

# Effects of management burning on prairie insect species richness within a system of small, highly fragmented reserves

Ron Panzer<sup>a,\*</sup>, Mark Schwartz<sup>b</sup>

<sup>a</sup>*Department of Biology and Exercise Sciences, Northeastern Illinois University, 5500 North St Louis Avenue, Chicago, IL 60625, USA*

<sup>b</sup>*Department of Environmental Science and Policy, University of California, Davis, CA, USA*

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## Abstract

A growing number of entomologists are expressing concern that insect species are being extirpated from fire-managed (F-M) reserves and are urging that management burning be substantially reduced. In accord with this view, the fire attrition hypothesis predicts that fire-excluded (F-E) sites will support greater species richness, greater mean population densities, and an inordinately large number of species that are absent from F-M sites. Comparative studies of remnant-dependent (r-d) species among F-M and F-E systems in northern Illinois, northwest Indiana, and southeast Wisconsin failed to support these predictions. Our results suggest that the fire-attrition model, based on prevailing burn practices, may be applicable to few if any species. We conclude that prevailing rotational, cool season burning practices have generally been compatible with the conservation of insect biodiversity within the highly fragmented prairie reserve system in the Chicago region. © 2000 Elsevier Science Ltd. All rights reserved.

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## 1. Introduction

Most of the Midwestern tallgrass prairie ecosystem has been destroyed (Schwartz and Hermann, 1997). Fortunately, numerous remnants persist. Unfortunately, most are very small, with most being less than 40 ha in size (Robertson et al., 1997).

Despite their small size, prairie remnants can support hundreds of insect species (Hendrickson, 1930; Evans, 1975; Panzer, 1988), scores of which are incapable of inhabiting the human-dominated landscapes within which reserves are situated (Panzer et al., 1995). Isolated as small populations on what are essentially small habitat islands, these remnant-requiring or “remnant-dependent” (r-d) organisms (see Panzer et al., 1995, 1997) must contend with frequent management fires conducted predominantly for the benefit of vascular plants.

Periodic fire has long been recognized as an important if not defining component of grasslands worldwide (Vogl, 1974; Kucera, 1981; de Van Booyen and Tainton, 1984; Towne and Owensby, 1984; Axelrod, 1985; Anderson, 1990; Collins, 1990). The historic role of frequent fires in the establishment and maintenance of tallgrass prairie

throughout Midwestern North America has been especially well documented (Gleason, 1913; Curtis, 1959; Moran, 1978; Pyne, 1986; Anderson, 1990; McClain and Elzinga, 1994; Robertson et al., 1997). Fire suppression in this region clearly favors the establishment of shrubs and trees, culminating in the degradation and eventual loss of native grassland ecosystems (Bourne, 1820; Ridgeway, 1873; Gleason, 1913; Curtis, 1959; Shelford and Winterringer, 1959; Gibson and Hulbert, 1987; Leach and Givnish, 1996). Given the historical importance of fire and its utility as a means by which to exclude invasive species (Anderson, 1972; Glass, 1991; Heidorn, 1991; Hutchison 1992; Wilson and Stubben-dieck, 1997), it is not surprising that the practice of prescribed burning has become firmly established among Midwestern conservation agencies.

### 1.1. *Insects and fire*

It seems reasonable to speculate that prairie-dwelling insects are, in some measure, capable of surviving fire, once a regular and frequently occurring feature of their environment. It may, however, be unwise to assume that adaptations that served to perpetuate extensive presettlement populations scattered over vast areas will serve to protect the comparatively minuscule insect populations

\* Corresponding author. Tel.: +1-708-687-6028.

E-mail address: rpanzer@earthlink.net (R. Panzer).

that survive on the small and widely scattered remnants that have remained intact. A growing number of entomologists are expressing concern that prairie insect species may be threatened by current prescribed burning practices (Orwig, 1992; Schlicht and Orwig, 1992; Moffat and McPhillips, 1993; Minno and Minno, 1996; Swengel, 1996; Pyle, 1997; Reed 1997; Williams 1997; Dietrich et al., 1998). All fear that repeated burning will culminate in a substantial loss of insect species richness. Some are convinced that myriad species have already been extirpated from fire-managed sites (Pyle, 1997; Schlicht and Orwig, 1999). All recommend that management burning be substantially reduced.

### 1.2. *The fire attrition hypothesis (FAH)*

The view that prevailing fire-management practices are incompatible with the conservation of insect species richness can be expressed in the form of a testable hypothesis. Tallgrass prairies are generally burned on a 2 to 5 year rotation (20–50% of each site is burned per fire), a common practice throughout much of the Midwest (Swengel, 1998). Simply stated, the FAH predicts that burning at this frequency reduces population sizes by allowing insufficient time for recovery. Over time, this process is expected to culminate in the loss of fire-sensitive species, particularly those incapable of recolonization from distant habitats.

We began a study in 1992 to test the predictions of the fire attrition hypothesis. This study extended over six seasons and focused exclusively on r-d species. The fire attrition model predicts that frequent fires will result in low population densities and reduced species richness on fire-managed sites. A system of fire-excluded sites is expected to support a suite of fire-sensitive species that are absent from fire-managed sites. We used a comparative approach to test these three predictions.

## 2. Methods

### 2.1. Sites

Fire-managed (F-M) sites are defined here as sites that have experienced prescribed and/or wild fires at least twice within the past 5 years and on at least eight occasions within the past 20 years. In contrast, fire-excluded (F-E) sites have experienced zero fires within the past 5 years and two or fewer fires within the past 20 years. Twenty-six F-M sites (mean fires per year = 0.55; S.D. = 0.19) and 18 fire-excluded sites (mean fires per year = 0.03; S.D. = 0.04) were included in this study (see Panzer, 1998); all are situated within the eastern portion of the tallgrass prairie biome (Fig. 1). Study sites ranged in size from 1.5 to 600 ha, with most falling within the 5–60 hectare range. Each site supports one or more

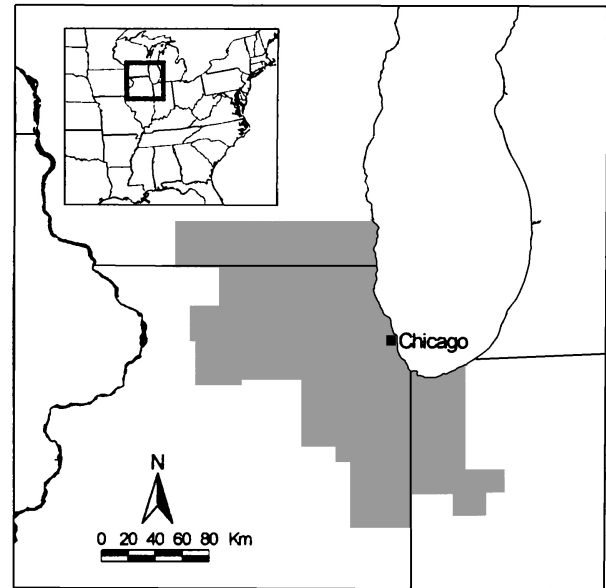


Fig. 1. Study area in southeastern Wisconsin, northeastern Illinois, and northwestern Indiana.

remnant plant communities ranging along a hydrological spectrum from xeric prairie to sedge meadow, with 41 of the 44 sites supporting mesic and/or wet prairie communities (see White, 1978). All of the sites included in this study are recognized as important natural communities by state and local conservation agencies. Thirty-five (80%) are currently managed as nature reserves. Six of the remaining nine are scheduled to be managed as reserves within the next 2 years.

### 2.2. Insect groups

Isolated as small populations within small, widely scattered reserves, r-d species are most likely to be eliminated by errant management practices. One hundred and five r-d species identified by Panzer et al. (1995) were included in this study.

Few insect groups have a greater potential for fire-sensitivity than do the remnant-requiring leafhoppers and butterflies (Panzer, 1998). Most species are univoltine and present as eggs or dormant larvae within the prairie duff in the spring and fall when prairies are burned. Both groups are widely considered to be sensitive indicators of habitat quality (New, 1991; Hamilton, 1995; Panzer et al., 1995).

Twenty-seven r-d butterfly and 67 r-d leafhopper species were included in this study. Along with 21 leafhopper and four butterfly species, three moth, three froghopper, three planthopper, one grasshopper, and one katydid species were included in the population density comparisons.

### 2.3. Leafhopper and butterfly inventories

Exhaustive leafhopper and/or butterfly inventories were conducted on 44 sites. Each site was visited on

numerous occasions ( $\geq 12$  visits) over multiple years ( $\geq 3$  years) before and during the course of this study. Both sweep nets and modified leaf blowers were used to collect leafhoppers. Butterflies were generally captured with aerial nets, identified, and released. Surveys were discontinued following six or more nonproductive visits (typically extended over two seasons) in which no new species were detected.

#### 2.4. Population density comparisons

Standard canvas nets 38 cm in diameter were used to collect leafhopper, frog hopper, planthopper, grasshopper, and katydid sweep samples randomly within similar paired F-E and F-M sites (see below). Adhesive-coated (Tanglefoot™) nine inch plastic plates were situated randomly within each treatment to sample leafhopper species. This destructive technique was used sparingly for a few days in the fall each year to sample ground-inhabiting species.

Three *Papaipema* moth species, present as eggs within the duff when prairies are burned, were sampled as late instar larvae along stratified transects that radiated in randomly chosen directions from the centers of large host plant patches. *Papaipema* moth larvae emerge and bore into plant stems each spring. The presence of a hole and frass at the base of a plant was considered to be evidence of current occupancy.

Butterfly detectability varies as a function of temperature, wind speed, and even time of day. A sampling approach that incorporated alternating sampling between treatments was employed to minimize this potential source of error. Five-minute counts were initiated from randomly chosen points by three or four investigators. Census takers proceeded at a uniform pace, conducted surveys between the hours of 10 AM and 5 PM, surveyed only on days with less than 50% cloud cover, surveyed only on days with light to moderate winds (2–6 kph), and recorded all individuals sighted.

Most sampling was done on a catch, count, and release basis. Voucher specimens were collected for less tractable species and have been deposited in the collection at Northeastern Illinois University.

#### 2.5. Data analysis

To control for variation in site size, species-area relations (log/log) were used to compare the distributions of  $r$ -d species richness among F-M and F-E sites. Slopes and heights of F-M and F-E species-area relations were compared and  $t$ -statistics were calculated following procedures outlined in Zar (1984). If the slopes and elevations of regression lines do not differ significantly, both lines can be treated as estimates of the same regression line (Zar, 1984). On this basis, the data from all sites were combined to produce joint equations for both taxa. A

Mann-Whitney two-sample test was then used to compare mean F-M and F-E residuals about each line.

Wetland species have been shown to be substantially less prone to extirpation from F-M sites (Panzer, 1998). Five butterfly and 19 leafhopper species strongly associated with sedge-dominated wetland habitats were excluded from consideration and the above procedures were repeated for both taxa to provide a more rigorous test of FAH.

Seven F-E sites, separated from similar F-M reserves by either roads, ruderal areas, residential areas, or a river were used for pair-wise comparisons of population densities between management classes. Sweep sample sizes ranged from 8 to 45 samples/treatment, with a mean of 19.5 (S.D. = 6.9;  $n = 63$ ). Moth densities were calculated based on presence/absence within host plant stems. Sample sizes, dictated by total number of stems present, ranged from 352 to 2947 stems examined/treatment (mean = 1166; S.D. = 1021;  $n = 6$ ). Mean sample densities were compared using Mann-Whitney two-sample tests to test for treatment effects within years and sites.

Finally, we examined the distribution of exclusive butterfly and leafhopper species (species found to occur in only one treatment) between systems. Randomization simulations that take into account disparities in species rarity, the distribution of species richness among sites, and the number of sites examined, were used to generate null distributions for this purpose.

### 3. Results

Comparisons between the slopes and heights of F-M and F-E species-area relations revealed no significant differences in the distribution of total leafhopper or butterfly species richness between these systems (Fig. 2a and b). Similarly, a comparison of the mean values of F-M and F-E residuals about common regression lines revealed no significant difference in the distribution of species richness between F-M and F-E systems ( $P = 0.448$ ;  $P = 0.462$ ).

The above procedures were repeated for both butterflies and leafhoppers with five and 19 wetland-associated species excluded respectively. As was the case with all species included, the slopes of the F-M and F-E species-area relations did not differ significantly for either taxon ( $P > 0.15$ ). However, the elevations of these lines were found to differ appreciably, indicating greater species richness among F-M sites (Fig. 2c and d).

Six of 27 butterfly species were restricted to two sites. Of these six species, four were found to occur exclusively on F-M sites while none were exclusive to F-E sites. We conducted 10,000 iterations of a randomization simulation to calculate the probability of this outcome. Estimating this probability is not straightforward since a greater number of F-M sites were sampled (21 versus 15) and

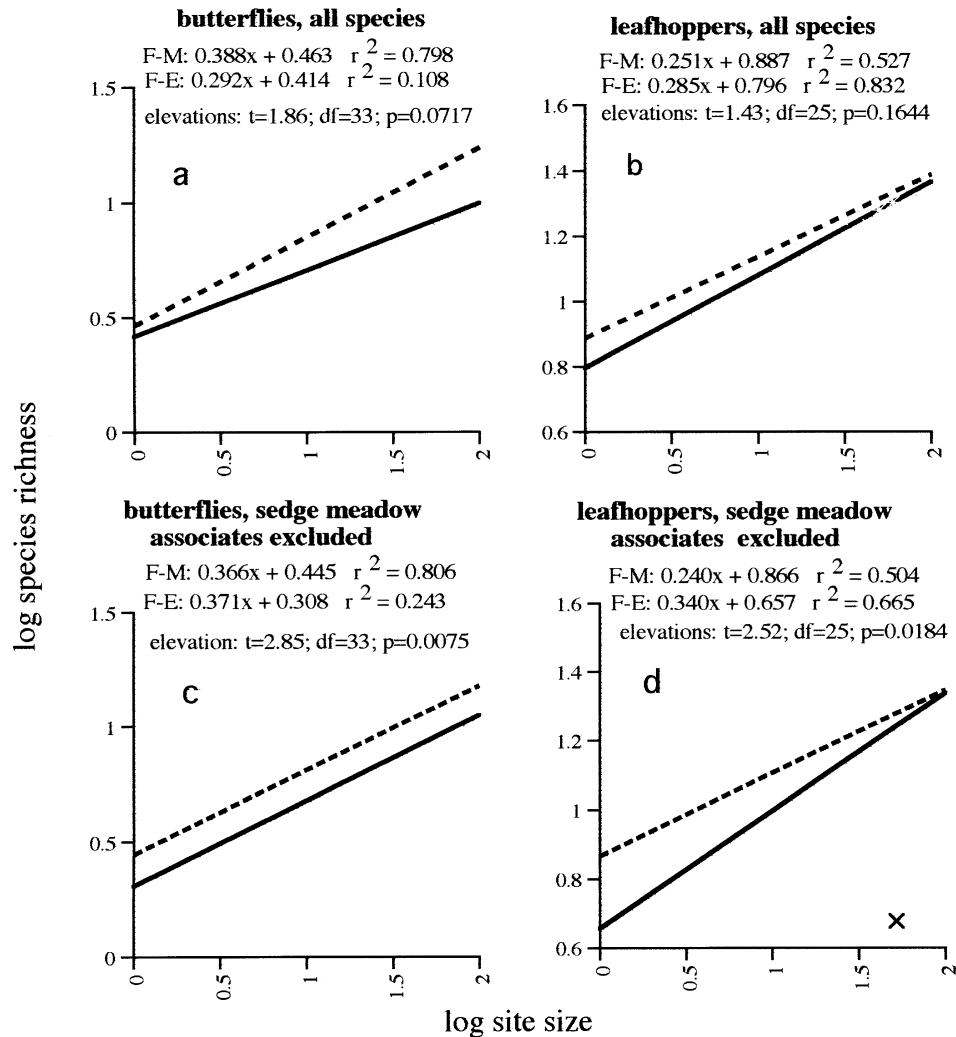


Fig. 2. Species area relations for r-d butterflies and leafhoppers on F-M and F-E prairie remnants. Broken lines represent F-M sites. Solid lines represent F-E sites. Lines representing F-M sites have greater elevation, indicating that species richness is greatest on F-M sites. Differences in elevation are significant in C and D, as indicated.

F-M sites tended to be slightly larger and to house more species. Taking these disparities into account, we distributed six species into any of two of the 31 sites. The probability of a species landing in a site was proportional to the number of species actually found on that site. The likelihood that four or more species would be found exclusively on F-M sites is  $P=0.27$ . In contrast, the likelihood that none of these 6 species would occur exclusively within the F-E system is low ( $P=0.023$ ).

Among 67 leafhopper species, 29 were found to occur exclusively in one treatment. A majority of these species (19) were limited in distribution to F-M sites. Repeating the methodology from above, we calculated the probability of finding 19 or more species only within F-M sites. Our observations included: (a) nine species found on four sites, three of which were exclusive to F-M sites and one that was exclusive to F-E sites; (b) four species found on three sites, two of which were restricted to

F-M sites; (c) six species found on two sites, one of which was restricted to F-M sites; and (d) 22 species found on a single site. The probability of finding at least 19 species to be restricted in occurrence to F-M sites is low ( $P=0.005$ ). The probability of finding 10 or fewer species on F-E sites is  $P=0.068$ . Thus, for both butterflies and leafhoppers, we find a pattern in which more species tend to be restricted to F-M sites and fewer species tend to be restricted to F-E sites than might be expected due to chance alone.

Finally, mean population densities within paired sites were found to be similar or higher on F-M sites in 79% of the tests conducted (e.g. Fig. 3). Among 25 species known to be fire-sensitive (Panzer, 1998), half of the populations examined (24/48) occurred in significantly greater numbers within F-M sites. Only 4/48 populations were found to occur in significantly greater numbers within F-E sites (Table 1).

4. Discussion

In disagreement with the FAH, both species richness and mean population densities were generally found to be similar or higher on frequently-burned sites as compared with F-E sites. When less fire-sensitive wetland species were excluded from regression analyses, F-M

sites supported greater numbers of both leafhoppers and butterflies, not less. In further disagreement with the FAH, the distribution of exclusive species indicates a strong trend toward species absence from F-E sites. Nine leafhopper species were found to occur exclusively on F-E sites; however, all are clearly wetland inhabitants (see DeLong, 1948; Panzer et al., 1995), reducing the likelihood that these occurrences can be attributed to the absence of fire (Panzer, 1998). These results, taken in total, suggest that the fire-attribution model, based on prevailing burning practices, is applicable to few if any insect species.

Experimental data indicate that many prairie-inhabiting insect species are not negatively impacted by rotational burning (Panzer, 1998). Among those that are, recovery is generally rapid (Panzer, 1998). The comparative data presented here suggest that 26 F-M sites, at least half of which have been rotationally burned 10 or more times within the past 20 years, have lost few, if any, r-d butterfly or leafhopper species. These diverse lines of evidence indicate that modern burn regimes have generally been compatible with the conservation of insect biodiversity and provide strong support for a F-M model where r-d species richness and population density are maintained under a management regime of incremental (rotational) burning.

Pyle (1997) and Schlicht and Orwig (1999) have speculated that the practice of rotational burning has resulted in the extermination of remnant-requiring insect species from numerous sites within the Midwest. The distribution of species richness among F-M and F-E sites, along with the under representation of unique species among F-E sites within the region studied here do not support this assertion. In contradiction to the predictions of the fire attrition scenario implied by these observers, the data presented here suggest that rotational burning has contributed to the protection of several species that would otherwise have been lost.

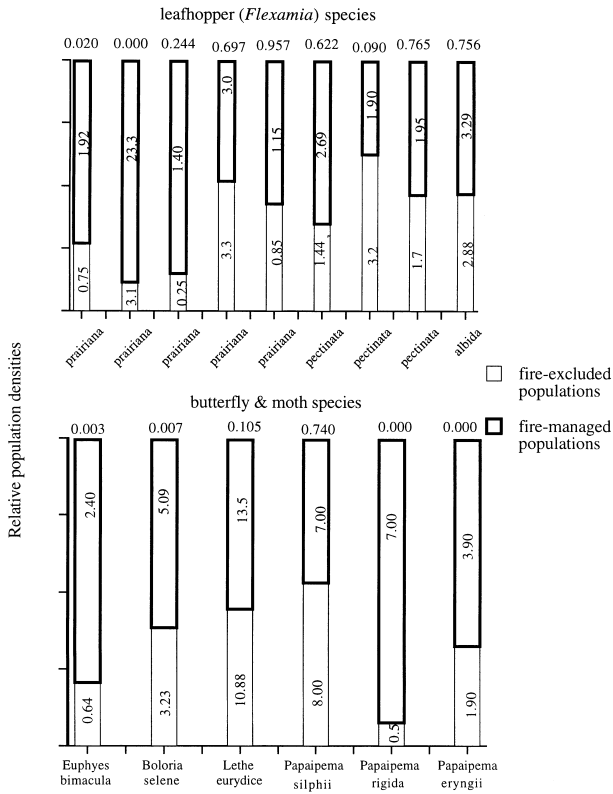


Fig. 3. Population density comparisons for r-d leafhopper (3), butterfly (3) and moth (3) species within one or more pairs of disjunct F-M/F-E prairie sites. Sample means are shown within bars. Significance values (Mann–Whitney) are shown above bars.

Table 1

Summary of 2-tailed Mann–Whitney and 2x2 chi square test results for 36 insect species on F-E and F-M prairie remnants in the Chicago region<sup>a</sup>

	No. of species considered	Greater population densities were recorded in:		
		F-M sites	No significant difference between sites	F-E sites
Family Acrididae: grasshoppers	1	–	1 (1)	–
Family Tettigoniidae: katydids	1	–	1 (1)	–
Family Cicadellidae: leafhoppers	21	20 (19)	22 (15)	2
Family Cercopidae: froghoppers	3	1 (1)	–	3 (3)
Family Issidae: planthoppers	3	2 (1)	1 (1)	1 (1)
Order Lepidoptera: butterflies	4	2 (1)	2 (1)	–
Order Lepidoptera: moths	3	2 (2)	1 (1)	–
Totals	36	27 (24)	28 (20) <sup>b</sup>	6 (4)

<sup>a</sup> Twenty-five of these species are known to be fire sensitive (Panzer, 1998). Data for these species is shown parenthetically.

<sup>b</sup> F-M population densities were larger in 21 of 28 cases where statistical power was insufficient to produce significant test results.

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