

## Fifty-five year changes in species composition on dry prairie remnants in south-central Wisconsin<sup>1</sup>

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KRASZEWSKI, S. E. (Land Resources Program, University of Wisconsin-Madison, Madison, WI 53706) AND D. M. WALLER. (Department of Botany, University of Wisconsin-Madison, Madison, WI 53706). Fifty-five year changes in species composition on dry prairie remnants in south-central Wisconsin. *J. Torrey Bot. Soc.* 135: 236–244. 2008.—Ten dry prairie remnants in south-central Wisconsin first sampled in 1950 were resampled in 2005 to determine changes in species presence and frequency and to characterize differences between species that increased and decreased in abundance. Combined species richness over all sites has increased (from 147 to 189), but this largely reflects the recruitment of exotic species, woody vegetation, and taller shade-tolerant species. Mean native species richness per site decreased (from 65.2 to 59.7) reflecting the loss of rarer and shorter forbs while mean exotic species richness increased greatly (from 0.3 to 10.1). Although the average number of habitat generalist species per site has remained constant (13.0 vs. 13.6), the number of habitat specialists has declined (from 27.4 to 21.7) contributing to declines in floristic quality. The five native species that decreased in occurrence and 15 that decreased in frequency tended to be short, non-clonal specialist forbs. The seven native increaser species tended to be tall, clonal, and/or woody generalists. Efforts to conserve dry prairies should thus focus on those species most at risk (i.e., rare, short-statured, and non-clonal) and the processes that sustain them. Burning and removing exotic species and woody vegetation appear to enhance our ability to conserve and maintain plant diversity in dry prairie remnants.

Key words: coefficient of conservatism, dry prairie remnants, exotic species, habitat generalist, habitat specialist, species richness.

Grasslands once covered 28% of the total land surface of Wisconsin with ~800,000 ha of prairie and ~3,000,000 ha of savanna (Curtis 1959). Less than 0.2% of those native grasslands remain in Wisconsin today (Henderson 1981). Significant changes in land use and management since the arrival of European settlers in the early to mid 1800s decimated these grasslands. These included converting prairies into farm fields, fragmenting grasslands by roads (Henderson 1981), and suppressing the large-scale fires that favor fire-adapted grasses and forbs over woody plants (Leach and Givnish 1996). Most prairie remnants remaining in Wisconsin are only a few hectares or less in size and are separated

from one another by a landscape which no longer experiences frequent fire. The majority of these remnants exist along railroad right-of-ways or along steep slopes that were too difficult to plow.

Dry lime prairies in Wisconsin occur on thin soils over dolomitic bedrock or calcareous gravel with high surface drainage (Curtis and Greene 1949). These prairies are frequent within or near the rugged topography of the “Driftless Area” never covered by recent glaciers. Dry prairies generally occur on south- and west-facing slopes exposed to the force of prevailing summer winds and the hot afternoon summer sun (Evers 1955). Although these remnants were often ignored by farmers as too difficult to plow, farming in surrounding lands still acted to disconnect dry prairies from the larger and more contiguous expanses of prairie and savanna habitat that once formed the matrix in this region.

To wisely manage these isolated remnants, we should understand how their vegetation is changing over time, which species need conservation attention, and whether plant traits can predict vulnerability to local extirpation. Here, we use a valuable set of historical data on species presence and frequency to assess

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long-term changes in the vegetation of ten dry lime prairie communities in south-central Wisconsin. In particular, we re-sampled ten dry lime prairie remnants first surveyed by scientists of the Plant Ecology Laboratory (PEL) at the University of Wisconsin-Madison in 1950 (Anderson 1954, Curtis 1959).

Other researchers have already used PEL data to track the long-term trajectories of plant communities. Leach and Givnish (1996) resurveyed the presence and absence of plant species on 54 mostly tallgrass prairie remnants in the late 1980s. They found native species losses of 7–50% particularly among rarer and shorter species and those fixing nitrogen. In contrast, taller herbs and species that were woody, heavy-seeded, or had vegetative reproduction tended to increase. In a succeeding study, Milbauer and Leach (2007) found that fire suppression facilitated increases in habitat generalist species and declines in once-dominant habitat specialists in 21 tallgrass prairie remnants. Resurveys of understories in the upland forests of northern Wisconsin reveal declines in native species density, increases in the ratio of exotic to native species, and community homogenization (Rooney et al. 2004). Here, “winner” species included a disproportionate number of abundant species, graminoids, and those that are abiotically pollinated. In contrast, “loser” species were rarer native forbs usually with biotic pollination (Wiegmann and Waller 2006).

Resurveys elsewhere reveal similar patterns of species loss and community change. Plant species that were rare in the past are often more likely to decline or disappear (Duncan and Young 2000, Fischer and Stocklin 1997). Habitat specialists decrease while generalist species increase (Fischer and Stocklin 1997). Species with short-stature appear more vulnerable to extirpation than taller species (Duncan and Young 2000, Turner et al. 1996), and clonal species fare better than non-clonal plants (Fischer and Stocklin 1997).

Here, we explore whether the long-term trends detected in these other plant communities extend to include the dry prairies in south-central Wisconsin. We examined changes in dry prairie species richness and frequency, determined which species are increasing and decreasing, and analyzed the characteristics of species that increased and decreased in abundance. We hypothesized: 1) that native species richness has declined while exotic species

richness has increased, 2) that the occurrence of rare species and those restricted to specialized prairie habitats has decreased, 3) that species that are herbaceous, lack clonality, and those with short stature, small seeds, and early flowering will have declined relative to species without these characteristics, and 4) that active management affects these trends, better sustaining plants specialized to prairie habitats.

**Methods. STUDY AREA.** The 10 dry prairie sites were located in Dane, Columbia, Sauk, and Iowa counties of south-central Wisconsin. The mean prairie area, measured on site with a GPS unit, was only 0.87 ha (2.15 acres), with sites ranging in size from 0.04 to 2.40 ha (0.1 to 5.9 acres). The perimeter to area ratio ranged from 0.03 to 0.22. The land owners and managers were interviewed to determine the management history of the 10 remnants. Six of the remnants have been actively managed via burning, removing woody vegetation, and/or controlling exotic species. Five of these six sites were managed to control woody vegetation and exotic species as well as being burned, while the other only had woody vegetation removed. This management only occurred on one of the sites over the entire interval with management not starting at the other five until at least the 1980s. The four unmanaged remnants were not grazed, burned, or managed to control woody or exotic vegetation since at least 1950.

**FIELD METHODS.** We compiled species lists in May, July, and late August/early September of 2005 to determine species richness. These lists were compiled by walking the remnant and recording all plant species present. In July we systematically sampled quadrats using the methods of the original sampler (Anderson 1954). That is, we located a total of twenty 1 m<sup>2</sup> quadrats along two or more transect lines at each site. The number of transect lines varied based upon the shape and size of each remnant. We noted all the species rooted in each quadrat for frequency analyses. We identified species in accordance with current taxonomic standards as found at the Wisconsin Vascular Plants Web Site (Wetter et al. 2003).

**DERIVED VARIABLES.** We tabulated the “coefficient of conservatism” (C) for each species and averaged these for each site and

sampling period. These C values were assigned for Wisconsin species by a core group of botanists (Bernthal 2003). They range from 0 to 10 in accord with their perceived fidelity to native plant communities (Swink and Wilhelm 1994). Plants with low fidelity to natural communities (including most exotics) receive a value of 0 while those with high fidelity receive higher values. We classify species with C values from 0 to 3 as habitat generalists and those with values from 8 to 10 as habitat specialists.

We tallied total species richness from the walk-through surveys at each site. We also tallied the number of different species found within the twenty 1 m<sup>2</sup> quadrats (species density at the 20 m<sup>2</sup> scale). We use these data to compare shifts in species occurrence and frequency (the number of quadrats a species was present in out of the 200 quadrats) between 1950 and 2005. We assigned each species to one of three categories (increaser, persister, or decreaser, or 1, 0, and -1, respectively) on the basis of statistical tests using McNemar's chi-squared test with continuity correction and paired *t*-tests of changes in frequency. We define persistent species as those present in both 1950 and 2005 that did not shift significantly in occurrence or frequency. We also compared community composition between the two survey periods using Sørensen's Index of Similarity (Bray and Curtis 1957).

To gain a more quantitative and continuous variable representing proportional changes in abundance we developed a new variable. Proportional declines are well-represented by:

$$(\text{Abund2005} - \text{Abund1950})/(\text{Abund1950})$$

which ranges from 0 to -1. However, this formula gives values that are unbounded and exceed 1 for increases above 100% and is undefined for species absent in 1950. We therefore define a new variable (change) as:

$$\frac{(\text{Abund2005} - \text{Abund1950})/}{(\text{Abund2005} + \text{Abund1950})}$$

This variable gives defined values bounded between -1 and +1 for all abundances. We also found it to be approximately normally distributed among 89 species (mean 0.018, S.D. 0.606).

We also sought predictor variables that would allow us to make more specific predictions about how species characteristics might affect shifts in species abundances. For example, we assigned all plants into one of six height classes: 1) < 0.25 m, 2) 0.25 < *x* < 0.5 m, 3) 0.5 < *x* < 1.0 m, 4) 1.0 < *x* < 2.0 m, 5) 2.0 < *x* < 4.0 m, and 6) > 4.0 m and predicted that increasers would be taller than decreasers (Leach 1990). We also assigned species to one of four life form classes: basal-leaved forb, erect forb, graminoid, or woody and predicted that woody species would be frequent among the increasers and herbaceous forbs (particularly basal-leaved species) among the decreasers. Because species vary considerably in seed weight and this affects the ability of species to persist and establish at a site, we expected increasers to have heavier seeds than decreasers (Leach 1990). Vegetative reproduction could also aid species in their ability to persist at a site so we predicted that clonal species would increase more often than they decreased. Species also differ in blooming season (Butler 1954) with taller species often blooming later than shorter species. We therefore classified species into four sequential categories and predicted that increasers would tend to bloom later than decreasers.

**METHODS OF ANALYSIS.** To test these hypotheses using the characters just defined, we used a combination of statistical tests chosen to match the variables used. When predictor variables were continuous, these tests included regression (for the continuous change variable) and logistic regression (for the discrete increaser, persister, decreaser categorical variable). When predictor variables were discrete, these included *t*-tests (with two categories) and ANOVA (for 3+ categories) for the continuous change variable and Fisher's exact test and chi-squared tests for the categorical change variable. We also used simple *t*-tests to evaluate differences between sets of sites that differed in management history. We checked residuals to ensure that they were normally distributed. To assess the relative and combined explanatory power of the predictor variables, we applied multivariate (general linear) models, sequentially eliminating variables that lacked significance (backward elimination). All analyses were conducted using R or JMP (version 6).

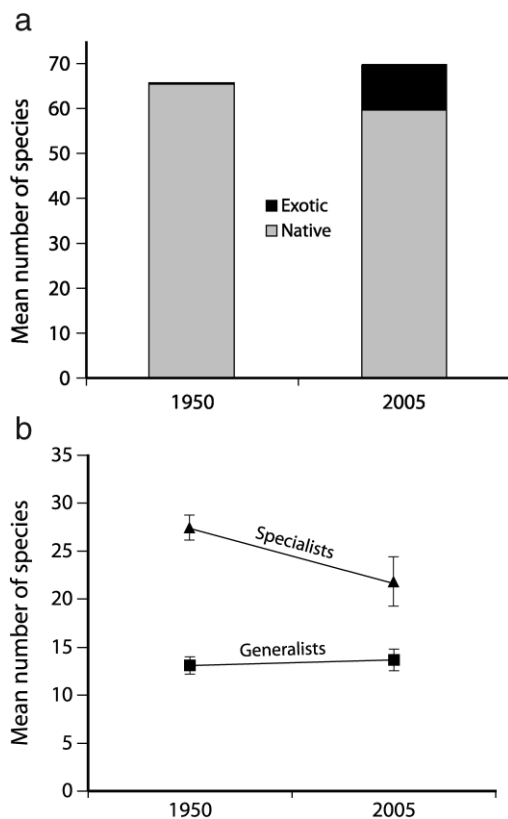


FIG. 1. a) Mean species richness per site ( $n = 10$ ) for 1950 and 2005. b) Mean number of habitat generalist and specialist species for ten dry prairie sites in 1950 and 2005. Error bars represent  $\pm 1$  SE.

**Results. SPECIES OCCURRENCES.** Anderson (1954) recorded 145 native and 2 exotic species over the ten sites in 1950. We observed 158 native and 31 exotic species at the same sites in 2005. The quadrat samples included 109 native species in 1950 and 119 in 2005 (plus 20 exotic species). Mean native species density (at the 20 m<sup>2</sup> scale) decreased 10% or about five species from  $65.2 \pm 11.2$  in 1950 (range: 49 to 81 species, Fig. 1a) to  $59.7 \pm 17.3$  in 2005. Total site richness ranged from 39 native species (49 total) to 97 native (110 total) species across sites. The mean Sørensen Index of Similarity between surveys at the same site was 0.92 indicating that sites retained most of the same species between 1950 and 2005.

Twenty-five native species found in 1950 at these sites disappeared by 2005. Eighteen of these (72%) were rare in 1950, each occurring at no more than one site. Two others (*Agalinis gattingeri* and *Asclepias lanuginosa*) have threatened status, three are woody trees or

shrubs, and three are shade-tolerant herbaceous species.

Of the 184 plant species observed in 2005, 67 were not observed in 1950. Eight of these bloomed early and were probably present in 1950 but not recorded during summer sampling. Of the other 59 newcomers, 29 are exotic. The other 30 are native and include six woody vines or trees and 14 shade-tolerant herbs distinct from open-grown prairie species (Leach 1996, Fifield-Murray 1978). Two other newcomer species, *Eupatorium sessilifolium* and *Orobanche uniflora*, are rare and of special concern in Wisconsin. Their apparent absence in 1950 could reflect either a true colonization or the fact that some rare species will always be missed in any finite sample of a rich community.

Anderson (1954) noted only two exotic species, *Poa* spp. at one site and *Convolvulus* spp. at two sites. Neither was found within the sampled quadrats. In contrast, we noted 31 exotic species with 101 site occurrences in 2005 (53 within quadrats). Every site in 2005 had exotic species, with an average of  $10.1 \pm 4.0$  exotic species present at a site. The most frequent exotic species were *Poa pratensis* or *P. compressa* (which together occurred in 46.5% of the quadrats sampled), *Euphorbia esula* (in 10.5%), *Rhamnus cathartica* (10.5%), *Taraxacum officinale* (7.5%), *Convolvulus arvensis* (6.5%), and *Pastinaca sativa* (6.5%).

**GENERALISTS AND SPECIALISTS.** The total number of specialist species ( $C$  values from 8 to 10) found at these sites declined from 31 in 1950 to 27 in 2005. Occurrences of these species also declined from 140 in 1950 to 107 in 2005 as did the average number of specialist species per site (from  $27.4 \pm 4.2$  species in 1950 to  $21.7 \pm 8.1$  in 2005, Fig. 1b). The number of native habitat generalist species ( $C$  values of 0–3) followed the opposite pattern, with many new species found (from 31 in 1950 to 39 in 2005) but only a small change in the number of occurrences (from 130 to 134). Within sites, the average number of generalists encountered also remained steady ( $13.0 \pm 2.9$  in 1950 versus  $13.6 \pm 3.4$  in 2005, Fig. 1b). Together, these shifts in specialists and generalists caused the mean coefficient of conservatism at a site to decline from  $5.6 \pm 0.1$  in 1950 to  $4.5 \pm 0.47$  in 2005. This decline in mean  $C$  (1.03) was highly significant (paired  $t$ -test,  $P < 0.001$ ) and remained significant even when

only native species were included (difference: 0.26,  $P = 0.035$ ).

**INCREASERS AND DECREASERS.** Three native dry prairie species declined in prevalence (*Anemone patens*, *Hedeoma hispida*, and *Potentilla arguta*, McNemar's test,  $P < 0.05$ ) and two others showed a tendency to decline (*Artemisia campestris* and *Solidago ptarmicoides*,  $P = 0.074$ ). Eleven native species decreased significantly in frequency (paired  $t$ -test,  $P < 0.05$ ) and four others tended to decrease over time ( $P < 0.10$ , Fig. 2a). No native species significantly increased in prevalence over sites, but two increased significantly in frequency (paired  $t$ -test,  $P < 0.05$ ) and seven tended to increase in frequency ( $P < 0.10$ , Fig. 2b). Sixty-five species persisted at similar levels of prevalence and frequency.

**ARE TRAITS RELATED TO SHIFTS IN ABUNDANCE?** We found several associations between the traits plant species displayed and their tendency to have increased or declined in abundance over the past 55 years. Most conspicuously, shorter plants were more likely to have declined in abundance and taller plants were far more likely to have increased. This held true for both categorical classifications of species as increasers, persisters, or decreasees ( $r^2 = 0.99$ ,  $P = 0.001$ , Fig. 3a) and the continuously distributed change variable reflecting species' proportional changes in abundance ( $r^2 = 0.24$ ,  $P < 0.0001$ , Fig. 3b). Decreasing species were, on average, less than 0.5 m tall with several species less than 0.25 m.

As expected, woody species were disproportionately represented among the increasers (44%) but rare among those that simply persisted (4.6%) or declined (0%, Fisher's exact test:  $P = 0.004$ , Fig. 4a). Graminoids were also far more common among the increasing species (22%) than among those that simply persisted (9.2%) or declined (0%). In contrast, declining species had a high proportion of basal-leaved forbs (31.6%) as did persisting species (23.1%) in contrast to their scarcity among increasers (11%). Decreasers also tended to have smaller seeds than persisters and increasers (means of 2.3 mg vs. 2.9 and 2.8 mg, respectively). This difference was significant between decreasees and persisters ( $F_{(2,84)} = 2.6$ ,  $P = 0.025$ ) but not between decreasees and increasers (probably due to small sample size). The ability to spread

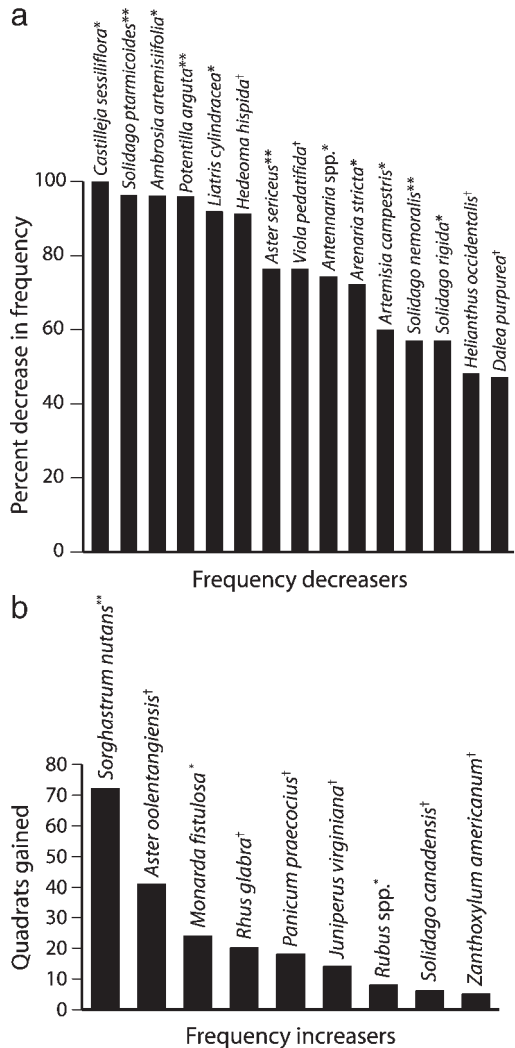


FIG. 2. a) Percent decrease in frequency from 1950 to 2005 for all ten sites for frequency decreasees ( $n = 15$ ). Species frequency decreasees were determined by a paired  $t$ -test ( $0.05 < P < 0.1$ ; \* $P < 0.05$ ; \*\* $P < 0.01$ ). b) Absolute increase in number of quadrats for increasers out of 200 total quadrats on ten sites between 1950 and 2005. Species frequency increasers were determined by a paired  $t$ -test ( $0.05 < P < 0.1$ ; \* $P < 0.05$ ; \*\* $P < 0.01$ ).

clonally also differed among the three groups, increasing consistently from decreasees (37.5% clonal), to persistent species (49.2%), to increasers (89%, Fisher's exact test:  $P = 0.043$ ), as predicted.

Declining species tended to be specialists with high site-specificity, as reflected in high C values. Decreasers had an average C of 5.9 versus 5.6 for persistent species and 3.6 for the increasers (Fig. 4b), with the first two values

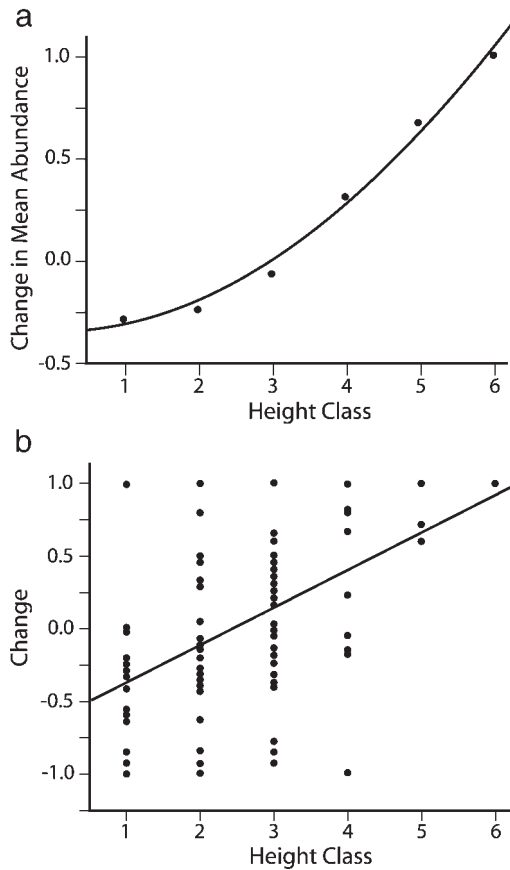


FIG. 3. Changes in abundance as a function of plant height. a) Mean changes in abundance based on classifying all species as 1 (increasers,  $n = 9$ ), 0 (persistent species,  $n = 65$ ), or -1 (decreasers,  $n = 16$ ). Quadratic fit:  $R^2 = 0.99$ ,  $P < 0.001$ . b) Proportional changes in abundance as a function of height class. Linear fit:  $R^2 = 0.24$ ,  $p < 0.001$ . Height class categories: 1)  $< 0.25$  m, 2)  $0.25 < x < 0.5$  m, 3)  $0.5 < x < 1.0$  m, 4)  $1.0 < x < 2.0$  m, 5)  $2.0 < x < 4.0$  m, and 6)  $> 4.0$  m.

significantly higher than the last (both  $P = 0.034$ ). Thus, many increasing species are already common and widespread. Blooming periods for all three groups were similar and occurred during late June and July. Five decreasers (31.2%) bloomed before June 1<sup>st</sup> and there was an association between early blooming and short stature ( $P = 0.025$ , Likelihood ratio test, logistic model).

In the succession of multivariate models that began with all plant trait predictor variables, only stature (height class) remained significant in the reduced model predicting proportional changes in plant abundance ( $r^2 = 0.239$ ,  $P < 0.0001$ ). In the corresponding set of

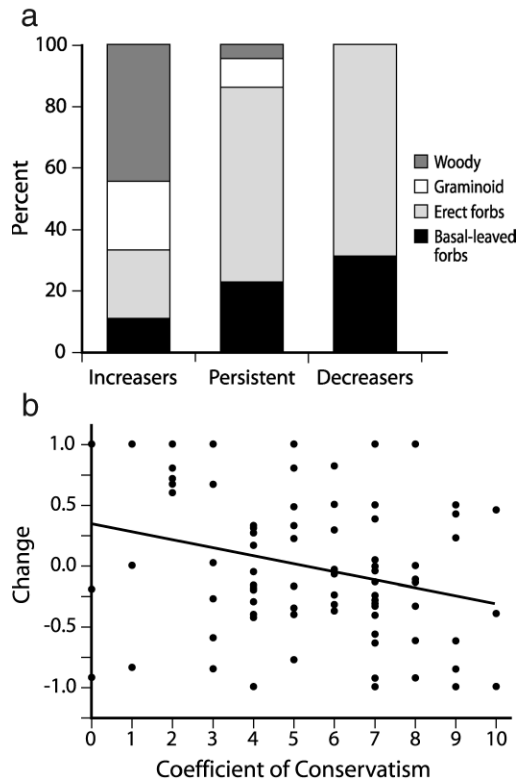


FIG. 4. a) Proportion of the four life forms exhibited by increasers ( $n = 9$ ), persistent species ( $n = 65$ ), and decreasers ( $n = 16$ ). Differences in life form between groups are significant (Fisher's exact test,  $P = 0.004$ ). b) Proportional long-term changes in abundance as a function of each species' C (coefficient of conservatism) value ( $r^2 = 0.076$ ,  $P = 0.01$ ).

backward elimination ordinal logistic models of the discrete change variable (increaser, persister, or decreaser), several variables remained significant or almost significant, but stature remained the most important predictor (Table 1,  $r^2 = 0.341$ ).

EFFECTS OF MANAGEMENT. Management practices affected both increasers and decreasers. Managed sites showed differential increases in frequency for *Monarda fistulosa* ( $t$ -test between management groups,  $P = 0.036$ ) and perhaps *Rhus glabra* ( $P = 0.085$ ) and *Panicum praecocius* ( $P = 0.071$ ). *Rhus* particularly increased its frequency on burned sites, probably reflecting its tendency to root sprout. *Panicum praecocius* only occurred on managed sites in 2005. *Liatris cylindrica* declined particularly on unmanaged sites ( $P = 0.035$ ). In contrast, *Juniperus virginiana* increased in

Table 1. Effect likelihood ratio tests for change in abundance. Results of the reduced ordinal logistic model predicting categorical changes in abundance (increaser, persister, or decreaser) as a function of plant traits. This model is highly significant (Chi-square = 34.8,  $r^2$  value = 0.341).

Source	Number of parameters	DF	L-R Chi-Square	$P >$ Chi-Sq
Height Class	5	5	20.252	0.0011 **
Life Form	3	3	7.809	0.0501 AS
Longevity	2	2	5.766	0.0560 AS
Blooming Season	4	4	10.192	0.0373 *
C value	1	1	1.858	0.1728

frequency more on unmanaged sites ( $P = 0.016$ ).

**Discussion.** Although total native species richness at these prairies tended to increase somewhat, this primarily reflects the recruitment of woody, taller, and more shade-tolerant species that do not typify dry prairie communities. The quadrat samples revealed that mean native species density declined 10% (five species) at the 20 m<sup>2</sup> scale. In contrast, exotic species have gained in number and abundance in these dry prairies. These trends mirror those found in Wisconsin tallgrass prairie systems (Leach and Givnish 1996, Milbauer and Leach 2007) and elsewhere as in Swiss Jura Mountain calcareous grasslands (Fischer and Stocklin 1997) and forest understories in northern Wisconsin (Rooney et al. 2004). These shifts in species composition were most pronounced on unmanaged sites. Even dry prairie remnants are being invaded by woody vegetation and shade-tolerant herbs, causing them to lose their open character. The lack of fire and more open sunny conditions, in turn, contributes to their losses of shorter and more heliophilic prairie species.

Species that declined over the past 55 years were likely to be rare, short plants closely adapted to these particular habitats. Those that increased tended to be generalists tolerant of a broader set of habitat conditions. These trends have been observed by others (Duncan and Young 2000, Fischer and Stocklin 1997, Leach and Givnish 1996, and Rooney et al. 2004). Many of the decreasing species are restricted to dry prairie habitats. Such habitat specialists now have few populations to serve as sources for local recolonization, and the small size of these remnants increases the likelihood of local extinction. Increases in common and invading exotic species combined with losses of habitat specialists contribute to the declines in community quality reflected in the decline in mean C across sites.

Nevertheless, despite large overall declines in occurrence, there was only a slight decline in overall (gamma) diversity with only four specialist species lost from these sites.

As already widespread species displace unique species with narrow habitat breadth adapted to particular sites, differences among sites (beta diversity) decline in a phenomenon referred to 'biotic homogenization' (McKinney and Lockwood 1999). Homogenization reflects widespread environmental changes such as habitat loss, degradation, and fragmentation that threaten native specialists and aid in the effective dispersal of opportunistic native and exotic species. Losses of specialists contribute to losses of biological diversity and uniqueness, raising concerns for the protection of remnant ecological communities. Because rarer and shorter dry prairie species are susceptible to woody and exotic species invasions, a common consequence of fire suppression, we should focus particular attention on management that helps to sustain these species.

Taller herbs, woody species, and those with clonal reproduction all tended to increase in abundance whereas many shorter, basal-leaved herbs declined as observed in other studies (Duncan and Young 2000, Fischer and Stocklin 1997, Leach and Givnish 1996, and Turner et al. 1996). Shorter and basal-leaved species are at a distinct disadvantage when fire suppression allows taller species to invade, compete for light, and overtop shorter species. Forbs with leaves close to the ground compete well on sparsely vegetated sites because they allocate less energy to supportive tissue but lose out to taller competitors on more productive, densely occupied sites (Givnish 1982). Short species also suffer as litter accumulates. When the taller invaders are woody, these trends are compounded. Humidity also rises, reducing the incidence and/or severity of fire. Leach and Givnish (1996) also found that species with smaller seeds tended to

decline, suggesting that smaller seeded species are at a disadvantage in thick swards where a dense thatch of litter hinders their establishment until the next disturbance. Our data support this trend in that decreaseers have 18% smaller seeds than increaseers.

Grasses and grass-like species usually persisted or increased. These species often reproduce vegetatively and often depend less on seed dispersal and establishment. These dry prairie remnants are also often isolated from other prairie landscapes, perhaps restricting pollinator availability for insect pollinated forbs and consequent seed set (Donaldson et al. 2002). Early blooming, short stature forbs may also have difficulty blooming when a thick litter layer retards or directly suppresses spring growth (Ehrenreich and Aikman 1963, Ehrenreich 1959, and Weaver and Rowland 1952).

Management affected increaseers more than decreaseers. Species that increased in response to management appeared to respond most to fire. For example, *Rhus glabra*, which increased on sites with burning, has been shown to increase in shoot density following fire (Knapp 1986) often increasing after both summer and winter burns (Adams et al. 1982). *Juniperus virginiana*, on the other hand, increased on sites lacking management. This species quickly invades unburned sites and suppresses native prairie species but can be controlled via burning, cutting, and fire (Hoffman and Kearns 1997).

**Conclusions.** The species composition of dry lime prairie remnants in south-central Wisconsin has changed as habitat specialists and species that were originally rare were lost and as habitat generalist and exotic species have invaded more sites. These changes probably reflect reductions in patch size and changes in the surrounding landscapes as well as fire suppression and the invasion of taller and woody plants. We should seek particularly to protect species that are currently rare and with traits associated with vulnerable species, including short-stature, basal-leaves, and forbs that depend on small seeds for propagation. Burning these prairies and removing woody vegetation would help to sustain vulnerable and characteristic short statured species. Ideally, we should conduct such management as controlled field experiments and carefully monitor these sites to assess how they affect

the population and community trajectories. Knowing the effectiveness of alternative management practices would do much to help us sustain this threatened but dynamic plant community.

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