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Managing for Habitat Heterogeneity in Grassland Agro-Ecosystems Influences the Abundance of Masked Shrews 

(Sorex cinereus)

Joseph J. Nocera
Kimberly L. Dawe

ABSTRACT. Many agri-environment programs prescribe modified farming practices and provision of heterogeneous habitat types, such as fallow field edges and shelterbelts. The conservation benefit of these actions to biodiversity has been well described. However, the response of insectivorous small mammals to habitat heterogeneity in agro-ecosystems has received little empirical attention. We compared the abundance of the masked shrew (Sorex cinereus) between hayfields and adjacent mesic old-fields in Nova Scotia, Canada. Masked shrews were almost twice as abundant in old-field habitat and were linked closely to abundance of Aranea (spiders), Coleoptera (beetles), and Malacostraca (slugs). This implies that mesic old-field habitat is beneficial to masked shrews and is a likely population source for shrews that occur in hayfields. Because masked shrews consume large quantities of invertebrate prey (that are often considered agricultural pests), we suggest that old-fields are an...
important habitat to include in agro-ecosystems that should benefit both wildlife and agricultural producers.

**KEYWORDS.** Agri-environment program, hayfield, masked shrew, prescribed habitat heterogeneity, *Sorex cinereus*

**INTRODUCTION**

Agricultural intensification is one of the greatest anthropogenic risks to animal conservation (Chamberlain et al., 2000; Green et al., 2005; Krebs et al., 1999). Many agri-environment programs have been implemented to counteract the typically negative effects of agricultural practices on wildlife. For example, the Conservation Reserve Program in the USA and the Permanent Cover Program in Canada retire environmentally sensitive farmlands through prescribed vegetation management.

The provision of non-cropped successional habitat (hereafter “old-fields”) is a common management prescription of programs directed at working agricultural landscapes, although the required degree of succession varies substantially—ranging from grassy margins mowed every few years to scrub-dominated fallow fields. Old-field habitats are valuable for increasing plant (Boutin et al., 2002; Jobin et al., 1997) and invertebrate diversity (Schmidt et al., 2005) and promoting the abundance and reproduction of breeding birds (Arnold, 1983; Benton et al., 2002; Vickery et al., 2001). The benefit of prescribed old-field habitat to insectivorous mammals has been less well studied. In particular, small mammals can be indicators of habitat quality and are thus an important group to investigate for response to such habitat management (Olson and Brewer, 2003).

Because small mammals are sensitive to habitat alterations, have short life-cycles, and are sampled easily, several aspects of their ecology in agro-ecosystems have been investigated, such as (1) species assemblages in managed prairies (Sietman et al., 1994); (2) community response to crop cover (Navo and Fleharty, 1983; Olson and Brewer, 2003) and configuration (Peles et al., 1997); and (3) the effects of agricultural practices such as mowing (Slade and Crain, 2006) and agriculturally-induced habitat fragmentation (Maisonneuve and Rioux, 2001; Millan de la Pena et al., 2003). Although the habitat used by small mammals is a function of landscape structure and cover vegetation (Churchfield et al., 1997;
Sietman et al., 1994), we know little about the response of insectivorous small mammals to prescribed habitat heterogeneity from agri-environment programs.

Renewed calls for attention to balancing agricultural production and biodiversity (Badgley, 2003; Sanford, 2006) have underscored the importance of learning how habitat prescriptions influence the diversity and abundance of wildlife, such as small mammals. Indeed, because many animal populations are becoming increasingly imperiled as more land becomes devoted to large-scale industrial agriculture, there is a need to study the wildlife benefits of non-cropped successional habitat types (Sutherland et al., 2006). However, the effectiveness of agri-environment programs is problematical to evaluate. For instance, it can be challenging to define and measure biodiversity responses (Kleijn and Sutherland, 2003). Suitable sites for comparative studies can be difficult to find because many programs are tailored to rare species of conservation concern. (e.g., Perkins et al., 2003). Therefore, in this study, we chose the masked shrew (Sorex cinereus [Kerr]), a generalist species that is common and distributed widely throughout North America (Whitaker, 2004), to examine whether habitat heterogeneity, common to prescriptions in managed agro-ecosystems, can promote the abundance of small mammals.

Abundance of masked shrews fluctuates yearly, and seasonally, with peaks in late August to early September during the breeding season. Variations in abundance have been related to rainfall and interspecific interactions (Bellocq and Smith, 2003; McCay and Storm, 1997). However, prey availability and, particularly, moisture and vegetation cover are the most important determinants of abundance for masked shrews (McCay and Storm, 1997; Pagels et al., 1994). When dense and persistent enough to produce a ground litter layer, vegetation provides protective cover and foraging microhabitats for insectivorous shrews (Dickman, 1988; Whitaker, 2004) and helps to retain favorable soil moisture levels (MacCracken et al., 1985). Although they are habitat generalists, masked shrews form prey preferences, and fluctuations in preferred invertebrate species may activate changes in abundance of shrews (McCay and Storm, 1997). Thus, we predicted that masked shrews would be more abundant in mesic old-field habitats than in drier hayfields and that this relationship would also be evident through differences in invertebrate assemblages between these habitats. If this prediction were correct, it would imply that provision of old-field habitat, in an agricultural grassland landscape, provides a population benefit to insectivorous small mammals. We test
these predictions by comparing the abundance of masked shrews, measured as capture success, between old-field and hayfield habitats. Each habitat type was characterized according to vegetation cover, height and density. We also modeled shrew abundance in these two habitat types with the abundance of eight orders of invertebrates.

**METHODS**

**Study Area**

We sampled three agricultural sites (centered around 44°45’N, 65°31’W) in the western Annapolis Valley of Nova Scotia, Canada: Belleisle Marsh Wildlife Management Area (hereafter “Belleisle”; 210 ha of terrestrial habitat), Upper Belleisle (116 ha of terrestrial habitat), and Queen Anne (180 ha of terrestrial habitat), which are all cultivated to support beef cattle operations. Historically, the areas were extensive salt marsh, but dykes built in the 17th century and a causeway in the 1960s converted and maintain these lands in a suitable state for agriculture.

Queen Anne is privately owned and managed exclusively by the land-owners. Belleisle and Upper Belleisle are predominantly government-owned and managed for wildlife and agriculture in a multi-faceted agri-environment program that integrates human land use and wildlife habitat. Active hayfields, mesic old-fields, and wetland impoundments constitute an equal amount of land cover at these two sites, but Queen Anne has no wetlands. Ditches of equal depth surround every field (both hayfield and old-field) and are the primary form of anthropogenic boundary features present.

Hayfields were planted (in the late 1980’s, early 1990’s) with large patches of grass mixtures of timothy (*Phleum pratense* L.), meadow fox-tail (*Alopecurus pratensis* L.), bluegrass (*Poa* spp.) and reed canary grass (*Phalaris arundinacea* L.). Except for a small amount (3.7 ha) of early-harvest silage produced at Queen Anne, harvest is wholly in the form of haylage. The date of first harvest at all sites is post 1 July, except for the small field cut for silage at Queen Anne.

Goldenrods (*Solidago* spp.), meadowsweet (*Spirea latifolia* W. Aiton), sedges (*Carex* spp.), elders (*Sambucus canadensis* L. and *S. pubens* Michx.), and wild rose (*Rosa virginiana* Mill.) typify the numerous old-fields at Queen Anne, Belleisle, and Upper Belleisle (96, 82, and 40 ha, respectively). Production at all old-fields ceased > 10 years prior to this study.
Study Species

Masked shrews are habitat generalists and show a great diversity in the habitats they occupy. In the central and southern Appalachians and Ohio River Valley, masked shrews prefer mature, mesic forest. However, we studied the subspecies *S. c. acadicus* (Gilpin), which is distributed throughout many habitat types (from forests to agro-ecosystems) in Nova Scotia and New Brunswick, Canada (Hartling and Silva, 2004; Stewart and Baker, 1997). Masked shrews live approximately 15 months and are active day and night (Churchfield, 1990). Their reproductive season begins in early spring and abundance peaks by late summer (Bellocq and Smith, 2003). During this period, populations consist of both over-wintering adults and young of the year, who themselves may breed before winter.

Field Procedures

From 1 June to mid-July of 2003 and 2004, we monitored the catch-per-unit-effort (capture abundance; # of individuals captured per 100 trap nights, where a trap night = one trap set for one night) of masked shrews using pitfall traps. Traps were haphazardly located among the old-field and hayfield habitats in each study site with the restriction that traps be placed > 20 m apart, as the typical home range of masked shrews is 0.04 ha (Whitaker, 2004). Traps were distributed equally between hayfields and old-fields (n = 16 in each), but were distributed unequally among sites, owing to differences in area (traps in hayfields: Belleisle = 8, Upper Belleisle = 4, Queen Anne = 4; traps in old-fields: Belleisle = 8, Upper Belleisle = 6, Queen Anne = 2). Traps were deployed in the same location each year.

Traps were checked weekly, totaling 736 trap nights in 2003, and 729 in 2004. We discontinued trapping when the field in which a point was located was harvested. If a field was not harvested by mid-July (e.g., all old-fields), we discontinued trapping by 21 July.

Pitfall traps were plastic cups (15 cm deep) inserted flush to soil surface, with ~5 cm of soap and water mixture on the bottom. Pitfall trapping is a successful (Hartling and Silva, 2004; Innes et al., 1990) and acceptable method to sample shrews under the guidelines established by the American Society of Mammalogists (American Society of Mammalogists, Animal Care and Use Committee, 1989).

Removal trapping is biased towards dispersers that move into the area when residents are removed and thus, may elevate absolute abundance measures (Bellocq and Smith, 2003; Sullivan et al., 2003). Because this
bias is consistent across sites, and we are interested in differences in capture abundance between sites, it is not expected to influence our results.

Ground-dwelling (epigeal) invertebrates were sampled with the same pitfall traps used to capture shrews. All invertebrate specimens captured during weekly checks were counted, and identified to order or further, when possible. A total of 19,900 epigeal invertebrates was sampled in 2003, and a further 20,975 individuals in 2004.

Grass height and density were simultaneously measured at each trap location with a visual obstruction “Robel-pole” (Robel et al., 1970). Robel-pole measurements were made in late May at least 5 m from the pitfall trap in each cardinal direction to establish a mean value for each trap. Mean vegetation cover and grass:forb ratio were estimated at trap locations using sampling frames (Daubenmire, 1959) thrown in each cardinal direction (a minimum of 5 m). Also within the frame, litter depth was measured (with a ruler against a profile cut from the soil).

Analytical Procedures

Our objective was to compare capture abundance of masked shrews between old-fields and hayfields, and to model capture abundance with a suite of habitat-related variables. Differences in data structure predisposed the analyses to two different types of models. We first modeled shrew abundance with discrete variables associated with sampling and habitat type; we did not include invertebrates or vegetative features at this point because we wanted to first ascertain if there was a difference between habitat types. Our second phase of modeling sought to answer why there might be a difference, so we then modeled shrew abundance with variables related to invertebrates. Because features differed so strongly between hayfields and old-fields, we did not include any other variables associated with vegetation in either modeling exercise. We simply identify these differences using descriptive statistics.

Model I: Capture Abundance and Habitat

We used linear mixed-effects models (LME, Mathsoft, 2000) to assess whether capture abundance of masked shrews differed between habitat types. We repeatedly sampled the same points, which creates the possibility of pseudoreplication, so we chose LME for our analyses because it accounts for the non-independence of errors created by such pseudoreplication. LME does this by estimating how the mean is influenced by predictive variables (fixed effects) and predicts how intra-group correlation (random
effects) influences the variance. As predictive variables, we used a binary habitat classification (hayfields vs. old-fields) as a fixed effect, and included trap (nested within study area and year) as a random effect. We compared variance components (the amount of variance attributable to particular variables) to assess model fit.

**Model II: Capture Abundance and Invertebrates**

Because variance in invertebrate capture was not a function of sample design (i.e., there is no reason to suspect that invertebrate capture differed systematically according to sample point layout), we used a standard linear regression model to examine the relationship between capture abundance of masked shrews and the abundance of individuals from eight orders of invertebrates. Diagnostic plots for models were visually assessed for adherence to normality assumptions, distribution of residuals, and presence of outliers. We removed outliers if the removal resulted in a reduction in spread of residuals (error variance) compared to the fitted model. Further, we removed terms with high collinearity (Graham, 2003; Hair et al., 1987) if an assessment of correlation coefficients showed that a predictor was more related to another better predictor than it was the dependent variable.

Stepwise model selection was based on Akaike’s Information Criterion (AIC; Akaike, 1974; see review by Burnham and Anderson, 2002). AIC is a goodness-of-fit parameter that identifies the model(s) with the fewest parameters that can best explain the data, which requires a stepwise comparative assessment of models. AIC is not based on statistical hypothesis testing, but ranks models according to difference in their AIC scores (the “evidence ratio”). Best-fit models show evidence ratios between 1 and 3. Models with evidence ratios between 3 and 6 earn moderate support, and > 6 have no support (Burnham and Anderson, 2002). The AIC model-selection procedure does not provide estimates of model accuracy or discrimination, so we provide F-values and coefficients of determination from our regression analyses. For all statistical analyses, we set the maximum probability of a type-I error at 0.05.

**RESULTS**

Measurements of all variables (including summary in Table 1) are presented as mean ± SD, unless specified otherwise. We detected masked shrews at 91% of sample points and capture abundance was 4.84 ± 4.94
We captured 63 masked shrews in 2003 (3.94 ± 2.79 per trap) and 92 in 2004 (5.75 ± 6.39 per trap), suggesting a moderate interannual population increase. Capture abundance differed between habitat types ($t = 5.15$, $p < 0.0001$; Figure 1). Overall, we detected a mean capture abundance of 3.31 ± 2.44 individuals per trap in hayfields, and 6.38 ± 6.28 individuals per trap in old-fields (Figure 1). The influence on capture abundance of repeatedly sampling a point was negligible both across sites (SD = 0.24) and between years (SD = 0.01) compared with the intercept (SD = 5.71).

Beetles comprised the most abundant invertebrate order (Table 1), followed by arachnids. Grasshoppers and harvestmen were the least abundant. The best-fit model of masked shrew capture abundance retained the terms of abundance of arachnids, beetles, and slugs ($R^2 = 0.50$, $F_{3,28} = 9.37$, $p = 0.0001$). No outliers or terms with high collinearity were detected. The second best-fit model had no support (evidence ratio = 18.9). The direction of the relationship was negative between masked shrew and beetle abundance ($t = -1.91$), but was positive with the abundance of arachnids ($t = 2.64$) and slugs ($t = 1.88$).

TABLE 1. Mean values ± standard deviation (SD) and range of 13 variables measured in relation to *Sorex cinereus* capture abundance

<table>
<thead>
<tr>
<th>Variable</th>
<th>Mean ± SD</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grass density / height (Robel pole reading) in May (dm)</td>
<td>2.81 ± 1.00</td>
<td>1.00–5.80</td>
</tr>
<tr>
<td>Litter depth (cm)</td>
<td>4.43 ± 3.12</td>
<td>0–11.5</td>
</tr>
<tr>
<td>Forbs (% of vegetation per quadrat)</td>
<td>26.30 ± 24.85</td>
<td>0–96.25</td>
</tr>
<tr>
<td>Grass (% of vegetation per quadrat)</td>
<td>73.70 ± 24.85</td>
<td>3.75–98.50</td>
</tr>
<tr>
<td>Cover (% of quadrat with vegetation)</td>
<td>63.12 ± 15.01</td>
<td>24.50–85.75</td>
</tr>
<tr>
<td>Aranea (spiders)*</td>
<td>8.12 ± 4.28</td>
<td>1.80–17.13</td>
</tr>
<tr>
<td>Coleoptera (beetles)</td>
<td>9.63 ± 11.29</td>
<td>0.98–47.82</td>
</tr>
<tr>
<td>Hemiptera (true bugs)</td>
<td>0.77 ± 2.50</td>
<td>0–13.07</td>
</tr>
<tr>
<td>Isopoda ( sowbugs)</td>
<td>3.69 ± 3.82</td>
<td>0–22.60</td>
</tr>
<tr>
<td>Lepidoptera (butterflies, moths)</td>
<td>0.09 ± 0.08</td>
<td>0–0.31</td>
</tr>
<tr>
<td>Malacostraca (slugs)</td>
<td>0.77 ± 1.55</td>
<td>0–7.02</td>
</tr>
<tr>
<td>Opiliones (harvestmen)</td>
<td>0.09 ± 0.17</td>
<td>0–0.65</td>
</tr>
<tr>
<td>Orthoptera (crickets, grasshoppers)</td>
<td>0.04 ± 0.05</td>
<td>0–0.24</td>
</tr>
</tbody>
</table>

*all invertebrate orders/classes are presented as catch-per-unit-effort (CPUE) per trap night.
Vegetation variables (Table 1) did not differ between years, but differed noticeably between habitat types (Table 2). A slightly deeper litter layer, greater grass height and density in May, and a greater proportion of forbs typify old-fields. Hayfields have a higher grass composition and a greater percentage of ground cover.

Although not discussed here, we note incidental capture of the following individuals from non-target species: meadow vole (*Microtus pennsylvanicus* (Ord); 7), short-tailed shrew (*Blarina brevicauda* (Say); 2), and pygmy shrew (*Sorex hoyi* (Baird); 1).
Habitat heterogeneity is essential to wildlife conservation on farmlands; it is associated with increased diversity of plants, invertebrates, and birds (Benton et al., 2003). Our study provides further evidence to support this relationship: the provision of non-cropped old-fields supported greater abundance of a small mammal, the masked shrew, than active hayfields did. This finding is consistent with the similar finding that masked shrew abundance can be higher in irrigated than in non-irrigated forest habitats (McCay and Storm, 1997). We posit that provision of old-field habitat offers benefits to wildlife. Depending on the wildlife species that benefit from habitat heterogeneity, the farmer may also benefit from the services they inadvertently provide.

A larger population of masked shrews in old-fields should represent a population source for individuals to disperse into less optimal hayfields. As a population source of small mammals, old-fields may also benefit predators of small mammals (e.g., owls [Swengel and Swengel, 1992] and weasels [Nichols and Nichols, 1935]). However, although old-fields may support or produce more shrews, they are also characterized by denser vegetation, which may make hunting shrews more difficult. Also, because masked shrews prey upon many invertebrates that are considered agricultural pests (e.g., chinch bugs [Blissus spp.], leafhoppers [Cicadellidae]; see review by Whitaker, 2004), old-fields benefit farmers as an in situ source of natural pest control.

Churchfield and colleagues (1991) found this to be true for a suite of small mammal species in England that included the shrew species Sorex araneus. Additionally, Maisonneuve and Rioux (2001) found insectivores,
such as the masked shrew, were in greater abundance in riparian zones that did not support rodent pest species, further representing the benefits to farmers of maintaining these habitats.

In turn, old-fields have a more complex and diverse vegetation structure and deeper litter than hayfields, all of which provide more substrate for insects and their reproduction; our results support this suggestion. The increase in insectivorous shrews is expected to counter-balance this, as they can consume up to 6800 prey/ha/day (Churchfield et al., 1991). Laboratory study has shown that masked shrews prefer arachnids in their diet (Bellocq and Smith, 1994), which is consistent with the relationship we detected. Our results also corroborate those of Snyder and Best (1988) who showed correlations between masked shrew abundance and vegetation density. However, because masked shrews in our study were more abundant in old-fields, which had less ground cover than neighboring hayfields, our results contrast with those of Innes and colleagues (1990), who found shrew abundance to be correlated with greater ground cover (and thus a more suitable microclimate).

In our study, we detected differences in abundance of masked shrews in response to managed agricultural habitats. Because the species is both abundant and widespread, our results are applicable beyond the boundaries of our study region, providing further incremental evidence to pressing questions in agro-ecosystem management (Sutherland et al., 2006). Caution is applied in this case, however, as landscape scale patterns can affect small mammal abundance, and thus should be considered when predicting responses across new landscapes (Mazerolle and Villard, 1999). Determining the management options to benefit wildlife in certain non-cropped habitats still requires much further study with other species in other agro-ecosystems as well. Nonetheless, our results support the suggestion (Butler et al., 2005) that integrating specific management options into agri-environment programs, such as prescribing habitat heterogeneity by maintaining old fields, will generally benefit most wildlife in a way that is not financially prohibitive and can be easily implemented by the farmer.

REFERENCES


