagule pressure (PP) and disturbance on change in nonnatives. (b) Same model with propagule pressure axis pointed away from reader.**

Ideally, the conceptual model should be four-dimensional, with separate axes for disturbance, propagule pressure, competition, and nonnative plant response; but for ease of comprehension, we will illustrate using a simplified hypothetical three-dimensional model, based on disturbance intensity and propagule pressure (Figure 1a). Because nonnative plants need resources to establish, persist, and expand in disturbed areas, it is resource availability that limits postdisturbance plant response (whether measured as species richness, frequency, cover, density, or biomass). Because resource availability increases with disturbance severity, potential postdisturbance plant response should also increase with disturbance (in this example, a concave function). Whether a particular nonnative plant achieves its postdisturbance potential depends on whether propagules are present. Obviously, if propagules are not present, the species will not occur on the disturbed site, regardless of resource availability. Thus, with no disturbance or no propagule pressure, there is no change in nonnatives. As disturbance and propagule pressure increase, nonnatives increase with a maximum at high disturbance and high propagule pressure, although in some cases nonnative propagules can saturate the environment resulting in a decreased rate of change (in this example, a convex function) (Figure 1a).

Unfortunately, managers usually have little information on propagule pressure or competitive ability. If we rotate Figure 1a until the propagule-pressure axis points toward the reader, this illustrates the information that may be available to managers: change in nonnatives as a function of disturbance (Figure 1b). This two-dimensional interaction results in a triangular (or wedge)-shaped postdisturbance response of nonnatives, bounded on the top by resource availability (potential response) but with realized response (the grided area) determined by propagule pressure (horizontal grid lines; Figure 1b).

A similar model could be constructed with propagule pressure replaced by competitive ability and, conceptually, a four-dimensional model could be created for disturbance, propagule pressure, and competition. If propagule pressure is high and the plant is a strong competitor, nonnative response should be set by resource availability. If propagule pressure is extremely low or the plant is a poor competitor, the response of nonnatives should be low regardless of resource availability. For many species and locations, however, propagule pressure and competitive ability are intermediate and nonnative response should be intermediate.

The concept of potential versus realized response can be illustrated by the threat of postdisturbance sedimentation. Disturbance creates the potential for erosion, but erosion is not realized unless there is a significant storm event. As vegetation recovers, the threat of erosion and sedimentation diminishes even though storms may occur (Elliot and Robichaud 2005). Likewise, disturbance creates conditions that are favorable for the colonization and growth of nonnative species. But these nonnative species will not establish unless their propagules are present and can compete with resident vegetation. As vegetation recovers, available resources diminish and the threat of invasion decreases.

## The Predictions

1. Because disturbance intensity, propagule pressure, and native–nonnative competition can vary widely after silvicultural treatments, the responses of nonnative plants to these treatments should also be variable.
2. Unless nonnative propagule pressure and competitive ability are extremely low, disturbance from harvesting should result in an increase in nonnative richness and/or abundance.
3. With uniform nonnative propagule pressure and competitive ability, increasing harvest intensity should result in an increase in nonnative richness and/or abundance.
4. Because prescribed fire is an additional disturbance that can also increase resource availability, harvesting and burning should result in a larger nonnative response than harvesting alone.
5. If disturbance is relatively uniform, nonnatives should increase with increasing propagule pressure.

## Methods for Testing the Model

We used data on posttreatment changes in nonnative plant abundance from the published literature to test our predictions about the responses of nonnative plants to silvicultural treatments. Silvicultural treatments were defined as any type of mechanical harvesting of trees including thinning, logging, or chaining. A nonnative plant was defined as a plant from outside of North America north of Mexico that has been released intentionally or unintentionally or has been disseminated or escaped as a result of human activity and

has become established somewhere within North America north of Mexico (after Kartesz 1999). Taxonomy and state noxious weed designation followed the *Synthesis of North American Flora* (Kartesz 1999).

Initially, we located articles based on keyword searches on combinations of "harvesting" and "thinning" and variations of nonnative, including "nonnative," "alien," "invasive," and "weed." However, because the literature on harvesting and nonnative plants is limited, these searches produced relatively few articles; we then used the literature cited sections of these articles, their "cited by" references, and our knowledge of the literature to identify additional articles. The criteria used to select articles were study sites in North America north of Mexico; quantitative data on changes in abundance of nonnative plants were included; and studies included preharvest and/or control (unharvested) data. This resulted in 42 articles (Table 1), with a geographic range from California to Virginia and Arizona to British Columbia. Forest types varied from western juniper to spruce and from pine-oak to northern hardwood. Interestingly, less than 25% of the publications had "nonnative," "alien," "invasive," or "weed" in the title or key word list. An article was scored as having a posttreatment increase in nonnative plants if abundance (frequency, cover, density, or biomass) of at least one nonnative species was larger on a treated site than on a pretreated or control site

Because of space limitations, we compared predictions from our model with (1) findings presented in an investigation by Nelson et al. (2008) of 70 ponderosa pine stands (15 untreated controls, 20 thinned, 19 thinned and prescribed burned, and 16 prescribed burned) on three National Forests in eastern Washington (the most complete article on the effects of thinning and thinning plus burning on nonnatives identified in our review) and (2) overall findings reported in the literature. See Martinson et al. (2008) for a detailed review of many of these studies.

## Results

### The Responses of Nonnative Plants to Treatments Should Vary

Nelson et al. (2008) found that two nonnative grasses and two nonnative forbs were more abundant in control stands and 12 nonnative grasses and 19 nonnative forbs were more abundant on thinned stands. Results from the remaining 41 studies also show variability among species in response to disturbance. Posttreatment nonnative species richness varied from 0 to 42. Not all nonnative species increased in abundance after treatment (Dyrness 1973, Griffis et al. 2001, Gray 2005). In fact, posttreatment change in frequency varied from 0 to 88% and change in cover varied from 0 to 18%. There also was a considerable amount of variation among sites (Scherer et al. 2000, Stickney and Campbell 2000, Selmants and Knight 2003). For example, Stickney and Campbell (2000) examined 25 clearcut conifer stands in Idaho and Montana. None of the stands had any nonnative plant species before harvesting. Eleven of the 25 stands had nonnative plant species after harvesting and burning (see propagule pressure later in this article).

### Nonnative Plants Should Increase with Treatment Disturbance

Nelson et al. (2008) found that nonnative species richness and cover were significantly higher on thinned stands than in control stands, although the magnitude of response was relatively small (changes of approximately 1.5 species and 1.5% cover). Of the remaining 41 studies, 37 reported an increase in nonnative species richness or abundance after treatment, 2 found no nonnative plants on either treated or control plots (Thompson and Gartner 1971, ponderosa pine in South Dakota; Wayman and North 2007, mixed conifers in California), 1 reported a loss of dandelion (*Taraxicum officinale*) 17 years after harvesting (Alaback and Herman 1988, spruce/western hemlock forests in Oregon), and 1 study did not have a harvest-only treatment (Fulé et al. 2005).

### Nonnative Plants Should Increase with Increasing Treatment Intensity

Thinning removed approximately 50% of the basal area in the eastern Washington study (Nelson et al. 2008). Three to 19 years after thinning, there was a significant negative relationship between basal area of residual trees and nonnative plant richness ($r = -0.27$; $P = 0.02$) and cover ($r = -0.25$; $P = 0.04$). Similarly, nonnative species richness and cover were negatively correlated with tree cover. Ten additional studies assessed the impact of treatment intensity on nonnative plants; all found a positive relationship between treatment intensity and increases in the abundance of nonnative plants. Nine of the studies found that nonnative plants increased with decreasing basal area or canopy cover (Uresk and Severson 1998, Thomas et al. 1999, Battles et al. 2001, Scheller and Mladenoff 2002, Abella and Covington 2004, Wienk et al. 2004, Gray 2005, Lindgren et al. 2006, and Dodson and Fiedler 2006). In addition, Aukema and Carey (2008) found that number of nonnative species increased with number of thinning entries.

### Harvesting and Burning Should Result in a Larger Nonnative Response than Harvesting Alone

Nelson et al. (2008) found that richness of nonnative plants was threefold higher on thinned and burned stands compared with stands that were only thinned and fourfold higher than on control stands. Nonnative cover was fourfold higher on thinned and burned stands compared with those that were only thinned and fivefold higher compared with control stands. Nonnative plant richness and cover was positively correlated with percent of area burned. Twelve additional studies compared treatments that combined harvesting and burning with those that included only harvesting; all reported that nonnative plants increased with the addition of prescribed burning.

By using the results from each study as an independent sample, a sign test can be used to test for a significant increase in nonnatives after treatment, a significant increase in nonnatives with increasing treatment intensity, and a significant increase in nonnatives with harvesting plus prescribed fire. The results are significant for all three comparisons ($P \ll 0.01$).

### Nonnative Plants Should Increase with Increasing Propagule Pressure

To test the effect of propagule pressure, Nelson et al. (2008) established transects along roadsides, on skid trails, and within stands—areas that often act as corridors for the spread of plant propagules (Parendes and Jones 2000, Trombulak and Frissell 2000, Gelbard and Belnap 2003). Nelson et al. (2008) found that nonnative plant cover was significantly higher along roadsides and skid trails than within stands (5.5, 5.0, and 1.5%, respectively, for roads, skid trails, and within stands), but concluded that propagule

**Table 1. Location, forest type, treatment types, response variables, and direction of response and authorship of investigations of the effects of silvicultural treatments on nonnative plants.**

| Location | Forest type | Metric | Harvest effect | Intensity effect | Harvest + burn versus harvest | Nonnative species | Change cover or frequency | Noxious species | Author |
|---|---|---|---|---|---|---|---|---|---|
| Arizona | PP | Frequency | Increase | Increase | | 3 | 1–14 | 3 | Abella and Covington 2004 |
| Oregon | S/WH | Frequency | Decrease | | | 1 | | | Alaback and Herman 1988 |
| Minnesota | A | Richness | Increase | | | 1 | | 1 | Alban et al. 1994 |
| Washington | DF | Richness | Increase | Increase | | 3–20 | | | Aukema and Carey 2008 |
| Oregon | DF/WH | Cover | Increase | | | | <1 | | Bailey et al. 1998 |
| | | Frequency | Increase | | | | <1 | | |
| | | Richness | Increase | | | 4–10 | | 5 | |
| California | Mixed conifer | Cover | Increase | Increase | | 1–7 | 3 | 1 | Battles et al. 2001 |
| | | Richness | Increase | Increase | | | | | |
| Virginia | Hardwood | Density | Increase | | | 1 | 0.3 | | Carter and Fredericksen 2007 |
| California | Mixed conifer | Cover | Increase | | Increase | 2 | 1 | 2 | Collins et al. 2007 |
| | | Richness | Increase | | Increase | | <1 | | |
| Washington | WH/DF, S/WH | Cover | Increase | | | | <1–18 | | DeFerrari and Naiman 1994 |
| | | Density | Increase | | | | <1–4 | | |
| | | Richness | Increase | | | 1–27 | | 13 | |
| Montana | PP | Cover | Increase | Increase | Increase | | <1–2 | | Dodson and Fiedler 2006 |
| | | Richness | Increase | | Increase | 1–10 | | 8 | |
| Washington | PP/DF | Cover | Increase | | Increase | | <1 | | Dodson et al. 2008 |
| | | Richness | Increase | | Increase | 0–7 | | 1 | |
| Oregon | DF | Cover | Increase | | Increase | 0–6 | 0–8 | 1 | Dyrness 1973 |
| | | Frequency | Increase | | Increase | | | | |
| | | Richness | Increase | | Increase | | | | |
| West | Conifers | Cover | Increase | | | | 0–7 | | Freeman et al. 2007 |
| | | Richness | Increase | | | 0–4 | | | |
| Arizona | PP | Cover | | | Increase | | | | Fulé et al. 2005 |
| | | Richness | | | Increase | | | 2 | |
| Oregon | Various | Cover | Increase | Increase | | | <1–5 | | Gray 2005 |
| | | Frequency | Increase | Increase | | 8 | <1–14 | 2 | |
| Arizona | PP | Forb cover | Decrease | | Increase | | <6 | | Griffis et al. 2001 |
| | | Forb richness | Increase | | Increase | 2 | <3 | 1 | |
| | | Gram cover | Increase | | Increase | | <5 | | |
| | | Gram rich | Increase | | Increase | 2 | <2 | 1 | |
| Oregon, Washington | DF | Cover | Increase | | | | 11–17 | | Halpern and Spies 1995 |
| | | Richness | Increase | | | 1 | <10 | | |
| Arizona | PP | Seeds | Increase | | | | | 1 | Korb et al. 2005 |
| British Columbia | LP | Richness | Increase | Increase | | 3 | | | Lindgren et al. 2006 |
| Montana | PP/DF | Cover | Increase | | Increase | | <3 | | Metlen and Fiedler 2006 |
| | | Richness | Increase | | Increase | | 1–10 | 2 | |
| Oregon | J | Biomass | Increase | | | | | 1 | Miller et al. 2005 |
| Arizona | PP | Biomass | Increase | | Increase | 1 | | 1 | Moore et al. 2006 |
| Arizona | PP | Abundance | Increase | | Increase | | <3 | 2 | Neary et al. 2002 |
| Washington | DF | Cover | Increase | | | | <1 | | Nelson and Halpern 2005 |
| | | Frequency | Increase | | | 5 | 2–31 | 2 | |
| Washington | PP | Cover | Increase | Increase | Increase | | <1 | | Nelson et al. 2008 |
| | | Richness | Increase | Increase | Increase | 42 | 2–6 | 11 | |
| Oregon, Washington | WH | Frequency | Increase | | | 8 | 1–88 | 3 | North et al. 1996 |
| British Columbia | DF | Cover | Increase | | | | | 1 | Page et al. 2005 |
| Oregon | Chaparral, oak | An grasses cover | Increase | | | | | | Perchemlides et al. 2008 |
| | | An forb cover | Decrease | | | | | | |
| Michigan, Wisconsin | Northern hardwood | Cover | Increase | Increase | | 1 | 0–5 | | Scheller and Mladenoff 2002 |
| | | Frequency | Increase | Increase | | | 2–34 | | |
| Washington | Mixed conifer | Cover | Increase | | Increase | 13 | <6 | 4 | Scherer et al. 2000 |
| Oregon | WH/DF | Cover | Increase | | | 5 | 0–3 | 3 | Schoonmaker and McKee 1988 |
| Wyoming | LP, SF | Presence | Increase | | | 7 | <1 | 2 | Selmants and Knight 2003 |
| Massachusetts | Mesic forest | Frequency | Increase | | | | | | Silveri et al. 2001 |
| Idaho, Montana | Various conifers | | Increase | | | | | 2 | Stickney and Campbell 2000 |
| Washington | DF | Cover | Increase | Increase | | 3 | <1% | | Thomas et al. 199 |
| | | Frequency | Increase | | | | | | |
| South Dakota | PP | Biomass | No exotics | | | | | | Thompson and Gartner 1971 |
| Washington | DF | Cover | Increase | | | | <1 | | Thysell and Carey 2000 |
| | | Richness | Increase | | | | | 3 | |
| South Dakota | PP | Biomass | Increase | Increase | | | | | Uresk and Severson 1998 |
| | | Richness | Increase | | | | | 3 | |
| Oregon | J | Biomass | Increase | | | | | 1 | Vaitkus and Eddleman 1987 |
| California | Mixed conifer | Frequency | No exotics | | | | | | Wayman and North 2007 |
| South Dakota | PP | Biomass | Increase | Increase | Increase | | | 1 | Wienk et al. 2004 |
| Oregon | PP/DF | Cover | Increase | | Increase | | <3 | 1 | Youngblood et al. 2006 |
| | | Frequency | Increase | | Increase | | | | |
| | | Richness | Increase | | Same | | | | |

A = aspen, *Populus tremuloides*; DF = Douglas-fir, *Pseudotsuga menziesii*; J = western juniper, *Juniperus occidentalis*; LP = lodgepole pine, *Pinus contorta*; PP = ponderosa pine, *Pinus ponderosa*; S = spruce, *Picea* spp.; SF = spruce-fir, *Picea* sp. and *Abies* sp.; WH = western hemlock, *Tsuga heterophylla*.

pressure was less important than environmental characteristics for determining within-stand abundance of nonnatives. Only one additional study examined the impacts of propagule pressure and logging roads on nonnatives. Silveri et al. (2001) found a higher frequency of *Celastrus orbiculatus* along logging roads than in adjacent logged areas in mesic forests in Massachusetts.

There is a general pattern of decreasing richness and/or abundance of nonnative plants with increasing elevation (Randall et al. 1998, Fisher and Fulé 2004). Presumably, propagule pressure also decreases with elevation. Stickney and Campbell (2000) examined 25 clearcuts in coniferous forests at elevations between 2,650 and 5,400 ft in the Northern Rockies. Their data indicate a negative correlation between elevation and the number of nonnative species ($r^2 = 0.80$; $P < 0.01$). There were no nonnative plants on logged stands above 4,900 ft.

### Nonnative Plants Should Decrease as Stands Recover from Disturbance

Although the model does not predict this result, it is often quoted as a generality (see Zouhar et al. 2008, for a review of nonnatives and postfire recovery). Nelson et al. (2008) found the opposite pattern. There was a significant positive correlation between time since thinning and nonnative richness and cover ($r = 0.37$ and $P = 0.04$; and $r = 0.36$ and $P = 0.04$, respectively). These results from thinning of ponderosa pine (*Pinus ponderosa*) dominated forests in the inland Northwest are in contrast to those observed after more intensive harvest activities in closed canopy forests of the Pacific Northwest. For instance, nonnative species cover peaked 2 years after clearcutting of western hemlock (*Tsuga heterophylla*) forests in the Western Cascades of Oregon (Schoonmaker and McKee 1988), 3–7 years after clearcutting of western hemlock forests of the Olympic Peninsula (DeFerrari and Naiman 1994), and 2 years after clearcutting of Douglas-fir (*Pseudotsuga menziesii*) forests of Oregon and Washington (Halpern and Spies 1995). Peaks in nonnative cover were variable for clearcut coniferous forests in the Northern Rockies, 2–3 years for some, 7–8 years for others, and 14–15 years postharvest for stands with sulfur cinquefoil (*Potentilla recta*)(Stickney and Campbell 2000).

## Discussion

The 42 articles reviewed in this investigation provide clear evidence that silvicultural treatments can increase the richness and abundance (whether measured as cover, frequency, or biomass) of nonnative plants. Nonnative plant response is positively correlated with intensity of disturbance. Thus, removing more basal area, making multiple thinning entries, or using prescribed fire for slash removal may exacerbate rates of invasion. As predicted by the model, there is high variation in nonnative plant response among species and sites; differences in response are likely driven by local variation in propagule pressure, condition of the forest community, or characteristics of the species in the local flora (native and nonnative species).

Our analyses considered the overall response of nonnative plants, but all nonnative plants are not equal with respect to ecosystem effects. Some nonnative plants can establish and persist in a community with little or no impact. Other nonnative plants, such as cheatgrass (*Bromus tectorum*), leafy spurge (*Euphorbia esula*), Dalmatian toadflax (*Linaria dalmatica*), and spotted knapweed (*Centaurea biebersteinii*), are invasive and can displace native vegetation, alter fire regimes, and, in extreme cases, completely change plant community composition. Unfortunately, there is no consistent definition of invasive species (Randall 1997). However, assessing the response of noxious weeds may provide some insight into whether the observed increase in nonnative plants poses a substantial management concern. Although noxious weed designation is a legal rather than a biological definition, nonnative noxious species generally spread aggressively and have adverse ecological or economic effects. Of 40 studies that reported positive posttreatment responses of nonnative plants, 30 reported at least one (and as many as 13) nonnative, noxious weeds.

Postharvest changes in nonnative cover were generally small, averaging 3% with a mode of <1%. This is not surprising because many of the untreated controls had no nonnative plants (e.g., Stickney and Campbell 2000). These posttreatment nonnatives may represent new local invasions. The invasion process typically has three phases: introduction, colonization, and naturalization (Cousens and Mortimer 1995). Commonly, there is a lag phase after introduction where the population size is small and stable (Radosevich 2008). In the lag phase, nonnative plants may be only a minor threat to the native community, but it is during this period that they are easiest to control. This may be the invasion phase that is driving the trends for most studies analyzed here. Korb et al. (2005) have indicated that timber harvest can result in an increase of nonnative plants in the soil seed bank. This can lead to an increase in nonnative propagule pressure with future disturbance. McGlone et al. (2006) reported that cheatgrass response to thinning was small during the first 5 years after harvest of ponderosa pine forests in Mount Trumball, Arizona. There was, however, a dramatic increase in cheatgrass cover 6 years after harvest because of a severe summer drought followed by heavy fall rain.

## Caveats

Using the literature as a data source introduces biases. First, it is easier to publish positive results (i.e., a significant posttreatment increase or decrease in nonnative plants) than no change. Using literature cited and "cited by" to find additional articles can bias toward publications that found similar or supportive results. In addition, although an article may report a significant trend for a particular nonnative species, all nonnative species may not have increased after harvest or the increases may not have been consistent among sites. Several of the studies used treatments of different ages to infer the impact of canopy closure on nonnative plants (Schoonmaker and McKee 1988, DeFerrari and Naiman 1994, Nelson et al. 2008); this approach assumes that conditions determining nonnative plant establishment have been constant over time. We know that this assumption is false. The number of nonnative species has increased exponentially in the past 2 centuries (Randall et al. 1998) and, presumably, nonnative propagule pressure has also increased over the same period. It could be argued that decreasing nonnative species in stands of increasing age reflects differences in propagule pressure at the time of disturbance rather than loss of nonnatives with canopy closure. Finally, although data in 38 of the studies indicated a posttreatment increase in nonnative species, many of the changes were not statistically significant. Because postharvesting changes in nonnative plants are small and variances tend to be large, a large sample size is needed to have the statistical power to detect changes (Kern et al. 2006). Many studies do not have adequate sample sizes to detect a significant change in postharvest nonnative

plants and pre- and postharvesting values for nonnative plants may not be reported.

## Conclusions

In general, if nonnative propagules are present, silvicultural treatments can lead to an increase in some nonnative plants (species and/or abundance). Nonnative plant response increases with treatment intensity (e.g., basal area removed or number of entries into stand) and the use of prescribed fire for slash disposal. Increasing nonnative propagule pressure leads to larger posttreatment increases in nonnative plants.

Although initial nonnative plant response to silvicultural treatments is small, this can lead to significant changes in the soil seed bank, increased nonnative propagule pressure, and subsequent increases in invasive, nonnative plants. Although coarse-scale responses (e.g., increase in nonnative species richness, cover, and/or biomass) to silvicultural treatments are predictable, fine-scale responses (e.g., species and location-specific changes in nonnative plants) are dependent on local conditions (i.e., propagule pressure and native and nonnative plant traits). Therefore, local knowledge becomes important for predicting these posttreatment changes and monitoring is essential for early detection of posttreatment invasions and/or expansions of nonnative plants.

The no-action alternative to silvicultural treatments may be wildfire rather than an undisturbed forest. Wildfire is a greater disturbance than harvesting and can result in large increases in invasive, nonnative plants (see Zouhar et al. 2008 for a comprehensive review). Economic and ecological costs and benefits need to be evaluated before making a decision to implement silvicultural treatments. If the decision is to treat stands, the impact of harvesting on nonnative plants needs to be anticipated and a plan to mitigate these impacts needs to be in place.

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