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Agriculture, Ecosystems and Environment 116 (2006) 235–243

Agriculture Ecosystems & Environment

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Plant and small mammal diversity in orchard versus non-crop habitats

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Received 21 January 2005; received in revised form 6 February 2006; accepted 23 February 2006
Available online 4 April 2006

Abstract

This study was designed to determine the abundance and diversity of vascular plant and small mammal communities in a mosaic of orchard and non-crop habitats in an agricultural landscape. Study areas were located at Summerland, British Columbia, Canada where seven replicate habitats: old field, sagebrush, dwarf, and conventional apple orchards, ponderosa pine forest, hedgerow, and riparian were intensively sampled for vascular plant (1999) and small mammal communities (1999–2003). Total plant species sampled included 104 herbs, 26 shrubs, and 9 trees. Mean crown volume index of herbs was similar among sites. Hedgerow and riparian habitats had substantial shrub layers, and the conventional orchard, pine forest, and riparian habitats had the highest biomass of trees. Total mean species richness of plants was similar, but did range from 12.3 species in old field sites to 32.3 species in sage sites. Overall plant species diversity and structural diversity were highest in the sage, hedgerow, and riparian habitats. Total structural diversity was positively related to total species richness and species diversity of vegetation. Mean total abundance of small mammals ranged from 28.1 to 37.0 ha^{-1} across old field, sage, and riparian habitats compared with a range of 6.2–16.7 animals/ha in the other habitats. Old field and sage habitats generally had the highest levels of species richness and diversity of small mammals, although the other non-crop habitats were similar to these in some years. Structural diversity of vegetation appeared to be a reasonable indicator of biodiversity, at least for vascular plants and small mammals, and should be included in future assessments of diversity in agroecosystems.

Keywords: Agroecosystems; Biodiversity; Non-crop and orchard habitats; Small mammals; Species richness; Vegetation

1. Introduction

The negative impact on native species and habitats by agricultural land use, primarily by agrochemicals (pesticides and fertilizers), grazing, introductions of exotic species, and modification of natural habitats, is well documented (Pimentel et al., 1992; McLaughlin and Mineau, 1995). Agricultural landscapes that have maintained mosaics of farmland habitats and remnant natural habitats of woodlots, hedgerows, shelterbelts, and riparian zones may offer an opportunity to conserve biodiversity while maintaining food production (Paoletti et al., 1992). Hedgerow and riparian habitats are particularly valuable for conservation of plant diversity in agricultural landscapes (Bunce and Hallam, 1993; Boutin et al., 2002). In addition, the varieties of habitats within agricultural landscapes help to shelter beneficial invertebrates (Altieri and Nicholls, 1999) and provide habitats for birds (Best, 1983).

Conservation of plant species diversity and structural diversity in both crop and non-crop areas would seem essential to maintenance of habitats for terrestrial wildlife species in agricultural landscapes. Freemark et al. (2002) reported on the importance of maintaining a diverse mosaic of crop and non-crop habitats in farmland for conserving species of vascular plants. Other studies have documented

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0167-8809/$ – see front matter © 2006 Elsevier B.V. All rights reserved.
doi:10.1016/j.agee.2006.02.010

the importance of non-crop habitats as refuges for those plant species indicative of natural regional vegetation (Boutin and Jobin, 1998).

Terrestrial small mammals are also common inhabitants of agricultural landscapes where they are an important prey source for a wide variety of avian, reptilian, and mammalian predators (Martin, 1994). Vegetative stratification providing for food and cover from predators appeared to be an important factor for habitats to support small mammal communities (Yahner, 1983; Fitzgibbon, 1997). However, Tattersall et al. (2002) found that the linear or non-linear character of habitats had little effect on abundance and diversity of small mammals. Clearly, the ability to predict a positive association between structural diversity of vegetative components and the abundance and diversity of small mammals would have considerable utility in enhancing plant and mammal conservation in agricultural landscapes.

Thus, this study was designed to determine the abundance and diversity of vascular plant and small mammal communities in a mosaic of orchard and non-crop habitats in an agricultural landscape. Another objective was to test the hypothesis that the abundance and species diversity of mammals would be positively related to the abundance and structural diversity of vegetation among these habitats.

2. Materials and methods

This study was located in the Okanagan Valley at the Pacific Agri-Food Research Centre and in Prairie Valley, Summerland, British Columbia, Canada (49°34′N; 119°40′W). Seven habitat types were distributed over these two adjoining valleys: old field, sagebrush, dwarf apple (*Malus domestica* Borkh.) orchard, conventional apple orchard, ponderosa pine (*Pinus ponderosa* Dougl.) forest, hedgerow, and riparian. The old field, sagebrush, and ponderosa pine forest sites were located at the Research Centre. The old field sites were abandoned ($\geq$25 years) hay fields composed of several grass (forage species) and other herbaceous species. These sites were each 2–3 ha in area. The sagebrush sites were natural habitats that have been relatively undisturbed, except for some grazing by feral horses. Some shrubs and herbs were present in this habitat type. These sites were each 5–6 ha in area. The ponderosa pine forest sites were also natural habitats which have had little disturbance, except for grazing by horses. Ponderosa pine was the major tree species with some herb and sporadic shrub species in the understory. These sites were each 5–6 ha in area.

The dwarf orchard, conventional orchard, hedgerow, and riparian habitats were located in Prairie Valley. The dwarf rootstock apple orchards were 2- to 3-year-old Imperial/Royal Gala cultivars in spacings of 0.5–1 m × 3–4 m. The conventional apple orchards were 10- to 30-year-old McIntosh and Red Delicious cultivars at spacings of 3–4 × 5 m. These orchards were mowed in alleys and sprayed with Roundup® herbicide for weed control along tree rows three to four times per growing season. Fungicides for control of powdery mildew (*Podosphaera leucotricha* (Ell et Everh.) Salm.) and apple scab (*Venturia inaequalis* (Cooke) Winter) were applied two to three times per season. A sterile insect release (SIR) program for codling moth (*Cydia pomonella* (L.)) has essentially eliminated the need for insecticide sprays for this pest. Nitrogen fertilizer was applied each spring either as ammonium nitrate (34-0-0) or urea (46-0-0) at rates dependent on tree growth and nutrition. Orchard floor vegetation was composed of several common herbaceous species. These orchard sites were each 4–7 ha in area.

The hedgerows formed border habitats between adjacent orchards or orchards and old fields. Vegetation along hedgerows was composed primarily of shrubs and some herbs. Hedgerows were usually up to 2 m wide and at least 100 m in length. The riparian habitats were located along a year-round flowing stream which bisected Prairie Valley. Tree and shrub species dominated these habitats with some herbs in the understory vegetation. Riparian sites were each 2–4 ha in area.

The study had a completely randomized design with three replicate sites of each habitat type. The 21 sites (7 habitat types × 3 replicates) were selected on the basis of availability of reasonably uniform vegetative characteristics of a given habitat type, proximity to one another, and availability of orchard sites that were the size of typical farming operations in the Okanagan Valley. All sites were far enough apart (0.12–0.62 km) to be statistically independent.

2.1. Vegetation

Three 25 m transects, consisting of five 5 m × 5 m plots were randomly located in each site (except for the hedgerows which each had two transects) following the method of Stickney (1980). Each plot contained three sizes of nested sub-plots: a 5 m × 5 m plot for sampling trees; a 3 m × 3 m sub-plot for sampling shrubs; and a 1 m × 1 m sub-plot for sampling herbs. Tree, shrub, and herb layers were subdivided into height classes. A visual estimate of percentage cover of the ground was made for each species height class combination within the appropriate nested sub-plot. Total percentage cover for each layer was also estimated. These data were then used to calculate a crown volume index (m^3/0.01 ha) for each plant species. The product of percentage cover and representative height gave the volume of a cylindroid which represented the space occupied by the plant in the community. Crown volume index values were then averaged by species for each plot size and converted to a 0.01 ha base to produce the values given for each species and layer (herbs, shrubs, and trees). Sampling was done once in June–July 1999. Plant species were identified in accordance with Hitchcock and Cronquist (1973) and Parish et al. (1996). Species richness, species diversity, and structural diversity were calculated for these data.

2.2. Small-mammal communities

Small-mammal populations were sampled at 4 week intervals from April to November 1999, April to October 2000, April to December 2001, and April to August 2002; and at 5–8 week intervals from December 2001 to March 2002 and from August 2002 to March 2003. One trapping grid (1 ha), with 49 (7 × 7) trap stations at 14.3 m intervals, and one Longworth live-trap at each station were located in each site (non-linear habitats), except for three sites that had irregular-shaped 1 ha area grids. One trapping line with seven trap stations at 14.3 m intervals with four Longworth live-traps at each station was located in each replicate site of the two linear habitats (hedgerow and riparian). Traps were supplied with whole oats and carrot, with cotton as bedding. Traps were set on the afternoon of Day 1, checked on the morning and afternoon of Day 2 and morning of Day 3, and then locked open between trapping sessions.

Terrestrial small mammals sampled by this procedure included the deer mouse *Peromyscus maniculatus* Wagner, montane vole, long-tailed vole *Microtus longicaudus* Merriam, northwestern chipmunk *Tamias amoenus* J.A. Allen, western harvest mouse *Reithrodontomys megalotis* Baird, Great Basin pocket mouse *Perognathus parvus* Peale, house mouse *Mus musculus* L., and the wandering shrew *Sorex vagrans* Baird. Although the northern pocket gopher also occurred in the study areas, it rarely appeared above ground and only two individuals were captured. All small mammals (except shrews) captured were ear-tagged with individually numbered tags, and released at point of capture (Krebs et al., 1969). There was, unfortunately, a high mortality rate for shrews in traps overnight, but this was unavoidable in practice. Shrews that died in traps were collected and identified according to Nagorsen (1996). Seasons were defined as summer (April–September) and winter (October–March) periods. Thus, there were four summer and two winter periods that had at least five trapping sessions.

Population estimates for the deer mouse, montane vole, Great Basin pocket mouse, and northwestern chipmunk, and in some cases for the western harvest mouse, were derived from the Jolly–Seber (J–S) stochastic model (Seber, 1982). The minimum number of animals known to be alive (MNA; Krebs, 1966) was used as the population estimate for the first and last sampling weeks of the study when the J–S estimate was not calculated. The reliability of the J–S model declines when population sizes are very low and no marked animals are captured (Krebs et al., 1986). Thus, the total number of individuals captured was used to compare populations of the long-tailed vole, house mouse, and wandering shrew.

2.3. Diversity measures

Species richness, species diversity, and structural diversity were used to measure ‘‘habitat diversity’’. Species richness was the total number of species sampled for the plant (herbs, shrubs, and trees) and small-mammal communities in each site (Krebs, 1999). Species diversity was based on the Shannon–Wiener index which is well represented in the ecological literature (Magurran, 2004). For the plant communities, species diversity was calculated using the crown volume index for each plant species averaged across the three transects (two transects in hedgerows) in a given site. This measurement of diversity was calculated separately for herbs, shrubs, trees, and for overall vegetation as total species diversity.

Structural diversity, or foliage height diversity, was based on the same indices as for species richness and diversity, with the height classes of each of the herb, shrub, and tree layers acting as ‘‘species’’. Therefore, structural richness was the total number of height classes occupied by the various vegetative layers. Structural diversity used the Shannon–Wiener index with plant species represented by height classes and the amount (crown volume index) of vegetation in each class.

Diversity for small mammals was calculated by using the estimated abundance of each species for a given sampling session and averaged over the total sampling sessions for each summer and winter period. Log-series alpha was also calculated for the small mammal communities as this index showed good discriminating ability in a wide range of circumstances (Southwood, 1978). This index is less affected by species dominance than the Shannon–Wiener index (Magurran, 2004).

2.4. Statistical analyses

A one-way analysis of variance (ANOVA) (Zar, 1999) was used to determine the effect of habitat type on mean crown volume index of herbs, shrubs, and trees. Mean species richness and diversity and mean structural diversity of the herb, shrub, and tree layers and totals were also compared using this ANOVA model. A repeated-measures ANOVA (SPSS Institute Inc., 1997) was conducted to determine the effect of habitat type on mean abundance of each species, total abundance, mean species richness, and mean species diversity of the small mammal communities. Data not conforming to properties of normality and equal variance were subjected to various transformations to best approximate the assumptions required by an ANOVA (Zar, 1999). Mauchly’s *W*-test statistic was used to test for sphericity (independence of data among repeated measures; Littel, 1989; Kuehl, 1994). For datasets that were correlated among years, the Huynh–Feldt correction (Huynh and Feldt, 1976) was used to adjust the degrees of freedom of the within-subjects F-ratio.

A simple linear regression analysis was used to determine the relationship between structural diversity and abundance (crown volume index), species richness, and species diversity of the plant communities. This same analysis was used to investigate the relationship between total abundance and species diversity of small mammals to the habitat variables of abundance, species diversity, and structural diversity of vegetation. An ANOVA was used to test the prediction that there was a linear relationship (i.e. $\beta \neq 0$) between the independent and dependent variables in these regressions.

Duncan's multiple range test (DMRT) was used to compare mean values. In all analyses, the level of significance was at least $P = 0.05$.

3. Results

3.1. Vegetation

Mean crown volume index of herbs was statistically similar ($F_{6,14} = 2.38$; $P = 0.09$) among sites. However, it may be biologically significant that the abundance of herbs was 3.0 times higher in the old field (42.5 m^3/0.01 ha) than ponderosa pine forest (14.0 m^3/0.01 ha) (Fig. 1A). Similarly, the old field had 2.2–2.4 times as much herb volume as the two orchard sites and riparian sites. In terms of volume, the old field sites were dominated by crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.) and Kentucky bluegrass (*Poa pratensis* L.), and the sagebrush sites by bluebunch wheatgrass (*A. spicatum* (Pursh) Scribn. & Smith), downy brome (*Bromus tectorum* L.), and diffuse knapweed (*Centaurea diffusa* Lam.). Dominant herbs in the orchard sites were quackgrass (*A. repens* (L.) Beauv.) and annual bluegrass (*P. annua* L.). In the ponderosa pine sites, bluebunch wheatgrass, downy brome, and balsamroot (*Balsamorhiza sagittata* (Pursh) Nutt.) were the dominant herbs. The hedgerow habitats had variable amounts of dominant herbaceous species such as redtop (*Agrostis gigantea* Roth), asparagus (*Asparagus officinalis* L.), burdock (*Arctium lappa* L.), northern bedstraw (*Galium boreale* L.), and Canada thistle (*Cirsium arvense* (L.) Scop.). Riparian habitats also had variable amounts of dominant herbs including scouring rush (*Equisetum hyemale* L.), field mint (*Mentha arvensis* L.), beaked sedge (*Carex rostrata* Stokes), and skullcap (*Scutellaria galericulata* L.).

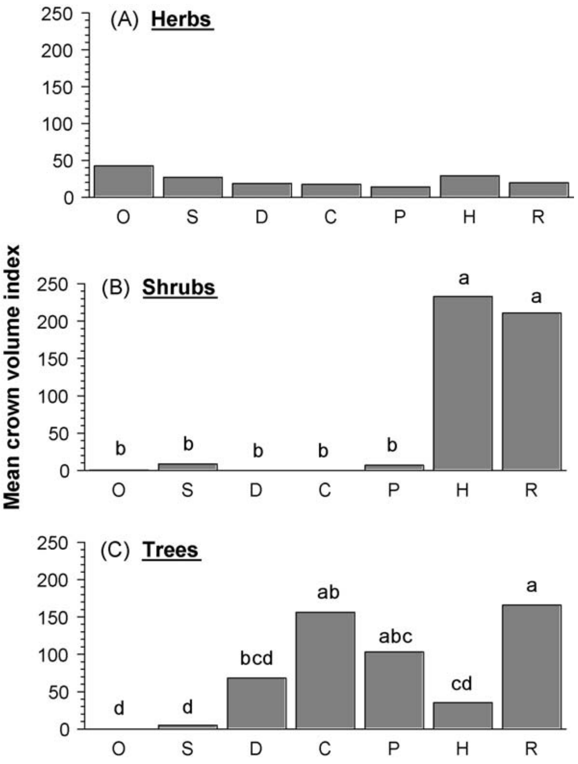

Fig. 1. Mean crown volume index (m^3/0.01 ha) of (A) herbs, (B) shrubs, and (C) trees in the seven habitats in 1999 at Summerland, British Columbia, Canada. Histograms of mean values with different letters are significantly different by Duncan's Multiple Range Test (DMRT). O = Old field; S = Sagebrush; D = Dwarf orchard; C = Conventional orchard; P = Ponderosa pine forest; H = Hedgerow; R = Riparian.

Mean crown volume index of shrubs was significantly ($F_{6,14} = 55.33$; $P < 0.01$) different among sites. The hedgerow and riparian habitats had substantial shrub layers (232.85 and 210.76 m^3/0.01 ha, respectively) compared with the other habitats (Fig. 1B). Prominent shrubs in the hedgerows included red-osier dogwood (*Cornus stolonifera* Michx.), Nootka rose (*Rosa nutkana* Presl), Virginia creeper (*Parthenocissus quinquefolia* L.), Douglas maple (*Acer glabrum* Torr.), and snowberry (*Symphoricarpus albus* (L.) Blake). Dominant shrubs in the riparian habitats were red-osier dogwood, Scouler's willow (*Salix scouleriana* Barratt), choke cherry (*Prunus virginiana* L.), woody nightshade (*Solanum dulcamara* L.), and snowberry. Shrub volumes were very low or non-existent in the old field and orchard sites, with big sagebrush (*Artemisia tridentata* Nutt.) dominating the sage habitat.

Mean crown volume index of trees was significantly ($F_{6,14} = 5.67$; $P < 0.01$) different among sites. The two orchard habitats and pine forest habitat had similar tree volumes, with the conventional orchard, pine forest, and riparian tree layers at the highest biomass levels (Fig. 1C). A tree layer was absent from the old field sites and of minor importance at 5.18 m^3/0.01 ha in the sage sites. As expected, apple trees were the dominant layer in both orchard sites, and ponderosa pine in the forest sites. Tower poplars (*Populus canascens* (Aiton) Sm.) dominated one hedgerow, with Sitka alder (*Alnus sinuata* (Regel.) Rydb.), paper birch (*Betula papyrifera* Marsh.), and quaking aspen (*Populus tremuloides* Michx.) as the dominant tree species in the riparian habitats.

3.2. Species and structural diversity

A total of 104 species of herbs, 26 species of shrubs, and 9 species of trees was sampled in this study. Of these plant species 37.3% (52 of 139) were introduced (exotic species), primarily among herbs where 44.2% (46 of 104) were not native species. Mean species richness of herbs was significantly ($F_{6,14} = 3.55$; $P = 0.02$) different among sites with the sage habitat having the most herbaceous species and the riparian the least (DMRT; $P = 0.05$) (Table 1).

Table 1
Summary of mean ($n = 3$ replicate sites) species richness, species diversity, and structural diversity for vascular plant communities in the seven habitat types and results of ANOVA

Parameter	Old field	Sage	Dwarf orchard	Conventional orchard	Pine forest	Hedgerow	Riparian	Analysis	
								$F_{6,14}$	P
Species richness									
Herbs	11.67^{bc} (1.86)	29.00^{a} (3.79)	18.00^{abc} (2.52)	18.67^{abc} (1.45)	22.00^{ab} (5.51)	12.33^{bc} (5.61)	8.67^{c} (2.85)	3.55	**0.02**
Shrubs	0.67^{c} (0.67)	3.00^{c} (0.00)	0.00^{c}	0.00^{c}	3.00^{c} (0.00)	7.00^{b} (2.08)	12.33^{a} (2.33)	14.04	**<0.01**
Trees	0.00^{c}	0.33^{bc} (0.33)	1.00^{b} (0.00)	1.00^{b} (0.00)	1.00^{b} (0.00)	0.67^{bc} (0.33)	4.00^{a} (0.58)	21.87	**<0.01**
Total	12.33 (1.20)	32.33 (3.76)	19.00 (2.52)	19.67 (1.45)	26.00 (5.51)	20.00 (4.73)	25.00 (5.69)	2.61	0.07
Species diversity									
Herbs	1.31 (0.17)	2.78 (0.34)	2.71 (0.26)	2.79 (0.28)	3.03 (0.39)	1.86 (0.91)	1.55 (0.51)	2.22	0.10
Shrubs	0.29^{bc} (0.29)	0.29^{bc} (0.04)	0.00^{c}	0.00^{c}	0.68^{b} (0.13)	1.98^{a} (0.37)	1.78^{a} (0.13)	19.13	**<0.01**
Trees	0.00^{b}	0.00^{b}	0.00^{b}	0.00^{b}	0.00^{b}	0.00^{b}	1.21^{a} (0.38)	10.29	**<0.01**
Total	1.37^{b} (0.20)	2.85^{a} (0.26)	1.48^{b} (0.36)	0.79^{b} (0.11)	1.24^{b} (0.33)	2.55^{a} (0.32)	2.67^{a} (0.27)	8.74	**<0.01**
Structural diversity									
Herbs	0.92 (0.19)	1.15 (0.03)	0.30 (0.16)	0.43 (0.09)	0.99 (0.26)	1.01 (0.33)	0.78 (0.39)	1.75	0.18
Shrubs	0.08^{c} (0.08)	1.27^{b} (0.12)	0.00^{c}	0.00^{c}	1.08^{b} (0.05)	1.57^{a} (0.04)	1.60^{a} (0.19)	62.34	**<0.01**
Trees	0.00	0.23 (0.23)	0.29 (0.20)	0.27 (0.06)	0.15 (0.13)	0.06 (0.06)	0.36 (0.09)	0.92	0.51
Total	0.92^{b} (0.19)	1.83^{a} (0.13)	1.07^{b} (0.07)	0.80^{b} (0.03)	0.90^{b} (0.31)	1.82^{a} (0.10)	1.35^{ab} (0.30)	5.28	**<0.01**

Parameters with P values in bold are considered to be significantly different. Within a parameter, mean values followed by different superscript letters (a, b, c) are significantly different according to Duncan's multiple range test. *Note*: Data entries are means with 1 S.E. in parentheses.

Mean values for species richness of shrubs and trees were also significantly ($F_{6,14} = 14.04$; $P < 0.01$ and $F_{6,14} = 21.87$; $P < 0.01$, respectively) different among sites. The riparian and hedgerow sites had the most shrub species, and the riparian site had the most tree species (DMRT; $P = 0.05$) (Table 1). Total mean species richness of plants was similar ($F_{6,14} = 2.61$; $P = 0.07$) among sites, but did range from a low of 12.33 species in the old field sites to 32.33 species in the sage sites.

Mean species diversity of herbs was similar among sites, but shrubs were significantly ($F_{6,14} = 19.13$; $P < 0.01$) different. Shrub species diversity was highest in the hedgerow and riparian sites, followed by the old field, sage, and pine forest habitats (DMRT; $P = 0.05$) (Table 1). Mean species diversity of trees was also significantly ($F_{6,14} = 10.29$; $P < 0.01$) different among sites, with the riparian habitat having the only recorded diversity of tree species (Table 1). Total species diversity was significantly ($F_{6,14} = 8.74$; $P < 0.01$) different among sites. This overall diversity measurement of plant species was highest in the sage, hedgerow, and riparian sites (DMRT; $P = 0.05$). Mean structural diversity of herbs and trees was similar among sites, but shrubs were significantly ($F_{6,14} = 62.34$; $P < 0.01$) different. Structural diversity of shrubs was highest in the hedgerow and riparian sites, followed by the sage and pine forest (DMRT; $P = 0.05$) (Table 1). The old field and orchard sites had negligible or no shrub layers. Total structural diversity was significantly ($F_{6,14} = 5.28$; $P < 0.01$) different among sites, with the sage, hedgerow, and riparian habitats dominating this measure of plant community structure (DMRT; $P = 0.05$).

Groups of plant species unique to each habitat type ranged from a low of three species in the old field and conventional orchard sites to 27 species in the riparian sites. Those habitats composed of, or associated with, agriculture: orchards, old fields, and hedgerows; had relatively similar numbers of native and introduced plant species (total ratio 11:18) in their unique groups. The natural habitats: sage, ponderosa pine forest, and riparian; had greater numbers of native than introduced species (total ratio of native to introduced species 36:8).

3.3. Small mammal abundance

The total numbers of individuals captured of all small mammal species are listed in Table 2, with a total of 5012 animals sampled on our sites. Deer mice accounted for 49.5% of all individuals captured, followed by montane voles 23.2%, Great Basin pocket mice 13.0%, western harvest mice 6.4%, house mice 4.8%, northwestern chipmunks 3.0%, and long-tailed voles 0.02%.

Mean abundance of deer mice was significantly ($F_{6,14} = 5.40$; $P < 0.01$) different among sites, with highest overall numbers (range of 11.6–22.9 ha^{-1}) in the riparian, sage, and old field habitats. The other four habitats had relatively similar mean numbers. Mean abundance of Great Basin pocket mice was significantly ($F_{6,14} = 15.82$; $P < 0.01$) different among sites, with highest overall numbers (19.0 ha^{-1}) consistently occurring in the sage habitat (DMRT; $P = 0.05$). *P. parvus* was not captured in the orchard, hedgerow, or riparian habitats.

Mean abundance of montane voles was significantly ($F_{6,14} = 9.74$; $P < 0.01$) different among sites, with the old field habitat consistently having the highest overall numbers (17.4 ha^{-1}) of this microtine. The hedgerow and riparian habitats also had voles, but at significantly lower abundance.

Table 2
Total numbers of individuals captured of each small mammal species in the three replicate sites of each habitat during the 4-year study at Summerland, British Columbia, Canada

Species	Old field	Sage	Dwarf orchard	Conventional orchard	Pine forest	Hedgerow	Riparian	Total
Peromyscus maniculatus	496	536	367	250	285	223	481	2638
Microtus montanus	780	63	24	34	40	129	94	1164
Perognathus parvus	139	256	0	0	103	0	0	498
Reithrodontomys megalotis	189	66	1	50	5	3	7	321
Tamias amoenus	2	46	0	0	100	1	0	149
Mus musculus	0	1	11	22	0	122	85	241
Microtus longicaudus	0	0	0	1	0	0	0	1
Total	1606	968	403	357	533	478	667	5012

The western harvest mouse also exhibited significantly ($F_{6,14} = 5.41$; $P < 0.01$) different mean numbers among sites, where it occurred primarily in the old field habitat at an overall mean abundance of 3.7 animals/ha.

Mean abundance of the northwestern chipmunk was significantly ($F_{6,14} = 39.70$; $P < 0.01$) different among sites, with this sciurid appearing predominantly in the ponderosa pine forest at an overall mean density of 6.5 animals/ha. Captures of chipmunks in the other habitats, except the sage at an overall mean density of 1.7 ha^{-1}, were negligible. The house mouse occurred at low numbers ($<5\ \text{ha}^{-1}$) with the hedgerow habitat clearly preferred ($F_{6,14} = 3.17$; $P = 0.04$). Mean abundance of the wandering shrew was significantly ($F_{6,14} = 93.56$; $P < 0.01$) different among sites, with highest overall numbers (1.7 ha^{-1}) in the riparian habitat.

Mean total abundance of small mammals per ha was significantly ($F_{6,14} = 13.44$; $P < 0.01$) different among sites with the old field, sage, and riparian habitats supporting the most (28.1–37.0) small mammals in terms of overall numbers (Fig. 2A). The ponderosa pine forest and hedgerow habitats had similar overall numbers (15.1–16.7) followed by the two orchard habitats (6.2–7.6).

3.4. Mammal diversity and habitat relationships

Mean species richness of the small mammal communities was significantly ($F_{6,14} = 18.45$; $P < 0.01$) different among sites, with highest richness in the old field and sage habitats (DMRT; $P = 0.05$). The orchard habitats had the lowest species richness according to overall mean values with non-overlapping 95% confidence intervals (Fig. 2B).

Mean species diversity (Shannon–Wiener, H') was significantly ($F_{6,14} = 23.78$; $P < 0.01$) different among sites, with the old field, sage, and pine forest habitats again having the highest diversity measurements. Diversity of small mammals in the orchard habitats also followed the pattern of species richness, with the lowest values during most years and overall. Log-series alpha was also significantly ($F_{6,14} = 4.38$; $P = 0.01$) different among sites with fewer dramatic differences among habitats. All non-orchard habitats had similar levels of species diversity (DMRT; $P = 0.05$), based on this measurement, which was less affected by species dominance.

Linear regression analysis indicated that total structural diversity of vegetation was positively related to total species richness ($r = 0.55$; $F_{1,19} = 8.30$; $P = 0.01$) and total species diversity ($r = 0.79$; $F_{1,19} = 30.86$; $P < 0.01$) of vegetation (Fig. 3A and B). There was no relationship between structural diversity and crown volume index of vegetation ($r = 0.18$; $F_{1,19} = 0.62$; $P = 0.44$).

Mean total abundance of small mammals was significantly ($r = 0.44$; $F_{1,19} = 4.45$; $P = 0.05$) related to total species

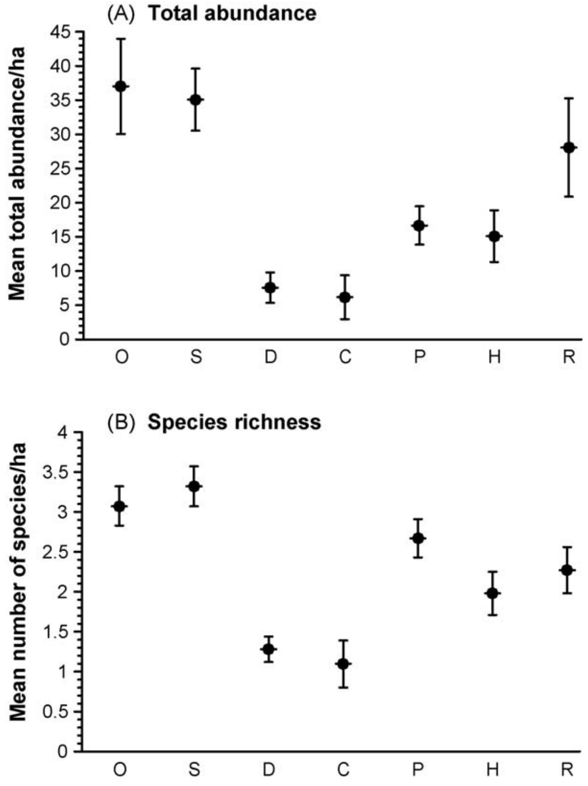

Fig. 2. Overall mean ($n = 18$; 3 sites × 6 seasons) ±95% confidence intervals for (A) total abundance and (B) species richness for the small mammal communities in the seven habitats at Summerland, British Columbia, Canada. O = Old field; S = Sagebrush; D = Dwarf orchard; C = Conventional orchard; P = Ponderosa pine forest; H = Hedgerow; R = Riparian.

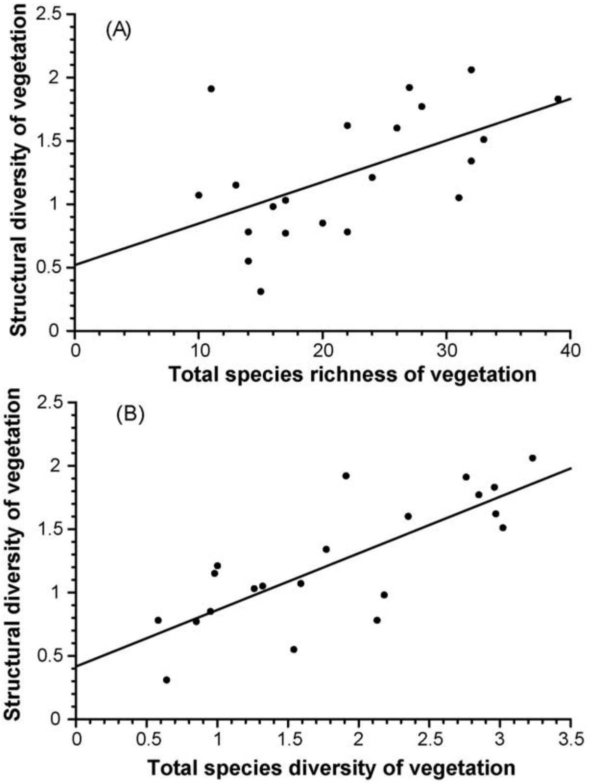

Fig. 3. Linear regression analysis relating mean total structural diversity of vegetation to (A) total species richness ($y = 0.52 + 0.03x$) and (B) total species diversity ($y = 0.42 + 0.45x$) of vegetation.

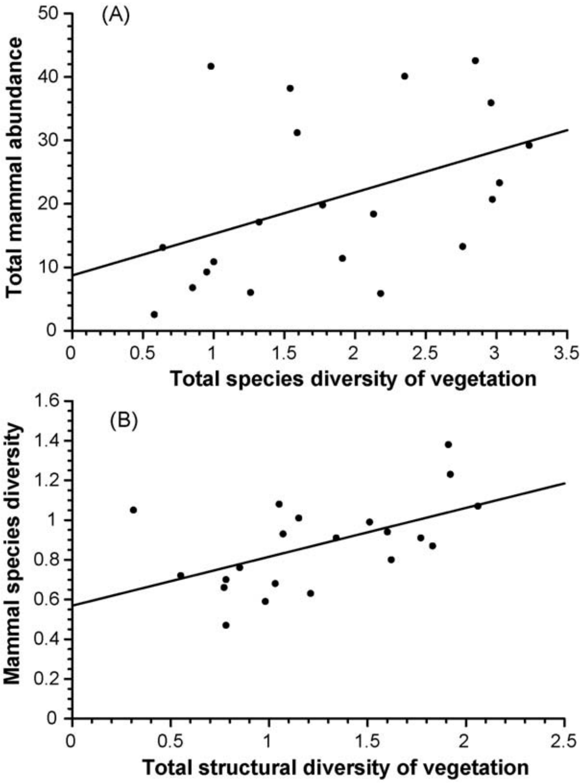

Fig. 4. Linear regression analysis relating (A) mean total mammal abundance to total species diversity of vegetation ($y = 8.71 + 6.55x$) and (B) mean mammal diversity (log-series) to total structural diversity of vegetation ($y = 0.57 + 0.25x$).

diversity (Fig. 4A), but not to volume ($r = 0.20$; $F_{1,19} = 0.82$; $P = 0.38$), total species richness ($r = 0.20$; $F_{1,19} = 0.76$; $P = 0.39$), or total structural diversity ($r = 0.34$; $F_{1,19} = 2.41$; $P = 0.14$) of vegetation. Mean species diversity (log-series) of small mammals was significantly ($r = 0.54$; $F_{1,19} = 8.00$; $P = 0.01$) related to total structural diversity of vegetation (Fig. 4B). This measure of mammal diversity was not related to total species richness ($r = 0.14$; $F_{1,19} = 0.36$; $P = 0.56$), total species diversity ($r = 0.38$; $F_{1,19} = 3.19$; $P = 0.09$), or volume ($r = 0.20$; $F_{1,19} = 0.82$; $P = 0.38$) of vegetation. Species richness of small mammals was not related to any of the vegetation parameters.

4. Discussion

The high structural diversity values in the sage, hedgerow, and riparian habitats and their positive linear relationship to species richness and species diversity, suggested that these sites had the greatest potential for conservation of native plant species. Despite the relatively high overall proportion (37.3%) of introduced plant species, the sage, ponderosa pine, hedgerow, and riparian habitats contributed 43 habitat-unique native species to the overall flora. This pattern of non-crop habitats, particularly hedgerow, riparian, and those of other linear configurations contributed the most to conservation of plant diversity in farming landscapes in the United Kingdom (Bunce and Hallam, 1993) and in Ontario and Quebec, Canada (Freemark et al., 2001; Boutin et al., 2002, 2003).

The contribution to the overall flora from the orchard habitats and those of agricultural origin: the old fields and hedgerows, was 29 unique species (11 native and 18 introduced) which was 39.7% (29 of 73) of the plant species unique to specific habitats. In addition, the similarity in mean total species richness among habitats was perhaps surprising with the dwarf and conventional orchards having 19.0 and 19.7 plant species, respectively, despite regular mowing of alleys and application of herbicide to tree rows three to four times each summer. This pattern was further supported by the general maintenance of herb species diversity in orchard sites despite the regular vegetation management regime (Sullivan and Sullivan, 2003). Although not likely to occur in intensively managed agro-ecosystems of annual crops, where non-crop vegetation is repeatedly reduced, the presence of these herbaceous plant species in the apple orchards contributed to the overall vegetative diversity in this landscape.

The abundance and diversity of small mammal communities were highest in those habitats where species and structural diversity of vegetation were highest, thereby providing a range of microhabitats and supporting our hypothesis. The major generalist species, the deer mouse, occurred in all habitats which was typical of this wide-ranging mammal (Baker, 1968). Deer mice have catholic feeding habits including seeds, berries, and invertebrates (Banfield, 1974). The northwestern chipmunk has also been described as a generalist species occupying a wide range of shrub-dominated habitats and an omnivorous diet (Sutton, 1992). However, in this study area, northwestern chipmunks occurred primarily in ponderosa pine forest and were rarely captured in other habitats except for the sagebrush sites.

The other members of our small mammal communities were specialists requiring certain habitat components. The montane vole was most common in old fields that provided perennial grassland habitat with both cover and food sources, such as grasses and forbs (Sullivan et al., 2003). This species occurs in native bunchgrass and sagebrush communities in the valley bottoms of the northern part of its range (Pearson et al., 2001). Montane voles occurred at low numbers in all other habitats in our study, but some grass-herb vegetation was required to sustain them. The house mouse is usually associated with farmland crops and stored grains where it can be a pest (Banfield, 1974). Thus, it was not surprising to find this commensal rodent in hedgerows and in lower numbers in the orchard and riparian habitats. The wandering shrew occurred at low numbers in the riparian sites where invertebrate food sources were likely most common in our semi-arid landscape (Nagorsen, 1996).

Some wildlife species such as the montane vole may feed on fruit trees during winter periods and cause considerable economic damage (Byers, 1984; Sullivan and Hogue, 1987). The lack of montane voles in the orchard sites was likely related to the vegetation management regime and contrasted markedly with patterns of vole abundance recorded in apple orchards during the 1980s at the nearby Research Centre (Sullivan et al., 2003).

The use of structural diversity of vegetation, which includes the species composition and layers of herbs, shrubs, and trees, has been promoted as an indication of biodiversity in both managed and natural forests (Sullivan et al., 2001; Lindenmayer and Franklin, 2002). This attribute of vegetation, as well as other structural features of an ecosystem, could be used as surrogates for biodiversity as well as act as indicators to monitor the success or failure of management practices designed to conserve biodiversity.

Acknowledgements

We thank the Pacific Agri-Food Research Centre, Agriculture and Agri-Food Canada, Monsanto Company, and the Applied Mammal Research Institute for funding support and provision of study areas; and commercial orchardists/farmers N. and L. Carter, J. and B. Geres, T. Gogol, D. Karlstrom, D. and D. Powell, and J. Young for their excellent cooperation and assistance. Thanks to H. Sullivan for able assistance with the fieldwork.

References

Altieri, M.A., Nicholls, C.I., 1999. Biodiversity, ecosystem function, and insect pest management in agricultural systems. In: Collins, W.W., Qualset, C.O. (Eds.), Biodiversity in Agroecosystems. CRC Press, Boca Raton, FL, USA, pp. 69–84.

Baker, C.H., 1968. Habitats and distribution. In: King, J.A. (Ed.), Biology of *Peromyscus* (Rodentia). Spec. Publ. 2. American Society of Mammalogists, Boston, MA, USA, pp. 125–167.

Banfield, A.W.F., 1974. The Mammals of Canada. University of Toronto Press, Toronto, Ont., Canada.

Best, L.B., 1983. Bird use of fencerows: implications of contemporary fencerow management practices. Wildl. Soc. Bull. 11, 333–347.

Boutin, C., Jobin, B., 1998. Intensity of agricultural practices and effects on adjacent habitats. Ecol. Appl. 8, 544–557.

Boutin, C., Jobin, B., Bélanger, L., Choinière, L., 2002. Plant diversity in three types of hedgerows adjacent to cropfields. Biodiv. Conserv. 11, 1–25.

Boutin, C., Jobin, B., Bélanger, L., 2003. Importance of riparian habitats to flora conservation in farming landscapes of southern Québec, Canada. Agric. Ecosyst. Environ. 94, 73–87.

Bunce, R.G.H., Hallam, C.J., 1993. The ecological significance of linear features in agricultural landscapes in Britain. In: Bunce, R.G.H., Ryszkowski, L., Paoletti, M.G. (Eds.), Landscape Ecology and Agroecosystems. Lewis Publishers, Boca Raton, FL, USA, pp. 11–19.

Byers, R.E., 1984. Control and management of vertebrate pests in deciduous orchards of the eastern United States. Hortic. Rev. 6, 253–285.

Fitzgibbon, C.D., 1997. Small mammals in farm woodlands: the effects of habitat, isolation and surrounding land-use patterns. J. Appl. Ecol. 34, 530–539.

Freemark, K.E., Bert, D., Villard, M.-A., 2001. Patch-, landscape- and regional-scale effects on biota. In: Gutwiller, K. (Ed.), Applying Landscape Ecology in Biological Conservation. Springer-Verlag, New York, NY, USA, pp. 53–58.

Freemark, K.E., Boutin, C., Keddy, C.J., 2002. Importance of farmland habitats for conservation of plant species. Conserv. Biol. 16, 399–412.

Hitchcock, C.L., Cronquist, A., 1973. Flora of the Pacific Northwest. University of Washington Press, Seattle, WA, USA.

Huynh, H., Feldt, L.S., 1976. Estimation of the box correlation for degrees of freedom from sample data in the randomized block and plot designs. J. Educ. Stat. 1, 69–82.

Krebs, C.J., 1966. Demographic changes in fluctuating populations of *Microtus californicus*. Ecol. Monogr. 36, 239–273.

Krebs, C.J., 1999. Ecological Methodology, 2nd edn. Addison-Welsey Longman, Menlo Park, CA.

Krebs, C.J., Gilbert, B.S., Boutin, S., Sinclair, A.R.E., Smith, J.N.M., 1986. Population biology of snowshoe hares. I. Demography of food-supplemented populations in the southern Yukon, 1976–1984. J. Anim. Ecol. 55, 963–982.

Krebs, C.J., Keller, B.L., Tamarin, R.H., 1969. *Microtus* population biology: demographic changes in fluctuating populations of *Microtus ochrogaster* and *M. pennsylvanicus* in southern Indiana. Ecology 50, 587–607.

Kuehl, R.C., 1994. Statistical Principles of Research Design and Analysis. Duxbury Press, Belmont, CA, USA.

Lindenmayer, D.B., Franklin, J.F., 2002. Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach. Island Press, WA, USA.

Littel, R.C., 1989. Statistical analysis of experiments with repeated measures. Hortscience 24, 36–40.

Magurran, A.E., 2004. Measuring Biological Diversity. Blackwell Publishing, Oxford, UK.

Martin, S.K., 1994. Feeding ecology of American martens and fishers. In: Buskirk, S.W., Harestad, A.S., Raphael, M.G., Powell, R.A. (Eds.), Martens, Sables, and Fishers. Biology and Conservation. Comstock Publishing Association. Cornell University Press, Ithaca and London, pp. 297–315.

McLaughlin, A., Mineau, P., 1995. The impact of agricultural practices on biodiversity. Agric. Ecosyst. Environ. 55, 201–212.

Nagorsen, D.W., 1996. Opossums, Shrews and Moles of British Columbia. The Mammals of British Columbia, vol. 2. UBC Press, Vancouver, BC, Canada.

Paoletti, M.G., Pimentel, D., Stinner, B.R., Stinner, D., 1992. Agroecosystem biodiversity: matching production and conservation biology. Agric. Ecosyst. Environ. 40, 3–23.

Parish, R., Coupé, R., Lloyd, D., 1996. Plants of Southern Interior British Columbia. Lone Pine Publishing, Vancouver, BC, Canada.

Pearson, D.E., Ortega, Y.K., McKelvey, K.S., Ruggiero, L.F., 2001. Small mammal communities and habitat selection in northern Rocky Mountain bunchgrass: implications for exotic plant invasions. Northwest Sci. 75, 107–117.

Pimentel, D., Stachow, U., Takacs, D.A., Brubaker, H.W., Duman, A.R., Meaney, J.J., O'Neil, J., Onsi, D.E., Corzilius, D.B., 1992. Conserving biological diversity in agricultural/forestry systems. Bioscience 42, 354–362.

Seber, G.A.F., 1982. The Estimation of Animal Abundance and Related Parameters, 2nd edn. Charles Griffin, London, United Kingdom.

Southwood, T.R.T., 1978. Ecological Methods. Chapman & Hall, London, UK.

SPSS Institute Inc., 1997. Statistical Programs for the Social Sciences. Chicago, IL, USA.

Stickney, P.F., 1980. Data base for post-fire succession, first 6 to 9 years, in Montana larch-fir forests. General Technical Report INT-62. USDA Forest Service, Intermountain Research Station.

Sullivan, T.P., Hogue, E.J., 1987. Influence of orchard floor management on vole and pocket gopher populations and damage in apple orchards. J. Am. Soc. Hortic. Sci. 112, 972–977.

Sullivan, T.P., Sullivan, D.S., 2003. Vegetation management and ecosystem disturbance: impact of glyphosate herbicide on plant and animal diversity in terrestrial systems. Environ. Rev. 11, 37–59.

Sullivan, T.P., Sullivan, D.S., Lindgren, P.M.F., 2001. Stand structure and small mammals in young lodgepole pine forest: 10-year results after thinning. Ecol. Appl. 11, 1151–1173.

Sullivan, T.P., Sullivan, D.S., Hogue, E.J., 2003. Demography of montane voles in old field and orchard habitats in southern British Columbia. Northwest Sci. 77, 228–236.

Sutton, D.A., 1992. Tamias amoenus. In: Mammalian Species, American Society of Mammalogists, Provo, UT, USA (Number 390).

Tattersall, F.H., MacDonald, D.W., Hart, B.J., Johnson, P., Manley, W., Feber, R., 2002. Is habitat linearity important for small mammal communities on farmland? J. Appl. Ecol. 39, 643–652.

Yahner, R.H., 1983. Small mammals in farmstead shelterbelts: habitat correlates of seasonal abundance and community structure. J. Wildl. Manage. 47, 74–84.

Zar, J.H., 1999. Biostatistical Analysis. Prentice-Hall, Upper Saddle River, NJ, USA.