Exotic Plant Species Diversity: Influence of Roads and Prescribed Fire in Arizona Ponderosa Pine Forests

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Abstract

Many studies have investigated the ecological effects of roads and roadsides as both habitat and dispersal corridors for exotic plant species. Several of these compared roadside exotic species richness and abundance with adjacent interior habitats, but we found no studies of individual exotic species' abundance between the two habitats in the context of prescribed fire. We measured exotic species richness and individual species' abundance along roadsides and in adjacent interior habitat (> 150 m) before and after prescribed fire at three ponderosa pine (*Pinus ponderosa* Douglas ex Lawson & C. Lawson) sites in northern Arizona. Eighteen of the 20 exotic plant species found in this study have been and continue to be intentionally introduced or are known agricultural seed contaminants. Roadsides had significantly higher exotic species richness than adjacent forest interior habitats, but only one site showed a significant (decreasing) fire effect on species richness. Four exotic plant species had significantly higher densities along roadsides at two of the three sites, and four species had no significant difference in abundance between habitats at any site supporting an individualistic species response hypothesis. Most exotic species showed no significant change in density post-prescribed fire suggesting that low-intensity prescribed fire may have minimal effect on exotic species diversity. Variability in total exotic species richness, composition, species' constancies, and species' densities between the three regionally similar sites suggests differing degrees and effectiveness of past management practices and policies such as intentional seeding.

Resumen

Muchos estudios han investigado los efectos ecológicos de las carreteras y los bordes de las carreteras como hábitats y corredores de dispersión para especies de plantas exóticas. Varios de estos compararon la riqueza y la abundancia de especies exóticas de los bordes de las carreteras con los hábitats interiores cercanos, pero nosotros no encontramos estudios de la abundancia individual de especies exóticas entre dos hábitats en el contexto de fuego prescrito. Nosotros medimos la riqueza de especies exóticas y la abundancia individual de especies a lo largo de los bordes de las carreteras y en un hábitat interior cercano (> 150 m) antes y después de fuegos prescritos en tres localidades de *Pinus ponderosa* Douglas ex Lawson & C. Lawson (pino ponderosa) en el norte de Arizona. Dieciocho de veinte especies de plantas exóticas halladas en este estudio han sido y continúan siendo introducidas intencionalmente o son conocidas como contaminantes de semillas agrícolas. Los bordes de las carreteras tienen riquezas de plantas exóticas significativamente más alta que los hábitats interiores cercanos del bosque, pero solo un sitio mostró un efecto significativo (reducción) al fuego en la riqueza de especies. Cuatro especies de plantas exóticas tuvieron densidades significativamente más alta a lo largo de los bordes de las carreteras en dos de los tres sitios, y cuatro especies no tuvieron diferencia significativa en la abundancia entre hábitats en cualquier sitio apoyando una hipótesis de respuesta individual de especies. Muchas especies exóticas no mostraron cambio significativo en densidad posterior a los fuegos prescritos, sugiriendo que los fuegos prescritos de baja intensidad deberían tener un efecto mínimo sobre la diversidad de especies exóticas. La variabilidad en la riqueza total de especies exóticas, la composición, las continuidades de especies y las densidades de especies entre los tres sitios regionalmente similares discrepan en grados y efectividad de las prácticas pasadas y políticas de manejo tales como la plantación programada.

Key Words: exotic species abundance, exotic species richness, intentional introduction, ponderosa pine, prescribed fire, seeding

INTRODUCTION

A number of studies in recent decades have examined factors contributing to successful colonization of exotic species in native habitats. Roads are often cited as contributing to the presence of exotic species. However, an understanding of exotic species' population dynamics along roadsides and in adjacent native habitats is often complicated by management practices such as prescribed burning. Studies attempting to account for both the role of habitat and prescribed burning on exotic species populations are lacking.

Recent literature reviews on the ecological effects of roads (Spellerberg 1998; Trombulak and Frissell 2000) focused on both the biotic and physical impacts of roads worldwide with particular emphasis on animal mortality, roadside soils, and exotic plant species. Roadsides provide both habitat and

Funding was provided by Rocky Mountain Research Station Research Work Units 4156 and 4251, the Joint Fire Science Program (01-1-3-25), and the National Fire Plan (01.RMRS.B.4).

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Manuscript received 18 June 2007; manuscript accepted 4 January 2008.

potential dispersal corridors for exotic plant species (Amor and Stevens 1975; Tyser and Worley 1992; Greenburg et al. 1997; Gelbard and Belnap 2003; Pauchard and Alaback 2006). Several studies have provided evidence of vehicles as accidental dispersal agents for exotic species along roads and railroads (Clifford 1959; Mühlenbach 1979; Schmidt 1989; Lonsdale and Lane 1994). Exotic plant species have been intentionally introduced via seed mixes planted to prevent soil erosion along roadsides (e.g., Tyser and Worley 1992; Parendes and Jones 2000) and postwildfire (e.g., Robichaud et al. 2000; Hunter et al. 2006) as well as encouraged by past range reseeding policy to enhance forage production in the more open ponderosa pine (*Pinus ponderosa* Douglas ex Lawson & C. Lawson) forests of the Southwest (Lavin and Springfield 1955). Several studies have focused on exotic plant species richness, distribution, and abundance along roadsides (e.g., Amor and Stevens 1975; Tyser and Worley 1992; Gelbard and Belnap 2003; Johnston and Johnston 2004).

There is a common perception that exotic species populations are enhanced by fire, but this perception is based largely on short-term studies of severely burned sites, such as wildfires (Crawford et al. 2001; Keeley et al. 2003; Hunter et al. 2006) or slash-pile burns (Korb et al. 2004). Milberg and Lamont (1995) have shown that populations of roadside exotic species increased after prescribed burning as well and suggested that seed availability and loss of sclerophyll woodland cover may be important drivers of postfire weed invasion. Kerns et al. (2006) also suggested that loss of overstory after prescribed fire is linked to increases in exotic species abundance. In spite of the tendency to group all exotic species together for abundance estimates (Milberg and Lamont 1995; Griffis et al. 2001; Keeley et al. 2003; Hunter et al. 2006; Kerns et al. 2006), inferring a similar response to changes in the abiotic or biotic environment for all exotic species, an individualistic species response (Gleason 1917, 1926) may be more insightful (e.g., Greenburg et al. 1997; Johnston and Johnston 2004). We hypothesized that at a regional scale, exotic species richness would be higher in roadside habitats, that low-intensity prescribed burns would have minimal effect on exotic species populations, and that exotic species' numerical response to prescribed fire and to habitat would vary by species. We quantified preburn and postburn exotic species richness, constancy, and density of individual exotic species in roadside and adjacent native habitats in order to determine the effects of prescribed fire within each habitat.

METHODS

Study Design and Field Methods

The study was conducted on three forested sites at 2 100–2 250 m elevation within ponderosa pine forests of northern Arizona, one each in the Coconino National Forest (CNF), Kaibab National Forest (KNF), and Apache–Sitgreaves National Forest (ASNF; Fig. 1). The forest understory and nearby meadows were dominated by perennial, mostly warm-season grasses, e.g., mountain muhly (*Muhlenbergia montana* [Nutt.] Hitchc.), blue grama (*Bouteloua gracilis* [Kunth] Lag. ex Griffiths), and Arizona fescue (*Festuca arizonica* Vasey), that show growth and flowering responses to monsoonal summer rains in July and August. Sites were sampled before monsoon rains in 2003 (preburn) and 2004 (postburn) to focus on cool-season exotics.

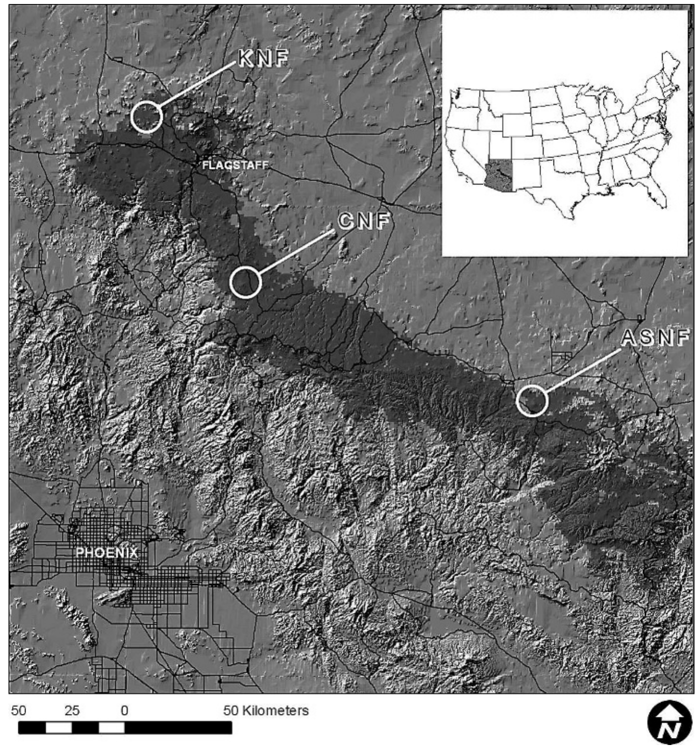

Figure 1. Study area map showing the locations of the three study sites in the ponderosa pine–dominated zone (dark shade) of the Mogollon Plateau, northern Arizona. KNF indicates Kaibab National Forest; CNF, Coconino National Forest; and ASNF, Apache–Sitgreaves National Forest.

In addition to understory exotic plant community structure, each study site was established to examine the response of forest and bird communities to large-scale prescribed fire experiments (see Saab et al. 2006, 2007, 2008). On each site, a 247–405-ha prescribed fire treatment (hereafter, burn) unit was paired with one (CNF) or two (ASNF, KNF) nearby (1–5 km) control (hereafter, no-burn) units of similar extent and with similar biota, geography, and climate. District fire personnel with the US Forest Service (USFS) were responsible for defining the extent and perimeter of the burned units. The geographic information system (GIS) ArcView® 3.2a (Environmental Sciences Research Institute 2000) and US Geological Survey digital orthophoto quads (1997–2000) were used to locate suitable areas for the placement of no-burn units on the landscape. Within the GIS, a randomly placed 125 × 125 m grid was used to systematically identify potential sampling points on each forest interior site when they met the following criteria: 1) points were ≥ 150 m from the unit edge, 2) a sampling point could be placed ≥ 250 m from its nearest neighboring point, and 3) points were stratified by canopy cover to ensure we sampled the proportions of forest canopy cover present in the burn–no-burn pairs (Table 1). We calculated average overstory canopy cover within a 100-m radius of sampling points using a GIS and methods described by Xu et al. (2006). A global positioning system was used to locate and permanently mark all sampling points on the ground and improved positional accuracy by differentially correcting the

Table 1. Total number of sampling point locations and average forest canopy cover values (SD) for burned and unburned units located on three sites in northern Arizona.

Site[1]	Forest interior N	Mean % canopy cover (SD)	Roadside N	Mean % canopy cover (SD)
KNF				
Burn	40	39.4 (19.4)	20	50.7 (10.3)
No-burn	40	46.7 (15.5)	20	56.4 (18.5)
Total	80	—	40	—
CNF				
Burn	40	51.5 (6.5)	11	42.0 (6.1)
No-burn	40	42.4 (7.2)	10	45.3 (4.3)
Total	80		21	
ASNF				
Burn	29	46.0 (7.0)	16	53.0 (9.5)
No-burn	29	54.6 (6.8)	14	55.1 (5.5)
Total	58	—	30	—
Grand total	218	—	91	—

[1]KNF indicates Kaibab National Forest; CNF, Coconino National Forest; and ASNF, Apache–Sitgreaves National Forest.

coordinates for each point using a local base station. One forest interior point on the KNF site was lost during the study because of debris flow.

From each forest interior sample point, a random azimuth was chosen to establish an 80-m transect centered at the 40-m mark with five subsamples at 20-m intervals (0, 20, 40, 60, and 80 m). T-square sampling (Diggle 1975; Krebs 1989; Engeman et al. 1994) was conducted for each exotic vascular plant species within a 5-m constraint on both the point-to-organism and the nearest-neighbor distances.

Roadside sampling points were established where the grid-lines intersected unpaved forest roads. A random starting point for an 80-m transect parallel to the roadside was chosen within 10–30 m from the line–road intersection. The 80-m transect was offset 5 m away from the edge of the road, defined as the vegetated edge or the center of the road ditch if one was present. Roadside transect sample sizes were constrained by the limited occurrence of roads at the CNF and ASNF sites and by the need to keep sampling intensity along the roadsides reasonably similar to the larger area of forest interior.

Density calculations for each exotic species present were based on modification of the T-square basic (TSBA) formula in Engeman et al. (1994) to allow for legitimate zeros (no plants in an area) within a 5-m search constraint (maximum radius of search area from subsample point or organism). For roadside sampling, it was possible for the nearest-neighbor half-circle sampling area to include part of the roadbed; if this occurred for a species, data were noted as edge plots and were given one half the normal weight of a nearest-neighbor, half-circle sample for that species. The density estimator for our modified TSBA formula was:

$$\mathrm{TSBA_{mod}} = (N_1 + N_2) / \left(\pi \sum \mathrm{R}^2_{(1)\mathrm{i}} + \pi \sum T^2_{(\mathrm{full})i} + 0.025\pi \sum T^2_{(\mathrm{edge})i}\right) \quad [1]$$

where N_1 = number of individuals found in point-to-organism radii with a 5-m maximum distance, N_2 = number of individuals found in nearest-neighbor radii with a 5-m maximum distance, $\mathrm{R}_{(1)\mathrm{i}}$ = point-to-organism radii or 5 for legitimate empty plots, $T_{(\mathrm{full})i}$ = nearest-neighbor radii for nonedge plots or 5 for legitimate empty plots, and $T_{(\mathrm{edge})i}$ = nearest-neighbor radii for edge plots or 5 for legitimate empty plots.

Individual plants were delineated by culms for perennial rhizotomous grasses and by single stems or center-of-the-rosette for forbs, annual grasses, and bunch grasses. Exotic plant species richness for each sample point was the number of species occurring at least once among the five subsamples.

Dormant-season prescribed fire treatments were applied in 2003 and 2004 by USFS fire crews in each national forest (Dickson 2006; Saab et al. 2006). Within 30 d postfire, we measured fire severity at each sample point in the burned units using visual estimates of ground char (Ryan and Noste 1985) within each of the 5-m radius T-square point-to-organism subsamples. Criteria for each ground-char class (unburned, light, moderate, deep) are given in Ryan and Noste (1985). The proportion of the surface area covered by each ground-char class was visually estimated using cover classes (Daubenmire 1959). Midpoints of these cover classes were used to calculate the percentage of each ground-char class for each sample point. To get a single value, ground char for each sample point was then calculated as a weighted average of light char (1x), moderate char (2x), and deep char (3x). This value has a possible range of 0–3 with higher values representing potentially stronger effects.

Voucher specimens for each exotic plant species are deposited at the USFS Herbarium at the Rocky Mountain Research Station Flagstaff Laboratory. Species circumscription and nomenclature follow Flora North America (2003–2007) and the Intermountain Flora (Cronquist et al. 1984, 1989, 1997).

Statistical Modeling and Analysis

Point-level exotic species richness and T-square density estimates for each species and each year were analyzed using multiple response permutation procedures (Mielke and Berry 2001) with an Excel® macro. Change-in-richness (2004 − 2003) data were analyzed by Proc GLM analysis of variance in SAS/STAT 9.1® (SAS/STAT 2003) for unbalanced designs. The relationship between ground char and the preburn vs. postburn change in exotic species richness and density was modeled with Proc REG linear regression in SAS/STAT 9.1®. The significance level for all statistical analyses was $\alpha = 0.05$.

RESULTS

Species Richness

A total of 20 exotic vascular plant species were found within the sample point transects at the three sites, and 18 of those have been intentionally introduced in North America (Table 2). *Salsola tragus* L. was found in only one plot at one site, but 11 species were found in multiple plots at all three sites (Table 2). Nine other exotic plant species were found on these sites outside the sample points.

Roadside habitats were richer in exotic species at all three sites (Table 3). CNF had higher median richness (Table 3) and

Table 2. Exotic vascular plant species found within forest interior (I) and roadside (R) samples at the Coconino National Forest (CNF), Kaibab National Forest (KNF), and Apache–Sitgreaves National Forest (ASNF) sites; total richness = 20 species. Numbers list citations for intentional introductions.

Family and species	CNF	KNF	ASNF	Intentional introductions[1]
Poaceae				
Agropyron cristatum	I, R	I, R		1, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 20
Bromus inermis	I, R	I, R	I, R	1, 3, 4, 5, 6, 7, 13, 14, 20
Bromus japonicus	I, R	R	—	—
Bromus tectorum	I, R	I, R	I, R	15, 18,[2] 22, 23, 25[2]
Dactylis glomerata	I, R	I, R	I, R	3, 4, 5, 6, 7, 10, 12, 13, 14, 20, 23
Phleum pratensis	R	—	—	6, 10, 13, 14, 20
Poa pratensis	I, R	I, R	I, R	1, 6, 7, 10, 20
Thinopyrum intermedium	I, R	I, R	R	2, 3, 4, 5, 7, 8, 11, 12, 13, 14, 20
Asteraceae				
Cirsium vulgare	I, R	I, R	I, R	16[3]
Lactuca serriola	I, R	I, R	I, R	—
Taraxacum officinale	I, R	I, R	I, R	16, 19, 23
Tragopogon dubius	I, R	I, R	—	24
Fabaceae				
Medicago lupulina	I, R	I, R	I, R	16,[3] 20, 23
Medicago sativa	I, R	—	R	5, 13, 14, 20
Melilotus officinalis	I, R	—	R	5, 11, 12, 13, 14, 20
Chenopodiaceae				
Salsola tragus	—	—	R	16[3]
Convolvulaceae				
Convolvulus arvensis	—	I, R	—	16,[2] 23
Geraniaceae				
Erodium cicutarium	I, R	I, R	—	1, 5, 23
Polygonaceae				
Polygonum aviculare	I, R	I, R	R	16[3]
Scrophulariaceae				
Verbascum thapsus	I, R	I, R	I, R	16,[2] 17, 19, 21, 23
Total site richness	19	16	14	

[1] 1, Parker and McGinnies 1940; 2, Lavin 1953a; 3, Lavin 1953b; 4, Reynolds et al. 1949; 5, Doran 1951; 6, Doran 1952; 7, Rich 1962; 8, Malechek 1966; 9, Springfield 1956; 10, Johnson and Hull 1949; 11, Pearson 1968; 12, Springfield 1965; 13, Lavin and Springfield 1955; 14, Anonymous 1973; 15, Cook and Dias 2006; 16, Mack and Erneberg 2002; 17, Gross and Werner 1978; 18, Mack 1981; 19, Mack 2003; 20, Anonymous 1948; 21, Mack and Lonsdale 2001; 22, Anonymous 1932, 1933, 1938; 23, Mack 1991; 24, Clements et al. 1999; 25, Upadhyaya et al. 1986.

[2] Both intentional and seed contamination.

[3] Seed contaminant.

total richness (Table 2) for both habitats. CNF was followed by KNF and ASNF in that order of decreasing exotic species richness.

One-year change in richness by sample point, 2004 minus 2003, was significant only for the burn/no-burn treatments ($F_{1,114} = 12.58$, $P = 0.0006$) at KNF. The burn treatment at KNF lost an average of 0.915 (SE $\pm$ 0.159) species $\cdot$ point^{-1} whereas the no-burn treatment lost an average of 0.051 (SE $\pm$ 0.173) species $\cdot$ point^{-1} from 2003 to 2004. Weighted ground-char values on the burned treatment did not explain a significant amount of that loss ($R^2 = 0.04$, regression slope of 0, $P = 0.1212$). Twelve of the 13 exotic species present before and after the burn treatment at KNF were the same with one unique species both preburn and postburn. Change in richness was not significantly different between burn and no-burn treatments, between roadsides and forest interior habitats, or for their interaction term at CNF ($F_{3,96} = 2.17$, $P = 0.0966$) or at ASNF ($F_{3,84} = 2.33$, $P = 0.0799$). It was also not significant between roadside and interior habitat or the interaction term at KNF.

Table 3. Median exotic species richness per sample point (quartile range) on roadside and forest interior habitats at Coconino National Forest (CNF), Kaibab National Forest (KNF), and Apache–Sitgreaves National Forest (ASNF) and the probability value (multiple response permutation procedures) of habitat richness differences being due to chance alone at each site.

Site	Roadside	Forest interior	Probability
CNF species $\cdot$ point^{-1}	10.0 (7.0–11.0)	4.0 (2.5–6.0)	< 0.0001
KNF species $\cdot$ point^{-1}	3.0 (1.5–5.0)	2.0 (1.0–3.0)	0.0078
ASNF species $\cdot$ point^{-1}	1.5 (0.0–2.0)	0.0 (0.0–1.0)	0.0043

Species Abundance and Occurrence

For 10 species found on multiple plots at all three sites with > 0.25 constancy on at least one habitat $\cdot$ site^{-1} (Table 2), comparisons of density estimates were made between habitats at

Table 4. Comparisons of 2003 exotic species densities (no. · ha^{-1}) and constancy by habitat and site. Note that a species may be present (constancy > 0) but have a median density $= 0$.[1]

Species	ASNF R	ASNF I	CNF R	CNF I	KNF R	KNF I
Thinopyrum intermedium						
Median density	0[2]	a	924[2]	0	0[2]	0
25th–75th density percentiles	0–0		473–1 858	0–0	0–63	0–0
Constancy	0.2	.00	0.9	0.06	0.25	0.03
Medicago lupulina						
Median density	0[2]	0	602[2]	0	0	0
25th–75th density percentiles	0–305	0–0	391–943	0–0	0–0	0–0
Constancy	0.27	0.02	0.9	0.16	0.03	0.03
Polygonum aviculare						
Median density	0	a	0[2]	a	0[2]	0
25th–75th density percentiles	0–0		0–0		0–365	0–0
Constancy	0.03	0.00	0.19	0.00	0.35	0.03
Bromus inermis						
Median density	0	0	0[2]	0	0[2]	0
25th–75th density percentiles	0–0	0–0	0–719	0–0	0–398	0–0
Constancy	0.03	0.09	0.38	0.06	0.48	0.06
Dactlyis glomerata						
Median density	0	0	393[2]	0	0	0
25th–75th density percentiles	0–0	0–0	0–1 174	0–0	0–0	0–0
Constancy	0.07	0.05	0.62	0.23	0.1	0.04
Taraxacum officinale						
Median density	0	0	248[2]	172	0	0
25th–75th density percentiles	0–0	0–0	110–697	0–506	0–153	0–191
Constancy	0.07	0.05	0.76	0.55	0.33	0.33
Poa pratensis						
Median density	0	0	894	0	0	0
25th–75th density percentiles	0–538	0–0	341–2 109	0–954	0–0	0–0
Constancy	0.33	0.21	0.86	0.44	0.03	0.05
Bromus tectorum						
Median density	0	0	1 114	599	0	0
25th–75th density percentiles	0–0	0–0	0–3 641	0–1 591	0–288	0–0
Constancy	0.03	0.03	0.71	0.66	0.38	0.1
Lactuca serriola						
Median density	0	0	100	0	0	0
25th–75th density percentiles	0–0	0–0	0–594	0–107	0–0	0–0
Constancy	0.07	0.02	0.53	0.26	0.21	0.21
Verbascum thapsus						
Median density	0	0	120	153	0	0
25th–75th density percentiles	0–0	0–135	0–205	0–487	0–217	0–243
Constancy	0.07	0.26	0.57	0.54	0.35	0.42
Cirsium vulgare						
Median density	a	0	0	0	0	0
25th–75th density percentiles		0–0	0–0	0–0	0–0	0–0
Constancy	0.00	0.02	0.05	0.04	0.05	0.03

[1]ASNF indicates Apache–Sitgreaves National Forest; CNF, Coconino National Forest; KNF, Kaibab National Forest; R, roadside; I, interior, and a, absent.
[2]Significantly higher-density habitat comparison within site (multiple response permutation procedures).

each site prior to prescribed fire. Intermediate wheatgrass (*Thinopyrum intermedium* [Host] Barkworth & D.R. Dewey) was significantly more dense and had higher constancy on roadside habitats than in the ponderosa pine understory at all three sites (Table 4). Black medick (*Medicago lupulina* L.), common knotweed (*Polygonum aviculare* L.), and smooth brome (*Bromus inermis* Leyss.) were more dense and with higher constancy on roadside habitats at two of the three sites, but note that *P. aviculare* was absent from the interior at ASNF. Two additional species were significantly more abundant and had higher constancy along roads at CNF: orchardgrass (*Dactylis glomerata* L.) and common dandelion (*Taraxacum*

Table 5. Changes in density (no. · ha^{-1}) and number of sample points occupied in 2003 and 2004 at Apache–Sitgreaves National Forest and Kaibab National Forest by habitat and prescribed fire treatment.[1]

	Apache–Sitgreaves National Forest				Kaibab National Forest			
	Interior		Roadside		Interior		Roadside	
Species	Burn	No burn	Burn	No burn	Burn	No burn	Burn	No burn
Thinopyrum intermedium								
Change in median density [interquartile distance]	a	a	0 [0]	0 [99]	0 [0]	0 [0]	0 [0]	0 [0]
2003/2004 Number of points occupied			1/1	5/6	0/2	2/1	4/4	6/6
Medicago lupulina								
Change in median density [interquartile distance]	0 [0]	0 [0]	0 [0]	0 [9]	a	0 [0]	a	0 [0]
2003/2004 Number of points occupied	0/1	1/0	2/0	6/7		2/1		1/0
Polygonum aviculare								
Change in median density [interquartile distance]	a	a	a	0 [0]	a	0 [0]	0 [0]	t [27]
2003/2004 Number of points occupied				1/1		3/2	2/1	12/9
Bromus inermis								
Change in median density [interquartile distance]	a	0 [0]	a	0 [0]	0 [0]	0 [0]	0 [7]	0 [3]
2003/2004 Number of points occupied		5/7		1/1	4/4	1/1	12/11	7/7
Dactylis glomerata								
Change in median density [interquartile distance]	0 [0]	0 [0]	0 [0]	0 [0]	0 [0]	0 [0]	0 [0]	0 [0]
2003/2004 Number of points occupied	1/7	2/3	0/1	2/2	3/0	0/3	2/1	2/1
Taraxacum officinale								
Change in median density [interquartile distance]	0 [0]	0 [0]	0 [0]	0 [0]	0 [0]	0 [19]	0 [15]	0 [0]
2003/2004 Number of points occupied	0/1	3/3	0/3	2/1	10/4	17/5	10/4	3/4
Poa pratensis								
Change in median density [interquartile distance]	0 [30]	0 [43]	0 [61]	0 [36]	a	0 [0]	0 [0]	0 [18]
2003/2004 Number of points occupied	7/10	5/13	5/8	5/5		4/5	1/0	0/2
Bromus tectorum								
Change in median density [interquartile distance]	a	0 [0]	0 [0]	0 [0]	0 [0]	0 [18]	0 [0]	0 [37]
2003/2004 Number of points occupied		2/4	0/1	1/1	3/4	6/16	5/2	10/13
Lactuca serriola								
Change in median density [interquartile distance]	0 [0]	a	0 [0]	a	0 [0]	0 [0]	0 [0]	0 [6]
2003/2004 Number of points occupied	1/2		2/2		6/1	11/1	2/1	6/1
Verbascum thapsis								
Change in median density [interquartile distance]	0 [0]	0 [13]	0 [15]	0 [1]	0 [0]	0 [36]	0 [0]	2 [26]
2003/2004 Number of points occupied	6/7	9/12	1/8	1/4	12/13	22/27	4/5	10/10

[1]a indicates absent from site; t, trace ($-0.5 > t < +0.5$).

officinale F. H. Wiggers). With the exception of *B. inermis* at ASNF, the above species also tended to occur more frequently (higher constancy; Table 4) along roadsides. Four species showed no significant differences in density estimates by habitat at any of the sites (Table 4): Kentucky bluegrass (*Poa pratensis* L.), prickly lettuce (*Lactuca serriola* L.), cheatgrass (*Bromus tectorum* L.), and mullein (*Verbascum thapsis* L.).

CNF had twice as many exotics species with significantly higher roadside densities than the other two sites (six vs. three), and had the highest density for four of these species (*T. intermedium, M. lupulina, D. glomerata,* and *T. officinale*; Table 4). *Thinopyrum intermedium* and *M. lupulina* are especially widespread along roadsides at CNF, both with a constancy of 0.90.

Fire Effects

No species at ASNF and KNF showed significant change in density or occurrence from pre- to post-prescribed fire (Table 5). However, with the exception of *V. thapsis* at both

Table 6. Changes in density (no. · ha^{-1}) and number of sample points occupied in 2003 and 2004 at Coconino National Forest by habitat and prescribed fire treatment.[1]

Species	Interior		Roadside	
	Burn	No burn	Burn	No burn
Thinopyrum intermedium				
Change in median density [interquartile distance]	0 [0]	0 [0]	6 [335]	5 [107]
2003/2004 Number of points occupied	4/3	1/2	11/11	8/9
Medicago lupulina				
Change in median density [interquartile distance]	0 [0]	0 [0]	7 [39]	10 [27]
2003/2004 Number of points occupied	9/9	4/6	10/10	9/10
Polygonum aviculare				
Change in median density [interquartile distance]	0 [0]	a	0 [0]	0 [0]
2003/2004 Number of points occupied	0/1		2/2	2/1
Bromus inermis				
Change in median density [interquartile distance]	0 [0]	a	0 [113]	0 [0]
2003/2004 Number of points occupied	5/7		8/10	0/1
Dactylis glomerata				
Change in median density [interquartile distance]	0 [10]	0 [0]	t [32]	0 [71]
2003/2004 Number of points occupied	15/15	3/5	10/10	3/6
Taraxacum officinale				
Change in median density [interquartile distance]	−2 [36]	0 [19]	0 [44]	−12 [538]
2003/2004 Number of points occupied	27/14	17/9	7/8	9/6
Poa pratensis				
Change in median density [interquartile distance]	35 [255]*	0 [38]1*	t [98]	25 [113]
2003/2004 Number of points occupied	15/29	20/22	9/8	9/10
Bromus tectorum				
Change in median density [interquartile distance]	0 [126]	0 [70]1	3 [87]1*	−35 [100]*
2003/2004 Number of points occupied	34/30	19/21	9/8	6/6
Lactuca serriola				
Change in median density [interquartile distance]	0 [13]	0 [0]	0 [19]	0 [26]
2003/2004 Number of points occupied	15/9	6/2	6/14	5/3
Verbascum thapsus				
Change in median density [interquartile distance]	0 [33]	0 [9]	4 [66]	0 [0]
2003/2004 Number of points occupied	30/31	13/18	9/9	3/3

[1]a indicates absent from site; t, trace ($-0.5 > t < +0.5$); 1, extreme outlier removed; and *, significantly different between treatments.

sites, *B. tectorum* and *T. officinale* at KNF, and *P. pratensis* at ASNF, occurrences were too few for meaningful statistical tests.

Most species at CNF also had little to no change in density or occurrence from pre- to post-prescribed fire (Table 6). However, in a comparison of burn to no-burn treatments for change in density from 2003 to 2004, two species had significant differences. *Poa pratensis* in the forest interior habitat had a significant increase in the burn treatment compared to the no-burn treatment as well as an increase in the number of sample points occupied and in the variability (interquartile distance) of the density estimates (Table 6). *Bromus tectorum* in the roadside habitat also had a slight increase in density compared to a significant decrease for the no-burn treatment between years. However, unlike *P. pratensis*, variability and number of sample points occupied showed no change (Table 6). The above change in density for *P. pratensis* and *B. tectorum* were not significantly explained by ground char ($R^2 = 0.0047$, regression slope of 0, $P = 0.6836$, and $R^2 = 0.0033$, regression slope of 0, $P = 0.8739$, respectively).

Table 7. Weighted means of ground char ± 1 SE. Ground char ratings are 0 (unburned), 1 (lightly burned), 2 (moderately burned), and 3 (severely burned).

Site[1]	Forest interior habitat	Roadside habitat
CNF	0.14 ± 0.01	0.14 ± 0.02
KNF	0.31 ± 0.02	0.28 ± 0.01
ASNF	0.22 ± 0.01	0.23 ± 0.02

[1]CNF indicates Coconino National Forest; KNF, Kaibab National Forest; and ASNF, Apache–Sitgreaves National Forest.

DISCUSSION

Our results concur with the results of seven other studies that show increased amounts of exotic species along roadsides when compared to adjacent native habitats. Five studies show increased exotic species richness (Western and Juvik 1983; Tyser and Worley 1992; Buckley et al. 2003; Watkins et al. 2003; Pauchard and Alaback 2006), three show increased frequency of occurrence for exotic species (Amor and Stevens 1975; Western and Juvik 1983; Tyser and Worley 1992), and two show increased canopy cover of exotic species (Johnston and Johnston 2004; Pauchard and Alaback 2006). Pauchard and Alaback (2006) suggested that this pattern may be the product of a long history of roadside disturbances and continuous propagule pressure. Roadside soils are disturbed during construction and maintenance, thus presenting opportunities for accidental plant dispersal by vehicles and human traffic (Clifford 1959; Schmidt 1989) and for intentional seeding.

Prescribed fire effects on species richness appear to be minimal at 1 yr postfire. The KNF site showed a small, but significant, 0.915 species · point^{-1} decrease in richness whereas the CNF and ASNF sites showed no change. Ground char was higher at KNF (Table 7) but had little explanatory power, so the decrease in richness at KNF cannot be attributed to differences in ground char. Overall, ground char was relatively light on all three sites. This may account for an opposing trend following wildfires in the western United States, which tend to produce an increase in exotic species richness (Crawford et al. 2001; Keeley et al. 2003; Kerns et al. 2006), possibly due to increased fire severity.

At the individual species level, eight of 10 exotic species with an adequate number of occurrences on at least one site (mostly at CNF; Tables 5 and 6) showed no significant change in density post–prescribed fire. At CNF, *B. tectroum* showed a significant increase in density and a stable occurrence pattern in the roadside habitat, but the pattern was not repeated on the interior at CNF or on either habitat at KNF. *Poa pratensis* had similar postfire density response, significantly higher on one habitat (interior) at one site (CNF) but not on the alternate habitat at that site or at the second site with an adequate number of occurrences (ASNF). Low-intensity prescribed fire in ponderosa pine understory may have little to no effect on exotic species richness, density, or occurrence. In the Australian sclerophyll woodland, Milberg and Lamont (1995) showed an increase in cover and constancy in weed species, including eight new annual species, post–prescribed fire for roadside plots. However, these plots in narrow strips of native vegetation were surrounded by annual pastures of grasses and subclovers (Milberg and Lamont 1995) contrary to our roadside plots, which were surrounded by native vegetation.

The role of management history in determining exotic species occurrence cannot be overlooked (Keeley et al. 2003; Huisinga et al. 2005). The differences in exotic species richness at the site level (Table 2), i.e., between the three national forests, may simply reflect different levels of seeding success by forest managers, the intensity at which they introduced exotic grasses and legumes, and the differences in seed contamination between seed sources. Eleven of the 20 exotic plant species we found are either cool-season grasses (eight Poaceae) or legumes (three Fabaceae; Table 2). Except for *Bromus japonicus* Thunb., these species have been intentionally introduced for soil erosion control (Lavin and Springfield 1955; Rich 1962), to improve forage production (Parker and McGinnies 1940; Johnson and Hull 1949; Reynolds et al. 1949; Lavin 1953a, 1953b; Lavin and Springfield 1955; Anonymous 1973), and to fill a perceived early season forage gap in the ponderosa pine zone of the southwestern United States (Parker and McGinnies 1940; Lavin 1953b; Malechek 1966). Many of the same species were also planted for improvement of elk (*Cervus elaphus*) forage (Pearson 1968; Kruse 1972), and continue to be used in seeding after wildfires in the Southwest (Wolfson and Sieg 2008). In addition, *M. lupulina, P. aviculare, V. thapsus, C. vulgare,* and *B. tectorum* are known seed contaminants (Novak and Mack 2001; Mack and Erneberg 2002).

Four cool-season grasses found in all of our sites (*T. intermedium, B. inermis, D. glomerata,* and *P. pratensis*; Table 4) were widely recommended for range reseeding in the Southwest (e.g., Parker and McGinnies 1940; Lavin and Springfield 1955; Anonymous 1973); three were more abundant and occurred more frequently at CNF. The greater abundance and higher constancy of *M. lupulina* at CNF may also reflect a more active seeding program by past forest managers. However, the low prostrate growth form of *M. lupulina* and *P. aviculare*, both of which had a significantly higher roadside abundance at two of the three sites (Table 4), may also indicate a roadside habitat preference.

Alternatively, increased exotic species diversity may be the result of higher livestock grazing pressure and thus increased seed dispersal by livestock into and within CNF compared to KNF and ASNF. However, even in these open range sites, grazing would not explain why roadside habitats have a more diverse exotics flora than the adjacent open forest interior. The greater exotic species diversity along roadsides is more easily explained by seeding or by vehicle and human traffic. Anecdotal observations during 2 yr of fieldwork at these sites would not support the hypothesis of greater vehicle traffic, human use, or livestock grazing at CNF.

MANAGEMENT IMPLICATIONS

Our study confirms the well-established pattern of higher exotic species richness in roadside habitats. However, not all exotic plant species are more abundant or occur more frequently in roadside habitats when compared to adjacent native habitats, thus supporting our hypothesis for species to have an individualistic numerical response to habitat.

The research results presented in this paper strongly suggest to managers that low-intensity prescribed burns, similar to the fuels-reduction fires in open ponderosa pine forests at our sites, may have minimal effect on exotic plant species diversity (richness, constancy, and abundance) in either roadside or forest interior habitats. However, this may not apply to higher-severity fires. Higher-intensity fire may also more strongly influence the numerical response of individual exotic species.

Our findings also suggest that past land management activities may explain site differences in exotic plant species diversity. Specifically, past attempts to increase forage production for livestock and wildlife by the intentional seeding of exotic forage grasses and legumes may also have introduced more invasive exotic species such as *Salsola tragus, Bromus tectorum,* and *Cirsium vulgare*. Current land managers need to consider the effects of possible seed contamination for any new seeding project.

ACKNOWLEDGMENTS

This work would not have been possible without the dedication of our field crew: Noah Barstatis, Brian Casavant, Matt Jedra, Barbara King, Scott McKenna, Andrea Meadows, John Pieper, Dale Rogers, Eric Tse, Amy Uhlenhopp, and Nina Wilson.

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