Response of Deeproot Sedge (*Cyperus entrerianus*) to Herbicide and Prescribed Fire in Texas Coastal Prairie

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Introduced accidentally from South America, deeproot sedge is rapidly expanding in a variety of habitats throughout the southeastern United States. Of particular concern is its rapid expansion, naturalization, and formation of monocultures in Texas coastal prairie, one of the most imperiled temperate ecoregions in North America. The objective of this research was to examine how deeproot sedge responds to prescribed fire, to the herbicide imazapic, and to treatment combinations of both. Combinations of prescribed fire and imazapic treatments and imazapic-only treatments effectively reduced deeproot sedge cover and frequency. However, plots exposed to dormant season fires (with no imazapic) had greater deeproot sedge cover after burn treatments were applied, indicating that coastal prairie management using only dormant season prescribed fire will not work toward reduction or management of this exotic invasive species. Although deeproot sedge cover was often reduced in fire–imazapic treatment combinations, it was still present in treatment plots. Moreover, desirable functional plant groups (i.e., native bunchgrasses) did not respond positively to the fire–imazapic treatments, but in some instances, woody plant coverage increased. Repeated, long-term approaches using integrated and coordinated efforts with multiple treatment options will be necessary to restore community structure to desired compositional levels. Such integrated approaches should be effective in reducing deeproot sedge frequency, cover, and extent to more manageable levels throughout its introduced geographic range.

Nomenclature: Imazapic; deeproot sedge, *Cyperus entrerianus* Boeck.

Key words: Coastal prairie, *Cyperus entrerianus*, deeproot sedge, herbicide imazapic, invasive, prescribed fire, Texas.

Habitats and ecosystems worldwide are being affected by exotic invasive plants (Hobbs 2000; Pimentel et al. 2000; Shadel and Molofsky 2002), which are considered to be among the most destructive threats to the future of natural ecosystems (Greenberg et al. 2001; Gurevitch and Padilla 2004). Such threats present a mounting challenge for land managers (Brooks et al. 2004) because many exotic invasive species do not exhibit the same ecological threats in native ranges that they do in newly naturalized areas (Hierro and Callaway 2003; Leger and Rice 2003). Such characteristics hamper identification of potentially problematic plants before naturalization (Wolfe 2002) and exacerbate challenges of control and management once naturalized (Hobbs and Humphries 1995; Levine et al. 2003). Of primary concern are exotic invasive species that have a homogenizing effect (McKinney 2004; Steidl and Litt 2009) in which ecosystem structure and function are altered (DiTomaso 2000; Zavaleta et al. 2001).

Many exotic invasive species, regardless of introduction pathway, often exist in small populations and never become problematic (Elton 1958; Wagner 1993). However, some species exhibit long lag phases, during which they exist at low densities in small populations, before they experience exponential growth and become problematic (Mack et al. 2000). Once naturalized, they often form large monotypic stands that alter (1) ecosystem productivity, function, structure, and stability; (2) microclimate, by shifting consumption and supply of water, minerals, nutrients, and sunlight; (3) historic fire regime frequency and intensity; (4) local competition dynamics; and (5) overall ecosystem stability (Claridge and Franklin 2002; DiTomaso 2000; Levine et al. 2003; Shadel and Molofsky 2002;

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Management Implications

This research is the first attempt, to our knowledge, to quantify the utility of prescribed fire, herbicide, and combinations thereof, to control deeproot sedge in coastal prairie. Prescribed fire alone, particularly during the time when most prescribed fires are performed (winter), did not work to reduce deeproot sedge cover or frequency. Specifically, combining imazapic and prescribed fire, with a long-term management goal of reimplementation of growing-season prescribed fire, should reduce deeproot sedge cover and frequency. Imazapic application can be effective throughout the growing season, both alone and in combination with prescribed fire, in reducing both the extent and the frequency of deeproot sedge. Both long- and short-term control will be dependent on effective monitoring of both aboveground plant and seed-bank response. As deeproot sedge produces a tremendous biomass (kg ha^{-1}) of highly germinable seeds, seed bank management will remain a key element in restoration of invaded sites. This research provides some of the first evidence that deeproot sedge control may be achieved through integrated management with both imazapic and prescribed fire.

Vila and Weiner 2004; Vitousek et al. 1996; Williams and Baruch 2000).

In contrast to many exotic invasive species that are easily identified and well studied in habitats dominated by grasses and forbs, deeproot sedge (*Cyperus entrerianus* Boeck.) has remained relatively unrecognized and understudied because of its similarity in appearance to native *Cyperus* congeners. Native to temperate Argentina, deeproot sedge is rapidly expanding throughout the southeastern United States in a variety of habitats (Carter and Bryson 1996; Rosen et al. 2006). Warning of its invasive potential, Carter (1990) originally reported deeproot sedge in 20 counties from 5 states in the southeastern United States, but recent surveys have verified its presence in an additional 37 counties, from Georgia and Florida west to Texas (Rosen et al. 2006). Clearly, deeproot sedge has emerged from a potential lag phase. Its regional expansion and naturalization have been linked to rice (*Oryza sativa* L.) production in both North and South America and with human disturbances (i.e., mowing, construction, agriculture, etc.) (Bryson and Carter 2004; Carter 1990; Carter and Bryson 1996; Rosen et al. 2006). Perhaps of greatest concern is its widespread, but relatively undocumented, presence in Texas coastal prairie, an ecoregion considered one of the most endangered in North America (Grace et al. 2000). Current estimates indicate that $< 1\%$ of an estimated 3.9 million ha (9.64 million ac) of this tallgrass prairie ecoregion remains in historic condition (Barrileaux and Grace 2000; Diamond and Smeins 1984; Grace et al. 2000). The additive or cumulative effects of alterations in historical disturbance regimes (i.e., fire, grazing, and flooding), land use (i.e., urbanization and agriculture), fragmentation, and patch size reduction have all contributed to coastal prairie degradation (Baldwin et al. 2007; Brennan and Kuvlesky 2005; Diamond and Smeins 1984).

Once established, deeproot sedge forms dense, monotypic stands and experiences high survival (Carter 1990; Carter and Bryson 1996). Moreover, it is capable of producing several culms per plant (Carter and Bryson 1996) and 1,300 to 3,200 kg seed ha^{-1} (526 to 1,295 seeds ac^{-1}), with $> 80\%$ germination rates (King et al. 2012). Because of these traits, continued expansion of deeproot sedge is likely and may further deteriorate this endangered ecoregion. However, neither its immediate nor long-term ecological effects are known (see Rosen et al. 2006). Universal control and management strategies have not yet been developed for deeproot sedge. Fire and herbicide treatments are commonly used for coastal prairie restoration and management efforts, particularly when attempting to control exotic invasive plants and restoring native plant communities (Barnes 2004; Grace 1998; Twidwell et al. 2012). Therefore, the objectives of this research were to quantify plant functional group response, changes in plant species richness and composition, and deeproot sedge percentage of cover and frequency changes in response to fire and imazapic application in Texas coastal prairie.

Materials and Methods

Study Area. This research was conducted at The Nature Conservancy of Texas' Texas City Prairie Preserve (TCPP) in Galveston County, TX (29°26′14″N, 94°57′10″W), and the U.S. Fish and Wildlife Service's (USFWS) Attwater Prairie Chicken National Wildlife Refuge (APCNWR), in Colorado County, TX (29°41′11″N, 96°18′3″W), and Anahuac National Wildlife Refuge (ANWR), in Chambers County, TX (29°39′22″N, 94°26′8″W). All study areas occurred within the coastal prairie ecoregion of Texas, each with varying levels of deeproot sedge infestation (see Table 1). The soils at the study sites included clays (ANWR, TCPP), silt loams (ANWR), silt clay loam (TCPP), and fine sandy loam (APCNWR). All three sites consisted of similar vegetation, including sedges (*Cyperus* spp. and *Carex* spp.), rushes and spikerushes (*Juncus* spp. and *Eleocharis* spp.), and a variety of grasses and forbs, such as paspalum (*Paspalum* spp.), woodsorrel (*Oxalis* spp.), cordgrass (*Spartina* spp.), and bluestem (*Andropogon* spp.). At the beginning of the study, all three study areas had very limited native woody plant cover (see Table 1). However, some woody invaders, such as Macartney rose (*Rosa bracteata* J.C. Wendl), Chinese tallowtree [*Triadica sebifera* (L.) Small], eastern baccharis (*Baccharis halimifolia* L.), and Drummond rattlebush [*Sesbania drummondii* (Rydb.) Cory] were present. The two former exotic invasive species were foci of already established fire-management protocols on each study area, and neither species occurred within the top 15 species (by

Table 1. Means ($\bar{x}$) and standard errors (SE) of deeproot sedge (DRS) cover (%), DRS frequency (%), species richness, and functional group cover (%) recorded in sampling plots at the Anahuac National Wildlife Refuge (ANWR) (Chambers County, TX), the Attwater Prairie Chicken National Wildlife Refuge (APCNWR) (Colorado County, TX), and the Texas City Prairie Preserve (TCPP) (Galveston County, TX) during May 2005 and May 2007.[a]

	ANWR		APCNWR		TCPP	
Variable	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
May 2005						
DRS cover (%)[b]	8.8 a	1.9	13.8 a	4.6	14.9 a	3.3
DRS frequency (%)[c]	56.7 a	8.2	32.9 b	7.3	54.0 ab	7.6
Species richness	22.7 a	1.1	14.1 b	0.8	21.3 a	0.8
Non-DRS exotic cover (%)	13.5 a	2.0	11.3 a	3.9	12.7 a	1.5
Woody cover (%)	6.6 a	1.0	3.8 a	2.4	3.7 a	1.0
Native bunchgrass cover (%)	9.8 a	1.0	21.4 b	2.4	14.1 a	2.1
May 2007						
DRS cover (%)	3.3 a	1.1	5.5 a	1.6	6.8 a	2.0
DRS frequency (%)	33.8 b	7.3	33.3 b	6.1	53.7 a	6.2
Species richness	25.6 a	0.8	23.1 a	1.4	24.6 a	1.0
Non-DRS exotic cover (%)	7.7 a	1.2	14.1 ab	3.0	20.7 b	3.6
Woody cover (%)	10.5 a	1.4	6.1 b	2.0	4.3 b	0.8
Native bunchgrass cover (%)	4.6 a	0.6	11.8 b	1.6	12.7 b	2.4

[a] Means followed by the same letter within the same row are not different ($P > 0.05$).

[b] DRS cover as estimated from Daubenmire cover classes in 10 sampling plots per treatment plot ($n = 6$) per block ($n = 13$) among study sites.

[c] DRS frequency calculated by the presence/absence in 10 sampling plots per treatment plot ($n = 6$) per block ($n = 13$) among study sites.

percentage of cover) at any study site at the initiation of this research (May 2005; see Appendix 1).

Plot Establishment and Data Collection. Thirteen 100-m by 90-m (328-ft by 295-ft) study areas were established among the three study sites (i.e., five at TCPP, four at APCNWR, and four at ANWR) during May 2005. Each study area was divided into six 50-m by 30-m treatment plots. The corners of each study area and each treatment plot were geographically positioned using a Trimble GeoExplorer 3 global positioning system unit (Trimble Navigation, 935 Stewart Drive, Sunnyvale, CA 94085), where the sides of each study area were positioned in cardinal directions. A corner of each treatment plot was randomly selected as a starting point, and a transect of randomly chosen length (10, 20, 30, or 40 m) was used to randomly position 10 1-m^2 (10.76-ft^2) quadrats, for a total of 780. Quadrats were used to quantify percentage of cover by plant species using the Daubenmire canopy–coverage technique in May 2005, 2006, and 2007. Data collected in May 2005 were before application of any treatments and were used to establish baseline plant-species composition in treatment plots, as well as deeproot sedge cover and frequency. Every species encountered in each 1-m^2 quadrat was estimated and recorded to the nearest 5% Daubenmire class.

Treatment Application Strategy. All treatments (i.e., imazapic, prescribed burn, imazapic and prescribed burn, and control) were randomly applied to treatment plots within each study area at each study site. At the onset of this research, there was considerable uncertainty in guaranteeing prescribed fire-treatment deployment because of a combination of potential weather, personnel, prescription, and logistical complications. Therefore, the herbicide treatment-application strategy was designed to ensure consistent imazapic treatments regardless of whether any prescribed fire treatments were ever applied. Therefore, each study area was established to receive a minimum of two treatment-plot replicates of the following: control (i.e., no imazapic and no prescribed fire), early growing-season imazapic application, and late growing-season imazapic application. Therefore, all study areas were established within preexisting burn units within each study site to (1) elevate the probability that some study areas would be burned, and (2) ensure entire study areas were burned if prescribed burns were executed within a particular burn unit. As such, all prescribed burn treatments were applied at the entire study area level (not at a treatment plot level), and all prescribed burns were performed within the goals of existing management strategies and burn prescriptions for the USFWS and The Nature Conservancy of Texas.

Table 2. Means ($\bar{x}$) and standard errors (SE) of species richness recorded in sampling plots among nine herbicide[a] and prescribed fire (burn) treatments applied to treatment plots in the Anahuac National Wildlife Refuge (Chambers County, TX), the Attwater Prairie Chicken National Wildlife Refuge (Colorado County, TX), and the Texas City Prairie Preserve (Galveston County, TX), May 2005 to May 2007.

		Species richness[b]							
		May 2005		May 2007		Raw change[c]		Absolute change[d]	
Treatment	Treatment description	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
1	Control	23.3	1.9	28.8	1.2	−5.5	2.1	5.8 b +	1.9
2	Herbicide April 2006 or herbicide August 2006	22.3	1.6	24.2	1.9	−1.8	1.2	3.5 c +	0.7
3	Burn October 2005; herbicide April 2006 or August 2006; burn February 2007	19.5	0.3	24.8	0.9	−5.3	1.0	5.3 b +	1.0
4	Herbicide April 2006 or August 2006; burn February 2007	18.5	1.2	24.3	1.5	−5.8	1.9	5.9 b +	1.9
5	Burn February 2006; herbicide April 2006 or August 2006; burn February 2007	13.5	1.4	23.6	2.4	−10.1	2.7	10.9 a +	2.2
6	Burn February 2006; burn February 2007	14.0	1.3	19.8	1.8	−5.8	2.9	9.8 ab +	1.3
7	Burn February 2006; herbicide April 2006 or August 2006	23.4	1.7	26.6	1.4	−3.2	2.4	8.3 ab +	0.7
8	Burn February 2006	24.3	1.7	24.7	0.8	−0.3	2.4	4.0 c +	1.6
9	Burn February 2007	19.0	1.9	25.0	2.4	−6.0	3.3	8.0 ab +	2.4

[a] All herbicide treatments were deployed using imazapic.

[b] Means followed by the same letter within the same column are not different ($P > 0.05$).

[c] Raw change in species richness was calculated by finding the difference in species richness for each sampling plot between May 2005 and May 2007. Reported means are calculated by averaging the difference for each sampling plot per treatment plot that were included within each specified treatment. Negative values indicate an *increase* in the number of species.

[d] Absolute change in species richness was calculated by finding the absolute difference in species richness for each sampling plot between May 2005 and May 2007. Any negative values were erased by calculating the absolute value of the difference. Reported means are calculated from means per treatment plot that were included within each specified treatment. Larger values indicate a greater change. Direction is indicated by a plus (+) for increases in species richness, or by a minus (−) for declines in species richness.

Prescribed Fire and Imazapic Application. Prescribed fires were applied to burn units containing study areas during three temporal periods: October 2005, and February 2006, and February 2007. Herbicide treatments were applied early (April 15, 2006, to May 15, 2006) or late in the growing season (August 15, 2006, to September 26, 2006) (see Table 2 for treatment descriptions). Imazapic (Plateau, BASF Corporation, 26 Davis Dive, Research Triangle Park, NC, 27709) was used for all herbicide applications because it is labeled for (among others) *Cyperus* spp. control. Treatment plots (0.15 ha) were uniformly sprayed until wet using either an all-terrain vehicle with a mounted boom-sprayer or backpack sprayers, using an application rate of 0.4 to 0.5 L ha^{-1} (6 to 8 oz ac^{-1}) of imazapic deployed in a water volume of 374 L ha^{-1} (40 gal ac^{-1} or 56 L 0.15 ha^{-1}; treatment plot size). All applicators calibrated application rate and volume to ensure consistency in imazapic delivery. All spraying was conducted during periods of no precipitation for several days before and after scheduled application, with a sustained wind of $<$ 16 km h^{-1} (9.94 mi h^{-1}). No treatments were applied between March 15 and June 15 in 2005 to 2007, to minimize impacts on breeding/nesting Attwater's prairie chickens (*Tympanuchus cupido attwateri*) at APCNWR and TCPP.

Data Analyses. The following Daubenmire cover classes were used to estimate percentage of cover: 0, 1 to 5.0, 5.1 to 15.0, 15.1 to 30.0, 30.1 to 50.0, 50.1 to 70.0, 70.1 to 85.0, 85.1 to 95.0, and $>$ 95%. A midpoint was calculated for each cover class, where midpoints for all species occurring within quadrats were summed and divided by the number of quadrats to estimate percentage of cover for each treatment plot (Knight 1978). Mean cover was also calculated for all species at each study site in May 2005 (pretreatment). Average deeproot sedge Daubenmire cover class values were calculated for each treatment plot (i.e., average cover values among 10 quadrats per treatment plot) for each sampling period. Deeproot sedge frequency (%) was calculated per treatment plot, where its presence/absence was recorded for each quadrat (n = 10) within each treatment plot and converted to a percentage.

Total species richness and both deeproot sedge percentage of cover and frequency were calculated for each quadrat in each sampling period. Percentage of cover was similarly calculated for species functional classes by summing cover of species within each class in each quadrat. Functional classes examined consisted of (1) all native and exotic woody perennials, (2) all exotic herbaceous species other than deeproot sedge, and (3) native bunchgrasses. Changes between May 2005 and May 2007 in species richness, deeproot sedge percentage of cover and frequency, and functional class cover were calculated for each quadrat and each treatment plot. Two approaches were used to calculate these change in percentage data. First, differences in species richness, deeproot sedge percentage of cover and frequency, and functional classes between May 2005 and May 2007 were calculated and are hereafter are referred to as *raw change* data. Second, differences in the absolute values of species richness, deeproot sedge percentage if cover and frequency, and functional classes between May 2005 and May 2007 were calculated and are hereafter are referred to as *absolute change* data. The latter approach was employed to ameliorate potential influences of negative data during calculations and subsequent analyses because potential differences among treatments may be masked when calculating variances of negative and positive change data. Both are reported herein for completeness.

Accounting for all realized applications of imazapic and prescribed fire, 15 treatment combinations were deployed during this study. However, because of variation in deployment of prescribed fire specifically, there were instances in which only a single treatment plot received a specific treatment combination (see King 2011 for full details), which rendered no replication for some treatments. We combined all treatments that received any imazapic applications (i.e., combined early and late treatment application). This approach was supported by two lines of evidence. First, King et al. (2014) demonstrated that total nonstructural carbohydrate cycling in deeproot sedge rhizomes was consistent from early through late growing season, indicating that imazapic application anytime during the growing season should provide similar results. Second, that prediction was borne out during preliminary analyses herein, where there were never any differences ($P > 0.05$) in deeproot sedge cover or frequency between early and late imazapic treatment applications. As such, nine treatment combinations (including a control) were used for all subsequent comparative analyses.

Initial analyses examined differences in species richness, deeproot sedge percentage of cover, deeproot sedge frequency, and functional class cover recorded during May 2005 (at the beginning of the research) among study sites using ANOVA. ANOVA was also used to examine differences in species richness, deeproot sedge percentage of cover, deeproot sedge frequency, and functional class cover among study sites in May 2007 (at the end of the research). All subsequent analyses examined differences in species richness change (raw and absolute), deeproot sedge percentage of cover change and frequency change (raw and absolute), and functional class cover change (raw and absolute) among treatments using ANOVA. Analyses were focused on evaluating the effects of the aforementioned nine treatment combinations on changes in species richness (raw and absolute), deeproot sedge percentage of cover (raw and absolute), deeproot sedge frequency (raw and absolute), and functional class cover (raw and absolute). Any differences ($P < 0.05$) occurring during ANOVAs were examined more closely using least-squares mean separation in SAS statistical software (SAS Institute Inc., 100 SAS Campus Drive, Cary, NC 27513-2414) (SAS Institute 2003; Zar 1999).

Results and Discussion

Starting and Ending Points: Cover and Frequency. At the initiation of this research (May 2005), species richness varied ($F = 25.27$; df = 2, 75; $P < 0.001$) among study sites, where fewer plant species were recorded in plots at APCNWR than were recorded in the other study sites (Table 1). However, both deeproot sedge cover ($F = 0.90$; df = 2, 75; $P = 0.410$) and frequency ($F = 2.68$; df = 2, 75; $P = 0.075$) were similar among study sites (Table 1), and deeproot sedge dominated or codominated all three study sites in May 2005 (Appendix 1). Smutgrass [*Sporobolus indicus* (L.) R. Br.] and carpetgrasses (*Axonopus* spp.) ranked behind deeproot sedge in greatest cover at TCPP, whereas winter bentgrass [*Agrostis hyemalis* (Walt.) B.S.P. and vaseygrass (*Paspalum urvillei* Steud.) followed deeproot sedge at APCNWR (Appendix 1). Smutgrass and erect centella [*Centella erecta* (L. f.) Fern.] were essentially codominant with deeproot sedge at ANWR before treatments (Appendix 1). Total cover by exotic species other than deeproot sedge was similar among study sites in May 2005 ($F = 0.19$; df = 2, 75; $P = 0.831$), as was cover by all woody species ($F = 1.08$; df = 2, 75; $P = 0.344$) (Table 1). However, cover by native bunchgrasses varied among sites ($F = 8.16$; df = 2, 75; $P = 0.001$), where APCNWR contained the greatest initial native bunchgrass cover ($> 20\%$; Table 1).

At the termination of data collection (May 2007), both species richness ($F = 1.29$; df = 2, 75; $P = 0.282$) and deeproot sedge cover ($F = 1.52$; df = 2, 75; $P = 0.226$) were similar among study sites (Table 1). However, deeproot sedge frequency ($F = 3.42$; df = 2, 75; $P = 0.038$) varied among study sites and was encountered in $> 50\%$ plots at both TCPP and ANWR at the end of this research (Table 1). Overall, deeproot sedge cover declined at all study sites following treatments but continued to rank high among the species with the greatest cover (Appendix 2). Percentage

Table 3. Means ($\bar{x}$) and standard errors (SE) of deeproot sedge percentage (%) of cover recorded in sampling plots among nine herbicide[a] and prescribed fire (burn) treatments applied to treatment plots among the Anahuac National Wildlife Refuge (Chambers County, TX), the Attwater Prairie Chicken National Wildlife Refuge (Colorado County, TX), and the Texas City Prairie Preserve (Galveston County, TX), May 2005 to May 2007.

		Deeproot sedge[b]							
		May 2005		May 2007		Raw change[c]		Absolute change[d]	
Treatment	Treatment description	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
		%							
1	Control	13.8	4.9	10.8	7.8	2.9 b	8.2	13.8 abc −	5.5
2	Herbicide April 2006 or herbicide August 2006	20.7	7.1	3.8	1.6	16.9 ab	6.2	17.2 ab −	6.1
3	Burn October 2005; herbicide April 2006 or August 2006; burn February 2007	23.1	7.2	3.3	0.9	19.8 ab	6.5	19.8 ab −	6.5
4	Herbicide April 2006 or August 2006; burn February 2007	4.5	1.7	1.6	0.6	2.9 bc	1.4	3.6 c −	1.2
5	Burn February 2006; herbicide April 2006 or August 2006; burn February 2007	29.0	9.9	4.8	1.7	24.3 a	10.3	26.25 a −	9.6
6	Burn February 2006; burn February 2007	9.7	4.4	8.0	3.4	1.6 c	2.4	5.5 bc −	1.7
7	Burn February 2006; herbicide April 2006 or August 2006	6.6	2.0	0.9	0.3	5.9 b	2.2	7.1 bc −	1.8
8	Burn February 2006	4.1	1.9	8.5	3.2	−4.4 c	4.0	7.9 bc +	2.8
9	Burn February 2007	7.3	5.1	11.7	5.0	−4.4 c	3.1	5.1 bc +	2.9

[a] All herbicide treatments were deployed using imazapic.

[b] Means followed by the same letter within the same column are not different ($P > 0.05$).

[c] Raw change in deeproot sedge percent cover was calculated by finding the difference in deeproot sedge percentage of cover for each sampling plot between May 2005 and May 2007. Reported means are calculated by averaging the difference for each sampling plot per treatment plot that were included within each specified treatment. Negative values indicate an *increase* in deeproot sedge percentage of cover.

[d] Absolute change in deeproot sedge percent cover was calculated by finding the difference in deeproot sedge percentage of cover for each sampling plot between May 2005 and May 2007. Any negative values were erased by calculating the absolute value of the difference. Reported means are calculated from means per treatment plot that were included within each specified treatment. Direction is indicated by a plus (+) for increases in deeproot sedge percentage of cover, or by a minus (−) for declines in deeproot sedge percentage of cover.

of cover of exotics other than deeproot sedge varied among sites in May 2007 ($F = 4.87$; df = 2, 75; $P = 0.010$), where ANWR contained less exotic cover than the other sites had (Table 1). At TCPP and APCNWR, exotics present at the start of research, including smutgrass and vaseygrass, maintained their high rank cover, whereas blackberry (*Rubus* sp.) and carpetgrasses became dominant at ANWR by the end of the study (Appendix 2). Woody plant cover also varied among sites ($F = 5.19$; df = 2, 75; $P = 0.008$), where ANWR contained close to twice the woody cover of APCNWR and TCPP (Table 1). Finally, the percentage of cover by native bunchgrasses varied among sites in May 2007, where APCNWR and TCPP contained more bunchgrass cover than did ANWR (Table 1).

Treatment Impacts: Species Richness. Raw change in species richness was similar ($F = 1.33$; df = 8, 67; $P = 0.246$) among all treatments, but absolute change in species richness ($F = 2.79$; df = 14, 63; $P = 0.010$) varied among treatments. Plots exposed to treatment 5 (burned in February 2006; growing season herbicide; burned in February 2007) gained nearly 11 species for the duration of the research (Table 2). However, one should view these results (as related to treatment impacts) with caution because species richness never declined for any treatment, including the control (Table 2).

Treatment Impacts: Deeproot Sedge Percentage of Cover. Both raw ($F = 3.40$; df = 8, 67; $P = 0.003$) and absolute ($F = 2.67$; df = 8, 67; $P = 0.013$) changes in deeproot sedge percentage of cover varied among treatments (Table 3). All but two treatments (treatment 8 [prescribed burn only in February 2006] and treatment 9 [prescribed burn only in February 2007] reduced deeproot sedge cover between May 2005 and May 2007 (Table 3). These two, one-time, prescribed, fire-only treatments

Table 4. Means ($\bar{x}$) and standard errors (SE) of deeproot sedge frequency (%) recorded in sampling plots among nine herbicide[a] and prescribed fire (burn) treatments applied to treatment plots among the Anahuac National Wildlife Refuge (Chambers County, TX), the Attwater Prairie Chicken National Wildlife Refuge (Colorado County, TX), and the Texas City Prairie Preserve (Galveston County, TX), May 2005 to May 2007.

		Deeproot frequency							
		May 2005		May 2007		Raw change[b]		Absolute change[c]	
Treatment	Treatment description	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
		%							
1	Control	68.3	20.1	51.2	13.5	16.7	19.6	36.7 −	13.1
2	Herbicide April 2006 or herbicide August 2006	63.3	12.3	44.2	10.1	19.2	10.9	27.5 −	9.1
3	Burn October 2005; herbicide April 2006 or August 2006; burn February 2007	87.5	6.3	62.5	12.5	25.0	13.2	25.0 −	13.2
4	Herbicide April 2006 or August 2006; burn February 2007	28.3	8.5	27.5	10.2	0.8	8.9	17.5 −	7.2
5	Burn February 2006; herbicide April 2006 or August 2006; burn February 2007	56.3	15.2	38.8	9.9	17.5	19.2	45.0 −	11.2
6	Burn February 2006; burn February 2007	28.0	9.2	33.0	8.8	−5.0	9.8	19.0 +	7.7
7	Burn February 2006; herbicide April 2006 or August 2006	44.2	11.6	20.0	7.7	24.2	17.5	55.8 −	8.7
8	Burn February 2006	45.0	16.1	56.7	16.9	−11.7	26.5	45.0 +	18.0
9	Burn February 2007	30.0	16.7	65.0	14.5	−35.0	14.8	35.0 +	14.8

[a] All herbicide treatments were deployed using imazapic.

[b] Raw change in deeproot sedge frequency was calculated by finding the difference in deeproot sedge frequency for each sampling plot between May 2005 and May 2007. Reported means are calculated by averaging the difference for each sampling plot per treatment plot that were included within each specified treatment. Negative values indicate an *increase* in deeproot sedge frequency.

[c] Absolute change in deeproot sedge frequency was calculated by finding the difference in deeproot sedge frequency for each sampling plot between May 2005 and May 2007. Any negative values were erased by calculating the absolute value of the difference. Reported means are calculated from means per treatment plot that were included within each specified treatment. Direction is indicated by a plus (+) for increases in deeproot sedge frequency, or by a minus (−) for declines in deeproot sedge frequency.

(February 2006 or February 2007) were the only treatments in which deeproot sedge cover increased over the duration of the study, including control plots, in which there was some reduction in deeproot sedge cover. However, the standard errors (May 2007) indicate that there was considerable variability in these control data. Interestingly, plots that were burned twice (February 2006 and 2007; treatment 6) did reduce deeproot sedge cover (Table 3). The greatest reduction in cover was observed from treatment 5 (prescribed burn February 2006; growing season herbicide; prescribed burn February 2007), but those plots also started (May 2005) with the greatest deeproot sedge cover (Table 3). Similarly, the other burn–herbicide–burn treatment (treatment 3) also had substantial reductions in deeproot sedge cover over time, as did the herbicide-only treatment (treatment 2) (Table 3). Deeproot sedge cover was effectively reduced using (1) a growing-season imazapic application, or (2) a burn–imazapic–burn sequence-application strategy (Table 3).

Treatment Impacts: Deeproot Sedge Frequency. Neither absolute changes ($F = 1.81$; df = 8, 67; P = 0.089) nor raw changes in deeproot sedge frequency varied among treatments ($F = 1.38$; df = 8, 67; P = 0.222). Relatively large standard errors for deeproot sedge frequency data likely precluded our ability to detect any clear changes in deeproot sedge frequency. Interestingly, of the nine treatment combinations, treatments 6, 8, and 9 (February burn-only treatments) were the only treatments in which deeproot sedge frequency increased for the duration of this study (Table 4), which mirrored the percentage of cover analyses (see Table 3). Deeproot sedge was never completely removed from any treatment (Table 4) and was still detected in treatments that had apparently reduced deeproot sedge cover (see Table 3).

Treatment Impacts: Functional Class Cover. No differences were detected in raw ($F = 1.31$, df = 8, 67; P = 0.256) or absolute changes ($F = 0.79$, df = 8, 67; P = 0.610) in percentage of cover by exotics other than

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Table 5. Means ($\bar{x}$) and standard errors (SE) of percentage (%) of cover of exotic species other than deeproot sedge recorded in sampling plots among nine herbicide[a] and prescribed fire (burn) treatments applied to treatment plots among the Anahuac National Wildlife Refuge (ANWR) (Chambers County, TX), the Attwater Prairie Chicken National Wildlife Refuge (APCNWR) (Colorado County, TX), and the Texas City Prairie Preserve (TCPP) (Galveston County, TX), May 2005 to May 2007.

		Exotic species other than deeproot sedge							
		May 2005		May 2007		Raw change[b]		Absolute change[c]	
Treatment	Treatment description	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
		%							
1	Control	7.5	1.7	13.6	3.6	−6.1	4.4	14.3 +	2.8
2	Herbicide April 2006 or herbicide August 2006	12.7	3.3	11.2	3.4	1.5	4.6	16.8 −	3.2
3	Burn October 2005; herbicide April 2006 or August 2006; burn February 2007	20.8	3.1	25.9	3.3	−5.2	2.9	18.8 +	5.7
4	Herbicide April 2006 or August 2006; burn February 2007	10.6	2.9	27.8	9.7	−17.2	7.1	20.9 +	6.2
5	Burn February 2006; herbicide April 2006 or August 2006; burn February 2007	8.4	6.6	6.7	3.1	1.8	6.1	9.3 −	5.0
6	Burn February 2006; burn February 2007	12.1	6.3	18.0	5.9	−5.9	8.8	20.9 +	6.2
7	Burn February 2006; herbicide April 2006 or August 2006	13.7	3.0	8.4	1.4	5.3	2.9	13.3 −	2.0
8	Burn February 2006	14.2	8.3	7.2	3.2	7.1	7.3	16.6 −	4.3
9	Burn February 2007	21.4	1.9	26.9	13.5	−5.6	15.1	24.1 +	9.0

[a] All herbicide treatments were deployed using imazapic.

[b] Raw change in percentage of cover of exotic species other than deeproot sedge was calculated by finding the difference in the total sum of the percentage of cover for exotic species other than deeproot sedge for each sampling plot between May 2005 and May 2007. Reported means are calculated by averaging the difference for each sampling plot per treatment plot that were included within each specified treatment. Negative values indicate an *increase* in percentage of cover for exotic species other than deeproot sedge.

[c] Absolute change in percentage of cover of exotic species other than deeproot sedge was calculated by finding the difference in total sum percent cover for each sampling plot between May 2005 and May 2007. Any negative values were erased by calculating the absolute value of the difference. Reported means are calculated from means per treatment plot that were included within each specified treatment. Direction is indicated by a plus (+) for increases in deeproot sedge frequency, or by a minus (−) for declines in deeproot sedge frequency.

deeproot sedge among treatments (Table 5). Likewise, no differences were detected in raw change of woody plant cover among treatments (F = 0.68, df = 8, 67; P = 0.711); however, absolute change varied among treatments (F = 2.81, df = 8, 67; P = 0.010) (Table 6). Changes in woody plant cover were minimal across all treatments; although woody plant cover tended to increase in nearly all treatments, no treatment exceeded 12% woody plant cover (Table 6). Raw percentage of cover change for native bunchgrasses was similar among treatments (F = 1.43, df = 8, 67; P = 0.200) but varied among treatments using absolute change (F = 3.88, df = 8, 67; P < 0.001) (Table 7). Although native bunchgrass cover generally declined among all treatments, the greatest declines were observed in treatments in which prescribed fire was applied twice (treatments 3, 5, and 6) (Table 7). However, all of these functional-class analyses should be viewed with some caution because nearly all estimates of raw or absolute changes had substantial estimates of standard errors, indicating some significant variation in functional-class estimates, even within each treatment.

Comparison to Remnant Prairies. No site used in this study resembled any ecological or biological community typical of what is considered to be pre-European, endemic, Texas coastal prairie because all sites had significant exotic invasive plant composition and a history of anthropogenically driven disturbances, including grazing, row–crop agriculture, mowing, fire prevention and reapplication, as well as physical fragmentation (Diamond and Smeins 1984; Grace et al. 2000; King 2011; Rosen 2007). Because < 1% of Texas coastal prairie remains in original and intact condition (Barrileaux and Grace 2000; Grace et al. 2000), restoration and recovery of even marginal coastal prairie patches is key for regional conservation and management (see Rosen 2007).

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Table 6. Means ($\bar{x}$) and standard errors (SE) of woody species percentage (%) of cover recorded in sampling plots among nine herbicide[a] and prescribed fire (burn) treatments applied to treatment plots among the Anahuac National Wildlife Refuge (Chambers County, TX), the Attwater Prairie Chicken National Wildlife Refuge (Colorado County, TX), and the Texas City Prairie Preserve (Galveston County, TX), May 2005 to May 2007.

		Woody species cover[b]							
		May 2005		May 2007		Raw change[c]		Absolute Change[d]	
Treatment	Treatment description	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
		%							
1	Control	2.0	0.6	8.7	2.4	−6.7	2.1	8.6 ab +	2.2
2	Herbicide April 2006 or herbicide August 2006	6.4	3.5	8.3	2.8	−1.9	2.8	8.1 ab +	2.5
3	Burn October 2005; herbicide April 2006 or August 2006; burn February 2007	2.9	1.8	2.5	0.8	0.4	2.1	4.9 bc −	1.7
4	Herbicide April 2006 or August 2006; burn February 2007	7.9	3.0	6.8	1.8	1.1	2.5	6.3 bc −	2.0
5	Burn February 2006; herbicide April 2006 or August 2006; burn February 2007	0.2	0.2	2.1	1.0	−1.9	1.1	2.3 c +	0.9
6	Burn February 2006; burn February 2007	0.9	0.8	2.8	1.4	−1.9	1.4	3.5 bc +	1.6
7	Burn February 2006; herbicide April 2006 or August 2006	8.0	1.8	9.6	2.2	−1.6	3.6	12.9 a +	1.3
8	Burn February 2006	5.1	1.1	11.5	2.7	−6.4	2.8	11.4 a +	1.7
9	Burn February 2007	3.1	1.7	5.7	3.4	−2.4	2.2	5.1 b +	2.6

[a] All herbicide treatments were deployed using imazapic.

[b] Means followed by the same letter within the same column are not different ($P > 0.05$).

[c] Raw change in woody species percent cover was calculated by finding the difference in the total sum woody species percentage of cover for each sampling plot between May 2005 and May 2007. Reported means are calculated by averaging the difference for each sampling plot per treatment plot that were included within each specified treatment. Negative values indicate an *increase* in woody species percentage of cover.

[d] Absolute change in woody species percentage of cover was calculated by finding the difference in total sum woody percent cover for each sampling plot between May 2005 and May 2007. Any negative values were erased by calculating the absolute value of the difference. Reported means are calculated from means per treatment plot that were included within each specified treatment. Direction is indicated by a plus (+) for increases in deeproot sedge frequency, or by a minus (−) for declines in woody species percentage of cover.

Thirty years ago, Diamond and Smeins (1984) described 15 endemic, Texas coastal-prairie sites that had no history of overgrazing, plowing, or use of herbicides or fertilizers, which established the baseline for representative plant-species composition, richness, and diversity for native, Texas coastal prairie. In their regional characterization, species richness ranged between 37 to 64, much greater than recorded in this study (23 to 26), even after fire and herbicide treatments. More recently, Rosen (2007) characterized the vascular flora of Nash Prairie, a 120-ha. pristine, coastal-prairie remnant (see Cohn 2006) in Brazoria County, TX, and reported 311 species. Similarly, Allen et al. (2001) reported > 500 species in small, remnant, coastal-prairie patches in southwestern Louisiana. Although species richness typically increased over time after deeproot sedge control and management treatments, study sites are species depauperate, possessing < 10% of the richness reported at Nash Prairie (Rosen 2007). It is clear that successful Texas coastal prairie restoration will require, in some form, a rather elaborate and lengthy restoration program, incorporating continuous monitoring of seed bank composition and development, intensive and aggressive plant management actions (i.e., herbicide, fire, removal, biological control, etc.), as well as direct and indirect seeding, reseeding, and interseeding of desired coastal-prairie species (DiVittorio et al. 2007).

Influence of Prescribed fire. Whether ignition naturally occurred from lightning or intentionally by humans, fire at least partially drives ecosystem structure and function throughout North America (DiTomaso et al. 2006a; Pyne et al. 1996). Current regional landscapes are dramatically different than they were historically, largely because of complex interactions among fragmentation, land-use changes, and establishment, spread, and naturalization of exotic invasive flora and fauna (Marx et al. 2008; Pyne et

Table 7. Means ($\bar{x}$) and standard errors (SE) of native bunchgrass species percentage (%) of cover recorded in sampling plots among nine herbicide[a] and prescribed fire (burn) treatments applied to treatment plots among the Anahuac National Wildlife Refuge (Chambers County, TX), the Attwater Prairie Chicken National Wildlife Refuge (Colorado County, TX), and the Texas City Prairie Preserve (Galveston County, TX), May 2005 to May 2007.

		Native bunchgrass cover[b]							
		May 2005		May 2007		Raw change[c]		Absolute change[d]	
Treatment	Treatment Description	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
		%							
1	Control	17.8	3.5	14.3	3.7	3.5	2.5	12.5 b −	1.9
2	Herbicide April 2006 or herbicide August 2006	14.6	1.9	14.4	3.6	0.2	3.7	15.0 ab −	1.9
3	Burn October 2005; herbicide April 2006 or August 2006; burn February 2007	18.9	8.2	6.3	0.7	12.6	7.9	16.9 a −	6.6
4	Herbicide April 2006 or August 2006; burn February 2007	16.7	4.7	13.9	3.9	2.8	4.2	11.5 b −	2.4
5	Burn February 2006; herbicide April 2006 or August 2006; burn February 2007	19.7	4.1	11.8	2.8	7.9	3.3	16.8 a −	2.5
6	Burn February 2006; burn February 2007	22.2	4.4	9.9	2.6	12.3	5.1	20.1 a −	1.8
7	Burn February 2006; herbicide April 2006 or August 2006	8.4	1.3	5.1	0.5	3.3	1.4	8.0 c −	0.5
8	Burn February 2006	9.5	2.4	2.5	0.7	7.1	2.9	9.9 bc −	1.6
9	Burn February 2007	5.8	2.6	3.8	1.1	1.9	2.7	6.3 c −	2.3

[a] All herbicide treatments were deployed using imazapic.

[b] Means followed by the same letter within the same column are not different ($P > 0.05$).

[c] Raw change in native bunchgrass species percentage of cover was calculated by finding the difference in the total sum native bunchgrass species percentage of cover for each sampling plot between May 2005 and May 2007. Reported means are calculated by averaging the difference for each sampling plot per treatment plot that were included within each specified treatment. Negative values indicate an *increase* in native bunchgrass species percentage of cover.

[d] Absolute change in native bunchgrass species percentage of cover was calculated by finding the difference in total sum native bunchgrass species percentage of cover for each sampling plot between May 2005 and May 2007. Any negative values were erased by calculating the absolute value of the difference. Reported means are calculated from means per treatment plot that were included within each specified treatment. Direction is indicated by a plus (+) for increases in deeproot sedge frequency, or by a minus (−) for declines in native bunchgrass species percentage of cover.

al. 1996). Consequently, the ecological role of fire has changed from a driver of ecosystem processes to one that is intentionally used to reestablish disturbance regimes, as a tool to manipulate succession, and to restore and rehabilitate systems dominated by exotic, invasive plants (Twidwell et al. 2012), but it may also be used unintentionally to encourage exotic invasive-plant naturalization (Towne and Kemp 2008). Fire regimes with regionally historical fire parameters (i.e., fire return interval, intensity, seasonal timing, etc.) implemented to manage exotic invasive plant species must mimic natural timing to promote the desired results for both exotic and native flora (Towne and Kemp 2008; Twidwell et al. 2012). High-intensity, warm-season burns may be beneficial in invasive plant management (Twidwell et al. 2012, 2013); however, most prescribed fires in the southeastern United States are conducted as cool-season burns and may not achieve desired management goals (see Brooks et al. 2004).

Each site in this study had preexisting fire-management plans, and the research was designed to fit into those established protocols. Two sites (APCNWR and TCPP) contained populations of the endangered Attwater's prairie chicken, and a primary focus at the third site (ANWR) was waterfowl management. Consequently, growing-season prescribed burns are not used in those areas, primarily to prevent destroying nests or killing broods of Attwater's prairie chickens or resident (nesting) waterfowl like mottled ducks (*Anas fulvigula*). Nearly all prescribed fires in coastal Texas are dormant-season burns, and all prescribed, burn-only treatments in this study were conducted in February (2006 or 2007 or both), except for one, which was burned in October 2005 and February 2007 (treatment 3). When only one burn was conducted, February 2006 or 2007,

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deeproot sedge cover increased, but when combined with the application of a second prescribed burn (treatment), deeproot sedge cover was reduced. Although reductions in deeproot sedge cover after multiple applications of prescribed fire would suggest long-term declines may be achieved, only temporary cover reduction was observed. In fact, deeproot sedge eventually responded with an increase in cover, even after a second burn application. Similarly, prescribed fire reduced the density of the exotic, invasive, perennial sulfur cinquefoil (*Potentilla recta* L.) the first year after a burn treatment, but no further significant reductions were measured between burned and unburned sites in the subsequent 5 yr (Lesica and Martin 2003). Long(er)-term monitoring of deeproot sedge's response to these dormant-season burns will be necessary because initial responses may be temporary. However, regular, intentional, and well-timed prescribed fire management is likely the most effective and cost-effective tool for coastal prairie management.

Although deeproot sedge cover response to prescribed fire varied, there was little or no effect by any prescribed fire treatment on species richness. Because of the nominal change in deeproot sedge cover, other species may not have been able to germinate and establish within treatment plots. However, if growing season burns were more effective at reducing deeproot sedge coverage, as demonstrated in other sedges (see Coppedge et al. 1998), then species richness might increase as native species germinate, establish, and recruit from remnant seed banks. Treatments that reduce cover of exotic species in coastal grasslands presumably promote native bunchgrass expansion by reducing competition and creating sites for seedling establishment, resulting in increased bunchgrass cover over time. However, native bunchgrasses, a targeted group for restoration purposes, did not respond positively, and tended to decline in coverage over the temporal window of this study. Perhaps, (1) a native seedbank did not exist for these bunchgrasses, (2) the fire return interval was too short to allow establishment because the greatest reductions occurring where prescribed burns were implemented twice during this study, or (3) cool-season prescribed fires (as used in this study) do not promote native bunchgrass establishment. In a 14-yr study, Towne and Kemp (2008) found that spring fires had little effect on species richness, but growing-season (summer) burns produced a 28% increase in Kansas tallgrass prairie. The effects of prescribed fire timing cannot be underestimated, and this study provides a basis to question the utility of dormant-season fires to purportedly manage exotic invasive species, such as deeproot sedge (see Twidwell et al. 2012). The results of this short-term study probably do not accurately reflect the presumed long-term positive effects of exotic plant removal and reintroduction of fire on the growth and establishment of native bunchgrasses.

The lack of widespread control of deeproot sedge or other exotic invasive species by prescribed fire alone was observed in the general increase in exotic species other than deeproot sedge in fire-only treatments. This functional group accounted for significant portions of the total plant cover at all three sites in this study, including many earlier introductions that have become ''naturalized,'' but continue to affect ecological functions in the coastal prairie. However, species composition of this group varied among sites, and dissimilar species-specific responses to fire may have contributed to the observed response patterns. Major species within this group largely consisted of perennial warm-season (C4) grasses, such as vaseygrass or bahiagrass (*Paspalum notatum* Fluegg), which are generally tolerant or promoted by fire. Thus, similar to deeproot sedge, control of these species may require combinations of fire and targeted herbicides (see Grace et al. 2001).

Another management priority in many coastal prairie remnants is controlling woody plant encroachment, which have increased in abundance since alteration of historic fire regimes and introduction of exotic woody invaders (see Bruce et al. 1997). Although it is generally well-established that periodic fires have the capacity to reduce woody cover in coastal grasslands (Twidwell et al. 2012, 2013), this study does not confirm that, at least in the short window examined herein.

Resprouting of woody species following removal of aboveground biomass by fire can be quite extensive, especially where cool-season fires leave root crowns virtually unharmed. Some species in particular, such as Chinese tallowtree and Macartney rose, are capable of significant recovery as soon as the summer after a winter burn (Grace et al. 2001). The lack of any substantial change in woody cover, even in treatments involving successive winter burns, indicates that winter burns were incapable of effectively controlling woody encroachment in these coastal prairies. Alternatively, more-intense growing-season burns, which more frequently damage belowground plant parts, would likely be more effective (Twidwell et al. 2012, 2013). Similarly, initial reductions in the aboveground biomass of deeproot sedge from dormant-season burns may be replaced by vigorous growth during subsequent growing seasons (as observed in this study), as belowground reserves were not affected by nongrowing season (cool-season) prescribed fire (see King et al. 2014). For example, deeproot sedge's rhizome nonstructural carbohydrate levels are generally static during winter (see King et al. 2014), when deeproot sedge appears to be at least partially dormant. Any prescribed fire applied during winter will remove standing matter, but the plant maintains substantial carbohydrate reserves to recover soon thereafter. Like cogongrass [*Imperata cylindrica* (L.) Beauv.], which can respond to fire via rhizomatous sprouting (Bryson and Carter 1993), deeproot sedge resprouting potential appears quite substantial.

Beyond resprouting, arguably the most important element of deeproot sedge management may be controlling seed production and minimizing seed bank development (King et al. 2012). In general, dormant-season prescribed fires effectively remove aboveground biomass but do not reside at sufficiently high temperatures for long enough periods to exert much effect on seed banks or rhizomes existing in dense deeproot sedge stands (J. R. King, unpublished data). In contrast, prescribed fires likely enhance regrowth of deeproot sedge by rhizomal sprouts and seed germination via removal of competition and physical and mechanical nutrient and mineral recycling. For example, prescribed burning has shown to stimulate seed germination of the yellow starthistle (*Centaurea solstitialis* L.) (DiTomaso et al. 2006b). Prescribed fires in deeproot sedge stands that have received an herbicide treatment produce greater heat but have short residual times. As such, seed bank management will likely be the primary challenge for any large- or small-scale control and management efforts (King et al. 2012). Studies examining seasonal timing, intensity, and frequency of prescribed burns needed to control aboveground deeproot sedge biomass and seed production are needed to understand the efficacy of prescribed fire as a management tool.

Influence of Herbicide. Exotic, invasive plants are often controlled using herbicides and are influenced by (1) the kind of herbicide (i.e., foliar-active, soil-active, etc.); (2) the mode of action; (3) the method of application (i.e., ground, aerial, etc.); (4) the direct and indirect effects on the ecosystem (i.e., both target and nontarget species mortality, cover, water and nutrient cycling, seed bank response, etc.); and (5) the physiological characteristics of the target species (i.e., reproductive stages and abilities, storage of carbohydrate, etc.) (Sosebee 1984). Imazapic applications were timed to determine whether the physiological characteristics of deeproot sedge during the growing season would respond to herbicide application. Unless deeproot sedge was physiologically ready to translocate foliar-applied herbicide to the roots, only a top-kill would be expected (Brady and Hall 1976; King et al. 2014; Sosebee 1984; Troxler et al. 2003). Therefore, the most favorable control rates and timing for the herbicide application should be correlated with different stages of plant growth and carbohydrate storage. For example, Blum et al. (2000) found that herbicide treatments can reduce purple nutsedge (*Cyperus rotundus* L.) and yellow nutsedge (*Cyperus esculentus* L.) density in bermudagrass [*Cynodon dactylon* (L.) Pers.] turf, although success varied by herbicide, application rates, and the combinations of different herbicides used together. In this study, growing-season imazapic applications reduced deeproot sedge cover. Future work should certainly consider other herbicides as well, to provide a larger suite of potential herbicides that may be applied to coincide with the time when deeproot sedge is most susceptible to herbicide applications.

Rhizome total nonstructural carbohydrate trends (King et al. 2014) corroborated these field-level responses because, when imazapic was applied during the growing season (either April or August), deeproot sedge was consistently increasing rhizome carbohydrates. As such, herbicides applied during any portion of the growing season should be assimilated and translocated to the rhizomes, exerting effective plant kill (Bariuan et al. 1999; Brady and Hall 1976; Sosebee 1984). Conversely, after carbohydrate concentrations in deeproot sedge's rhizomes peak in August, there is a period of decline, followed by relative stagnation, (September–January) in carbohydrate levels (King et al. 2014). Herbicides applied during that time will be ineffective in achieving total plant kill because the translocation processes needed for the herbicide to reach the rhizomes are either absent or reduced (Brady and Hall 1976; Sosebee 1984).

Integrated/Repeated Treatments. Integrated control approaches are critical to achieve the desired reduction in frequency and cover of exotic, invasive plant species (DiTomaso et al. 2006a,b; Hobbs and Humphries 1995; Paynter and Flanagan 2004). Reduction of deeproot sedge frequency and cover was greatest when a combination of burn–herbicide–burn treatments were used. DiTomaso et al. (2006b) also found that integrating prescribed fire and herbicide treatments reduced the density of the exotic invasive yellow starthistle in California better than either treatment applied alone. Paynter and Flanagan (2004) found that varying combinations of herbicide, prescribed fire, and bulldozing were effective in the reduction of catclaw mimosa (*Mimosa pigra* L.), although neither were effective alone. In this study, plots that were only burned had greater deeproot sedge cover after burn treatments, indicating that using prescribed fire exclusively (particularly during the dormant season) did not reduce deeproot sedge coverage or frequency. Increases in deeproot sedge cover in response to dormant season prescribed fire was likely based on (1) its ability to quickly recover its aboveground biomass from rhizomal sprouts (see DiTomaso et al. 2006a), or (2) the inability of fire, at times, to carry through a live stand of deeproot sedge (J. R. King, personal observation), or both. Although dormant season prescribed fire may be effective on plants that were previously killed by growing-season herbicide, fire may be an effective tool to remove aboveground portions of deeproot sedge (regardless of herbicide treatment). However, the response of deeproot sedge to (other) herbicides integrated with experimentally implemented growing-season prescribed fires should be a focus of future deeproot sedge control efforts.

A singular control approach, such as growing-season herbicide application may accomplish plant-kill, but dead vegetative matter remains (as observed in this study) and inhibits establishment of other potentially desirable plant species. Although prescribed fire may reduce the percentage of cover of an invasive plant, greater seed germination from remnant seedbanks may occur after fires (J. R. King, unpublished data; see Dyer 2002). Conversely, if both herbicides and prescribed fire (either dormant or growing season) are used integratively, greater plant-kill and more thorough removal of dead plant matter should be achieved, along with increases in native species richness and cover. However, any change in species composition or richness will be partially determined by the composition of the remnant seed bank, and each species' seed response to the treatments (Buhk and Henson 2006; Jutila and Grace 2002; Vermeire and Rinella 2009). Deeproot sedge's seed-bank response to prescribed fire may be tremendous, as is its seed production and germination rates (King et al. 2012). As such, seed-bank management will remain a key element in long-term control and management strategies.

Cover reductions do not necessarily equate with complete eradication. Seven of the nine (including controls) treatments decreased deeproot sedge cover (Table 3), and six treatments had some reduction in frequency; however, no treatment eradicated deeproot sedge (Table 4). Complete eradication is not likely an attainable goal for deeproot sedge; however, long-term management may eventually reduce both its frequency and extent to a manageable or negligible level.

Effective deeproot sedge control and management may be achieved through careful combinations of prescribed fires and herbicide applications. Dormant-season prescribed fire seems to accelerate cover expansion, but, when combined with herbicide treatments, substantial reduction in coverage can be achieved. This suggests that prescribed fire during late fall through winter, followed by a growing season (i.e., April to August) herbicide application, targets rapidly regrowing and reproductive plants. Because it is unlikely that deeproot sedge has been removed, another prescribed burn would be helpful to remove remnant dead vegetation, which might affect the remaining native seed banks. It may be necessary to initially broadcast spray (i.e., aerial, tractor boom spray), but, as deeproot sedge diminishes, spot-spray methods may prove to be more economically feasible and friendlier to nontarget species. This method, if repeated, should remove deeproot sedge from the site or at least reduce its population to acceptable levels.

Continued monitoring will be required, and land managers may be required to adjust their management practices to respond to deeproot sedge and nontarget species. Long-term control and management of deeproot sedge will require repeated, integrated efforts and an exceptional awareness of seed-bank management (see King et al. 2012). Focus on any single control treatment may be detrimental to management efforts with deeproot sedge. As with many invasive species, the key to preventing a problematic expansion of deeproot sedge is early detection, an expeditious control response, eradication of small satellite populations, and integrated control methods (Bryson and Carter 2004; DiTomaso et al. 2006a; Hobbs and Humphries 1995; McNeely et al. 2005; Paynter and Flanagan 2004). It is clear that a single treatment application will not be effective with deeproot sedge. Conversely, repeated, long-term approaches, using coordinated efforts and multiple treatment options, will be effective, but only if land managers are committed to such extended and prolonged management efforts.

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Appendix 1. Means ($\bar{x}$) and standard errors (SE) for percentage (%) of cover for the top 15 species at Anahuac National Wildlife Refuge (ANWR) (Chambers County, TX), Attwater Prairie Chicken National Wildlife Refuge (APCNWR) (Colorado County, TX), and Texas City Prairie Preserve (TCPP) (Galveston County, TX) in May 2005 (pretreatment).

ANWR			APCNWR			TCPP		
Species	$\bar{x}$	SE	Species	$\bar{x}$	SE	Species	$\bar{x}$	SE
Cyperus entrerianus[a]	8.8	0.6	*Cyperus entrerianus*[a]	13.8	1.7	*Cyperus entrerianus*[a]	14.9	1.2
Sporobolus indicus[a]	8.1	0.5	*Agrostis hyemalis*	8.9	0.8	*Sporobolus indicus*[a]	7.7	0.7
Centella erecta	7.8	0.5	*Paspalum urvillei*[a]	6.8	1.2	*Axonopus* spp.	7.6	0.9
Paspalum sp. 4*	5.6	0.4	*Juncus marginatus* Rostk.	4.1	0.4	*Paspalum* sp. 4*	4.8	0.5
Cynodon dactylon[a]	4.1	0.3	*Iva angustifolia* Nutt. ex DC	3.0	0.6	*Eleocharis* spp.	4.2	0.4
Iva angustifolia Nutt. ex DC	4.1	0.3	Unknown 2	2.8	0.6	*Centella erecta*	3.0	0.3
Euthamia spp.	3.5	0.2	*Juncus brachycarpus* Engelm.	2.7	0.4	*Cynodon dactylon*[a]	2.7	0.4
Rubus spp.	3.3	0.2	*Polygonum pensylvanicum* L.	2.7	0.5	*Spartina patens* (Aiton) Muhl.	2.7	0.5
Mimosa spp.	3.2	0.2	*Iva annua* L.	2.5	0.4	*Iva angustifolia* Nutt. ex DC	2.3	0.2
Unknown 1	3.2	0.2	*Rosa bracteata*[a]	2.4	1.0	*Juncus marginatus* Rostk.	2.2	0.2
Baccharis halimifolia L.	2.5	0.2	*Andropogon* sp. 1	2.2	0.5	*Spartina* sp. 1*	2.2	0.4
Juncus marginatus Rostk.	2.3	0.2	*Paspalum* sp. 2*	2.1	0.5	*Neptunia lutea* Benth.	2.1	0.2
Solidago spp.	2.0	0.1	*Paspalum plicatulum* Michx.	1.9	0.7	*Paspalum* sp. 1*	2.0	0.5
Unknown 3	1.4	0.1	*Eleocharis* spp.	1.7	0.3	*Paspalum urvillei*[a]	2.0	0.3
Andropogon glomeratus	1.4	0.1	*Paspalum* sp. 3*	1.6	0.7	*Baccharis halimifolia* L.	1.9	0.4

[a] Exotic, invasive species.

* Numbered species indicate those unidentified but distinguished from congeners.

Appendix 2. Means ($\bar{x}$) and standard errors (SE) of percentage (%) of cover for the top 15 species by cover at Anahuac National Wildlife Refuge (ANWR) (Chambers County, TX), Attwater Prairie Chicken National Wildlife Refuge (APCNWR) (Colorado County, TX), and Texas City Prairie Preserve (TCPP) (Galveston County, TX) in May 2007 (posttreatment).

ANWR			APCNWR			TCPP		
Species	$\bar{x}$	SE	Species	$\bar{x}$	SE	Species	$\bar{x}$	SE
Rubus spp.	6.6	0.5	*Paspalum urvillei*[a]	5.8	1.0	*Paspalum notatum*[a]	11.8	1.4
Axonopus sp.p	6.3	0.9	*Paspalum* sp. 3*	5.7	0.6	*Sporobolus indicus*[a]	7.2	0.6
Iva annua	4.9	0.4	*Cyperus entrerianus*[a]	5.4	0.9	*Cyperus entrerianus*[a]	6.1	0.8
Solidago sp. 1*	4.5	0.6	Unknown 4	5.4	0.4	*Paspalum* sp. 3*	5.2	0.6
Sporobolus indicus[a]	4.0	0.5	*Paspalum notatum*[a]	4.7	0.8	*Eleocharis* spp.	4.9	0.4
Ambrosia spp.	3.9	0.3	*Ambrosia* spp.	4.7	0.4	*Spartina* sp. 1*	4.1	0.7
Cyperus entrerianus[a]	3.3	0.5	*Iva annua*	3.5	0.6	*Mimosa* spp.	3.6	0.3
Mimosa spp.	3.1	0.4	*Polygonum* spp.	3.3	0.5	*Ambrosia* spp.	3.0	0.3
Baccharis halimifolia	2.8	0.5	*Rubus* spp.	3.2	0.7	Unknown 5	3.0	0.4
Dichanthelium spp.	2.8	0.3	*Euthamia* spp.	2.7	0.7	*Limnosciadium* sp.	2.5	0.3
Polygonum spp.	2.6	0.3	*Juncus marginatus*	2.6	0.3	*Axonopus* spp.	2.2	0.5
Phyla lanceolata (Michx.) Greene	2.5	0.3	*Andropogon* spp.	2.4	0.4	*Phyla lanceolata* (Michx.) Greene	2.1	0.3
Rumex spp.	2.1	0.3	*Eleocharis* spp.	2.3	0.3	*Iva annua*	1.8	0.2
Rhynchospora caduca Elliott	2.1	0.3	*Panicum* spp.	2.2	0.3	*Sesbania* spp.	1.6	0.3
Andropogon spp.	2.0	0.3	*Phyla lanceolata* (Michx.) Greene	1.5	0.4	*Paspalum denticulatum* Trin.	1.6	0.2

[a] Exotic invasive species.

* Numbered species indicate those unidentified but distinguished from congeners.