CONSERVATION ISSUES

Application of an Expert System Approach for Assessing Grassland Status in the U.S.-Mexico Borderlands: Implications for Conservation and Management

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ABSTRACT: Grasslands around the world have experienced dramatic decline for more than a century. The spatial extent and condition of grasslands in many regions of the world are poorly understood because they lack conservation priority relative to other ecological systems. We developed a simple, yet broadly applicable rapid assessment expert system approach that can be used to assess these aspects of grassland status for improved conservation planning and management. Here we apply our approach to the semi-arid grasslands of the Arizona, New Mexico, and Mexico borderlands. We specifically used a combination of expert input and field verification. The accuracy of the expert input was 71%-77%. Native grasses dominate nearly 50% of identified grasslands. However, over two-thirds of these grasslands have experienced a moderate level of shrub encroachment. These areas may be restorable to open grassland with use of prescribed fire. Non-native perennial grasses are common to dominant on 12% of the study area. More than 67% of native grasslands occur on private and state land where there is little legal protection. In a region experiencing substantial development pressure, grasslands are at risk of becoming increasingly fragmented and vulnerable to non-native species invasion. Now, with our approach to assessing grassland status, conservation practitioners and partners are better positioned to protect and restore the remaining grasslands in the U.S.-Mexico borderlands and elsewhere.

Index terms: conservation planning, expert mapping, restoration potential

INTRODUCTION

Grasslands around the globe have long been and continue to be affected by both natural disturbances and anthropogenic influences (Archer 1989; Whitford 1997; White et al. 2000). Between 1700 and 1992, approximately 20% of the world's grasslands were converted to other land uses and cover types (Ramankutty and Foley 1999). In the once expansive Great Plains of the United States, surveys suggest that 99.9% of tallgrass prairie, 46.4% of mixed prairie, and 21.1% of shortgrass prairie have been lost or otherwise degraded since 1830 (Samson and Knopf 1994). Consequently, many species native to North American grasslands are in decline (Parmenter and Van Devender 1995). Vegetation change or loss in grasslands also negatively affects associated riparian systems by reducing aquifer recharge in the surrounding watershed, exacerbating erosion, and facilitating sedimentation of streams (Debano et al. 1984; Schlesinger et al 1990; Abrahams et al. 1994).

A method for gaining a comprehensive understanding of grassland extent and status is critical for effective conservation planning and management. Satellite imagery has been used to evaluate regional grassland extent with limited success (Kepner et al. 2000; Muldavin et al. 2001). However, technical problems involving the identification of distinct spectral signatures for plant species and geologic substrates, difficulties in distinguishing certain shrubs from perennial grasses, and the timing of image acquisition in relation to vegetation phenology often undermine the overall accuracy of image-based analyses. Cloudless image acquisition, preparation, and field calibration also can be time-consuming and costly (Warren and Hutchinson 1984; Elvidge and Lyon 1985; Huete and Jackson 1987; Huete 1988; Ringrose et al. 1999). The accurate interpretation and assessment of grassland status, such as the extent of shrub encroachment and the ratio of native versus non-native grasses, is often not possible with remotely-sensed data.

Here we describe a relatively simple, yet broadly applicable, field-based approach that can be used to conduct a regional scale, rapid assessment of the ecological status of grasslands. We applied this approach to the borderlands of south-central Arizona, southwestern New Mexico, and northern Mexico. A number of studies have contributed to the understanding of site level grassland dynamics in this region (Hennessey et al. 1983; Bahre and Shelton 1993; Brown et al. 1997; Valone and Kelt 1999; Valone et al. 2002; McClaran 2003). However, few have attempted to more fully characterize the spectrum of grassland changes across the borderlands. The method specifically allows for the mapping and assessment of the extent of shrub encroachment, non-native grass invasion, and overall condition of grasslands, including those with restoration potential, across the region. Our study is timely in that the borderlands' grasslands are being lost to development at an increasingly rapid rate. To evaluate the risk of future degradation and loss, we conducted an analysis of the ecological status of grasslands relative to their land management and legal protection status. The final results of this study provide a cost-effective and spatially explicit framework for grassland conservation planning and management in the borderlands of the southwestern United States, northern Mexico, and elsewhere.

METHODS

Study region

The arid and semi-arid grasslands of the U.S.-Mexico borderlands harbor a suite of biodiversity from all taxonomic groups (McClaran 1995; Parmenter and Van Devender 1995). These grasslands have experienced dramatic and spatially non-uniform vegetation changes during the last 130 years. Changes include decreased native perennial grass abundance and cover, conversion to shrubland, increased presence of non-native species, and fragmentation (Humphrey 1958; Buffington and Herbel 1965; Hastings and Turner 1965; Bahre 1991). Explanations for alteration of grasslands include conversion to farmland, poorly-managed livestock grazing, regional climate change, suppression of wildfire, and other activities related to human development (Hastings and Turner 1965; Swetnam 1990; Bahre 1995; Brown et al. 1997; McPherson and Weltzin 2000). Exacerbating the problem, demographic projections and population movement patterns within the southwestern United States suggest that grassland habitats will face particularly strong development pressures in the next decade (U.S. Bureau of the Census 2000; Theobald 2003). Suburban and ex-urban development fragment habitat and migration corridors linking adjacent mountain ranges (Theobald et al. 1997) and prevent the restoration of important ecological processes, such as fire, that are critical to maintaining grasslands (Heckert 1994; McPherson 1995).

We specifically studied the grasslands included in the Apache Highlands ecoregion, a 12 million hectare planning unit straddling the border of southeastern Arizona, southwestern New Mexico, northern Sonora, and northwestern Chihuahua (Marshall et al. 2004) (Figure 1). The Apache Highlands encompasses the Verde River and Big Chino valleys of central Arizona to the Mogollon Rim in the north and south through the isolated mountain ranges of northern Mexico to the main cordillera of the Sierra Madre Occidental. The ecoregion was defined by The Nature Conservancy based on Bailey (1995) and is roughly composed of the western portion of Bailey's Chihuahuan Desert ecoregion and the northern portion of the Madrean ecoregion. The southern boundary corresponds to the northern distributional limit of Sinaloan tropical deciduous thorn scrub (Brown and Lowe 1980). The Apache Highlands ecoregion encompasses the sky island mountain ranges of the borderlands and the Southwest's largest expanse of plains and semi-desert grasslands (Felger and Johnson 1995).

Mapping grasslands in the U.S. portion of study region

An overview of the steps taken to conduct the borderland grassland assessment is provided in Figure 2. We first interviewed 24 range management specialists from 10 federal and state agencies, institutions, and non-governmental organizations (Table 1). All experts had considerable knowledge of grassland extent and conditions in their local jurisdiction. We asked the experts to base their delineation on their knowledge of local conditions, including site potential for supporting grassland, and the existence of other data sources (e.g., historical accounts, survey points, repeat aerial and still photography, and historical monitoring data). To standardize and facilitate the expert-based characterization of grasslands, we developed five grassland classes, or condition types, using information from range management sources and experts (McPherson 1997; D. Robinett, Natural Resources Conservation Service, pers. comm.) (Table 2). Experts delineated polygons representing the five grassland condition types on hard copy regional maps referenced at scales between 1:100,000 and 1:250,000 containing elevation contours and location. The GAP vegetation system, a widely-employed vegetation classification for Arizona and New Mexico, was also printed on the maps to serve as a general reference to vegetation type (Thompson et al. 1996; Halvorson et al. 2001). However, it should be noted that the developers of the GAP map did not incorporate information on the amount of shrub cover nor presence of non-native grasses as a means of classifying grassland association types. For our study, status-related information was derived from the expert's knowledge of the landscape and other data sources. The expert-drawn grassland condition polygons were digitized using ArcView GIS.

Field verification of U.S. grasslands

We conducted 17 field trips in the U.S. portion of the study area to verify the expert-derived grassland maps. To ensure that selection of sampling sites was as random as possible, we first generated 25-30 random points on the grassland maps (printed at reference scales of 1:100,000 and 1:150,000). On average, we sampled 11 points per trip within close proximity to the random points. When sampling in the vicinity of a road, we walked 100-200 m perpendicular to the road before sampling to eliminate the effect of any disturbance from the road on vegetation. We estimated percent canopy cover of all shrubs, of mesquite (*Prosopis velutina* Woot.) and juniper (*Juniperus sp*.), and of perennial grasses to the nearest 5%. In order to calibrate cover estimations, we estimated percent canopy cover using 3-6 pace transects, each 150-200 meters in length, for a total of 400-600 sampling points per site for the first seven field trips (Avery 1975; Bonham 1989). On later trips, ocular estimates of canopy cover were made, interspersed periodically with pace transects to maintain consistent cover estimates. Others have also implemented visual estimation to expedite the sampling process, particularly when the study area encompasses a large geographic area (Anable et al. 1992; Muldavin et al. 2001). In total, we sampled 190 points throughout the study area.

The abundance of non-native perennial lovegrasses, primarily Lehmann (*E. lehm-*

Figure 1. The Apache Highlands ecoregion study area.

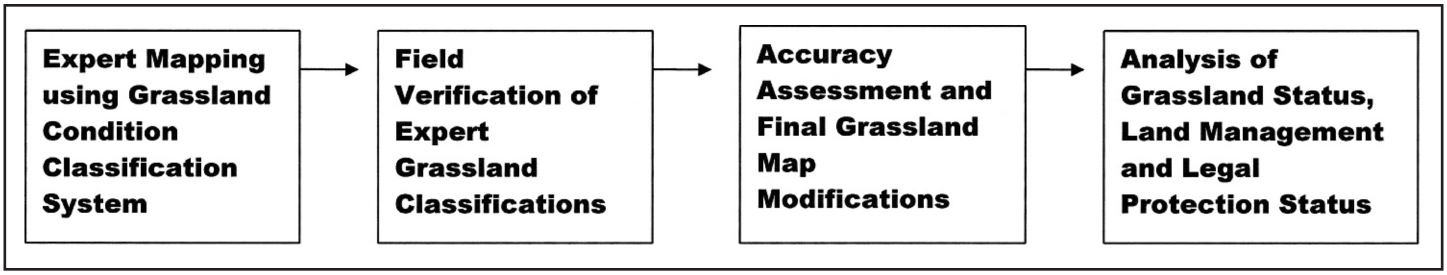

Figure 2. Flow chart of the U.S.-Mexico borderlands grassland assessment methodology.

anniana Nees.) and Boer (*E. curvula var. conferta* Stapf.*)*, was ranked at each sampling site on an ordinal scale as dominant, common, scattered, rare, or absent (Shaver et al. 2000; see Table 2 for a detailed explanation of ranks). To make this scale more quantitative and enhance consistency between observers and sampling trips, we used a modified frequency sampling method to obtain a combined measure of species density and dispersion. We recorded the presence or absence of the non-native grasses in four 3-m x 3-m quadrats placed at 15 m intervals along a transect line (48 quadrats per transect). Transects were approximately 180 m in length, with a total of 3-4 transects and 144-192 quadrats per field site. This sampling method was used during the first seven field verification trips. As with the canopy cover estimations, we subsequently employed ocular estimations of rank abundance interspersed with periodic quantitative sampling.

If a discrepancy between the expert map and our assessment of the condition class arose, we attempted to clarify the discrepancy with additional mapping using 1:100,000 scale topographic maps. A portion of the field verification was conducted during the spring-summer dry season following several years of below-average precipitation. Although this was likely to affect perennial grass cover more than shrub cover, the grassland types were distinguished primarily on the basis of shrub canopy cover. Many shrub species undergo cycles of population increase and decline in response to variation in precipitation, even on an intra-decadal time scale (McClaran 2003). However, shrub mortality at our field sampling sites, as indicated by dead, dried individuals, was low and insufficient in magnitude to cause a change in our determination of grassland condition class. Furthermore, at three long-term monitoring sites in southeastern Arizona, periodic eruptions and declines of short-lived shrub species did not change how grassland sites were classified over a 14- to 36-year time period. This was due to the presence of other shrub species whose populations were stable or slowly increasing over time (McClaran 2003; D. Gori, unpubl. data). Furthermore, we emphasize that our final map of grassland status was generated with the intention of relatively short-term use, with the additional aim of updating the map every 10 years.

Accuracy assessment

We assessed the accuracy rate of the expert grassland designations for the U.S. portion of the study area by generating an error matrix of omission and commission. In ArcView, we conducted an overlay operation using the field sampling points and coincident expert designated grassland polygons. We then compared our field-verified assessment classifications with those of the experts in a matrix format, a method that has been used to assess the accuracy of remotely-sensed vegetation classifications (U.S. Department of Agriculture, Forest Service 1995). In total, we evaluated 182 sample points, after removing eight points first classified by experts as "unknown." To derive a percent accuracy rate, we simply divided the number of commissions by the total number of sampling points.

We also generated a second error matrix using data from 52 sample sites collected from long-term monitoring areas in the borderlands between 1998 and 2002, in addition to our 182 points (for a total of 234 points). We justified the use of these

Table 1. Agencies and organizations represented by expert consultants.

Arizona State Lands Department, USA
Instituto del Medio Ambiente y el Desarollo Sustentable del Estado de Sonora (IMADES), Mexico
Instituto Nacióinal de Investigacióhes Forestales, Agricolas y Pecuarias (INIFAP), Mexico
New Mexico Natural Heritage Program, USA
Pima County, Tucson, AZ, USA
The Nature Conservancy in Arizona, USA
United States Bureau of Land Management (BLM)
United States Department of Agriculture's Forest Service
United States Department of Agriculture's Natural Resources Conservation Service (NRCS)
University of Arizona, USA
University of Arizona Cooperative Extension Service, USA

Table 2. Classification of grassland condition. Experts were asked to base their opinion on their field-based experience, knowledge of local soil conditions and potential for supporting a grassland, and awareness of other data sources (historical accounts, survey points, photographs, etc.). Experts were also asked to identify grasslands with unknown condition types.

GRASSLAND CONDITION	DESCRIPTION
OPEN-NATIVE	< 10% shrub cover whose herbaceous component is entirely or predominantly native perennial grasses and herbs (D. Robinett, pers.comm; modified from Anderson et al. 1998).
RIPARIAN NATIVE	Riparian grassland dominated by giant sacaton (*Sporobolus wrightii* Munro ex Scribn.).
RESTORABLE NATIVE	Native perennial grasses and herbs dominate with 10-35% total shrub cover and mesquite or juniper cover < 20% (D. Robinett, NRCS, pers. comm.; McPherson 1997). A key characteristic of this type is its restoration potential back to open-native grassland by using prescribed burning to reduce shrub cover. In some cases, grazing rest may be required to accumulate sufficient fine fuels to carry a fire (Gori and Backer 2005).
NON-NATIVE	Non-native perennial grasses (primarily *Eragrostis lehmanniana* Nees and *Eragrostis curvula var. conferta* Stapf.) are classified as common or dominant. The grasses were considered rare at a site if they occurred in <10% of the frequency quadrats sampled, scattered if the frequency was 10-20%, and common or dominant if the frequency was >20%. Common was distinguished from dominant based on whether the combined canopy cover of non-native lovegrasses was or was not greater than any other single species at the site. Total shrub cover could be either < 10% for an open-non-native grassland or between 10-35% and mesquite or juniper cover < 20% for a shrub-invaded non-native grassland (D. Robinett, NRCS, pers. comm.).
HISTORIC	Greater than 20% canopy cover of mesquite and juniper combined and/or > 35% total shrub cover; (D. Robinett, NRCS, pers. comm.; McAuliffe 1995; McPherson 1997); type conversion to shrubland is either permanent or will require 40+ years of livestock exclusion for partial recovery of perennial grasses (Hennessey et al., 1983; Valone et al., 2002). The historical time scale relates to vegetation accounts provided by vegetation inventories from the mid to late 1800s (Gehlbach 1993).

previously collected sample points because the methods used to estimate canopy cover of shrubs and perennial grasses were similar to those we employed in the field, allowing us to categorize grassland condition class at these sites. There was an absence of non-native perennial grasses at these 32 sites.

Expert map modifications

We overlaid the digitized expert-drawn grassland maps with a hexagonal grid (50 ha cell resolution) as a way to facilitate the modification of expert grassland classifications. We then modified the incorrect grassland designations hexagon by hexagon using the data collected at the field sampling points and our supplemental field mapping. After all modifications were made, we aggregated the individual hexagon expert maps by grassland condition types. The resulting expert map files were merged to generate a single grassland map for the U.S. portion of the study area.

Grasslands in northern Mexico

Field staff applied the previously described expert-based mapping approach, in addition to supplemental analysis of Landsat TM satellite data (30 m cell resolution), to delineate the extent of different grassland types in Mexico (Aguirre et al. 2002). This mixed approached was employed because: (1) the extent of non-native love-grass invasion is minimal in northern Mexico as compared to the U.S. and (2) many of the areas occupied by grasslands are remote, privately owned, and not easily accessed by vehicle. Field sampling and verification were conducted by staff from the Mexican state agency, the Instituto del Medio Ambiente y el Desarollo Sustentable del Estado de Sonora (IMADES). Training sessions with field personnel were conducted in the United States and Mexico to ensure that vegetation sampling techniques would be consistent across the border and that the grassland condition classification system was uniformly applied (Table 2). The final digital map was subsequently appended to the U.S. grassland map to create a seamless cross-border spatial data set.

Analysis of land management and legal protection status

To evaluate the land management and legal protection status of grasslands in

Table 3. Results of the accuracy assessment of the original expert designated grassland condition class polygons. (a) Error matrix of omissions and commissions using field sampling data points (182 points), and (b) error matrix of omissions and commissions using field sampling points in addition to previously collected data from long-term monitoring sites (234 points in total) in the Apache Highlands ecoregion study area. Commissions are highlighted in bold, omissions are in regular type. Accuracy rate is calculated by dividing total commissions by the total number of sampling points.

(a)

EXPERT VERIFIED FIELD SAMPLING POINTS (columns); EXPERT DESIGNATED POLYGONS (rows)

GRASSLAND CONDITION	HISTORIC	NON-NATIVE	OPEN-NATIVE	RESTORABLE	RIPARIAN	TOTAL
HISTORIC	**23**	0	1	11	0	35
NON-NATIVE	4	**18**	0	5	0	27
OPEN-NATIVE	0	2	**32**	4	0	38
RESTORABLE	11	12	1	**55**	2	81
RIPARIAN	0	0	0	0	**1**	1
TOTAL	38	32	34	75	3	**182**

(b)

EXPERT VERIFIED FIELD SAMPLING POINTS (columns); EXPERT DESIGNATED POLYGONS (rows)

GRASSLAND CONDITION	HISTORIC	NON-NATIVE	OPEN-NATIVE	RESTORABLE	RIPARIAN	TOTAL
HISTORIC	**25**	0	1	11	0	37
NON-NATIVE	4	**18**	0	5	0	27
OPEN-NATIVE	0	2	**50**	4	0	56
RESTORABLE	11	12	1	**86**	2	112
RIPARIAN	0	0	0	0	**2**	2
TOTAL	40	32	52	106	4	**234**

the borderlands relative to their ecological status, we conducted a spatial analysis of the final grassland map and previously available GIS data delineating land management boundaries and legal protection levels (ALRIS 1998; Halvorson et al. 2001; Weinstein 2002). We also examined the relationship between the number of different land managers and the size of contiguous grassland patches. We did this to quantify the complexity of future conservation planning and management efforts across increasingly large geographic scales.

RESULTS

Accuracy assessment

We found that experts correctly identified grassland types in 129 of 182 field sampling points, giving an overall accuracy rate of 71% in an error matrix of omissions and commissions (Table 3a). In a second error matrix where we used sampling points from previous studies, we found correct identifications in 181 of 234 field sampling points, giving an accuracy rate of 77% (Table 3b).

Extent of borderland grasslands

Our study identified 4,928,448 ha as either extant or historic grassland, accounting for 41% of the Apache Highlands ecoregion (Figure 3). Of this, 64% (3,146,000 ha) is extant grassland and the remaining 36% (1,782,000 ha) is considered historic, or former, grassland. Our sampling data indicated that perennial grass canopy cover was always < 3%, but typically < 1% in all of our identified historic grasslands. Over 150,000 ha of potential grassland, or 4% of the area identified as extant or historic

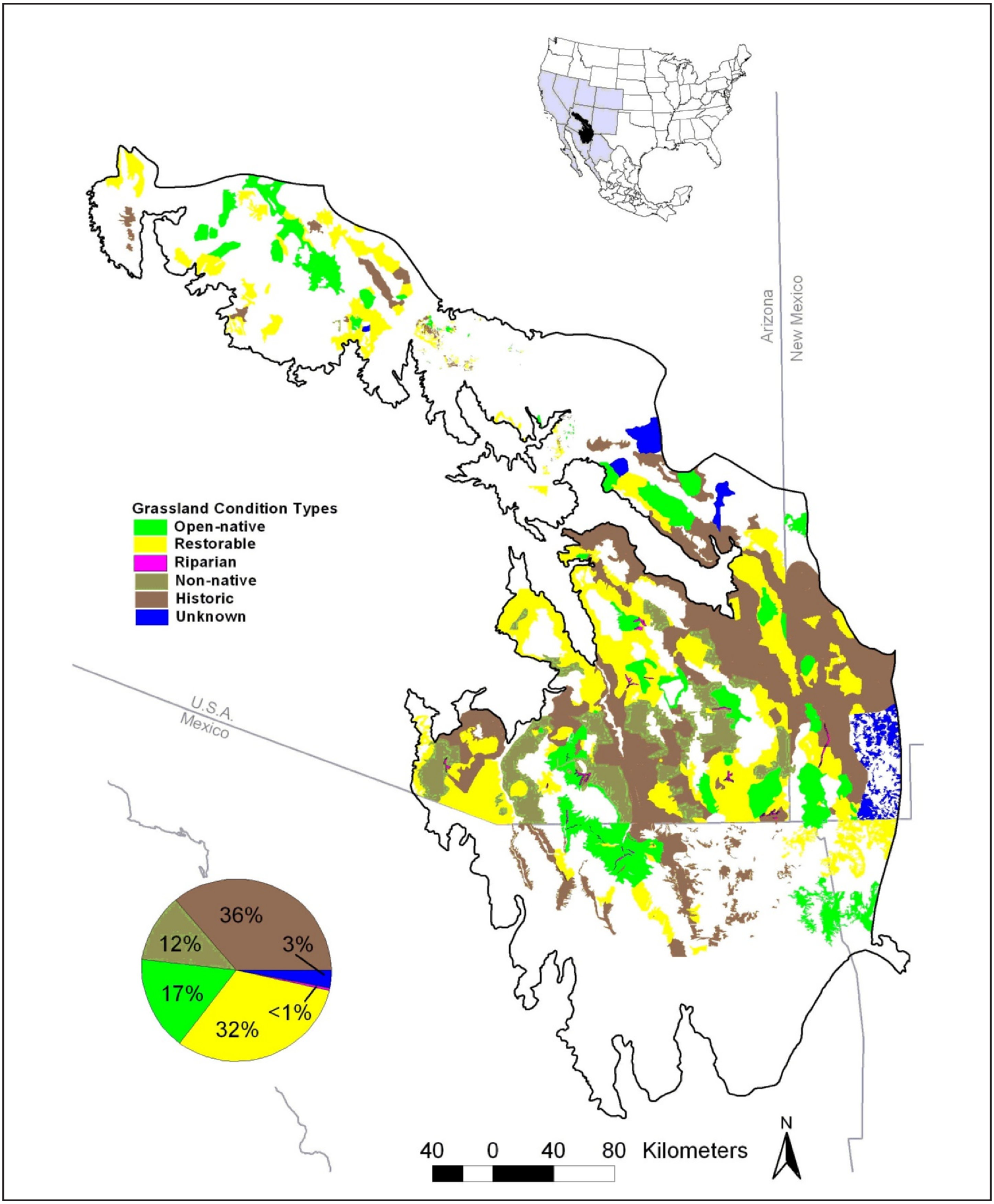

Figure 3. Final map of grassland condition types in the Apache Highlands ecoregion study area.

grassland, was classified as an unknown grassland condition type by experts due to a lack of current information or experience in the area. Total area values for individual condition types are provided in Table 4.

Extent of extant grasslands

The open-native type covers 26% of extant grasslands, the restorable native type covers 50%, and the riparian type covers < 1 %. Together, all native grassland types (open-native, restorable, and riparian) account for 76% of these grasslands.

Non-native lovegrass species (Lehmann and Boer) are now common or dominant on 19% of extant grasslands. The distribution of non-native grasslands was not uniform but was restricted to southeastern Arizona. We found no regional-scale non-native grass invasion on the Mexico portion of the study area.

Land management and legal protection status

As expected, we found that the number of land management entities increases significantly with grassland patch size (N = 375, patches < 40 ha removed from analysis; $r^2 = 0.43$; $p < 0.0001$; $F = 280.3$). The greatest proportion of identified grasslands in the U.S.-Mexico borderlands is under private ownership (Table 4). Federal and state-managed lands are ranked second, comprising roughly equal proportions of all identified grasslands, while the remainder is under tribal or another jurisdiction.

The distribution of native grassland types (open-native, restorable, and riparian) by land manager is provided in Figure 4. Most open-native and riparian grasslands are on private lands. Restorable native grassland is almost evenly distributed among federal, private, and state land management. Overall, native grasslands comprise approximately three-quarters of total extant grasslands on private, federal, and state-managed lands.

The overall distribution of native grassland types by legal protection status is provided in Figure 5. Nearly 90% of identified grasslands in the borderlands have low to no legally mandated land protection status that would prevent land cover conversion (Table 5). The same is true for open-native grassland on private land. On state-managed land, this value increases to nearly 100%. Most riparian grassland also occurs on private land, of which less than 5% has a high level of protection. The vast majority (over 90%) of riparian grassland has low to no level of legal protection. Slightly more than one-third of restorable grassland occurs on federally managed lands, indicating at least some legal protection. The remaining two-thirds of restorable grassland (in nearly equal proportions on private and state-managed lands) have almost no legal protection.

DISCUSSION

Expert mapping and accuracy assessment

Many researchers do not view the use of expert opinion in scientific studies as a sufficiently rigorous method. However, we believe that our use of the approach was warranted given the absence of viable alternatives for meeting our specific objectives (e.g., the development of a rapid and cost-effective assessment of regional grassland status for immediate conserva-

Table 4. Land manager by grassland condition type in area units of hectares. Percentages were calculated based on sum totals of each column. The percentages in the bottom row represent overall proportions of grassland condition types.

MANAGER	OPEN-NATIVE	RIPARIAN	RESTORABLE	NON-NATIVE	HISTORICAL	UNKNOWN	TOTAL
FEDERAL	109,305	1,808	526,296	131,294	516,170	49,490	1,334,363
	(13%)	(8%)	(34%)	(22%)	(29%)	(32%)	(27%)
PRIVATE	487,728	15,368	499,344	259,649	599,171	42,039	1,903,299
	(59%)	(72%)	(32%)	(44%)	(34%)	(27%)	(39%)
STATE	140,019	4,262	480,151	202,772	526,055	6,048	1,359,308
	(17%)	(20%)	(31%)	(34%)	(30%)	(4%)	(28%)
TRIBAL	85,968	0	49,891	0	125,914	57,462	319,235
	(10%)	(0%)	(3%)	(0%)	(7%)	(37%)	(6%)
OTHER	47	0	3,222	447	8,528	0	12,244
	(<1%)	(0%)	(<1%)	(<1%)	(<1%)	(0%)	(<1%)
TOTAL	823,067	21,438	1,558,904	594,163	1,775,837	155,039	4,928,448
(% of row)	(17%)	(<1%)	(32%)	(12%)	(36%)	(3%)	(100%)

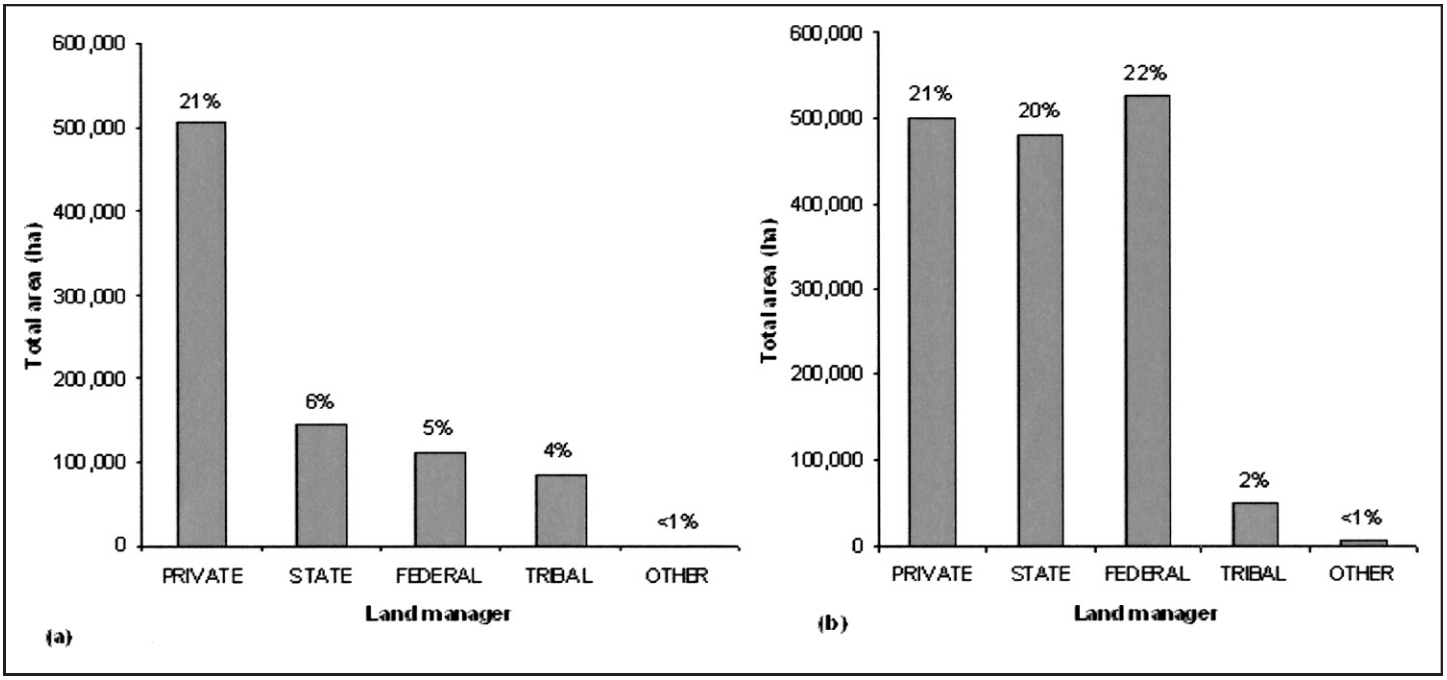

Figure 4. Land management status of native grasslands in the Apache Highlands study area: (a) open-native and riparian grasslands, and (b) restorable native grasslands. Percentages are the proportion of the sum total area of all native grasslands in the study region (2,403,409 ha) by land manager.

tion planning and management). Moreover, expert-based methods have been employed with success in other studies reported in the literature (Hannah et al 1998; Cowling et al. 2003; Brown et al. 2004).

We also conducted a rigorous and repeatable field verification process of our expert-derived data. In a follow-up study to the present study, the method was extended to the remainder of the state of Arizona (Schussman and Gori 2004). With a set of completely different experts, the accuracy rate was comparable at 74%. These estimates compared favorably to those for land cover maps without condition classes

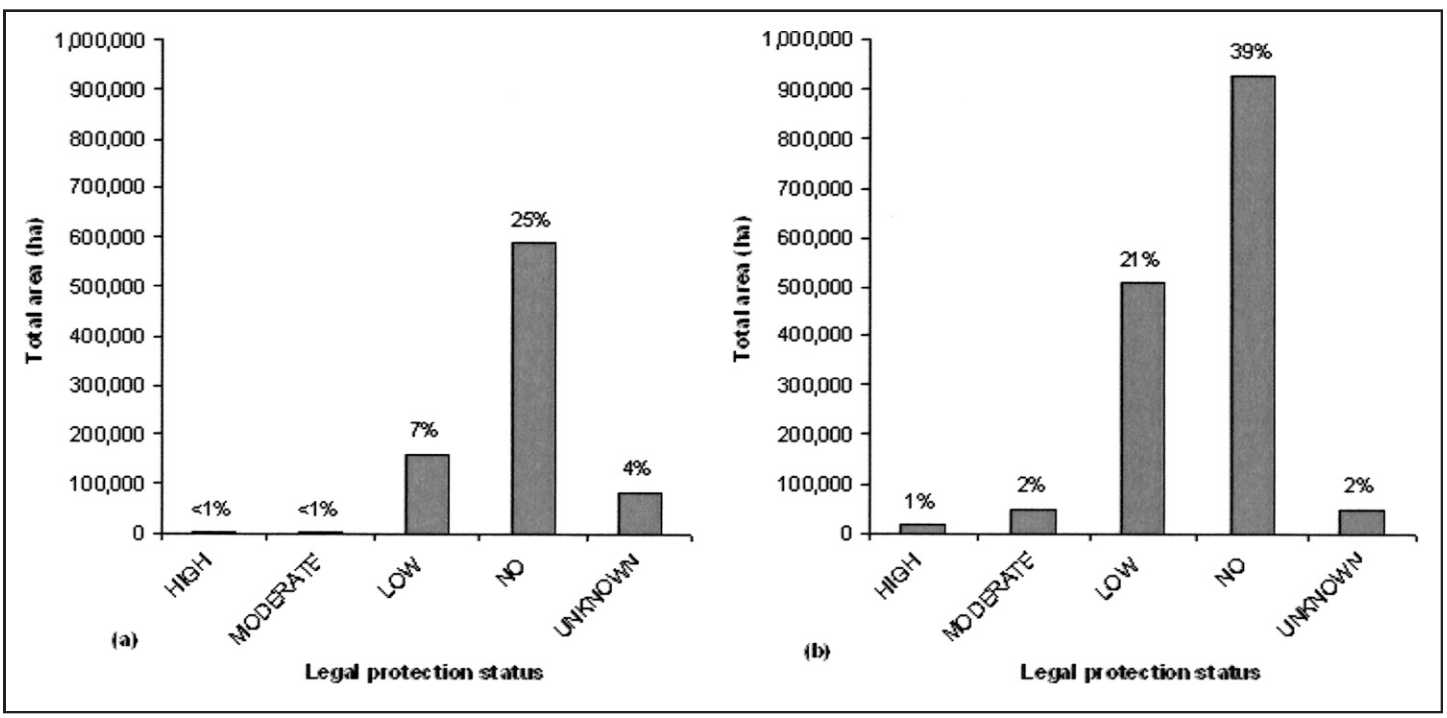

Figure 5. Legal protection status of native grasslands in the Apache Highlands study area: (a) open-native and riparian grasslands, and (b) restorable native grasslands. Percentages are the proportion of the sum total area of all native grasslands in the study region (2,403,409 ha) by legal protection status. See the caption of Table 5 for definition of protection status level.

Table 5. Legal protection status by grassland condition type in area units of hectares. Percentages were calculated based on sum totals of each column. The percentages in the bottom row represent overall proportions of grassland condition types. High = USGS Gap protection level 1 or the highest level of legally mandated permanent land protection, Moderate = USGS Gap protection level 2 or lands with permanent protection but which may receive uses or management practices that degrade the quality of existing natural communities; Low = USGS Gap protection level 3 or lands with some degree of protection but that are subject to extractive uses which may involve land cover clearing; No = USGS Gap protection level 4 or lands without legal protection (USGS GAP Analysis Program 2002; Weinstein 2002).

PROTECTION STATUS	OPEN-NATIVE	RIPARIAN	RESTORABLE	NON-NATIVE	HISTORICAL	UNKNOWN	TOTAL
HIGH	2,793	1,701	18,098	42,846	26,671	0	92,108
	(<1%)	(8%)	(1%)	(7%)	(2%)	(0%)	(2%)
MODERATE	5,204	202	53,578	3,808	22,532	3,181	88,505
	(1%)	(1%)	(3%)	(1%)	(1%)	(2%)	(2%)
LOW	154,815	4,158	508,506	107,597	483,701	50,865	1,309,643
	(19%)	(19%)	(33%)	(18%)	(27%)	(33%)	(27%)
NO	574,236	15,376	928,518	439,911	1,111,116	43,530	3,112,687
	(70%)	(72%)	(60%)	(74%)	(63%)	(28%)	(63%)
UNKNOWN	86,020	0	50,205	0	131,818	57,462	325,504
	(10%)	(0%)	(3%)	(0%)	(7%)	(37%)	(7%)
TOTAL	823,067	21,438	1,558,904	594,163	1,775,837	155,039	4,928,448
(% of row)	(17%)	(<1%)	(32%)	(12%)	(36%)	(3%)	(100%)

previously derived from analysis of Landsat satellite imagery, which ranged between 57% and 78% (e.g., Lauver and Whistler 1993; Ringrose et al. 1999; Kepner et al. 2000; Halvorson et al. 2001). With subsequent field-verified map modifications, the accuracy of our final grassland map for the Apache Highlands ecoregion was further improved. We emphasize that our accuracy rate of 71%-77 % accounts not just for the correct classification of a grassland polygon as a vegetation type, but it also accounts for the correct assignment of the grassland's "condition" class, including shrub cover and abundance of non-native perennial grasses. To date, there are no other studies that include this important measure of grassland status in their classification of vegetation types.

Status of borderland grasslands

Open-native grassland constitutes less than 20% of grasslands identified in our study region, the Apache Highlands ecoregion. Taken together, however, all native types (open-native, riparian, and restorable) comprise nearly 50% of this area. Although shrubs have invaded nearly 70% of all identified grasslands, nearly one-third exhibit restoration potential with the combined use of grazing rest (for fine fuel accumulation) and subsequent prescribed fire.

Over the course of a century, we found that 36% of grasslands in the ecoregion experienced a conversion to shrubland ("historic grassland") as a result of factors ranging from climate change (Brown et al. 1997) to fire suppression (Bahre 1991). Although this figure may be an overestimate because some of this area may have always been shrubland – and future work should focus on refining this estimate – it is the first known attempt to identify the extent of grassland loss regionally. Furthermore, based on our field data, this value does not appear to depend on our use of a cover value for mesquite to distinguish restorable versus historic grassland. In fact, 37 of 39 field sampling sites classified as historic had greater than 35% total shrub cover. This suggests that mesquite invasion is usually accompanied by an increase in other shrub species as well. Accordingly, our study underscores the loss of grasslands in the borderlands as a function of shrub invasion and related factors (e.g., persistent low perennial grass cover due to drought and livestock grazing followed by soil erosion). Other site-based studies corroborate this conclusion (Buffington and Herbel 1965; Hennessey et al. 1983; Dick-Peddie 1993; Archer 1994; Brown et al. 1997; Muldavin et al.1998; Rich et al. 1999; Kepner et al. 2000). Additionally, our study quantifies shrub encroachment in a spatially explicit manner on a regional scale. We specifically did not quantify historic grassland that has been lost to agriculture, urbanization, and other activities directly attributable to humans. Observations suggest that, until recently, these impacts have been minimal (Archer 1989). We recommend that future research should consider these anthropogenic forces, given the rapid rate of development (Theobald 2003) and regional climate change now occurring in borderland grasslands (SRAG 2000).

Lehmann and Boer lovegrass species are now common or dominant in nearly 20% of extant grasslands. The distribution of these species is currently restricted to the U.S. portion of the study area. In addition to the fact that the U.S. Department of Agriculture made non-native lovegrass seed available and encouraged its planting beginning in the 1930s, this distribution is at least partially a function of their physiological tolerances and soil type preferences (Cox and Ruyle 1986). Recent modeling work using refined physiological parameters suggests that the two non-native lovegrass species will continue to expand in the United States and across the border into Mexico's grasslands (Geiger et al. 2003; Schussman et al. 2006). The two species are likely to invade grasslands that are becoming increasingly subject to human-induced fragmentation and disturbance (Burke and Grime 1996; D'Antonio et al. 1999). This is particularly true of grasslands located in the vicinity of the growing population centers of southeastern Arizona, where the construction of new roads may serve as conduits to invasion (Gelbard and Belknap 2003). While our results do not offer an explicit prediction of the path of invasion, they provide a basis for vegetation modeling or informed speculation on the future spread of non-native perennial grasses in grasslands.

Fire and the restoration of shrub-invaded grasslands

Both direct and indirect evidence point to the historical importance of frequent fire in maintaining borderland grasslands and limiting the expansion of shrubs (Humphrey 1958; Bahre 1985, 1991; McPherson 1995; Kaib et al. 1996). However, evaluating the use of fire as a restoration tool following more than a century of fire suppression and shrub encroachment is complex. Sufficient fine fuels must accumulate and, thus, grazing rest is often required prior to prescribed burning. In addition, the variable effects of prescribed fire across soil types and precipitation regimes are not well understood (Buffington and Herbel 1965; Dick-Peddie 1993; Gosz and Gosz 1996; Drewa and Havstad 2001; McClaran 2003; Geiger and McPherson 2005). Nonetheless, many studies have shown that prescribed burning can reduce the abundance and cover of dominant shrubs in borderland grasslands and can have beneficial effects on nutrient cycling and the availability of herbaceous forage to herbivores (Reynolds and Bohning 1956; Cable 1967; White 1969; Wright 1974; Bock and Bock 1992, 1997; McPherson 1995; Valone and Kelt 1999; Blank et al. 2003; Ruthven et al. 2003; Gori and Backer 2005).

Implications for grassland conservation and management

Here we have demonstrated a rapid, regional assessment approach to evaluating the extent and condition of grasslands that is straightforward in application and is instructive in its ability to produce meaningful results for conservation planning and management in a time and cost effective manner. Moreover, our method recently benefited from further refinement and wider application in a vegetation study conducted for the Bureau of Land Management by The Nature Conservancy in New Mexico and, in Arizona, in an assessment of the remainder of the state's grasslands (Schussman and Gori 2004).

Our results suggest that private land owners are critical to native grassland conservation in the U.S.-Mexico borderlands. However, conducting conservation-based work on private land can involve a host of legal and political complexities (Hilty and Merenlender 2003). Others have discussed the use of conservation-oriented tools, such as conservation easements, as a means of influencing management practices on private land (Wright 1994; Bowers 1999). Our results also suggest that federal conservation partners, particularly the U.S. Forest Service and U.S. Bureau of Land Management, should be engaged in the restoration of the borderlands' native, shrub-invaded grasslands.

Ultimately, the conservation and restoration of native grasslands will require that conservationists work with a variety of land managers. As reported, the number of land management entities increases with grassland patch size. This suggests that the complexity of conservation planning, restoration, and management is likely to increase as a function of the size of unfragmented grasslands. This is particularly true for open-native sites in our study region, possessing the second largest median patch size (4570 ha, after 9626 ha for non-native grassland patches). Nevertheless, with the results of this study in mind, conservation workers can now be more strategic in priority setting, in the establishment of conservation partnerships between different land managers, and in the effective allocation of scarce financial resources for land management and restoration activities.

Few studies have directly addressed the need to understand land management and legal protection status of their study areas, particularly those offering a regional perspective, as a means of informing conservation practitioners in the priority setting process (Sanderson and Redford 1997; Bowers 1999; Hilty and Merenlender 2003). This is particularly true when compared to the body of work focused on conservation methods and analyses (e.g., Noss 1983, 1996; Bedward et al. 1992; Soulé and Noss 1998; Margules and Pressey 2000; Myers et al. 2000; Groves et al. 2002; Sanderson et al. 2002; Redford et al. 2003). This is surprising given that 71% of land in the contiguous United States is privately owned (Hilty and Merenlender 2003) and, thus, unlikely to afford higher levels of legal protection. This is also alarming in view of the paucity of conservation-based studies conducted on private lands even though these lands contain habitat for 95% of federally-listed endangered species (Hilty and Merenlender 2003). Our work is unique in that it highlights the need for two distinct yet parallel conservation activities: (1) the legal protection of native grasslands from land cover fragmentation and (2) the restoration and management of these grasslands using fire to prevent further loss of grasslands to shrub encroachment. Each of these is equally important because without legal protection, grasslands are at risk of becoming increasingly fragmented, making these and adjacent lands more difficult to restore and manage with fire.

Perhaps most importantly, the results of our study already have been used to direct conservation action. For example, the assessment facilitated the implementation of land protection in northern Mexico (e.g., acquisition of Los Fresnos Ranch, a significant grassland site) and was used to evaluate priority sites for endangered species reintroduction (e.g., the black-tailed prairie dog (*Cynomys ludovicianus)* by the Arizona Game and Fish Department (AGFD)). It also has been used to identify a large and unfragmented grassland in the Chino Valley of Central Arizona where, in a collaborative effort between a private land owner, AGFD, and The Nature Conservancy, prescribed burning for restoration purposes is now being conducted. Furthermore, this work has directed funding to grassland restoration and purchase of conservation easements in Arizona through the National Resource Conservation Service's Farm Bill programs.

CONCLUSION

The classic basin and range physiography of the Apache Highlands ecoregion is often described as being composed of mountain islands surrounded by desert seas (Gehlbach 1993). This analogy underestimates the importance of semiarid grasslands, or seas, in comparison to the mountain "sky" islands. As with grasslands found throughout the world, grasslands in the Apache Highlands are at risk of continued shrub encroachment and becoming increasingly fragmented and vulnerable to non-native species invasion as the ecoregion experiences increasing pressure from development. Now, with a new expert system approach for assessing grassland status, conservation practitioners and partners are better positioned to protect and restore the remaining native grasslands in the United States-Mexico borderlands and elsewhere.

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