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RESEARCH ARTICLE

Factors Affecting Revegetation of Oil Field Access Roads in Semiarid Grassland

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Abstract

We assessed vegetation recovery on access roads removed after well abandonment in an active oil-producing region of northern Great Plains grasslands. We compared extant vegetation on 58 roads, restored 3–22 years previously, to records of species seeded on each and to adjacent, undisturbed prairie, to evaluate main differences between the restored and adjacent community and to explore patterns in the restored plant community over time. The restored plant community was dominated by low richness of seeded non-native and native grasses and forbs, whereas adjacent prairie had numerous, abundant native graminoids and shrubs and higher richness of native forbs. Cover of seeded species on roads was double that of colonizing species. Disparity in cover of dominant native grasses between the adjacent community and relatively narrow restored roadway suggests that conditions for germination and survival in roadbeds are poor. This is at least partly due to persistence of seeded species. Differences in restored plant composition over time were best explained by changes in species seeded, from non-natives to natives, and secondarily by successional shifts from ruderal to perennial non-seeded species. Of the 30 species seeded at least once on these roads, only 10 were commonly used. The long-term influence of seeding choices in grassland road restorations implies that improvements in these practices will be critical to reversing ecological impacts of roads.

Key words: **colonization, establishment, long-term outcomes, mixed-grass prairie, native versus non-native species, revegetation, road removal, road restoration, seed selection.**

Introduction

Roads, as pervasive and usually long-term landscape features, can have cumulative negative effects on biological communities and processes that are spatially disproportionate to the extent of their linear corridor (Forman & Deblinger 2000; Trombulak & Frissell 2000). Effects on plant communities include shifts in species composition, reduced native species richness, and invasion of non-native plant species (Tyser & Worley 1992; Angold 1997; Gelbard & Belnap 2003; Watkins et al. 2003) caused by alterations in factors such as hydrology, soil properties, nutrient and light availability, and dispersal (e.g. Greenberg et al. 1997; Parendes & Jones 2000). Many of these effects extend to surrounding intact communities, if present (Gelbard & Harrison 2003). Restoring roads to a pre-disturbance condition that does not prolong ecological impacts is therefore essential (Zink et al. 1995; Switalski et al. 2004).

Reversing road effects through removal and restoration is beginning to be tested (Switalski et al. 2004; McCaffery et al. 2007). Research has focused on techniques for short-term minimization of erosion and sedimentation and establishment of vegetative cover on logging roads in forests of the United States and Canada (e.g. Kolka & Smidt 2004). It is unclear how these findings will apply to other ecosystems and whether results will be long-term.

Road removal has also been done in association with reclamation of oil production disturbances across much of the Great Plains of North America (USDI & USDA 2006), although no published studies exist. Recent acceleration of petroleum industry activity has been estimated at 50–60% above the 10-year average in some areas (M. Sexton 2006, USFS, Dickinson, ND, personal communication). The accompanying upswing in construction and eventual restoration of extensive networks of roads could have significant impacts on surrounding native grasslands, depending on its success.

Studies of restoration and revegetation in grasslands have typically taken place in former agricultural fields or surface-mined lands (Wali 1999; Bakker et al. 2003; Polley et al. 2007; Fagan et al. 2008). In these settings, basic grassland functions, such as soil stabilization, wildlife habitat, and livestock forage, can be restored by planting standard mixtures of native and non-native plants (Schuman et al. 1990; McLauchlan et al. 2006). However, some species or cultivars used in these mixtures can compete with local native species, invade native prairie, or depress or alter genetic diversity (Christian & Wilson 1999; Bakker & Wilson 2001; Hammermeister

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2001; Gustafson et al. 2004). Locally collected native seed mixtures may circumvent some of these problems. Yet, even when native species are used, the ability to restore diversity of native grassland remains elusive because dynamics of seed addition, seed dispersal, resource use, competition, and disturbance are not fully understood (Pyke & Archer 1991; Martin et al. 2005; Polley et al. 2005; Dickson & Foster 2008).

Roads, in contrast to mined or agricultural lands, are linear corridors with a higher ratio of edge to disturbed area. This characteristic may speed spontaneous colonization and succession because propagule sources have greater contact with disturbed soils, implied by studies of naturally recolonized abandoned roads (Jaynes & Harper 1978; Cotts et al. 1991). These processes, however, could be impeded by the semiarid and highly variable climate of the Great Plains (Call & Roundy 1991). In mixed-grass prairie restoration experiments, Bakker et al. (2003) found that significant variation in establishment of seeded native grasses by site and year could be explained by differences in summer precipitation. Similarly, studies of road cuts in semiarid environments have found climate-related limitations to establishment of colonizing plants (Bochet & García-Fayos 2004).

This study takes advantage of a multi-aged set of 58 previously restored roads in the Little Missouri National Grasslands (LMNG), United States, where restoration after oil and gas development is intended to minimize the environmental impact of these activities. We compared extant vegetation on these roads to records of species seeded on each, as well as to the surrounding undisturbed plant community. The specific objectives were to (1) investigate the extent to which revegetation practices are resulting in plant communities similar to the surrounding grasslands; (2) identify factors that limit reassembly of these plant communities; and (3) evaluate the selection and compare the establishment of seeded species. More precise information on long-term restoration outcomes will inform restoration decisions for removed grassland roads.

Methods

Study Area and Restoration Background

Our study area was the LMNG (405,000 ha; between lat 46°16′N and 48°08′N, long 102°39′W and 104°02′W) in the North American Great Plains. The vegetation is predominantly mixed-grass prairie on rolling uplands or alluvial flats, but also includes *Artemisia* (sagebrush)-dominated floodplain terraces and lowlands, shrubby hardwood draws and north-facing slopes, and sparsely vegetated badlands breaks (Barker & Whitman 1988). Soils vary with topographical features and sedimentary parent materials. Besides oil and gas development, grazing by cattle for beef production is the primary use of the grasslands.

The region has a continental to semiarid climate, with warm, windy summers (mean July temperature 21.1°C) and frigid winters (mean January temperature −10.7°C) (NOAA 2004). Mean annual precipitation is 378 mm, most of which is rainfall between May and July, although there is high variability within and among years in rainfall and snowfall (NOAA 2004). A summary of monthly Palmer Drought Severity Index data for west-central and southwest North Dakota for the years included in this study showed the following climatic variability: 1983—wet; 1984–1987—normal; 1988–1993—dry; 1994–1996—wet; 1997–1998—normal; 1999–2000—wet; 2001—normal (NOAA 2006).

Restoration activities on the LMNG are directly tied to petroleum extraction, which began in this region in the 1950s. When oil or natural gas wells are permanently abandoned, oil companies are required to restore the well pad and associated access road to approximate original vegetation and landform (USDI & USDA 2006). This process generally involves removing or burying surface materials, recontouring subsoil to approximate original contours, distributing stored topsoil, and seeding 3–7 grass and forb cultivars (USDI & USDA 2006). The composition of seed mixes shifted from non-native to native cultivars around the mid-1990s, following regulatory directives, although unauthorized use of non-native species still occasionally occurs (LMNG Botany Program, USFS, Bismarck, ND, unpublished study). Seeding rates range from about 10 to 20 kg/ha of pure live seed.

"Partial" or "interim" restoration often precedes the "final" restoration process described above (USDI & USDA 2006). This entails revegetation of road verges and exposed soil surrounding newly constructed well pads after drilling. Since this process is often done 10–20 years before final restoration, the plant species used are often different from those used for final restoration.

Site Selection

The focus of this research was to restore access roads associated with permanently abandoned oil wells from across the LMNG, within the U.S. Forest Service (FS) Medora and McKenzie management districts, western North Dakota, United States (Fig. 1). Road restoration information is typically stored with records of the associated well. We sought to minimize confounding historical and landscape-scale factors using four successive selection criteria, following these steps separately for each management district. First, we queried an FS database of all petroleum facilities on the LMNG for "Plugged and Abandoned Producer," that is, wells that previously produced oil but are now permanently abandoned and thus usually restored. Second, we eliminated wells that were 2 years old or less or that were described in other records as being non-producers ("dry holes") so as to focus on roads in use for more than 2 years before being restored. This step caught database errors of misclassified non-producing wells. Third, we used LMNG geographic information system (GIS) data and aerial photos to eliminate roads that crossed private or state land, old-fields of the introduced *Agropyron cristatum* (Crested wheatgrass), or extensive human-disturbed areas. Fourth, we eliminated short access roads (<100 m) so there would be sufficient area to sample between road intersections and well pads.

To distinguish planted from naturally colonizing species, we sought seeding records (seed tags or seed certification

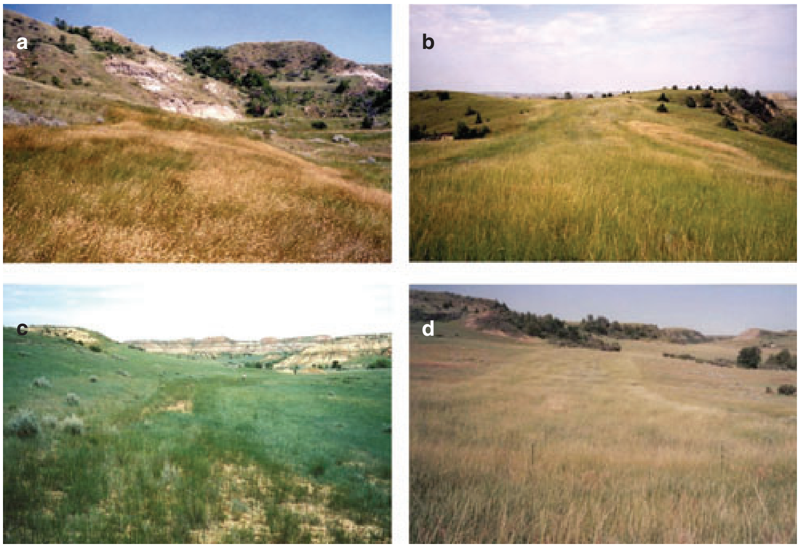

Figure 1. Four examples of restored road study sites. Final restoration was in 1986 (a), 1992 (b), 1995 (c), and 1996 (d). Little Missouri National Grasslands, North Dakota, United States.

lists) from the final road restoration for the 100 Medora and 170 McKenzie sites selected in our initial screening. Although required in LMNG well files, these records were often lacking: approximately 50% of Medora and 4% of McKenzie files contained seeding records. When these records were lacking, we interviewed LMNG personnel and seeding contractors to obtain details about seeding for as many sites as possible. Thirty Medora and 28 McKenzie sites met all criteria after field verification and were sampled ($n = 58$).

Vegetation Sampling

Fieldwork was done from mid-June to late August 2004 and 2005. At each study road, we measured its length, then placed 3–5 transects at random points along the road, with more transects on longer roads (100–500 m long = 3 transects, 501–1000 m = 4, >1000 m = 5). Transects crossing potentially confounding features (streams, road intersections, steep bluffs, wooded draws) were excluded. Transects were 100 m long and perpendicular to the road, with the midpoint intersecting the center of the road. On each half-transect, we sampled the vegetation within 1-m^2 plots at 3, 10, 25, and 50 m. As average width of disturbance was 21 m, this approach captured the recontoured and seeded roadbed, the boundary region, and undisturbed, adjacent prairie. In each plot, we identified all vascular plants and estimated cover for each species following Mueller-Dombois and Ellenberg (1974) using these classes: 1 (<1% cover), 2 (1–5%), 3 (5–25%), 4 (25–50%), 5 (50–75%), and 6 (>75%). Plants with distinguishing characteristics that could not be field-identified were collected. These specimens and additional vouchers were deposited at the University of Minnesota Herbarium (St. Paul, MN, U.S.A.). Although we expect some forbs were missed due to desiccation and grazing before sampling, we do not consider this problematic because our analyses excluded rare occurrences and focused on overall patterns.

We also assigned cover classes (as described above) for total lichen, total moss, and bare ground (exposed soils and rocks not covered by vegetation). Cover of cyanobacterial/green algal crust was not explicitly quantified, and is thus contained in cover of lichen and bare ground. Litter, rare because of frequent grazing, was included in cover estimates of individual plant species. Although we did not record total plant canopy cover, our estimates of bare ground are a measure of the area not occupied by obvious vegetative material. We recorded elevation and coordinates of each plot with GPS (Garmin GPSMAP® 76S), and measured slope and aspect using a compass with built-in clinometer. Within and surrounding each plot, we described general plant community, soil surface texture, and evidence of human disturbance or livestock grazing. Preliminary analyses indicated that many of these factors were not important to overall patterns and are thus not presented here. We noted whether the plot was within (on road), on the boundary (which was usually sharp), or beyond the directly disturbed area of restoration (off road).

Seeding Assumptions

Species listed on seed tags or certification slips from final restoration (including impurities) were assumed to be seeded. Records from partial restorations or follow-up seedings were included when available. To minimize underestimation of seeding because of missing records and to corroborate assumptions not based on hardcopy records (e.g. interviews), we used the following visual clues in the field to note which species had likely been seeded at each road: sharp contrast in abundance within compared to beyond the disturbance boundary, obviously growing in rows, or those commonly introduced for revegetation that were not present in the immediate surroundings. We eliminated 14 of 58 sites from analysis of seeding establishment because seeding information was inconsistent with the composition of species likely to have been seeded based on field observations. For all other analyses, we retained the data from all 58 sites.

Data Analyses

Comparison of Restored to Adjacent Communities. We used nonmetric multidimensional scaling (NMS) ordination to summarize differences in sample plots based on species distributions. For each road, plot data for all transects were grouped and averaged by their distance from transect midpoint (i.e. each road had four subsample groups for 3, 10, 25, and 50 m from its midpoint). Abundance data were averaged for each road using cover-class midpoint percentages, then converted back to cover classes which were used directly for NMS (McCune & Grace 2002). Infrequently occuring species (present at <5% of roads) were excluded. The NMS consisted of the relative abundance of 107 species in each of 232 subsample units. We used PC-ORD (version 4.37, McCune & Mefford 1999), which automates methods of Mather (1976) and Kruskal (1964), with default specifications of the "slow and thorough" procedure using a Sorensen distance measure. Because the final stress and final instability were acceptable, but not ideal, two additional rounds of the "slow and thorough" procedure were done. All rounds resulted in a three-dimensional final solution, final stress between 19 and 21, final instability between 0.0043 and 0.0055, and very similar ordination graphs, thus suggesting overall patterns were stable enough to warrant interpretation. We present the most stable of these solutions here.

We also compared diversity and abundance components of strictly on and off road plots (excluding border plots). For each road, total species richness per plot and richness per plot partitioned by life-form and species origin (native/non-native) were averaged for on and off road plots; these means were considered independent samples ($n = 58$) which were compared using two-sample t tests (assuming unequal variances) (JMP version 7.0.1, SAS Institute, 2007). Likewise, species evenness, calculated as Pielou's J' (Pielou 1977) following Zar (1999, pp. 41–42), and cover of bare ground, using the midpoint value of the cover class, were averaged and tested as above for on and off road plots. Relative abundance of each species as a percent of the total vascular plant cover in on versus off road plots was averaged per road and summed within each life-form/origin category.

Relationship of On-Road Vegetation to Explanatory Factors. A separate NMS ordination was done to determine whether species composition on restored roads was related to age of restoration, seeding practices, or other variables. Species abundance data for only on-road plots were averaged as described previously and infrequently occurring species (those present in <5% of subsamples) were excluded. Thus, we used the relative abundance of 91 species within the 58 on-road sample units for the NMS analysis, as above. Several quantitative and categorical variables were evaluated using NMS ordination graphs, including cover of lichen, moss, and bare ground; age of restoration; and management district. We also grouped roads into three time periods using year of final restoration (1983–1987, 1988–1994, and 1995–2001) which correspond to known changes in LMNG revegetation policy (i.e. mostly non-natives, a mix of natives and non-natives, only natives; Fig. 1) (M. Sexton 2004, USFS, Dickinson, ND, personal communication).

Seeding Establishment and Colonization on Restored Roads. To assess seeding establishment and colonization without seeding, we compared species seeded to extant vegetation for each restored road. For these purposes, we only considered species present in sample plots within the area directly disturbed by restoration, excluding plots on the boundary and plots off the road in adjacent prairie. To assess the overall contribution of seeding to existing plant cover, we averaged by road the absolute percent cover per plot of all seeded species and of all presumably colonizing species, comparing these means using a two-sample t test (as above). We also partitioned this information by origin and year-restored time period. For each seeded species, relative establishment (proportion of roads where a given species was seeded and currently present with at least one individual out of all roads where that species was seeded) and relative colonization (proportion of roads where a given species was currently present but had not been seeded of all roads) were calculated. We averaged each species' frequency (percent of plots present) and abundance (average of cover-class midpoints) separately for those roads where it was seeded and those where it was not seeded.

Results

Main Differences Between Restored and Adjacent Plant Communities

Current restoration efforts have not yet achieved the goal of diverse, native plant communities on these restored roads. Instead, roadbed flora was dominated by a low number of seeded species; over 65% of average relative cover of vascular plants were accounted for by six species (Table 1), four of which were exotics (*Agropyron cristatum*, *Thinopyrum intermedium* [Intermediate wheatgrass], *Bromus inermis* [Smooth brome], and *Melilotus officinalis* [Yellow sweetclover]) and all but *B. inermis* were also distinctive of the on-road plant community (Fig. 2). In contrast, the six most dominant species in adjacent prairie (*Bouteloua gracilis* [Blue grama], *Pascopyrum smithii* [Western wheatgrass], *Carex filifolia* [Threadleaf sedge], *Hesperostipa comata* [Needle-and-thread], *Schizachyrium scoparium* [Little bluestem], and *Nassella viridula* [Green needlegrass]) accounted for only 50% of the relative cover, the remainder of which was composed of higher richness and cover of native graminoids and shrubs and higher richness of annual and perennial native forbs compared to roadbeds (Table 1). This community-level distinction represented 36% of variation in species abundance along the first NMS axis (Fig. 2).

Table 1. Mean relative cover and mean species richness per plot in on versus off road plant communities (excluding border plots; $n = 58$) by life-form and origin (native/non-native).

	Relative Cover (%)		*Species Richness*		
Plant Life-form	*On Road*	*Off Road*	*On Road*	*Off Road*	*p-value*
Graminoids (perennial)					
Native total	40	65	2.74	4.35	<0.0001
***Pascopyrum smithii* (Rydb.) A. Löve**	17.0	9.4			
***Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths**	1.3	20.0			
***Nassella viridula* (Trin.) Barkworth**	12.0	4.1			
Hesperostipa comata (Trin. & Rupr.) Barkworth	3.5	5.2			
Carex filifolia Nutt.	0.1	7.4			
***Schizachyrium scoparium* (Michx.) Nash**	0.8	4.9			
***Koeleria macrantha* (Ledeb.) Schult. [Prairie Junegrass]**	0.9	3.1			
Carex inops L.H.Bailey ssp. *heliophila* (Mack.) Crins[a] [Sun sedge]	0.1	3.1			
Distichlis spicata (L.) Greene [Saltgrass]	2.8	0.8			
***Calamovilfa longifolia* (Hook.) Scribn. [Prairie sandreed]**	0.3	1.6			
Bouteloua curtipendula (Michx.) Torr. [Sideoats grama]	0.1	1.6			
***Elymus lanceolatus* (Scribn. & J.G. Sm.) Gould [Thickspike wheatgrass]**	1.1	0.9			
Muhlenbergia cuspidata (Torr. ex Hook.) Rydb. [Plains muhly]	0.0	1.2			
Non-native total	36	7.4	1.04	0.34	<0.0001
***Agropyron cristatum* (L.) Gaertn.**	16.0	1.1			
***Thinopyrum intermedium* (Host) Barkworth & D.R. Dewey**	11.0	0.1			
***Bromus inermis* Leyss.**	4.4	2.1			
***Poa pratensis* L.**	0.8	3.6			
Graminoids (annual)					
Native total	0.0	0.0	0	0.0042	0.1037
Non-native total	0.1	0.3	0.10	0.06	0.1196
Forbs (perennial or biennial)					
Native total	9.8	9.0	2.23	3.55	<0.0001
Artemisia frigida Willd.	3.4	1.2			
Artemisia ludoviciana Nutt. [White sagebrush]	0.7	0.5			
Achillea millefolium L. **[Common yarrow]**	0.2	0.7			
Symphyotrichum oblongifolium (Nutt.) G.L. Nesom [Aromatic aster]	0.2	0.7			
***Linum lewisii* Pursh[b] [Prairie flax]**	0.7	0.1			
Grindelia squarrosa (Pursh) Dunal [Curlycup gumweed]	0.8	0.0			
Non-native total	7.0	1.0	0.73	0.44	0.0010
***Melilotus officinalis* (L.) Lam.**	5.5	0.3			
Taraxacum officinale F.H. Wigg. [Common dandelion]	0.7	0.5			
Forbs (annual)					
Native total	0.4	0.5	0.23	0.49	0.0004
Non-native total	0.5	0.1	0.23	0.05	0.0013
Woody Plants					
Native total	6.2	17.0	0.53	0.93	<0.0001
Artemisia cana Pursh [Silver sagebrush]	2.4	3.0			
Symphoricarpos occidentalis Hook. [Western snowberry]	0.6	4.0			
Juniperus horizontalis Moench [Creeping juniper]	—	3.9			
Gutierrezia sarothrae (Pursh) Britton & Rusby [Broom snakeweed]	2.3	0.5			
Rhus trilobata Nutt. var. *trilobita* [Skunkbrush sumac]	0.0	1.3			
Ericameria nauseosa (Pall. ex Pursh) G.L. Nesom & Baird [Rubber rabbitbrush]	2.1	0.3			
Non-native total	—	—			

Relative cover for individual species as percent total vascular plant cover was averaged using cover-class midpoints. Species are listed from highest to lowest total absolute cover; those in bold were seeded and established on at least one road. Only species present either on or off roads with more than 1% relative cover are shown, except for forbs which were limited to those present either on or off roads at more than 0.5% relative cover. Total relative cover values are sum of relative cover of all species within group. Mean species richness was compared with two-sample t tests. Nomenclature and origin follow USDA, NRCS (2010). Common names are in brackets unless mentioned in text.

[a] Also likely includes *C. duriscula* C.A. Mey. [Needleleaf sedge] and possibly *C. torreyi* Tuck. [Torrey's sedge].

[b] Records specified "Appar Lewis" cultivar, a collection native to the Great Plains.

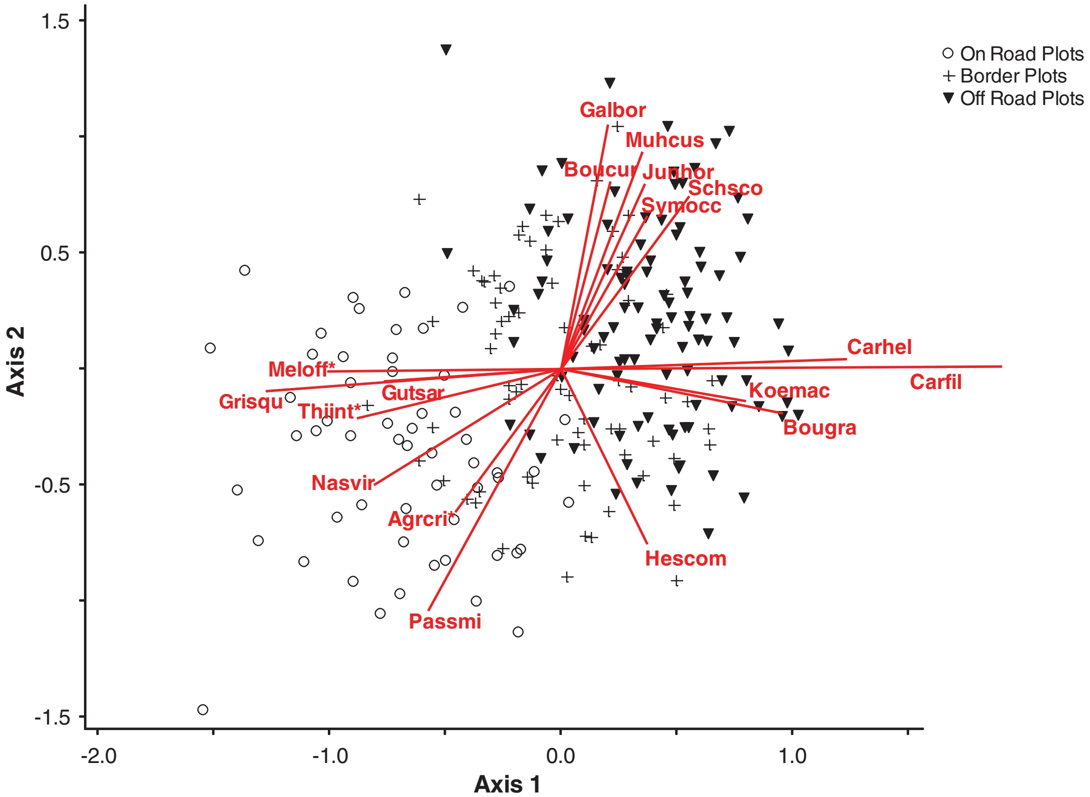

Figure 2. NMS ordination of relative average abundance of 107 species in plots at 3, 10, 25, and 50 m from midpoint for each road, showing the first two axes. Symbols identify whether all plots at a given distance for each road were within the restored road area (○), beyond the restored road (▼), or were on the border or had a combination of on/off road plots (+). Angle and length of vectors express direction and strength of species' relationships to each axis for those species abundance correlations with $r^2 > 0.20$ for either axis. Agrcri* = *Agropyron cristatum*, Boucur = *Bouteloua curtipendula*, Bougra = *Bouteloua gracilis*, Carfil = *Carex filifolia*, Carhel = *Carex inops* ssp. *heliophila*, Galbor = *Galium boreale*, Grisqu = *Grindelia squarrosa*, Gutsar = *Gutierrezia sarothrae*, Hescom = *Hesperostipa comata*, Junhor = *Juniperus horizontalis*, Koemac = *Koeleria macrantha*, Meloff* = *Melilotus officinalis*, Muhcus = *Muhlenbergia cuspidata*, Nasvir = *Nassella viridula*, Passmi = *Pascopyrum smithii*, Schsco = *Schizachyrium scoparium*, Symocc = *Symphoricarpos occidentalis*, Thiint* = *Thinopyrum intermedium*. *Non-native. Nomenclature follows USDA, NRCS (2010).

Total species richness and amount of bare ground also differed between on and off road communities. Richness was lower on restored roads, averaging 7.8 species/m^2, compared to 10.2 species/m^2 off roads ($t_{114} = -5.42$, $p < 0.0001$). Species evenness, however, did not differ significantly on versus off roads, at 0.58 and 0.59, respectively ($t_{1033} = -0.46$, $p = 0.6486$). Bare ground was greater on roads compared with adjacent prairie, at 25% compared to 11% average cover, respectively ($t_{114} = 5.14$, $p < 0.0001$).

Factors Influencing On-Road Vegetation

The plant community on restored roadways was influenced by the time elapsed since restoration; however, differences in age were clearly attributed to changes in seed selection as well as expected successional patterns. Older restorations (1983–1987 and 1988–1994) were strongly associated with the non-native grasses *A. cristatum*, *T. intermedium*, and *B. inermis* (Fig. 3a & b). More recent restorations (1995–2001) were strongly associated with the native grasses *N. viridula* and *P. smithii* (Fig. 3a & b). Except for *B. inermis*, these species were documented as commonly seeded during those respective time periods (Table 2). Of secondary importance over time were typical colonization patterns of species with no record of seeding. The most recently restored plant communities were defined by native and non-native ruderal species characteristic of disturbed areas, including *Artemisia frigida* (Prairie sagewort) and *Schedonnardus paniculatus* (Tumblegrass) (Fig. 3b). In contrast, plant communities of mid-aged and oldest restorations were associated with several native shrubs, such as *Ericameria nauseosa* (Rubber rabbitbrush), and non-ruderal native herbs, including *C. filifolia* and *Lithospermum incisum* (Narrowleaf stoneseed) (Fig. 3b).

Bare ground was a persistent problem for the restorations. Many roads from each time period were associated with greater areas of bare ground (Fig. 3a), representing 39% of variation in species abundance in the first NMS axis. Age of restoration accounted for 27% of variation in the second NMS axis.

Seeding Selection and Effectiveness

Seeding choices have a long-term effect on species composition of restored roadways. The cover of seeded species on roadbeds was double that of species with no record of having

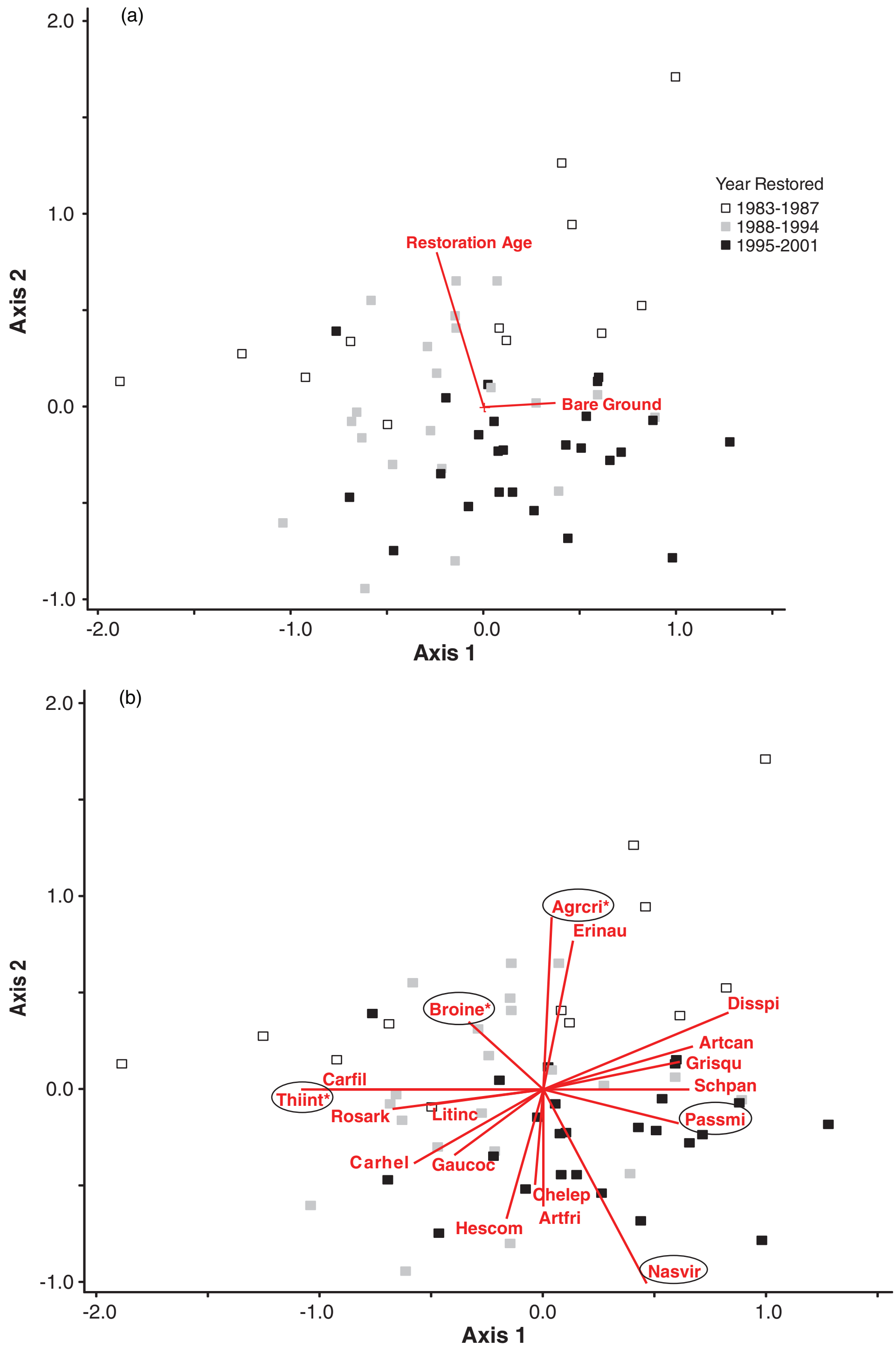

Figure 3. NMS ordination of relative average abundance of 91 species within on-road plots for each road, showing the first two axes. Symbols denote groups based on the year each road was restored, corresponding to shifts in revegetation species. Angle and length of vectors on graphs express direction and strength of each variable's relationship to each axis. (a) Explanatory variables having r^2 values more than 0.10 for either axis. (b) Species abundance correlations with r^2 more than 0.18 for either axis. Circled species are those with any record of seeding. Agrcri* = *Agropyron cristatum*, Artcan = *Artemisia cana*, Artfri = *Artemisia frigida*, Broine* = *Bromus inermis*, Carfil = *Carex filifolia*, Carhel = *Carex inops* ssp. *heliophila*, Chelep = *Chenopodium leptophyllum*, Disspi = *Distichlis spicata*, Erinau = *Ericameria nauseosa*, Gaucoc = *Gaura coccinea*, Grisqu = *Grindelia squarrosa*, Hescom = *Hesperostipa comata*, Litinc = *Lithospermum incisum*, Nasvir = *Nassella viridula*, Passmi = *Pascopyrum smithii*, Rosark = *Rosa arkansana*, Schpan = *Schedonnardus paniculatus*, Thiint* = *Thinopyrum intermedium*. *Non-native. Nomenclature follows USDA, NRCS (2010).

Table 2. Frequency of selection over time for species in mixes.

Species	*Frequency of Seeding (%)* 1983–1987 *n* = 6	1988–1994 *n* = 17	1995–2001 *n* = 21
Non-native			
Thinopyrum intermedium	100	88	76
Agropyron cristatum	83	65	67
Melilotus officinalis	50	65	52
Festuca ovina L. [Sheep fescue]	67	18	10
Astragalus cicer L. [Chickpea milkvetch]		41	
Bromus inermis		6	5
Poa pratensis			10
Medicago sativa L. [Alfalfa]		6	5
Thinopyrum ponticum (Podp.) Z.-W. Liu & R.-C. Wang [Tall wheatgrass]		12	
Psathyrostachys juncea (Fisch.) Nevski [Russian Wildrye]	17		
Festuca rubra L. [Red fescue]			5
Lolium perenne L. [Perennial ryegrass]			5
Native			
Pascopyrum smithii	50	82	100
Nassella viridula	17	82	100
Calamovilfa longifolia	17	18	76
Elymus lanceolatus	67	35	33
Linum lewisii			52
Ratibida columnifera		6	33
Elymus trachycaulus	33	71	52
Sporobolus cryptandrus (Torr.) A. Gray [Sand dropseed]	50	24	10
Dalea purpurea Vent. [Purple prairie clover]	17	6	14
Schizachyrium scoparium	17	12	48
Achillea millefolium			43
Bouteloua gracilis	17		19
Koeleria macrantha			10
Artemisia ludoviciana			14
Bouteloua curtipendula	17	6	
Sporobolus airoides (Torr.) Torr. [Alkali sacaton]	17		
Krascheninnikovia lanata (Pursh) A. Meeuse & Smit [Winterfat]			10
Vicia americana Muhl. ex Willd. [American vetch]		6	

Species are grouped by origin and listed from most to least commonly seeded. Frequency is the percentage of roads where seeded and is reported by time period. Data presented are for the 44 roads analyzed for seeding. Nomenclature and origin follow USDA, NRCS (2010). Common names are in brackets unless listed previously.

been seeded, with absolute mean cover of 32 and 15%, respectively ($t_{83} = 7.08$, $p < 0.0001$). When year of restoration was considered, the shift in seed choice was apparent as absolute cover of seeded non-natives decreased and that of seeded natives increased over time; but no matter which was used, the total proportion of seeded species was always greater than non-seeded species (Fig. 4). The cover of non-seeded native species was relatively equal over time, regardless of whether native or non-native species were used (Fig. 4).

A limited number of species were seeded repeatedly, but with varying success. Although 30 species, in total, were intentionally seeded at least once, 6 of those species were seeded at more than one-half of the roads; an additional 4 were seeded at between one-fourth and one-half of the roads (Table 3). Most of these regularly planted species had at least a 50% establishment rate and an average frequency on those roads over 20% (Table 3). However, three native grasses were used repeatedly with low success: *S. scoparium*, *Elymus trachycaulus* (Slender wheatgrass), and *Calamovilfa longifolia* (Prairie sandreed) had poor establishment and/or were infrequent where they were seeded.

Interestingly, several seeded species that were not as commonly used established well when seeding was attempted. These included the native grass *B. gracilis*, the non-native grasses *B. inermis* and *Poa pratensis* (Kentucky bluegrass), and the native forb *Ratibida columnifera* (Prairie coneflower), that established at over 50% of the roads where they were seeded with at least 20% frequency (Table 3). These species were also present at roads where they were apparently not seeded, although their average frequency of occurrence in those instances was less (Table 3).

Some species were more common on roads where they were not seeded compared with where they were seeded, although all were relatively uncommon. This included five native grasses, two native forbs, and an exotic annual forb (Table 3).

Discussion

After nearly 20 years, seeding species-poor mixes of grasses and forbs on recontoured access roads has not achieved a community comparable in species richness or composition to adjacent prairie, an outcome consistent with findings from similarly restored grasslands (Kindscher & Tieszen 1998; Baer et al. 2002; Fagan et al. 2008). Undisturbed prairie was largely composed of native, perennial graminoids, with higher richness of native forbs and abundant native shrubs compared to revegetated roads. The restored roadbed community was dominated by native and non-native seeded grasses. For the on-road community, differences in species composition over time were primarily explained by changes in species seeded for restoration rather than successional patterns. Colonization by the surrounding native community into restored roadbeds appears to be limited, despite its close proximity and high ratio of edge to disturbed area. Our findings suggest several key explanations for the observed lag in recovery, including pre-emption by seeded species, inhibitory soil conditions, and climatic limitations.

Limitations to Vegetation Recovery

Dispersal constraints are well-documented in grasslands, and at grassland restoration sites in particular (Coffin et al. 1996;

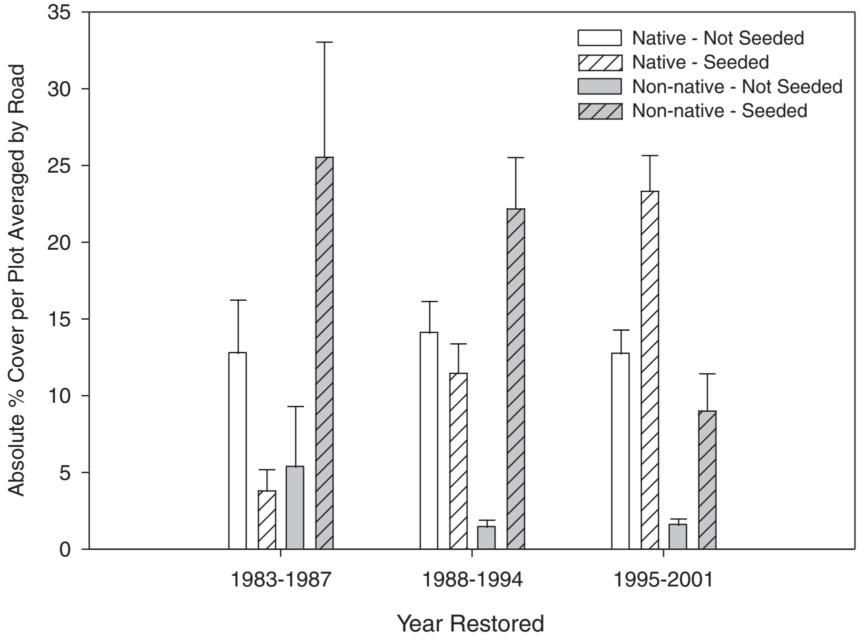

Figure 4. Absolute percent cover per plot (averaged by road) of seeded and not seeded species within on-road plots, partitioned into native and non-native components by year restored. Bars shown with standard error of the mean. Number of roads in analysis were as follows: 1983–1987 ($n = 6$), 1988–1994 ($n = 17$), 1995–2001 ($n = 21$).

Kindscher & Tieszen 1998; Kirkman et al. 2004; Bischoff et al. 2008). However, dispersal challenges that plague many restorations, such as distance from remnant seed sources and extensive, contiguous areas of bare ground, are not the case here, where relatively narrow, reworked roadbeds pass through a matrix of undisturbed native prairie. There should be many opportunities for propagule dispersal, and in fact several species with no record of seeding (e.g. *Hesperostipa comata*, *Distichlis spicata*, *Artemisia frigida*) achieved comparable or higher relative cover on roads. Yet, for the majority of the most abundant species in adjacent prairie, unless they were seeded, their cover on roads was considerably lower or they were not present at all.

Seeding had the most obvious effect on species composition of on-road plant communities, and may also be limiting establishment of surrounding plants. A large body of research has shown that non-native perennial species introduced for revegetation in North America, including many used on these roads, persist and compete with native species (Tyser & Worley 1992; Christian & Wilson 1999; Bakker & Wilson 2001; Wilson & Pärtel 2003). The noticeable influence of *Agropyron cristatum* and *Bromus inermis* in plant communities on roads despite more than 20 years since being planted is consistent with these studies. Furthermore, the presence of these and other seeded non-native species in immediately adjacent prairie suggests these species are spreading. Similarly, Hammermeister (2001) showed that native wheatgrass cultivars (*Elymus lanceolatus*, *Elymus trachycaulus*, *Pascopyrum smithii*) used for revegetation of oil well sites consistently dominated seeding experiments with other native species, although whether this was due to high-performance traits exclusive to the cultivars or to early successional/rhizomatous traits of the wheatgrasses was not tested. Indeed, native cultivars are not always more competitive or productive than local seed sources, at least in short-term transplant experiments (Wilsey in press). In this study, use of cultivars cannot be confirmed for every project, but seeding records did specify repeated use of named cultivars for many species (S. Simmers, unpublished data). We found that the native grasses *P. smithii* and *Nassella viridula* on the youngest restorations had dominance patterns similar to non-native species on older restorations and were more abundant on versus off roads. Forbs *Ratibida columnifera* and *Linum lewisii* were also commonly used in recent restorations and were more frequent on versus off roads. In addition, the shift from seeding non-native to native species apparently did not make a difference in the establishment of non-seeded native species. These patterns indicate similar mechanisms of competition or pre-emption of space and resources by these species. However, persistent high cover of bare ground on restorations of all ages points to two further limitations.

As space is available, altered soil or physical conditions may be preventing establishment of surrounding species, as well as certain seeded species. Although the recontouring process involves tearing up roadbed structure and has proved effective on forest roads (Kolka & Smidt 2004), soil on roads in this study remained visibly compacted within at least the top 5–10 cm of exploratory soil cores taken within restored areas of several study roads in contrast to cores taken in adjacent, undisturbed areas (M. Gonzalez 2006, USFS, Bismarck, ND, personal communication). Unrelieved compaction could

Table 3. Summary of species seeded on 44 restored roads, ordered by life-form and total number of roads where seeded.

Species	# Roads Seeded	# Roads Present	Relative Establishment	Relative Colonization	Seeded: Average Frequency (%)	Seeded: Average % Cover Class	Not Seeded: Average Frequency (%)	Not Seeded: Average % Cover Class
Graminoids (perennial)								
Pascopyrum smithii	38	43	1.00	0.11	80 ± 22	5–25	34 ± 24	1–5
**Thinopyrum intermedium*[a]	37	33	0.89	0	46 ± 33	5–25	—	—
Nassella viridula	36	41	0.97	0.14	80 ± 27	5–25	30 ± 28	<1
**Agropyron cristatum*[b]	30	33	0.93	0.11	56 ± 33	5–25	6.3 ± 9.5	<1
Elymus trachycaulus	25	5	0.20	0	9 ± 23	<1	—	—
Calamovilfa longifolia	20	12	0.55	0.02	13 ± 19	<1	0.6 ± 3.1	<1
Elymus lanceolatus	17	16	0.59	0.14	24 ± 27	1–5	4 ± 10	<1
Schizachyrium scoparium	13	11	0.23	0.18	3.6 ± 7.5	<1	4.3 ± 8.7	<1
Sporobolus cryptandrus	9	22	0.56	0.39	9 ± 10	<1	14 ± 22	<1
**Festuca ovina* L.[c]	9	5	0.56	0	8.0 ± 8.5	<1	—	—
Bouteloua gracilis	5	21	0.60	0.41	38 ± 44	1–5	12 ± 19	<1
Koeleria macrantha[d]	2	30	1.00	0.64	12 ± 1	<1	15 ± 15	<1
**Bromus inermis*	2	19	1.00	0.39	44 ± 47	1–5	7 ± 12	<1
**Poa pratensis*	2	19	0.50	0.41	32 ± 45	1–5	8 ± 13	<1
Bouteloua curtipendula	2	4	0	0.09	—	—	1.5 ± 6.4	<1
**Thinopyrum ponticum*	2	1	0.50	0	12 ± 16	<1	—	—
Hordeum jubatum L.[U] [foxtail barley]	1	4	0	0.09	—	—	1.5 ± 5.0	<1
Forbs (perennial)								
**Melilotus officinalis*	25	31	0.84	0.14	51 ± 34	1–5	18 ± 29	1–5
Linum lewisii[e]	11	21	0.73	0.30	27 ± 28	<1	10 ± 18	<1
Achillea millefolium	9	14	0.33	0.25	9 ± 17	<1	6 ± 15	<1
Ratibida columnifera	8	24	0.75	0.41	27 ± 30	<1	12 ± 19	<1
**Astragalus cicer*	7	5	0.71	0	14 ± 13	<1	—	—
Dalea purpurea	5	11	0.80	0.16	16 ± 13	<1	2.3 ± 5.2	<1
Artemisia ludoviciana	3	10	0	0.23	—	—	4.2 ± 8.5	<1
**Medicago sativa*	2	6	0.50	0.11	5.6 ± 7.8	<1	3 ± 11	<1
Vicia americana	1	16	0	0.36	—	—	8 ± 16	<1
Forbs (annual)								
**Salsola tragus* L.[U] [Russian thistle]	1	3	0	0.07	—	—	1.8 ± 8.8	<1

For a given species, relative establishment is the proportion of roads where it was seeded and is currently present of all roads where seeded; relative colonization is the proportion of roads where it is currently present but had not been seeded of all roads; average frequency is the percentage of plots where it was present per road; and average percent cover class is its average abundance per road. Nomenclature follows USDA, NRCS (2010). Common names are in brackets unless listed previously.

*Non-native. [U]Species unintentionally included in mixes (impurities or weed seeds).

[a] Includes *Agropyron intermedium* var. *trichophorum* and *A. intermedium* var. *intermedium*.

[b] Includes *Agropyron desertorum* and *A. cristatum*.

[c] The cultivar of *F. ovina* used here was likely introduced from Turkey (Aubry et al. 2005).

[d] Records specified "Appar Lewis" cultivar, a collection native to the Great Plains.

[e] Other species listed in seed mixes at least once but not present within any on-road plots were *Psathyrostachys juncea**, *Festuca rubra**, *Lolium perenne**, *Sporobolus airoides*, *Bromus tectorum* L.*[U] [cheatgrass], *Avena fatua* L.*[U] [wild oat], and *Krascheninnikovia lanata*.

explain slow rhizomatous colonization and seedling establishment, because compaction is known to restrict root growth on reconstructed surface mine soils (e.g. McSweeney & Jansen 1984). Apparently, many seeded species were able to establish despite this condition, which Lesica and Allendorf (1999) suggest could be due to traits cultivars possess because of selection in more agronomic-like settings. However, it may also be that seeded species are able to suppress later establishment and species turnover simply because they are there first and are able to exploit limited resources and microsites within available space. It will be crucial for future research to include measurements of soil properties so that the importance of differences in compaction, texture, nutrient availability, soil biota, etc. can be better understood in relation to plant establishment.

Another overarching constraint to rates of recovery in this grassland system is its annual and seasonal climatic variability (Call & Roundy 1991). This is known to affect regularity of seed and rhizome production, dispersal, germination, and seedling establishment for grassland plants (Briske & Wilson 1977; Bakker et al. 2003; MacDougall et al. 2008). However, secondary successional patterns, apparent when we considered restoration age, suggest that late seral species can eventually colonize restored roads under prevailing semiarid conditions. Future post-treatment monitoring should consider local measures of precipitation, temperature, and other climatic variables to determine the extent these factors influence long-term outcomes.

Species Selection in Seed Mixes

Seed mixes chosen for revegetation in this study included a strikingly low number of species compared with the flora of the surrounding community. In contrast to the nearly 200 species observed in adjacent mixed-grass prairie, only 30 species, in total, were ever included in seedings. Of these, 10 were used repeatedly on most of the roads. Furthermore, seed mixes contained only five species on average. The practical reasons for choosing such a limited number of species are many, and include wide availability of cultivars and their dependable establishment (Aubry et al. 2005), lack of commercial availability of many native species (Bohnen & Galatowitsch 2001), and risks in sowing high diversity mixes in which the majority of species sown do not establish long-term (Ward et al. 1996; Sluis 2002). Yet, when compared with low diversity mixes, higher diversity mixes have been shown to result in target plant communities that maintain higher diversity and function (Pywell et al. 2002; Foster et al. 2007; Piper et al. 2007). In order to overcome seed production and seed dispersal limitations in grasslands and to reach long-term goals, it may be necessary to invest in higher diversity seed mixes, moderating high costs and risks with more appropriate species selection and more suitable seedbed preparation and seeding methods.

Our findings also demonstrate the need for optimizing seeding choices. Although most regularly planted species had good long-term establishment, three native grasses were common in mixes but were seldom present where planted. Despite poor establishment, these species continued to be chosen. We also found that some native species characteristic of adjacent grassland (e.g. *Bouteloua gracilis*) were not often included in seed mixes, although they established well. This suggests some species are being underutilized. Perhaps the most important point here is that without adequate follow-up evaluation, restoration practices are more ineffective than they need to be and time does not compensate for these failures.

Adequate evaluation of restoration outcomes requires the ability to compare what was done during the restoration to the ecological outcome. Perhaps our most startling finding was the lack of sufficient records to support this comparison. Insufficient records forced us to eliminate sites from the potential selection pool or make assumptions about what was seeded based on secondary evidence. In one of the districts studied, 96% of the records were deficient and relied on secondary evidence. The value of this comparative evaluation to improve restoration practice is evident from this study; refining species selection in mixes through adaptive management techniques and improving roadbed preparation are likely to substantially improve project success. As road restoration is widespread, the potential for oil development to have lower environmental impact is significant. This study adds to others that have called for improving baseline data gathering and monitoring (e.g. Kondolf 1995; Block et al. 2001).

Implications for Practice

- Expect the composition and richness of seed mixes, whether non-native or native, to persist at least 20 years on restored grassland roads, so select species thoughtfully.
- The few natives that disperse and establish well will not need planting, but the majority of species in adjacent grassland will need either direct seeding to overcome dispersal barriers or improvements in seedbed preparation to overcome apparent abiotic constraints.
- The utility of long-term data collected to improve future restoration practice is limited without detailed records of species sown and techniques used during restoration; without records we cannot distinguish consequences of our actions from biological processes.

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