Prescribed fire and invasive plants in the western United States: A meta-analysis approach to evaluating successes and failures

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Introduction

Fire effects on plant communities and individual species have been a topic of interest for many years, particularly in light of renewed interest in the use of prescribed burns to reduce fuel loads, control exotic, invasive plant populations, or restore native plant communities. A number of literature reviews have presented in-depth discussions on the responses of invasive plants to fire, and an abundance of quantitative data have been generated over the years from experimental and post-wildfire field studies (Gill 1981; Keeley 1981, Rowe 1983, D'Antonio 2000, Brooks and Pyke 2001, Keeley 2001, Rice 2004, DiTomaso et al. 2005). To date, however, no one has attempted to use synthetic statistical tools to evaluate the magnitude of fire effects on exotic plant species across studies.

In this paper, we present the third component of a research project funded by the Center for Invasive Plant Management that examines the use of fire as a tool to control invasive plants. The first two phases involved (1) a literature review on fire and invasive species (Rice 2004) and (2) a report, based in part on a workshop held in 2004 (DiTomaso et al. 2005). Our goal here is to quantitatively examine and compare data available on fire effects on invasive species to determine if there are overall patterns that can be used to develop a decision tool for using prescribed burning to control invasive plants. Although qualitative literature reviews provide a wealth of information, a quantitative model based on empirical data that could predict outcomes of using prescribed burning would be of practical use to land managers. Land managers must make decisions about the benefits and risks of using prescribed burns on a regular basis and a rigorous analysis that seeks generalities as well as an understanding of the sources of variability in outcome would be of great value to them.

Life histories and growth forms of plants can determine their vulnerability to fire. Knowledge of a plant’s life history, location of perennating buds, seed dispersal and dormancy mechanisms, and seed longevity is required to predict which species are susceptible to fire (Brooks and Pyke 2001). Pyke et al. (2003) have presented a preliminary flow chart to predict outcomes of fire treatments on several life forms of invasive plants. These life forms are based on the work of Raunkiaer (1934), who classified plants on the basis of the position of the perennating buds or meristem. Pyke et al.’s flow chart indicates that the most important factor to consider in using fire to manage annual, monocarpic plants is seed longevity. If seeds live longer than two years, then fire alone may not be an effective management strategy. Invasive, annual grasses with shorter lived seeds may be successfully controlled with fire and follow-up treatments. The response of perennial or polycarpic plant species to fire is largely dependent on the location of the buds relative to a fire. Shrubs and trees (phanerophytes) are dependent on protection of vascular tissue and resprouting to survive post-fire.

One tool for synthesizing results of multiple studies and deriving general outcomes of given treatments, such as prescribed burning, is meta-analysis (Osenberg et al. 1999). Meta-analysis is a statistical tool useful for determining general relationships influenced by predictor variables and it offers an alternative and more objective approach to lengthy qualitative literature reviews (Hedges et al. 1999, Gurevitch et al. 2001, D’Antonio et al. 2003). In meta-analysis, the outcomes of different studies are examined by calculating “effect sizes” and testing whether they are consistent with one another across studies and whether the overall magnitude of the effect size is large, moderate, small, or not significant

(Gurevitch and Hedges 2001). When effect sizes are not consistent across studies, sources of heterogeneity are sought.

The interactions between fire and invasive plants are complex. Through meta-analysis of multiple independent studies, we sought to discover general patterns that could be useful for developing guidelines for using prescribed burning to control invasive species. We asked the following specific questions: 1) What is the overall effect size of fire on different life forms of invasive plants in the western USA and how much does that effect vary? 2) How do season of burn, number of burns, and plant life form influence effect size and its variation? 3) What are the magnitudes of effect sizes based on geographic location? And, 4) Can we develop a practical and useful decision tool for using prescribed burning to control invasive plants?

Methods

Literature reviews

We conducted this study in several phases. Initially, we selected three species of exotic, invasive plants – *Bromus inermis* (smooth brome), *Centaurea solstitialis* (yellow starthistle), and *Cytisus scoparius* (scotch broom) – as representatives of three distinct plant groups – grasses (hemicryptophytes), forbs (chamaeophytes), and perennial shrubs/trees (polycarpic phanerophytes), respectively. We selected these species because we knew that they had been studied in prescribed burning experiments and all are management concerns in several states. We began an extensive search for published studies on fire effects on these species. We utilized Biological Abstracts/BIOSIS (1991-present, provided by Ovid), Agricola (1970-present, provided by EBSCOhost), Agricola (Silver Platter, 1984-present), the Web of Science (1996-present, provided by Thomson Scientific), and WorldCat (present, provided by First Search) at Oregon State University and used the common and scientific names for each plant species in conjunction with search terms “prescribed and fire,” “prescribed and burn,” “fire,” and “burn.” We used the term “fire” to include wildfire studies in the searches.

When it became obvious that the number of studies suitable for a meta-analysis was limited, we added more representative species to the database. Species added to our study included *B. tectorum* (cheatgrass), *B. japonicus* (Japanese brome), *B. rubens* (red brome), *B. mollis* (soft brome), *C. maculosa* (spotted knapweed), *Taeniatherum caput-medusae* (medusahead), *Genista monspessulana* (french broom), and *Tamarix spp*. (tamarisk). We expanded our research efforts and traveled to the USDA Forest Service Rocky Mountain Research Station Fire Sciences Laboratory in Missoula, Montana to obtain gray and unpublished literature not available at the Oregon State University library. The Fire Sciences Laboratory (FSL) has developed the Fire Effects Information System (FEIS) that contains English-language literature on about 900 plant species, including extensive synoptic reviews on fire effects and fire ecology for a number of invasive plants. References in this system are derived from the Citation Reference System (CRS), a searchable database of citations developed by the Fire Effects Research Work Unit. We took advantage of their extensive reference lists and reviewed papers for *Bromus inermis* (n=127 references), *B. tectorum* (n=497), *B. japonicus* (n=81), *B. madritensis* (n=81), *Centaurea solstitialis* (n=164), *C. maculosa* (n=236), *Taeniatherum caput-medusae* (n=125), *Cytisus scoparius* (an initial list of 115 references), and *Tamarix spp*. (n=266). The FSL was marking their 50,000th reference at the time of our visit.

We also examined on-line abstracts from the Ecological Society of America annual meetings for the past three years to determine if any researcher might have unpublished data that would be appropriate for inclusion. We contacted such researchers and a number of them generously provided data sets. In addition, a workshop report on the use of fire as a tool for controlling invasive weeds, held in March 2004, and a review paper on the use of fire as a tool for controlling non-native invasive plants, both part of a shared grant from the Center for Invasive Plant Management in Montana, were useful in providing information to this study (DiTomaso et al. 2005, Rice 2004).

Building the database

The next phase of the study consisted of building the database. Our criteria for including a study in the database included: a) species that represented one of three Raunkiaer life forms (although we separated annual and perennial grasses) b) studies that reported means, standard deviations or standard errors of species responses to fire, and c) studies that had repeated control (unburned) and treatment (burned) plots or areas, or pre- and post-burn data of the same sites. We used percent cover, seedling density, seed bank counts, and biomass measures as acceptable response variables. Because multiple measures from the same study are not independent, we generally included only one response measure from each study. Exceptions occurred for two studies in which we included two response variables that represented separate life stages of a species (seeds and seedlings). For all studies we included independent potential predictor variables such as season of burn, number of burns, number of years since burn (identified as post-burn year), fuel information, and place of study (as a very general surrogate for climate). Not all of the studies had information on all of the variables, but we included any information that was provided. Many of the studies reviewed had unique information that could not be utilized in the meta-analysis. Most of the studies that we reviewed lacked either replication or true control plots and so could not be used in this analysis.

Analysis

We used MetaWin (Rosenberg et al. 2000) to conduct a categorical meta-analysis of the data. Meta-analysis uses effect sizes calculated for each study to determine the magnitude of the response to a given treatment. Effect sizes are calculated from response ratios – the ratio of the mean value of a particular response variable in an experimental group to that in the control group for that study. The response ratio therefore quantifies the proportionate change that occurs from a treatment (Hedges et al. 1999). We calculated effect sizes using a log response ratio because these ratios quantify proportionate changes and eliminate inherent differences among sites and populations (Hedges et al. 1999). The overall effect size is the mean derived from all of the independent studies. A negative response ratio value indicates that there is a smaller number in the burn treatment than in the control, and a positive value indicates a greater number in the burn treatment than in the control. An effect size is generally interpreted as “small” if it is 0.2, “medium” if it is 0.5, and “large” if it is 0.8, and “very large” if it is greater than 1.0 (Cohen 1969).

After calculating effect sizes for each study, we used a mixed-effects model for analysis because we made the assumption that there are additional sources of variation in fire effects and that there are fixed differences in life forms. Mixed models are more general than either fixed or random-effects models because they become a fixed-effects model if there is no variation remaining after sampling error and differences among categories have been accounted for or they become a random-effects model if there is only one descriptive variable (Gurevitch and Hedges 1999). We conducted the analysis in a hierarchical scheme. First, we first calculated the grand mean effect size across all studies and all treatments (without incorporating data structure) to determine if there was a common true effect size and whether the response ratio was relatively consistent or homogeneous. Then, we proceeded to group comparisons across broad categories, first by life form, genus, and species. Season of burn, number of burns, post-burn year (when measures were taken), and location were used as independent variables, with effect size as the dependent variable. We continued to make comparisons between control and treatment groups at finer scales (e.g., number of burns by season, number of burns by post-burn year, number of burns by life form, etc.).

Within each analysis, we considered the cumulative effect size a true estimate of the overall magnitude of fire effect if the bias-corrected bootstrapped 95% confidence interval (hereafter CI) of the calculated mean effect size did not overlap zero ($p < 0.05$). We used the bias-corrected C.I.s because meta-analyses generally do not conform to normal distribution criteria (Rosenberg et al. 2000). The homogeneity of effect sizes within a cumulative effect size for a group was determined using the weighted sum of squares statistic, Q_T, which represents the total heterogeneity in a sample (Rosenberg et al. 2000). A significant Q_T (using a chi-square table) indicates that there is greater variation among effect sizes than expected by sampling error and that other factors should be considered (Rosenberg et al. 2000). Measures of heterogeneity between groups (Q_B) indicated whether fire effects are significantly different among treatment and plant life forms or species. Within group heterogeneity is expressed as Q_W.

Using Rosenthal's equation, we calculated the number of records needed to generate a reliable result for mean effect size (Rosenthal 1979, cited in Rosenberg et al. 2000). This "fail safe" number is actually the number of additional non-significant studies required to change a significant result to a non-significant one. It is based on the idea that there is a publication bias towards publishing studies that show significant results. If the fail safe number is much greater than the actual number of records used, then the results are considered to be a reliable estimate of the true effect. If the number is near or less than the number of records, then the result is not considered reliable.

Results

The database

The complete data set for the meta-analysis contained only 21 studies from which effect sizes could be calculated, with 73 comparisons between control and treatment means (Table 1). Included in these were six studies that compared unburned and burned plots after wildfires. This low number reflects a general lack of replication, lack of reporting of variation, or lack of adequate controls within the otherwise vast literature on fire. The distribution of records (or comparisons) for each species was: *B. inermis* (n=9), *B. tectorum* (n=23), *B. japonicus* (n=5), *B. mollis* (n=2), *B. rubens* (n=5), *Taeniatherum caput-medusae* (n=1), *Centaurea solstitialis* (n=6), *Tamarix spp.* (n=4), *Genista monspessulana* (n=16), and *Cytisus scoparius* (n=2). Of the 21 studies, six unpublished data sets were generously provided by researchers, and standard error measures were provided by two authors of papers that had not reported them in their publications. The data were sorted by life form, species, season of burn, number of burns, and post-burn year. We were unable to test the importance of fire intensity characteristics, fuel manipulations, or post-fire precipitation patterns because there was not enough information in the majority of studies to characterize the conditions.

A matrix of categorical grouping variables used in the meta-analysis is shown in Table 2. Sample sizes within life forms and treatments were often too small to carry out reliable analysis.

Outcomes of meta-analyses

Importance of life form: The grand mean effect size (E^{++}) across all studies and plant groups was first calculated without incorporating data structure (using a random effects model). E^{++} across all of the studies was -0.8747, a large negative effect of burning on all species in the analysis, and the bias C.I. ranged from -1.1815 to -0.5849. However, the measure of heterogeneity was significant (Q_T = 91.71, df=71, p = 0.049), indicating that that there was greater variation among effect sizes than expected by sampling error.

Grand mean effect size for all records of woody shrubs and trees was -1.2629, a very large effect. Total heterogeneity (Q_T = 25.11, df = 21, p = 0.242) in this group was not significant, indicating a homogeneous negative response to fire. The grand mean effect size (E^{++}) of fire on all annual grasses was -0.6688. This is considered a medium response (Cohen 1969), but the high heterogeneity indicated that other variables should be explored as predictors (Q_T = 50.98, df=34, p = 0.031). The only perennial grass in this analysis was smooth brome and the grand mean effect between 3 studies and 9 records was -0.8506, a large negative effect. The total heterogeneity (Q_T = 20.37, df = 8, p = 0.009) indicated a large variation in effect sizes among the records. The only forb represented in the analysis was yellow starthistle, with 2 studies and 6 records of data. The grand mean effect size (E^{++} = -0.559) indicated a medium negative effect and heterogeneity was not significant (Q_T = 5.08, df = 5, p = 0.405). However, with this small sample size, results are not reliable. Rosenthal's fail safe number was only 7.9.

When we calculated effect sizes using a mixed model grouped by life form (separating annual and perennial grasses), fire treatment had the greatest effect (E^+ = -1.3265) on phanerophytes, the shrubs and trees, although heterogeneity within groups was significant (Q_W = 108.35, df=68, p = 0.001) (Figure 1). The effect size for smooth brome was large (E^+ = -0.8823), and medium for all annual grasses (E^+ = -0.6705) and yellow starthistle (E^+ = -0.5952). Total heterogeneity in this analysis (Q_T) was 113.99 (df = 71, p = .001). This suggests that other sources of variation should be examined.

Effects of number of burns: The number of burn treatments had a predictable negative effect on exotic species abundance. Although there were unequal records for one, two, and three burn treatments, plots burned three times across studies had the largest mean effect size (E^+ = -1.4240, n = 5). This result was driven by one of the few studies that carried out burning for three consecutive years for yellow starthistle control (Kyser and DiTomaso 2002). One time burns had the next greatest effect (E^+ = -0.8041, n = 51), and two burns had a medium effect size (E^+ = -0.6869, n = 11). French broom was the only plant species for which there were studies that included two years of burning treatments and two years of post-burn data (Odion and Haubensak 1999). The heterogeneity within groups (Q_W = 99.36, p = 0.003) and total heterogeneity (Q_T = 100.98, p = 0.004) indicated that other sources of variation were also important. Some studies included fuel addition or removal treatments, but samples sizes were too small for reliable analysis and there were no significant test outcomes with the small sample sizes.

All of the annual grass studies involved one burn treatment, so effects could not be compared with multiple years of burning. There were no significant differences in effect sizes among annual grass species' responses to one time burn treatments, although the effect size was smallest for *B. tectorum* (E^+ = -0.3659, n = 23) and larger for other species. Burning had the greatest effect on *B. rubens* (E^+ = -1.1850, n = 5), followed by *B. mollis* (E^+ = -1.0331, n = 2), and *B. japonicus* (E^+ = -0.7491, n = 5). Total heterogeneity was not significant. Eight studies followed annual grass post-burn response for two years. In the second year after burning, the effect on *B. tectorum* decreased further (E^+ = -0.2227, n = 6). For *B. japonicus*, the effect slightly increased (E^+ = -0.8986, n = 4). The differences were not significant between the species.

We could not separate species of phanerophytes for individual analysis. Studies in which woody plants were burned once or twice indicated that the first burn had the largest effect (E^+ = -1.632, n=4; E^+ = -0.2644, n = 10, respectively). The first post-burn year also had a higher effect size than the second post-burn year (E^+ = -1.2239, n= 6 and E^+ = -0.2295, respectively). However, these results were not significantly different and sample sizes were too small to be reliable.

Effects of season of burn: The season of burn proved to be an important predictor of outcome. When we grouped seasons across all studies, burns that occurred in the late spring had the greatest negative impact on species (E^+ = -1.7567, n = 4) (Figure 2). Heterogeneity between seasons (Q_B = 9.21, df = 4) was significant (p = 0.05), and total heterogeneity (Q_T) was not significant, indicating some consistency among responses. Spring, late summer, and fall burns also had large effect sizes (E^+ = -0.9147, n = 8; -1.4629, n = 3; and -0.9717, n = 34, respectively). Summer burns had the least effect size (E^+ = -0.1439, n = 18). We further examined effect sizes of season by analyzing only studies with a single burn treatment. The pattern was consistent (Figure 2).

Late spring was the most effective season for burning annual grasses ($E^+ = -1.78$, n=4), although late summer and spring burns also had very large effect sizes ($E^+ = -1.4626$, n=3; $E^+ = -1.343$, n = 5) (Figure 3). Late spring burn studies did not include any of the cheatgrass records, but did include data for medusahead, red brome, and soft brome. Summer burns essentially had no effect ($E^+ = 0.0720$, n = 12) and fall burns had a small effect ($E^+ = -0.4265$, n = 10). Between group heterogeneity (Q_B) was significant (df=4, p = 0.045). Sample sizes vary greatly in this analysis, but Rosenthal's fail safe number is 67.8, indicating some reliability. Total heterogeneity was not significant.

There were 23 data base entries for fire effects on cheatgrass. The grand mean effect size was negligible ($E^{++} = -0.3421$, n = 23). Because there were more records for this species than any other and because of it's ecological importance, we evaluated whether differences in effect sizes were predictable based on season. The greatest number of studies documented burn effects in the summer, with fewer records for fall, and less than 5 each for late summer and spring burns. With these vastly different sample sizes, there were no significant differences between seasons and results were not reliable. There were, however, large differences in the magnitude of effect sizes. Spring burning had the greatest effect ($E^+ = -2.1221$, n = 2), followed by late summer ($E^+ = -1.4617$, n = 3), fall ($E^+ = -0.1629$, n = 8), and summer ($E^+ = 0.1795$, n=10).

When we combined all annual grass entries and *B. inermis* studies, differences within and between seasons were significant (df = 4, p = 0.04; df = 38, p = 0.01, respectively). Four *B. inermis* records were added to the fall burn records and three were added to the spring burn records in the analysis and the other season records remained unchanged (i.e., late spring, summer, and late summer). Late spring burning still had the greatest negative effect (E^+ = -

1.6736, n = 4) and summer burning essentially had no effect (E^+ = 0.0621, n= 12). Also, with all grasses considered together, the negative effect size for fall burning increased (E^+ = -0.7544, n = 17) while the negative effect size of spring burning decreased (E^+ = -0.8939, n = 8). This result indicates that burning *B. inermis* in the spring may not be as effective as burning the species in the fall.

When we examined studies with summer burns, only the annual grasses (n = 12) and yellow starthistle (n = 6) were represented. There was essentially no effect of fire on annual grasses (E^+ = 0.0323) and a medium effect size on yellow starthistle (E^+ = -0.5763). There was a trend that summer burns had a stronger effect on yellow starthistle than on annual grasses (df = 1, p = 0.09).

Fall burning had different effects among phanerophytes, annual grasses, and *B. inermis*, the perennial grass. Fall burning had the largest effect on phanerophytes (french and scotch broom) (E^+ = -1.2042, n= 18) and the least effect on annual grasses (E^+ = -0.4353, n = 11). It should be noted that all of the phanerophyte records/studies involved fall burning. Fall burns also had a very large negative effect on *B. inermis* (E^+ = -1.1853, n=5). However, within group heterogeneity was significant, and total heterogeneity (p = 0.03) suggests that other variables should be considered. No seasonal burn comparisons could be made for yellow starthistle, the only forb in the database, because it was only burned in the summer when it is in fruit.

Effects of geographic location: The only species in the database for which we could examine effects of geographic location were *B. tectorum* and *B. inermis*. Eight studies on *B. tectorum* were conducted in Sierra Nevada, Utah, or Oregon and Idaho. The Oregon and Idaho studies were grouped because of generally similar weather patterns between northeastern Oregon and western Idaho. There were no significant differences either within or between groups, based on place of study. The mean effect size was very low (E^{++} = -0.3427, n = 23). A study on *B. inermis* in northern Arizona was compared to one in S. Dakota and total heterogeneity was significant (Q_T = 16.80, df = 8, p = 0.03), suggesting that other factors may be playing a role in responses. In fact, a drought occurred in northern Arizona during the study on *B. inermis* in 2002, which may have led to a large decrease in the control plots, and lessened the effect of light burning in the treatment plots (Levine, pers. comm.). The two studies did not have comparable post-year measures (less than one year vs. one year), so this may also explain some of the variation. Rosenthal's fail safe number was 13.0. The total number of records was 6, so the results may not be reliable.

Discussion

Our meta-analysis indicates that prescribed fire has a generally large negative effect on invasive plants, and that this is a true estimate of the measured effect because the confidence interval does not overlap zero. However, the large variation among studies limits any conclusion that all studies share a common effect size. Also it should be emphasized that the measured effects are short term because most studies monitored only first or second year post-fire responses.

The most important predictor variable in this meta-analysis was the season of burn. However, season of burn and life form may co-vary in the analysis because annual grasses were the only life form burned in the late spring and this time period had the largest negative effect size. Similarly, fire had the greatest negative effect on woody species that were burned only in the fall. The tamarisk study did not include season of burn and was not included in the analysis.

Effect sizes for burning annual grasses one time varied considerably, with the least effect on *B. tectorum* and the greatest effect on *B. rubens*, although differences were not significant. The variation in effect sizes among annual grasses is not surprising. Winter annual species such as *B. rubens* and *B. japonicus* can mature from spring through early summer. With this range of phenological variation, it may be difficult to time fires so that seeds on the plants are in the most susceptible developmental stage to burning (e.g., not too green, but not yet dispersed on the ground).

The impact of cheatgrass invasions in the West has been much discussed (Mack 1981, Mack 1986, D'Antonio and Vitousek 1992) and experiments to control its abundance are underway. The Joint Fire Sciences Program has funded two proposals that involve the use of fire to restore sagebrush communities influenced by cheatgrass (see proposals by Brooks 2001, McIver et al. 2005 on JFSP web site). Despite a plethora of literature on this species, including 497 references listed in the FEIS database, very few studies contained quantitative data on cheatgrass responses to fire and so few met the criteria for inclusion in a meta-analysis. Nonetheless, those 23 studies that did meet meta-analysis criteria suggest little to no negative effects of fire on cheatgrass. There were only two records for spring burning effects on cheatgrass in the database and the majority of records in our database documented the effects of summer and late summer burns (n = 13). Although the result was not reliable due to a low fail safe number (zero), spring burning had the greatest effect on *B. tectorum*. This is the time when the new seed crop is on the plants, supporting the idea that burning before seed dispersal may be an effective strategy for reducing the active seed bank. Nonetheless, the available studies suggest fire is not likely to be a reliable tool for reducing cheatgrass.

The perennial cool season grass, *B. inermis*, was susceptible to both spring and fall prescribed burns in two studies (Blankespoor and Larson 1994, Levine, pers. comm.), but fall burning had the largest effect size overall ($E^{+} = -1.1511$ vs. $E^{+} = -0.3151$). Late spring burning to control *B. inermis* has often been used in conjunction with efforts to control other exotic cool season species, and has been only marginally effective (Blankespoor and Larson 1994). However, Willson (1990) found that burning smooth brome later in the spring (mid- to late-May), when tillers have elongated and are exposed above ground, significantly reduced tiller density later in the year.

Among the life form groups we evaluated, fire had the greatest effect on woody species. Studies of french broom suggest that multiple years of burning are important for reducing broom density because fire produces high seedling densities from a persistent seed bank (Odion and Haubensak 1999, Alexander and D'Antonio 2003b). However, a single fire can effectively deplete the seed bank and repeated burning does not reduce the seed bank beyond the density observed after one fire (Alexander and D'Antonio 2003). Stand age may play a role in these results, and it may be easier to eliminate the seed bank in young stands where the seed bank can be more rapidly depleted (Odion and Haubensak 1999). However, Alexander and D'Antonio (2003) found no strong relationship between stand age and seed bank size.

In contrast to the brooms, tamarisk recovers from fire largely by resprouting (Busch 1995). Although the effects of fire on tamarisk foliar cover and biomass removal were evident from 1 to 11 years post-burn (Harms, pers. comm.), the ability to resprout suggests that additional control methods are needed.

The only forb species for which we had data was yellow starthistle, an annual forb species, and the majority of data came from a single study (Kyser and DiTomaso 2002). Although this study was excellent because of the data collected on multiple burns, post-burn monitoring (four years), and native species, we could not use all of the data from a single study, and there were not enough studies on this species for the meta-analysis. Searches for data on fire effects for other forb species, particularly perennial forbs, were unsuccessful. There was a medium effect size for summer burning ($E^+ = -0.5763$, n = 6) that included two records from one other study (Gucker 2004). Seed longevity of yellow starthistle requires multiple burns to reduce the seed bank by 99%, but the effect is only temporary and the seed bank rebounds after a few years (Kyser and DiTomaso 2002). Hence, the need for repeated burning is clear.

The lack of studies on the use of fire to control perennial forbs is notable. Although we reviewed the literature for spotted knapweed (a perennial forb), no studies presented data required for meta-analysis. Fire can kill spotted knapweed above ground, but the taproot below ground can survive and resprout and the seed bank survives low severity fires (Lacey et al. 1992). Temperatures above 200^{o} C. for a period of two minutes reduce knapweed seed germination (Abella and MacDonald 2000). Although the responses of spotted knapweed to fire are variable, it seems likely that the below ground perennating buds reduce the effectiveness of fire in controlling knapweed infestations.

Challenges in creating decision tools for managers

The two greatest obstacles we encountered to developing a decision tool for using prescribed fires to control invasive plants were the lack of studies that provided essential response criteria (means, variances, and sample sizes between control and treatment groups) and the unique methodologies of many studies, including manipulation and characteristization of fire treatments. Even when we expanded our search to include a broad range of species, studies that had the essential criteria for meta-analysis were sparse. A third important consideration was the lack of studies that included data on the responses of native vegetation to the burning treatment. Such data are important because the native species are the ones that are presumably more desirable and thus treatments that both reduce the invaders while enhancing the natives would be the most valued.

Fire characteristics are extremely variable and are difficult to precisely measure or control during prescribed fire. When variable fires interact with heterogeneous landscapes and different land-use histories, even studies on the same plant species may yield different results. Additional sources of variation come with different experimental designs and methods. Added or altered fuel treatments, pre and post-fire grazing regimes, time of day, and the use of backing fires instead of head fires, also affect plant responses to burning (Vogl 1974, DiTomaso et al. 2005). A better understanding of the influence of these factors will improve our ability to predict desired outcomes when using prescribed fire as a management tool.

Prescribed burning to control or eliminate exotic, invasive plants has met with limited success, particularly when used to treat annuals that are favored by disturbance. In forest systems, where one restoration goal has been to return to historical fire regimes, the presence of exotic species creates a new obstacle. These exotic species that were not historically a part of the landscape may make it difficult to choose between more frequent fires or a fire frequency that favors native species for successful restoration (Keeley 2005). In addition, given the constraints of planning prescribed burns both legally and logistically, it may not be feasible to carry out frequent or multiple burns that might control some species (DiTomaso et al. 2005). Because of the complex nature of the interrelationships between fire and plants, it may be difficult to account for all of the variability in any one system.

We understand the intrinsic bias in a database developed from a small set of studies and unpublished data, but synthesizing results across multiple studies can lead to a stronger decision tool than relying on the results of individual studies. As more studies are added to the database, and more data on native vegetation are collected, the effects of prescribed burning will be better quantified and we will be better able to pinpoint the sources of context specific responses to this management tool. Information on timing, climate, and fuel treatment that is collected in a consistent manner across studies will be extremely useful to understanding the causes of variation in outcome and will enhance the usefulness of tools such as meta-analysis. Monitoring beyond one or two years post-burn treatment will also contribute to understanding the long-term consequences of using prescribed burn treatments.

While our analysis overall suggests that there are many circumstances where fire is indeed a useful tool for reducing invasive species, for the present, we cannot yet provide a detailed quantitatively based decision tool for managers. Instead, managers must continue to decide what information about prescribed burning is most relevant to their particular situation. We hope that our data base will at least provide an overview of the most rigorous studies we could find on fire impacts on this selected group of invasive plant species in the western United States.

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