Using a diverse seed mix to establish native plants on a Sonoran Desert burn

| **Scott R Abella, John L Gunn, Mark L Daniels, Judith D Springer, and Susan E Nyoka**

ABSTRACT

Revegetating burns is a major challenge facing resource managers in the low- and unpredictable-precipitation deserts of the southwestern US. We monitored the effectiveness of using a diverse, 28-species seed mix for establishing native plants on a 1.5-ha (3.7-ac) burn in the northern Sonoran Desert. Our objective was to compare species performances, which we assessed by measuring species frequencies and cover on 5 sampling dates to capture variation during a 32-mo period following seeding. By 15 mo after seeding, desert senna (*Senna covesii* (Gray) Irwin & Barneby [Fabaceae]) established best, with a frequency of 91% (based on 22, 10-m^2 plots) and a relative cover of 19%. Four other seeded species also became established in ≥ 50% of plots by 32 mo after seeding. Several seeded species, including desert senna (which flowered only 7 wk after seeding) and purple threeawn (*Aristida purpurea* Nutt. [Poaceae]), were observed with seed heads during one or more sampling periods. Although precipitation was only 67% of normal for 21 mo following seeding and 71% of species established in < 10% of plots, we consider the seeding to have met short-term management objectives because of the subset of highly successful species. Our results also illustrate the caution that should be used when evaluating seeding success: conclusions would have differed if the diversity of the seed mix had not included the successful species, and longer term monitoring was needed to detect some species in the seed mix that did not establish until 32 mo after seeding.

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KEY WORDS

fire, revegetation, seeding, species selection, mulch, *Senna covesii*

NOMENCLATURE

USDA NRCS (2007)

Revegetating burned areas is a formidable challenge facing resource managers in arid lands of the southwestern US. Southwestern deserts, such as the Sonoran, are not generally thought to have a history of frequent burns. Multiple ignition sources combined with increased fuel loads (often resulting from invasion of exotic annual grasses and increased density of nonpalatable shrubs), however, have increased frequencies, sizes, and severities of wildfires (Schmid and Rogers 1988). Many long-lived native species in these deserts are not considered fire adapted (Brown and Minnich 1986). Natural revegetation of desert burns by native species may be slow or dominated by exotic annual grasses that perpetuate a frequent-fire regime (Cave and Patten 1984). For example, Guo (2004) found that species richness of native perennial plants continued to increase up to 60 y following protection from disturbance in the Sonoran Desert. This implies a long recovery time for these native perennial communities. In another example, Brown and Minnich (1986) reported little to no recovery of pre-burn native perennial communities and 7 times greater exotic grass cover than native perennial cover at 4 Sonoran Desert sites 3 to 5 y after fire. Resource managers may have several reasons for actively revegetating burns with native species: to provide competition with exotic species, minimize soil erosion and dust pollution, improve aesthetics, forestall further land degradation such as that caused by unauthorized off-road driving, and increase native species diversity (Roundy and Biedenbender 1995; Bean and others 2004).

Although not widely evaluated for its effectiveness or value in revegetating burns in southwestern deserts, seeding has long been assessed for revegetating other disturbed areas, such as abandoned farmland or depleted rangelands (Cox and others 1982). Many past seeding projects used cultivars or exotic species, however, including relatively recent examples of seeding the highly flammable and invasive exotics known as Mediterranean grass (*Schismus barbatus*) and red brome (*Bromus rubens*) (Jackson and others 1991; Grantz and others 1998) (see Table 1). The use of native species has been limited by a lack of available seeds and by findings that native desert species are difficult to establish (Judd and Judd 1976; Bainbridge and Virginia 1990; Banerjee and others 2006). Consequently, several syntheses have discouraged seeding as a revegetation option in southwestern deserts (Cox and others 1982; Monsen and others 2004; Bainbridge 2007). Yet, seeding (for example, by aircraft) may be one of only a few feasible options for reintroducing propagules to large desert burns covering thousands of hectares in remote or rugged terrain. Furthermore, many factors (other than climate) that resource managers can manipulate can influence the outcome of seeding projects (Glendening 1942; Jackson and others 1991). Species selection, seeding rate and timing, method of seeding (for example, bare seed versus pelleting), site preparation, and supporting treatments (such as, tilling, mulching, and grazing protection) all can influence seeding success (Montalvo and others 2002). These observations suggest that seeding has the potential for meeting revegetation objectives, given some combinations of favorable factors, but these combinations remain poorly understood.

Our study objective was to assess the factor of native species selection by monitoring the outcome of a 28-species operational seeding on a northern Sonoran Desert burn. We quantify seeding success by measuring species-specific frequencies using permanent monitoring plots sampled 5 times to capture intra- and inter-annual variability during 32 mo following seeding.

SITE AND BURN DESCRIPTION

This study occurred on a 1.5-ha (3.7-ac) burn (UTM, NAD83: 408300 m E, 3741370 m N, zone 12, 595 m [1952 ft] elevation) in the southeastern corner of the 928-ha (2293-ac) Cave Creek Regional Park, managed by the Maricopa County Parks and Recreation Division. This park is 5 km (3.1 mi) west of the town of Cave Creek and 15 km (9.3 mi) north of the city of Phoenix suburbs. The study site lies within the Arizona Upland subdivision of the Sonoran Desert (Brown 1994), which is characterized by a diverse vegetation structure including giant saguaro (*Carnegiea gigantea* (Engelm.) Britton & Rose [Cactaceae]), various small trees (for example, yellow paloverde [*Parkinsonia microphylla* Torr. (Fabaceae)]), cacti (for example, *Opuntia* Mill. [Cactaceae] and *Cylindropuntia* spp. [Cactaceae]), shrubs (such as creosote bush [*Larrea tridentata* (DC.) Coville (Zygophyllaceae)]), and annual and perennial herbaceous plants. At the closest long-term weather station, the Carefree station 8 km (5 mi) east of the study site but 176 m (577 ft) higher in elevation, precipitation has averaged 33 cm/y (13 in/y) for the past 46 years (Western Regional Climate Center, Reno, Nevada). Average temperatures range from a July high temperature of 39 °C (102 °F), to a January low temperature of 5 °C (41 °F). Except for the driest months of April to June, which receive only 7% of the total annual amount, precipitation is relatively evenly distributed throughout the year. Soils at the study site are mapped in the Carefree series and classified as fine, mixed, superactive, hyperthermic Vertic Haplargids (Camp 1986).

The human-caused burn started on private land on 1 June 2005, burned 1.5 ha (3.7 ac) of the park, and was contained the same day it started. Weather conditions at the time of the fire included an air temperature of 40 °C (104 °F), relative humidity of 8 to 10%, and > 30 km/h (18.6 m/h) wind speeds. As observed by JL Gunn, flame heights during the fire were 1 to 1.3 m (3.3 to 4.3 ft). Based on observations following the fire, most shrubs were top-killed or killed entirely, including creosote bush, triangle bur ragweed (*Ambrosia deltoidea*), and foothill paloverde (*Parkinsonia* L.) (Figure 1).

SEEDING PROCEDURES

To assess feasibility of direct seeding and to avoid impacts to soil and archeological resources, no site preparation was done before seeding. Seeding was performed as a hydroseeding directly on the burn surface on 31 July and 1 August 2005, about 60 d after the fire. The hydroseed slurry, applied by truck using a hose connected to a continuously blending tank, consisted of seeds, tackifier (ground *Plantago* spp.), and fertilizer (112 kg/ha [100 lb/ac] of 16N:20P_2O_5:0K_2O ammonium phosphate). A mulch was applied after seeding, consisting of 3900 kg/ha (3470

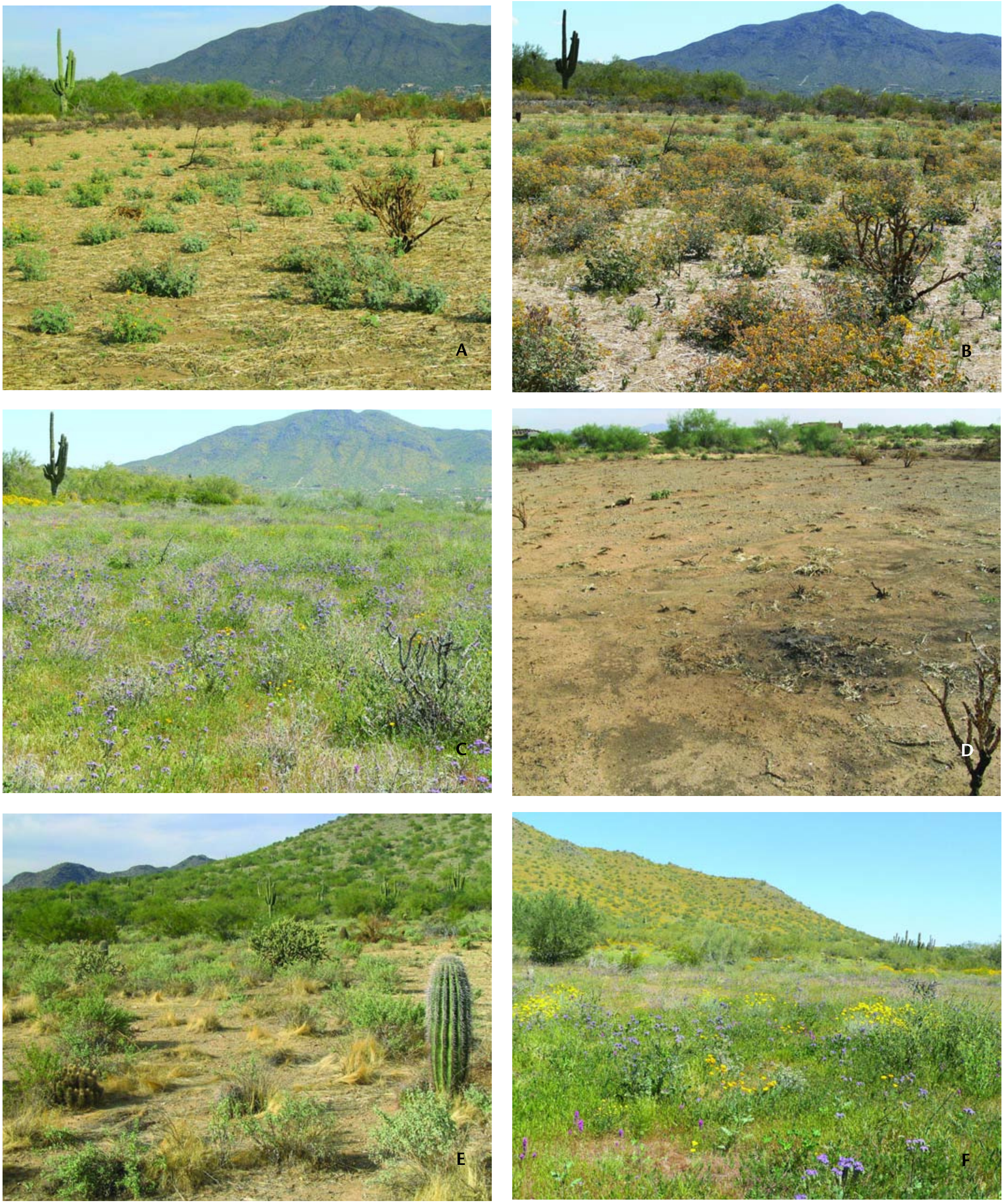

Figure 1. Seeded burn on the same point 3 (A), 9 (B), and 32 (C) mo after seeding (Nov 2005, May 2006, and Mar 2008, respectively). The major plant visible in the seeded area of photo A is desert senna. A non-seeded burn (D) and an adjacent non-burned area (E) are shown in November 2005 for comparison with photo A. Desert marigold (*Baileya multiradiata*) and desert bluebells flowering in the seeded burn in March 2008 (F).
Photos by Scott R Abella (a, d, e) and Mark L Daniels (b, c, f).

lb/ac) of straw stabilized by a top coat of 560 kg/ha (500 lb/ac) of wood fiber and 168 kg/ha (150 lb/ac) of tackifier.

A total of 28 species (Table 1) were in the seed mix, with each species seeded at rates of 0.3 to 7.5 kg/ha (0.27 to 6.7 lb/ac) pure live seed (based on tetrazolium tests). Species were chosen based on 4 main factors. First, all species were native to the Arizona Upland subdivision, which we determined using published floras and range maps (Shreve 1951; Kearney and Peebles 1960; Turner and others 1995). Second, species were chosen based on their occurrence at the site prior to the fire, or on their historical presence at the site, which managers suspected had declined in abundance due to past land uses. For example, managers included many of the perennial grasses in the seed mix because these species were thought to have declined because of past grazing or other ecological changes (Guo 2004). Third, managers wished to include a diverse array of plant lifeforms and longevities (annuals and perennials) in the seed mix. Fourth, the practical constraint of seed availability partly dictated composition of the seed mix. Seeds were purchased from a variety of vendors, and unfortunately the precise genetic origin of the seeds was unknown. The seeding rates were chosen for practical reasons to include the greatest representation of species as possible within funding and availability constraints. Samples of seed matter of native species were analyzed for possible exotic species content, with only a trace amount (0.16% by weight) of buffelgrass (*Pennisetum ciliare* (L.) Link [Poaceae]) detected in one sample (Arizona Department of Agriculture, State Agricultural Laboratory, Phoenix, Arizona, and Hulsey Seed Laboratory Inc, Decatur, Georgia). Exotic seeds were removed from the seed mix prior to seeding.

MONITORING AND ANALYSIS

To assess species establishment on the seeded burn, we established two 100-m (328-ft) long transects that started 15 m (49 ft) inside the burn and were separated by 15 m (49 ft). At the beginning of each transect and at 10-m (33-ft) intervals, we established a 1.785-m circular (10 m^2) plot, resulting in a total of 11 plots per transect. In each plot, we recorded live plant species rooted in plots and visually categorized aerial cover of each species using cover classes (Peet and others 1998): 1 = trace (0.1%), 2 = 0.1 to 1%, 3 = 1 to 2%, 4 = 2 to 5%, 5 = 5 to 10%, 6 = 10 to 25%, 7 = 25 to 50%, 8 = 50 to 75%, 9 = 75 to 95%, and 10 = > 95%. Cover classes were used only to rank-order species coverages by computing relative percent cover, summing to 100% for all species including both seeded and non-seeded volunteer species. To measure seasonal and yearly variation in species composition after seeding, we conducted sampling during a 32-mo period on 2 November 2005, 1 May 2006, 12 October 2006, 2 May 2007, and 27 March 2008. On the first sampling, some plants, especially grasses, were in cotyledon or early seedling stages, making accurate identification difficult. We collected, pressed, and identified unknown specimens to species when possible, but sufficient plant material was not available for identifying 15 of the collected specimens from the first sampling. Fewer than 5 specimens could not be accurately identified on the other 4 sampling dates. Unidentified specimens were not included in the data set. We were also not able to reliably differentiate the 2 seeded *Penstemon* species, so we grouped these species for analysis. For both seeded and non-seeded species, we calculated percentage frequency based on presence/absence in 22 plots. We classified non-seeded species as native or exotic following USDA NRCS (2007). On each plot, we also visually categorized mulch cover in 5% classes.

On the last monitoring date (27 March 2008), we also sampled five 10-m^2 plots in each of a burned, non-seeded, and an unburned, non-seeded area for comparison with the burned, seeded plots. These areas were immediately adjacent to the burned, seeded area, and the area available for sampling limited the number of plots to 5 spaced every 10 m (33 ft) along a 50-m (164-ft) long transect. We sampled these plots using the same methods as for the burned, seeded plots. Because of the close proximity of the seeded and non-seeded burn areas, it is possible that seeds of some seeded species were transported between areas by way of wind, animals, or other factors.

ASSESSMENT CONTEXT

As is often the case in studies of unplanned wildfires, our study is limited by a lack of quantitative pretreatment data and the fire is not replicated (van Mantgem and others 2001). Our results apply only to the particular weather conditions characterizing our study period. In 21 mo following seeding, precipitation was only 67% of normal at the long-term Carefree weather station, and only 53% of normal if the unusually wet month of September 2006 is not included (Figure 2). In addition, several exceptionally severe dry periods occurred, such as a 3-mo period (December 2005 to February 2006) beginning 4 mo after seeding, when only 0.025 cm (0.01 in) of precipitation fell. This period normally receives 33% of the total annual precipitation. A weather station with shorter term records, but only 3 km (1.9 mi) from the study site and with a more comparable elevation and weather pattern, registered even less precipitation (Maricopa County Flood Control District, Phoenix, Arizona). This station recorded only 70% (29 cm

[11.4 in]) of the already below-average precipitation recorded at the longer term station during 21 mo after seeding. However, in months 22 through 32 after seeding and preceding our last monitoring date, precipitation was 132% of the long-term average.

SEEDED SPECIES ESTABLISHMENT

Among the 28 species that were seeded, species exhibited several different establishment patterns through time on the burn (Table 1). Some species, such as several of the seeded grasses including tanglehead (*Heteropogon contortus*) and curly-mesquite (*Hilaria belangeri*), had emerged by the first fall sampling 3 mo after seeding, yet did not result in established plants at the time of the last sampling 32 mo after seeding. Other species (for instance, creosote bush) were not observed or were rarely observed in all 5 sampling periods, although it is possible that seedlings emerged and died between monitoring dates. The perennial *Penstemon* species (*eatonii* and *parryi*) typified an episodic emergence pattern in which seedlings were recorded the first spring after seeding, subsequently disappeared, were recorded again during the second spring, then were absent during the third spring. Another category of species, exemplified by the perennial desert senna (*Senna covesii*) and the annual desert bluebells (*Phacelia crenulata*), had established plants recorded throughout the monitoring period.

Considering seeded plant groups (grasses, forbs, shrubs), forbs established best by the end of the monitoring period (Table 1). Only one grass (purple threeawn [*Aristida purpurea*]) exhibited substantial establishment by the end of the study. Purple threeawn was among the most successful of all species, however, because not only did it become established on 27 to 50% of the plots on the last 2 sample dates but also those plots contained fruiting individuals. Because most other grass species germinated but did not produce mature plants, apparently they are more establishment- than seed-limited at this site (Turnbull and others 2000). There could be many causes for this, such as herbivory, inhospitable soil conditions, unfavorable post-seeding weather, or possibly genetic stock poorly adapted to the site.

No seeded shrubs became established in more than 9% of burned + seeded plots during any monitoring period. One of the seeded shrubs not detected on any plots, creosote bush, is a dominant species in North American warm deserts, but it reestablishes slowly after disturbances and is considered a late-successional species (Brown and Minnich 1986). This species is thought to germinate only sporadically and has exhibited mixed success in seeding trials (Bean and others 2004; Banerjee and others 2006). Creosote bush may not be well matched to our site's early successional conditions, although it also is possible

TABLE 1

Establishment of seeded and non-seeded species on 5 monitoring dates on a Sonoran Desert burn, Cave Creek Regional Park, Arizona.

Species [z]	Burned, seeded					Burned, non-seeded	Non-burned, non-seeded
	2 Nov 2005	1 May 2006	12 Oct 2006	2 May 2007	27 Mar 2008	27 Mar 2008	27 Mar 2008
	Frequency (%) [y]						
SEEDED GRASSES							
Aristida purpurea Nutt. (Poaceae)	0	91	4	50	27	0	0
Heteropogon contortus (L.) P. Beauv. ex Roem. & Schult. (Poaceae)	95	45	9	0	0	0	0
Hilaria belangeri (Steud.) Nash (Poaceae)	86	9	5	0	0	0	0
SEEDED SHRUBS							
Atriplex canescens (Pursh) Nutt. (Chenopodiaceae)	0	9	0	9	5	0	0
Olneya tesota A. Gray (Fabaceae)	9	0	0	5	0	0	0
SEEDED FORBS							
Argemone hispida A. Gray (Papaveraceae)	68	5	14	0	0	0	0
Baileya multiradiata Harv. & A. Gray ex A. Gray (Asteraceae)	14	23	0	18	14	0	0
Castilleja exserta (A. Heller) T.I. Chuang & Heckard (Scrophulariaceae)	0	5	0	0	95	0	0
Eschscholzia californica Cham. ssp. *mexicana* (Greene) C. Clark (Papaveraceae)	82	77	0	0	73	20	0
Glandularia gooddingii (Briq.) Solbrig (Verbenaceae)	5	0	0	41	0	0	0
Lesquerella gordonii (A. Gray) S. Watson (Brassicaceae)	91	82	0	59	59	20	0
Lupinus sparsiflorus Benth. (Fabaceae)	27	5	0	0	41	0	0
Penstemon eatonii A. Gray / *parryi* (A. Gray) A. Gray (Scrophulariaceae)	0	68	0	28	0	0	0
Phacelia crenulata Torr. ex S. Watson (Hydrophyllaceae)	95	100	23	73	95	80	0
Senna covesii (A. Gray) Irwin & Barneby (Fabaceae)	59	68	91	82	91	40	0
Sphaeralcea ambigua A. Gray (Malvaceae)	36	55	0	9	5	20	0

	Burned, seeded					Burned, non-seeded	Non-burned, non-seeded
Species [z]	**2 Nov 2005**	**1 May 2006**	**12 Oct 2006**	**2 May 2007**	**27 Mar 2008**		
	Frequency (%) [y]						
MAJOR NON-SEEDED SPECIES							
Ambrosia deltoidea (Torr.) Payne (Asteraceae)	5	0	0	0	14	60	100
Amsinckia menziesii (Lehm.) A. Nelson & J.F. Macbr. (Boraginaceae)	0	86	41	0	91	40	100
Astragalus nuttallianus DC. (Fabaceae)	0	9	0	0	64	60	60
Bromus rubens[x] L. (Poaceae)	0	5	0	0	41	40	0
Crassula connata (Ruiz & Pav.) A. Berger (Crassulaceae)	0	0	0	0	73	20	80
Cryptantha spp. (Boraginaceae)	46	23	0	9	32	0	40
Erodium cicutarium [x] (L.) L'Hér. ex Aiton (Geraniaceae)	100	95	0	91	91	100	100
Lepidium lasiocarpum Nutt. (Brassicaceae)	0	64	0	0	73	80	40
Logfia arizonica (A. Gray) Holub (Asteraceae)	0	0	0	0	27	20	80
Malva parviflora[x] L. (Malvaceae)	95	32	0	45	95	80	0
Oncosiphon piluliferum[x] (L. f.) Källersjö (Asteraceae)	0	0	0	5	86	80	100
Parkinsonia florida (Benth. ex A. Gray) S. Watson (Fabaceae)	0	0	0	0	5	0	40
Pectocarya recurvata I.M. Johnst. (Boraginaceae)	0	91	0	9	68	80	100
Plantago ovata Forssk. (Plantaginaceae)	73	82	0	18	73	80	60
Schismus barbatus[x] (Loefl. ex L.) Thell. (Poaceae)	0	95	82	100	95	100	100
Vulpia octoflora (Walter) Rydb. (Poaceae)	0	0	0	0	23	60	60

[z] Other species seeded but exhibiting little or no establishment in the first 4 samplings and no establishment during the last sampling (27 Mar 08) were the following: grasses: *Bothriochloa barbinodis, Bouteloua curtipendula, Bouteloua rothrockii, Eragrostis intermedia, Muhlenbergia porteri, Panicum obtusum, Setaria vulpiseta, Sporobolus cryptandrus*; shrubs: *Calliandra eriophylla, Larrea tridentata*; forb: *Allionia incarnata*.

[y] Based on 22, 10-m^2 plots for the burned + seeded area, and 5 plots each for the burned + non-seeded and the non-burned + non-seeded areas.

[x] Exotic species.

that the particular seeds used were not suited to germinate at our site.

We judge desert senna to be the most successful species, not just among forbs but among all species (Table 1). In addition to occupying ≥59% of plots in all 5 sampling times, this species exhibited the highest relative cover of any seeded species, ranging from 19 to 64% among monitoring dates. Furthermore, we observed desert senna flowering just 7 wk after seeding and during each monitoring period. By 21 mo after seeding, this species had attained a density of approximately 2000 plants/ha (810/ac).

Several other seeded forbs exhibited establishment more episodic than desert senna but were still successful during multiple monitoring periods. Except for one sampling date in which no plants were detected, for example, Gordon's bladderpod (*Lesquerella gordonii*) occupied ≥ 59% of plots (Table 1). California poppy (*Eschscholzia californica*) became established in ≥ 73% of plots in 3 of 5 monitoring periods. Coulter's lupine (*Lupinus sparsiflorus*) and *Penstemon* spp. occurred in ≥ 27% of plots in 2 sampling times. Underscoring the importance of not prematurely drawing conclusions about the potential establishment of a species through seeding, exserted Indian paintbrush (*Castilleja exserta*) was detected on only one plot in only one sampling time during the first 21 mo after seeding. This species, however, occupied 21 of the 22 plots by 32 mo after seeding. Presumably seeding allowed this species to create a persistent seedbank that awaited suitable emergence conditions (Bowers and others 2004).

NON-SEEDED SPECIES

Non-seeded species composition on the last monitoring date was generally similar on the seeded and non-seeded burn area. Native annuals such as curvenut combseed (*Pectocarya recurvata*) and desert Indianwheat (*Plantago ovata*) occurred at high frequencies in both areas (Table 1). Exotic species occurrences were similar between the 2 areas. Exotic species richness also was similar between the seeded and non-seeded burn but was proportionally lower on the seeded burn due to its higher total richness (Figure 3). Although exotics were present on the seeded burn, the suite of treatments (mulching, seeding, and fertilization) did not increase frequencies above those on the non-treated burn. Nevertheless, a concern is whether the abundance of Mediterranean grass and red brome will increase, possibly competing with native plants and accruing fuel. By 32 mo after seeding, Mediterranean grass attained a frequency on both burned areas similar to its frequency on the surrounding non-burned area.

MULCHING AND SEEDING PROCEDURE

Mulching and hydroseeding were chosen as the revegetation treatments for this burn, and it would be useful for future research to examine whether these relatively intensive methods improved establishment relative to less-intensive methods, such as simply broadcast seeding. Percentage of mulch cover was not strongly correlated (Pearson's $r^2 < 0.11$) with the total number of species occurring/plot in any of the sampling periods. Mulch persisted at 2 to 50% cover among plots 21 mo after seeding and still averaged 5% cover in our plots 32 mo after seeding. Jackson and others (1991) found that mulching with 900 kg/ha (800 lb/ac) of straw increased plant establishment after seeding 3 to 4 times on abandoned Sonoran Desert farmland. Similarly, Glendening (1942) found that straw mulching enhanced establishment of 10 seeded grasses in southern Arizona. It also is possible that our seeding method (hydroseeding) could have differentially affected seeded species. For example, Montalvo and others (2002) concluded that small-seeded species established better from hydroseeding than from drill seeding, whereas the opposite was true for large-seeded species. Drill seeding was not an option in our study because equipment running over the site was considered undesirable.

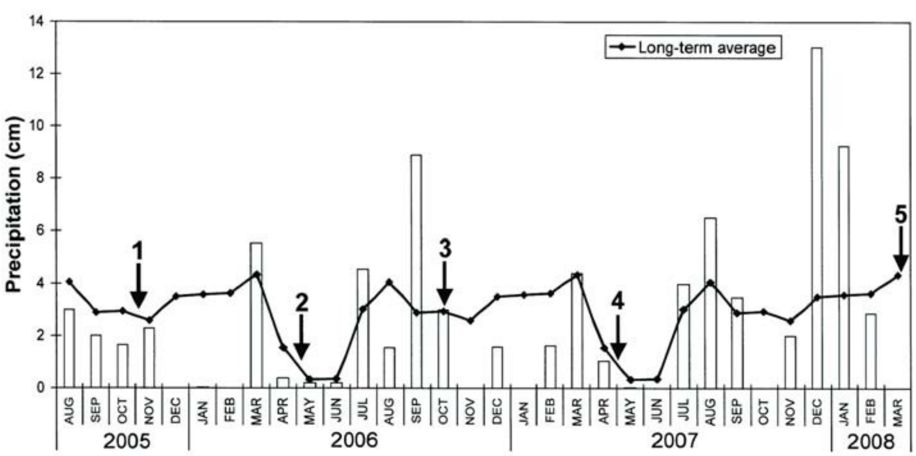

Figure 2. Monthly precipitation after seeding on 1 August 2005, through March 2008 after the last sampling, compared with the 1962–2007 average. Arrows indicate sampling dates. Precipitation recorded at the Carefree, Arizona, weather station, 8 km (5 mi) east of the study site (Western Regional Climate Center, Reno, Nevada).

COSTS AND OVERALL SEEDING EFFECTIVENESS

By ascribing a monetary value to large perennial plants (such as cacti and shrubs) that were killed on park land by the fire, an insurance settlement provided funding for the seeding in the amount of US$ 9900/ha ($4000/ac). Although only 7 (25%) of the 28 seeded species became established in 25% or more of plots by the end of the monitoring period, the site appeared well vegetated and contained flowering individuals (see Figure 1). This aesthetic component, and the avoidance of barren land that may encourage illegal dumping or other abuses, is a positive outcome of this seeding for park management. Ecologically, desert senna, which has been cited to live for 10 y (Bowers and others 2004), may have initiated the establishment of a fertile-island structure that is key for plant regeneration and ecosystem function in these desert systems (Carrillo-Garcia and others 2000). Seeded species comprised 54% of total plant richness per plot 21 mo after seeding and 29% of richness after 32 mo even with elevated richness of non-seeded annuals following a moist period (Figure 3). This suggests that the seeding treatment substantially boosted species richness relative to no management. Long-term success of the seeding depends on the ability of seeded species to persist aboveground or in the seedbank (Judd and Judd 1976; Price and Reichman 1987), or to facilitate the establishment of other native species. One concern is that the exotic annual Mediterranean grass is the most prominent volunteer (see Table 1), and it is unclear whether competition from seeded plants has affected, or will affect, its abundance. Longer term monitoring is needed to ascertain outcomes of interactions between seeding and community succession on this burn. Future research also could compare community functional variables (for example, insect community composition or small mammal abundance) between seeded and non-treated areas to assess possible benefits of revegetation beyond the plant community.

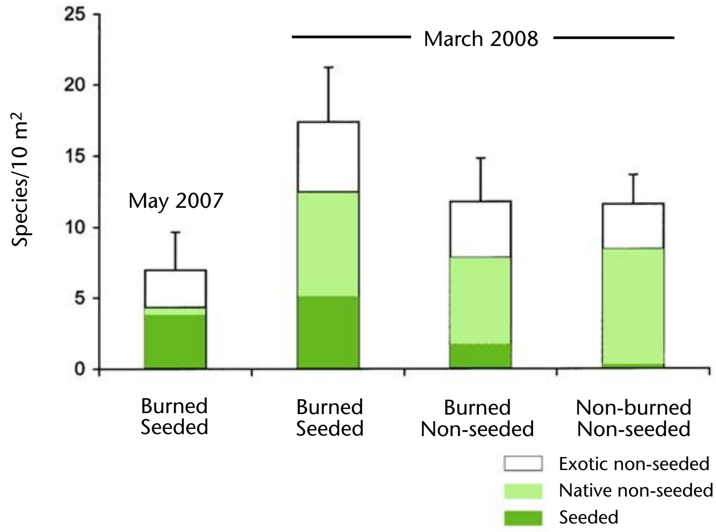

Figure 3. Species richness during the last 2 sampling dates of seeded and non-seeded species on a Sonoran Desert burn, Cave Creek Regional Park. Error bars are one standard deviation for average total richness.

CONCLUSION

Many variables can affect seeding success, such as species selection, genetic stock and germinability, associated treatments (such as mulching and weed control), environmental site conditions, timing of seeding, and precipitation. For example, if we had not selected any of the approximately 7 species that we consider productive in this particular seed mix, our conclusion about the effectiveness of the entire seeding would probably be different. Seeding has typically been largely discouraged as an option for revegetation of native species in North American warm deserts (Bainbridge 2007), but this area of research needs further study. Our data suggest that seeding native species may have potential as a revegetation tool in these deserts. Our viewpoint is that in evaluating seeding as an option, we need a better understanding of which species are amenable to seeding given current technology, which treatments can promote success, and under which environmental site conditions particular combinations of seeding components can be effective. This seeding also could be viewed as only an initial intervention designed to align the site on a desirable successional trajectory, and further treatments could build on the productiveness of seeded species that did establish.

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