Research Article

# Decreased small mammals species diversity and increased population abundance along a gradient of agricultural intensification

Stefania Gentili[a,*], Maurizia Sigura[a], Laura Bonesi[b]

[a]Department of Agricultural and Environmental Sciences, University of Udine, Via delle Scienze 208, 33100 Udine, Italy
[b]Department of Earth Sciences, University of Cambridge, Downing Street, CB2 3EQ Cambridge, UK

## Abstract

Agricultural intensification has been widespread worldwide over the last decades and has lead to a loss of semi-natural habitats. These changes are likely to have affected both the composition and diversity of small mammal communities living in agricultural landscapes. In this context, we compared population abundance (expressed as an index of relative abundance), species richness (S), and species diversity (expressed as Shannon and Pielou indexes) of small mammal assemblages (i.e. sub-sets of the entire small mammal community) living in three areas in North-East Italy positioned along a gradient of agricultural land-use intensification (measured with the Landscape Conservation Index).

We expected that assemblages would be less diverse and dominated by generalist species where the landscape was more intensely cultivated and where semi-natural habitats were less common. In the three areas, from a total of 4630 trap-nights, 668 individuals were captured with Sherman traps, belonging to four species (*Apodemus agrarius*, *A. flavicollis*, *A. sylvaticus* and *Microtus arvalis*). The results showed that population abundance, type of species present (but not species richness), and species diversity were affected by agricultural intensification and landscape naturalness. In particular, moving from less natural to more natural landscapes, we observed no effect on richness of species but increasing diversity due to a greater abundance of the more specialist species, such as *A. agrarius* and, partially, *A. flavicollis*, the latter only present in the most natural area. Generalist species, namely *A. sylvaticus*, and those associated with disturbed environments, such as *M. arvalis*, were instead more abundant in less natural landscapes. When considering population abundance, the highest overall abundance of small mammals was found in the most disturbed landscape.

The results were consistent with those of research carried out in other agroecosystems of Europe and highlight the controversial effect of the anthropogenic impact on small mammal assemblages, since a decrease in species diversity may be associated with an increased overall population abundance, due to the success of few generalist species.

## Introduction

In the past century, rural areas have undergone a process of intensification of agricultural practices that altered the structure of the landscape and the composition of plant and animal communities (Burel et al., 1998; Robinson and Sutherland, 2002) leading to a widespread decline in farmland biodiversity measured across many different taxa (Benton et al., 2003). These practices have resulted in modified ecosystems that are characterized by a high level of disturbance and homogeneity. In spite of these unfavorable conditions, often species are capable of significant levels of adaptation and find ways to survive (New, 1997; Vickery et al., 2002; Hoffmann et al., 2003; Jeanneret et al., 2003; Marshall et al., 2006; Dormann et al., 2007). Several species of small mammals are adapted to life in agro-ecosystems (Fitzgibbon, 1997; Ouin et al., 2000) and they often represent an important element of the ecosystem as they can significantly affect the diversity of higher trophic levels (Giraudoux et al., 1997; Moore et al., 2003). In this context, it is interesting to understand how agricultural intensification affects aspects of small mammal communities such as population abundance, species richness, and species diversity.

Studies that have analyzed the abundance of individuals, species richness, and species diversity of small mammal communities in agricultural landscapes found that they can be influenced by the gradient of structural complexity of the landscapes and by the different agricultural management techniques but the trends observed by these studies are by no means uniform. Some studies have found a tendency of greatest richness and diversity in the less intensely cultivated landscapes. For example, Silva et al. (2005) showed that the structural complexity of landscapes, as measured by coverage and shape of residual forest patches, was positively correlated with greater species richness, while species diversity was positively correlated with macrohabitat characteristics, such as the increasing gradient of different land-use cover. However, not all studies found that both richness and diversity were affected by agricultural intensification. Michel et al. (2006) demonstrated that, in areas where land-uses are relatively homogeneous, species diversity but not species richness, was negatively affected by the increasing level of agricultural land-use intensification. Millán de la Peña et al. (2003) found that agricultural intensification did not affect species richness, but that it had an effect on species diversity instead, with rare and habitat-specialist species being negatively influenced and habitat-generalist species being favored. Moreover, the kind of farming practices can also mediate the effect of structural landscape complexity on small-mammal communities. In a study that compared conventional vs. organic farming practices, Fischer et al. (2011) found that in conventional farming complex landscapes, characterized by a low percentage of arable lands, significantly increased small mammal abundance, species richness and diversity, whereas in organic farming small-mammal abundance was found to be highest in simple land-

*Corresponding author
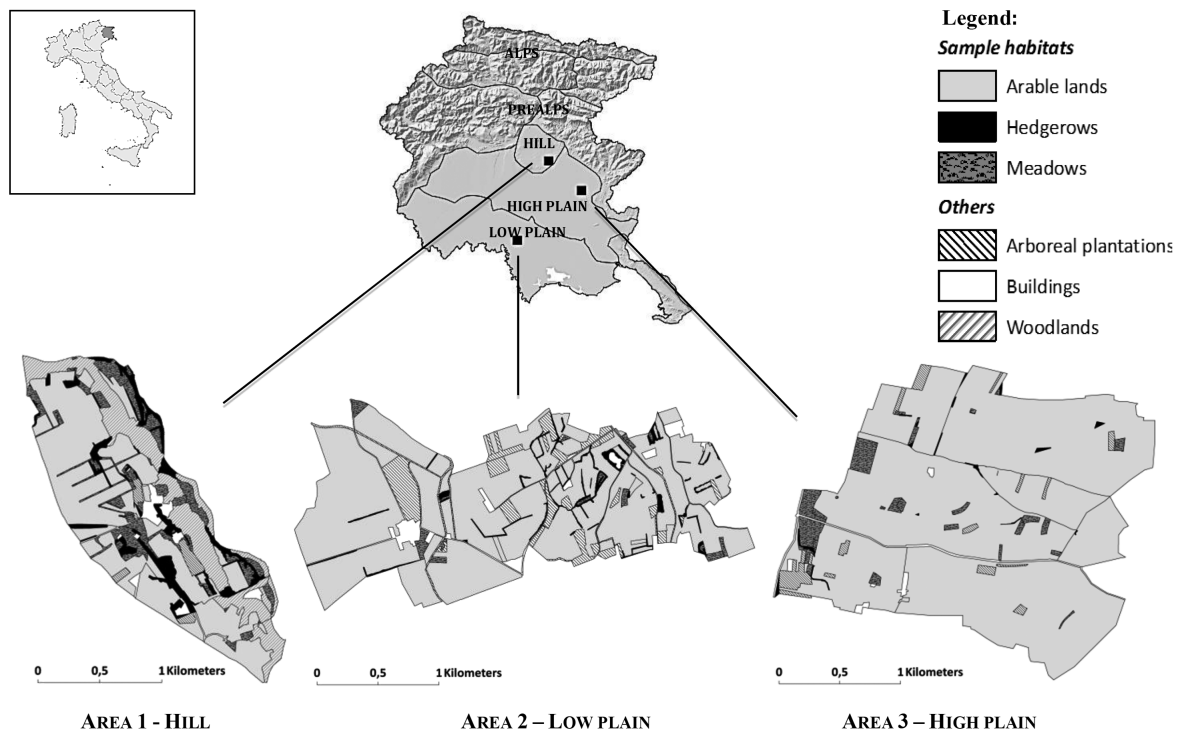Email address: ste.g@virgilio.it (Stefania Gentili)

**Figure 1** – Localization of the three study areas along the geographical zones of the Friuli Venezia Giulia region. Land-use cover maps of the three study areas (AREA 1, AREA 2, AREA 3). In the legend, the sample habitats (arable lands, hedgerows, meadows) and other habitat typologies (arboreal plantations, buildings, woodlands) are shown.

scapes. A greater abundance of small mammals in the most intensely cultivated, i.e. simplest, landscape has also been observed by Michel et al. (2006) when assessing small mammals in hedges.

The purpose of this study was to compare small mammal assemblages in three areas along a gradient of increasing agricultural intensification and decreasing naturalness. We compared overall (including all species) and relative (comparing different species) population abundances, species richness, and species diversity amongst the study areas, expecting that small mammal assemblages would be less diverse and dominated by generalist species where the landscape was more intensely cultivated and where semi-natural habitats were less common.

The study took place in North Eastern (NE) Italy where agricultural landscapes, hosting a high diversity of small mammals, are widespread (Lapini et al., 1995). In NE Italy, the alpine and continental bio-geographical regions meet allowing alpine species of small mammals, such as *Sorex alpinus* Schinz, 1837, to coexist with continental ones such as *Suncus etruscus* Savi, 1822 (Lapini et al., 1995). The high diversity of small mammal species and of habitats makes this area particularly suitable for studies of the composition of small mammal communities in relation to the landscape.

## Methods

### Study area

In NE Italy, small mammal communities of rodents in agricultural landscapes are dominated, in terms of abundance, by *Microtus arvalis* Pallas, 1779, and by a generalist species, namely *Apodemus sylvaticus* Linneus, 1758. Other rodents, such as *A. flavicollis* Melchior, 1834, and *A. agrarius* Pallas, 1771, are also common in the area and well adapted to live in heterogeneous agricultural landscapes (Lapini et al., 1995).

Three study areas were chosen having a decreasing level of agricultural intensification due to different agricultural land-uses (Figure 1). All areas were dominated by conventional rather than organic farming.

These areas were located in the region Friuli Venezia Giulia (north east Italy, 46°4′0″N, 13°14′0″E) and they were all below 180 m a.s.l.. The region is characterized by an annual rainfall of 800-1000 mm and by an average annual temperature of 13° C.

The first area (Area 1 – 250 ha) was located in the hills and was characterized by an heterogeneous agricultural landscape in which crops covered 54% of the entire landscape (maize, soybean, vineyards, artificial woodlots), followed by riparian woodlands and other semi-natural habitats (20% - hedgerows, meadows, set asides). Small anthropogenic structures, such as rural villages and farms, were found within the area (4%).

The second area (Area 2 - 350 ha) was located in the lower part of the regional plain. It was characterized by intermediate levels of agricultural intensification, with crop fields covering 82% of the entire landscape, while the presence of tree crops (7% - poplar plantations and artificial woodlots) and other semi-natural habitats (6% - meadows, set-asides, hedges and some wet habitats) covered most of the rest. The remaining 4% was occupied by rural villages. This landscape was heterogeneous and fragmented.

The third area (Area 3 - 600 ha) was located in the high part of the regional plain and was characterized by an intensely cultivated landscape influenced, in some sections, by land consolidation works (i.e. operations carried out because of the introduction of irrigation). The dominant matrix was composed of arable land (maize, soybean, winter cereals) for a total of 91% of land cover, in which there were isolated patches of semi-natural habitats (hedges and meadows) covering 3% of the landscape. The remaining land-use was covered by artificial woodlots and rural villages (6%).

### Small mammals sampling strategy

Within each of the three study areas, sampling was stratified by habitat (Table 1). The following three habitat types were considered: hedgerows, meadows, and cornfields. In Area 1 and Area 3, each habitat type was replicated four times for a total of 12 sampling patches in each area distant from each other 400 m as a minimum. The distance corresponded to the capacity of instantaneous movements of the most abundant small mammal species (*A. sylvaticus*) (Szacki and Liro, 1991). In Area 2, each habitat type was replicated only twice due to logistic constraints (for a total of 6 sampling patches). We decided to carry out the lowest number of replicates in Area 2 as this area presents intermediate levels of agricultural intensification when compared with

**Table 1** – Graphic representation of the sampling strategy. Each circle corresponds to one sampling patch.

| Sampling strategy | AREA 1 | AREA 2 | AREA 3 |
|---|---|---|---|
| *Cornfields* | | | |
| *Hedgerows* | | | |
| *Meadows* | | | |
| Average patch size | 0.91 ha | 0.88 ha | 0.53 ha |
| % of study area covered by sampling patches | 4% | 1% | 1% |



Area 1 or 3 (see below) and we preferred to maximize sampling in those areas that were more distant along the landscape gradient.

As a trapping method, we used Sherman traps (LFA type: cm 8 x 9 x 24. H.B. Sherman Traps©). This method of capture tends to be selective because it traps mostly small mammals weighing at least 5 grams (Umetsu et al., 2006; Caceres et al., 2011) therefore precluding the capture of small insectivorous mammals. Within each of the 30 sampling patches, 10 Sherman traps, spaced 10 m from each other, were placed along 100 m long line-transects. The traps were baited with a mixture of wheat, oats, sunflower seeds, a piece of carrot and hay, and activated for four consecutive nights. For all areas, sampling sessions were repeated in four consecutive seasons: summer 2011, autumn 2011, winter 2011/12, and spring 2012. Traps were checked once a day in all seasons except for summer when they were checked twice a day to prevent animals from dying because of the heat. Animals trapped were marked individually with ear tags (National Band & Tag Company®), weighed, aged, and assessed for reproductive status, and then released at the point of capture.

## Quantifying naturalness

Landscape naturalness was quantified by means of the Landscape Conservation Index (LCI) proposed by Pizzolotto and Brandmayr (1996), based on the calculation of the relative coverage area of land-uses classified within four different naturalness classes (arable lands, arboreal plantations, grasslands, natural lands). Land-uses were evaluated and classified based on their naturalness (most natural = less intensely managed) and ranked from the most artificial to the most natural (Batzella et al., 2012; Ferrari et al., 2008) as follows: 1) *Arable lands*: maize, soybean, winter cereals; 2) *Arboreal plantations*: artificial woodlots, vineyards; 3) *Grasslands*: meadows, set asides; 4) *Natural lands*: hedges, groves. Land-uses were interpreted from aerial images of the area (year 2006, resolution 0.5 x 0.5 m, National Geoportal - Friuli Venezia Giulia region). Based on the aerial images, land-use maps were then created by GIS (ArcGIS, ESRI© software) at the scale 1:3000, using the Corine Land Cover categories (IV level) to classify them. Each land-use, within each of the three study areas, was quantified in terms of coverage (ha). The Landscape Conservation Index was then calculated as follows (Ferrari et al., 2008): if $x_i$ is the relative area mapped for each naturalness class (expressed as a proportion or percentage) and $c_i$ is the cumulative relative value of the $i^{th}$ class, the sum of these cumulative values is

$$A = \sum_{i=1}^{n} c_i \qquad (1)$$

where $n$ is the number of naturalness classes (in our case four). To obtain cumulative relative areas, the classes were ordered according to their naturalness from the most artificial ($i = 1$) to the most natural ($i = n$) as described above. In this way, classes with a higher degree of naturalness have greater weight. In fact, if $x_1, ..., x_n$ are the relative areas occupied by the classes, the cumulative values are:

$$c_1 = x_1, \quad c_2 = x_1 + x_2, \quad ... , \quad c_n = x_1 + ... + x_n \qquad (2)$$

and

$$A = c_1 + c_2 + ... + c_n = x_1 + (x_1 + x_2) + ...(x_1 + ... + x_n) \qquad (3)$$

In this way the value $A$ can be assumed as a measurement of the degree of artificiality of the land-use classes (Ferrari et al., 2008).

In the case of a landscape consisting entirely of the most degraded category (only 1 artificial class) $A_{max}$ is calculated as

$$A_{max} = (n - 1) \qquad (4)$$

The Landscape Conservation Index ($LCI$) is calculated following the formula

$$LCI = 1 - \frac{A}{A_{max}} \qquad (5)$$

The index ranges between 0 and 1. Values close to 1 indicate landscapes characterized by high naturalness, while values close to zero indicate landscapes strongly influenced by human activities (i.e. agriculture).

## Characterizing small mammal assemblages

To characterize small mammal assemblages we used a series of measures that quantify population abundance, species richness, and species diversity. To quantify population abundance, we calculated an index of relative abundance ($T$) based on the number of individuals captured in relation to the number of nights the traps were open for (Cagnin et al., 1998; Ouin et al., 2000; Klaa et al., 2005), according to the following formula

$$T = 100 \frac{N_a}{n_a \times t_a} \qquad (6)$$

where $a$ is the area (Area 1, Area 2, Area 3), $N$ is the number of individuals of a particular species, $n$ is the number of nights a trap is open for, and $t$ is the number of active traps.

Species richness was quantified by counting the number of different species ($S$), while species diversity was quantified by means of two indexes, the Shannon index and the Pielou index. The Shannon index ($H'$; Magurran 1988) was calculated as

$$H' = -\sum_{i=1}^{s} p_i \times \ln p_i \qquad (7)$$

where $p_i$ is the ratio between the number of individuals of a particular species and the number of total individuals captured ($n_i/N$). The Pielou (or evenness) index ($J$; Magurran 1988) was calculated as

$$J = \frac{H'}{\ln S} \qquad (8)$$

where $H'$ is the number derived from the Shannon diversity index and $S$ is species richness.

These indexes were calculated for each of the 30 sampling patches and for all the trapping sessions, for a total of 120 cases and all the values were used for the analyses. The resulting values were checked for normality with the Anderson-Darling test (Minitab v.16® software). As conditions of normality were not satisfied, we applied non-parametric tests to examine the differences in assemblages' composition between the study areas. In particular, we used the non-parametric Kruskal-Wallis test. When the test resulted significant, showing a difference between the three study areas, we presented in the text only the results of the post-hoc Mann-Whitney test to highlight how the three study areas were different from each other. We accepted a result as significant when $p < 0.05$.

The three indexes ($T$, $H'$, $J$) were also calculated for each of the three study areas by pooling all the data for all patches and for all seasons and reported in a tabular format.

## Results

### Characterization of naturalness

The three study areas were all characterized by the prevalence of agricultural land-uses but, on the basis of the Landscape Conservation Index, they were well differentiated along a gradient of decreasing naturalness corresponding to agricultural intensification with Area 1 being the most natural and Area 3 the least natural (Table 2).

### Small mammal assemblages

Sherman traps were active for a total of 4630 trap-nights (calculated as the number of traps open by the number of sampling nights, excluding the number of non-active traps) and captured 668 individuals of four different species: *A. sylvaticus* (68% of total captures), *A. agrarius* (21%), *M. arvalis* (6%), and *A. flavicollis* (5%). In Area 1, we observed a significantly lower overall abundance of small mammals than in Area 3 (Mann-Whitney test: $T_{Area1}$ vs. $T_{Area3}$: W = 2054, $p = 0.03$), but greater than in Area 2 (Mann-Whitney test: $T_{Area1}$ vs. $T_{Area2}$: W = 1556, $p = 0.01$), whereas the difference in abundance between Area 3 and Area 2 was not significant (Mann-Whitney test: $T_{Area2}$ vs. $T_{Area3}$: W = 1652, $p = 0.23$). In this analysis, the number of cases was given by the number of patches x the number of seasons (sample sizes for the above tests: Area 1 = 48; Area 2 = 24; Area 3 = 48), hence the values of the index of relative abundance used for the Mann-Whitney tests are not directly comparable with those in Table 3, where abundances were calculated for each study areas by pooling all the data. Species richness was the same in all three areas (Table 3). Species diversity, calculated with the Shannon Index, and evenness, calculated with the Pielou Index, were not significantly different amongst the three areas when considering all the single patches as sample units (Kruskal-Wallis test - Shannon: H = 3.02, $p = 0.22$; Pielou: H = 2.01, $p = 0.36$), but they were when the patches were taken in groups of three (see Appendix S1).

### Naturalness and associated assemblages

To evaluate the association between rodent assemblages' composition in the three study areas and the gradient of agricultural land-use intensification, we plotted the relative abundance of the species captured in each area, against the Landscape Complexity Index (Figure 2).

We observed two different trends. *A. sylvaticus* tended to markedly decrease from the least natural to the most natural area while, on the contrary, *A. agrarius* tended to increase, albeit only slightly, from the least natural to the most natural area. *M. arvalis* tended to decrease between more and less natural areas, but this species was absent from the very most natural area, Area 1. *A. flavicollis* was absent in two of the three study areas, so no trend could be observed.

## Discussion

The results of our study showed that composition and diversity of small mammal assemblages in agricultural areas were significantly affected by the degree of naturalness and landscape complexity. In particular, species diversity decreased and the number of generalist species increased with increasing agricultural intensification, confirming our initial expectations.

We found that the method of capture that we have used, namely Sherman traps, has been useful for obtaining ecological information concerning presence/absence, spatial distribution, relative and overall abundance of species, providing a good spatial resolution with comparable low sampling effort, as demonstrated also by other studies (Flowerdew et al., 2004; Gurnell and Flowerdew, 1994).

### Small mammal richness and distribution of specialists/ generalists

The composition of the small mammal assemblages reflected what we expected in terms of the kind of species that we would find in relation to the level of naturalness. Indeed, moving from less natural to more natural landscapes, we observed, on one hand, a clear trend of increasing relative abundance of the specialist species associated with more natural habitats, such as *A. flavicollis* and *A. agrarius*, and, on the other hand, an opposite downward trend of relative abundance of the more generalist species (*A. sylvaticus*).

*A. flavicollis*, a species with specific requirements in terms of habitat preferences as it prefers heavily wooded areas with good tree cover and shrubs (Capizzi and Santini, 2007), was only found in Area 1. While *M. arvalis*, a species typical of open habitats such as meadows and set asides (Capizzi and Santini, 2007) was only found in Area 2 and Area 3. Some studies have shown that this species is associated with intensely cultivated landscapes and, for this reason, it can be considered a good indicator of agricultural intensification (Delattre et al., 1996; Burel et al., 2004). However, even if the species tends to be effectively present in highly cultivated environments, it still prefers fields and meadows arranged in heterogeneous patches (Heroldova et al., 2007; Millán de la Peña et al., 2003), a landscape that characterized Area 2 more than Area 3, where maize and other cultivated fields were abundant and organized in large and compact patches (Figure 1). Due to the relatively low levels of captures of *M. arvalis* we were not able to discriminate whether the species was more abundant in Area 2 or 3.

### Small mammal diversity

Regarding the diversity of small mammal assemblages in agricultural landscapes, we found that the Shannon and Pielou indexes varied in the three areas according to the level of naturalness of the area (Table 3 and Appendix S1). Area 1, the most natural area, presented the highest Shannon index, indicating high species diversity. The Pielou index was

**Table 2** – Landscape Conservation Index (*LCI*, see Methods Eq. 1, Eq. 4 and Eq. 5) (Pizzolotto and Brandmayr, 1996) to measure landscape naturalness. $L$ = land cover area as a percentage; $C_i$ = cumulative percentage; $A$ = sum of the cumulative relative areas of the naturalness classes; $A_{max}$ = ($n$ - 1), where n is the number of naturalness classes. It represents the maximum potential value of artificiality obtainable in the case of a landscape consisting entirely of the most degraded category, and it is the same for all the three study areas.

**Landscape Conservation Index**

| Classes | AREA 1 | | AREA 2 | | AREA 3 | |
|---|---|---|---|---|---|---|
| | $L$ (%) | $C_i$ | $L$ (%) | $C_i$ | $L$ (%) | $C_i$ |
| Arable lands | 54 | 0.54 | 88 | 0.88 | 96 | 0.96 |
| Arboreal plantations | 2 | 0.56 | 9 | 0.96 | 3 | 0.99 |
| Grasslands | 12 | 0.68 | 1 | 0.97 | 0.4 | 0.99 |
| Natural lands | 32 | 1 | 3 | 1 | 0.6 | 1 |
| | | | | | | |
| $A$ | | 0.17 | | 0.28 | | 0.29 |
| $A_{max}$ | | 3 | | 3 | | 3 |
| $LCI$ | | 0.41 | | 0.06 | | 0.02 |

**Table 3** – Trapping results for Sherman traps (data pooling for all patches and for all seasons). $N$ = number of individuals; $T$ = index of relative abundance; $Tot\ T$ = index of relative abundance considering all the individuals captured across all species (not the sum of $T$ of each species); $S$ = total number of species; $H'$ = Shannon index; $J$ = Pielou index.

**Trapping results for Sherman traps**

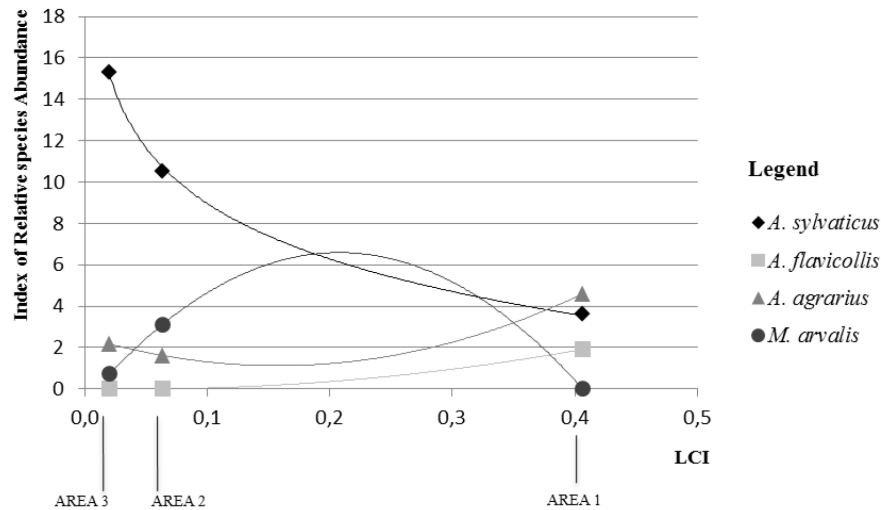| Species | AREA 1 | | AREA 2 | | AREA 3 | |
|---|---|---|---|---|---|---|
| | $N$ | $T$ | $N$ | $T$ | $N$ | $T$ |
| *A. sylvaticus* | 66 | 3.6 | 91 | 11.0 | 295 | 14.8 |
| *A. flavicollis* | 35 | 1.9 | 0 | 0 | 0 | 0 |
| *A. agrarius* | 84 | 4.6 | 14 | 1.7 | 42 | 2.3 |
| *M. arvalis* | 0 | 0 | 27 | 3.3 | 14 | 1.9 |
| | | | | | | |
| *Tot N / Tot T* | 185 | 10.2 | 132 | 19.4 | 351 | 18.0 |
| | | | | | | |
| **Indexes** | | | | | | |
| $S$ | | 3 | | 3 | | 3 |
| $H'$ | | 1.04 | | 0.82 | | 0.53 |
| $J$ | | 0.95 | | 0.76 | | 0.48 |

**Figure 2** – Landscape conservation index (*LCI*) in relation with the relative abundance of species captured in each study area with Sherman traps. Curves are fitted as polynomial (II order) to show trends.

also relatively high in Area 1 and 2 compared to Area 3, suggesting that the assemblages of the two most natural areas were relatively even, while that of the most man-modified area, Area 3, were dominated by fewer species which may be an indication of the impoverishment and homogenization of assemblages. Indeed, Area 3 was dominated by cultivated fields and while it exhibited the highest number of individuals captured, most of these individuals belonged to a single species, *A. sylvaticus*, the most generalist and adaptable species amongst the ones found in this study (as described for the rest of Italy by Capizzi and Filippucci 2008).

Area 2 was positioned in the middle between Area 1 and Area 3 in its values of species diversity, as expected if considering its intermediate level of naturalness (Table 3 and Appendix). The assemblage in Area 2 was dominated by *A. sylvaticus*, while *A. agrarius* was present with the lowest percentage of captures compared to the other areas. However, the distribution of captures was even, indicating a good level of assemblage diversity.

Our findings, for both species richness and species diversity, were consistent with other studies in Europe. In France in three different sites, Michel et al. (2006) found that Shannon and Pielou indexes for small mammal assemblages decreased as the intensification of agricultural practices increased. In agricultural areas of the Czech Republic, the species most widely spread in a mosaic of hedges, fields and small woods interspersed in a 91% matrix of cultivated land, was *A. sylvaticus*, followed by *M. arvalis*, with *A. flavicollis* abundant only in wooded areas (Heroldova et al., 2007). Fischer et al. (2011) in Southern Lower Saxony, found a trend in mean species richness similar to our study, with increasing richness in landscapes with percentage of arable land lower than 80%. In their study and similarly to our study, Fischer et al. (2011) found that *A. agrarius* was absent from most simplified landscapes; on the contrary, *M. arvalis* was absent from most complex and heterogeneous landscapes. The same results were found also by Kozakiewicz et al. (1999), according to which *A. agrarius* was positively influenced by the presence of extensive wooded patches.

**Naturalness and associated abundance**

It is well known that agricultural intensification tends to lead to a loss of diversity (Robinson and Sutherland, 2002) as revealed by the increase of common and generalist species such as *A. sylvaticus* observed also by Love et al. (2000), Kleijn and Sutherland (2003) and Millán de la Peña et al. (2003). Our findings are consistent with the general assumption that the diversity of assemblages is highest in landscapes with limited anthropogenic disturbance and that the diversity and composition of communities depends on landscape naturalness (Bates and

Harris, 2009; Michel et al., 2007). When the level of naturalness in a landscape is decreased by increasing the number and extent of human modified habitats, small mammal assemblages tend to become impoverished, less diverse and dominated by generalist species that are more able to adapt to human-induced alterations. However, we have found that the decrease of small mammal diversity that accompanies agricultural intensification is matched by a corresponding increase in the overall abundance of small mammals, due mainly to the higher population abundance of the generalist species. This trend is confirmed by Butet et al. (2006), who have also found that in highly intensified landscapes a crop matrix provides high levels of rodents' abundance but low diversity. In the same direction, Jennings and Pocock (2009), considering three aspects of agricultural intensification related both to the cultivation techniques (increased use of agrochemicals) and to the structural alteration of the landscape (change in grassland management and increased field size due to hedgerow loss), found that increased intensification leaded to changes in abundance of insectivorous small mammal species and their prey. The authors found that sites characterized by the highest level of intensification had a great influence on small mammal abundance, increasing number of species that can make use of changing resources (generalists), either by being highly mobile or by reproducing rapidly, and limiting the survival of less mobile and slow breeding species. In our study, we observed a much higher number of individuals in the less natural area, Area 3, compared to the most natural one, Area 1 (n = 351 in Area 3 and n = 185 in Area 1). This can probably be attributed to the level of resources found in Area 3 compared to Area 1. Due to the abundant presence of maize fields, Area 3 was characterized by a great but seasonal availability of seeds and grains encouraging seed-eater and generalist species, such as *A. sylvaticus*, to be particularly abundant (Schoener, 1983). Area 1 instead, was likely to provide a less abundant but more constant level of food resources throughout the year due to the presence of stable semi-natural patches. If we examine the seasonal abundances of small mammals in Area 3 compared to Area 1, we notice that there is a much greater variation between the winter and the summer abundances in Area 3 than in Area 1. In Area 3, summer abundance is five times greater than in winter, while in Area 1 summer abundance is only twice greater than in winter, suggesting a much stronger effect of the seasonal abundance of resources in Area 3. The pattern of small mammals boom-and-bust vs. a more stable pattern of abundances is likely to influence also the fitness and abundance of the predators (Love et al., 2000; Bosè and Guidali, 2001; Millán de la Peña et al., 2003).

## Conclusions

In many areas of Europe there is a strong trend toward an intensification of agriculture with a loss of naturalness (Henle et al., 2008). Agricultural intensity strongly influences ecosystem services and biodiversity (Stoate et al., 2001; Reidsma et al., 2006; Kleijn et al., 2009). Our study suggests that this process of land-use change may lead to a loss of diversity and complexity of the small mammal assemblages but not necessarily to a loss of overall abundances of individuals. Although we expected a high diversity of species in the small mammal assemblages of the studied region, due to the particular biogeographical characteristics of the area whereby the continental region meets the alpine one (Lapini et al., 1995), we found a relatively low diversity in terms of type of species, supporting the observation that assemblages inhabiting agro-ecosystems tend to be dominated by only a few species.

# References

Bates F.S., Harris S., 2009. Does hedgerows managment on organic farms benefit small mammal populations? Agr. Ecosyst. Environ. 129: 124–130.

Batzella M., Balvis T., Muntoni F., Marini A., 2012. Vegetation map and assessment of naturalness degree of the Ogliastra territory (Sardinia, Italy). Forest@ - Rivista di Selvicoltura ed Ecologia Forestale 9: 130–136.

Benton T.G., Vickery J.A., Wilson J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? Trends Ecol. Evol. 18: 182–188.

Bon M., Paolucci P., Mezzavilla F., De Battisti R., Vernier E., 1995. Atlante dei Mammiferi del Veneto. Lavori Soc. Ven. Sc. Nat., Suppl. vol. 21.

Bosè M., Guidali F., 2001. Seasonal and geographic differences in the diet of the barn owl in an agro-ecosystem in northern Italy. J. Raptor. Res. 35: 240–246.

Burel F., Baudry J., Butet A., Clergeau P., Delettre Y., Le Coeur D., Dubs F., Morvan N., Paillat G., Petit S., Thenail C., Brunel E., Lefeuvre J.-C., 1998. Comparative biodiversity along a gradient of agricultural landscapes. Acta Oecol. 19: 47–60.

Burel F., Butet A., Delettre Y.R., Millàn de la Pena N., 2004. Differential response of selected taxa to landscape context and agricultural intensification. Landsc. Urban Plan. 67: 195–204.

Butet A., Paillat G., Delettre Y., 2006. Factors driving small rodents assemblages from field boundaries in agricultural landscapes of Western France. Landsc. Ecol. 21: 449–461.

Caceres N.C., Nápoli R.P., Hannibal W., 2011. Differential trapping success for small mammals using pitfall and standard cage traps in a woodland savannah region of southwestern Brazil. Mammalia 75: 45–52.

Cagnin M., Aloise G., Garofalo G., Milazzo C., Cristaldi M., 1996. Les communautes de petit Mammifères de terre des trois "fiumare" de la Calabre (Italie du sud). Vie Milieu 46: 319–326.

Cagnin M., Moreno S., Aloise G., Garofalo G., Villafuerte R., Gaona P., Cristaldi M., 1998. Comparative study of Spanish and Italian terrestrial small mammals coenoses from different biotopes in Mediterranean peninsular tip regions. J. Biogeog. 25: 1105–1113.

Capizzi D., Filippucci M.G., 2008. *Apodemus sylvaticus* (Linnaeus, 1758). In: Amori G., Contoli L., Nappi A. (Eds.) Fauna d'Italia. Mammalia II. Erinaceomorpha, Soricomorpha, Lagomorpha, Rodentia. Calderini, Bologna, pp. 604–618.

Capizzi D., Santini L., 2007. I roditori italiani. Ecologia, impatto sulle attività umane e sugli ecosistemi. Delfino Editore.

Churchfield S., Hollier J., Brown V.K., 1997. Community structure and habitat use of small mammals in grasslands of different successional stage. J. Zool. 242(3): 519–530.

Delattre P., Giraudoux P., Baudry J., Quéré J.P., Fichet E., 1996. Effect of landscape structure on Common Vole (*Microtus arvalis*) distribution and abundance at several space scales. Landsc. Ecol. 11: 279–288.

Dormann C.F., Schweiger O., Augenstein I., Bailey D., Billeter R., De Blust G., DeFilippi R., Frenzel M., Hendrickx F., Herzog F., Klotz S., Liira J., Maelfait J.-P., Schmidt T., Speelmans M., Van Wingerden W.K.R.E., Zobel M., 2007. Effects of landscape structure and land-use intensity on similarity of plant and animal communities. Global Ecol. Biogeogr. 16: 774–787.

Ferrari C., Pezzi G., Diani L., Corazza M., 2008. Evaluating landscape quality with vegetation naturalness maps: an index and some inferences. Appl. Veg. Sci. 11: 243–250.

Fischer C., Thies C., Tscharntke T., 2011. Small mammals in agricultural landscapes: Opposing responses to farming practices and landscape complexity. Biol. Conserv. 144: 1130–1136.

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.

Flowerdew J.R., Shore R.F., Poulton S., Sparks T.H., 2004. Live trapping to monitor small mammals in Britain. Mammal Rev. 34: 31–50.

Giraudoux P., Delattre P., Habert M., Quéré J.P., Deblay S., Defaut R., Duhamel R., Moissenet M.F., Salvi D., Truchetet D., 1997. Population dynamics of fossorial water vole (*Arvicola terrestris scherman*): a land use and landscape perspective. Agr. Ecosyst. Environ. 66: 47–60.

Gurnell J., Flowerdew J.R., 1994. Live trapping small mammals: a practical guide. The Mammal Society, London.

Henle K., Alard D., Cobb P., Firbank L.G., Kull T., McCracken D., Moritz R.F.A., Niemela J., Rebane M., Wascher D., Watt A., Young J., 2008. Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe: a review. Agr. Ecosyst. Environ. 124: 60–71.

Heroldova M., Bryja J., Zejda J., Tkadlec E., 2007. Structure and diversity of small mammal communities in agriculture landscape. Agr. Ecosyst. Environ. 120: 206–210.

Hoffmann J., Greef J.M., Kiesel J., Lutze G., Wenkel K.O., 2003. Practical example of the mosaic indicators approach. Agr. Ecosyst. Environ. 98: 395–405.

Jeanneret P., Schüpbach B., Luka H., 2003. Quantifying the impact of landscape and habitat features on biodiversity in cultivated landscapes. Agr. Ecosyst. Environ. 98: 311–320.

Jennings N., Pocock M.J.O., 2009. Relationships between sensitivity to agricultural intensification and ecological traits of insectivorous mammals and arthropods. Conserv. Biol. 23: 1195–1203.

Klaa K., Mill P.J., Incoll L.D., 2005. Distribution of small mammals in a silvoarable agroforestry system in Northern England. Agroforest. syst. 63: 101–110.

Kleijn D., Kohler F., Báldi A., Batáry P., Concepcion E.D., Clough Y., Diaz M., Gabriel D., Holzschuh A., Knop E., 2009. On the relationship between farmland biodiversity and land-use intensity in Europe. Proc. R. Soc. B. 276: 903–909.

Kleijn D., Sutherland W.J., 2003. How effective are European agri-environment schemes in conserving and promoting biodiversity? J. Appl. Ecol. 40: 947–969.

Kozakiewicz M., Gortat T., Kozakiewicz A., Barkowska M., 1999. Effects of habitat fragmentation on four rodent species in a Polish farm landscape. Landsc. Ecol. 14: 391–400.

Lapini L., 2009. Micromammiferi della riserva naturale "Lago di Cornino" (Forgaria nel Friuli, Udine, Italia nord-orientale). Gortania, Botanica, Zoologia 31: 143–170.

Lapini L., Dall'Asta A., Dublo L., Spoto M., Vernier E., 1995. Materiali per una teriofauna dell'italia Nord-orientale. Gortania - Atti Mus. Friul. St. Nat. 17: 107–119.

Love R.A., Webbon C., Glue D.E., Harris S., 2000. Changes in the food of British Barn Owls (*Tyto alba*) between 1974 and 1997. Mammal Rev. 30: 107–129.

Maddock A.H., 1992. Comparison of two methods for trapping rodents and shrews. Israel J. Zool. 38: 333–340.

Magurran A., 1988. Ecological Diversity and its Measurement. Chapman & Hall, London.

Marshall E.J.P., West T.M., Kleijn D., 2006. Impacts of an agri-environment field margin prescription on the flora and fauna of arable farmland in different landscapes. Agr. Ecosyst. Environ. 113: 36–44.

Michel N., Burel F., Butet A., 2006. How does landscape use influence small mammal diversity, abundance and biomass in a hedgerow networks of farming landscapes? Acta Oecol. 30: 11–20.

Michel N., Burel F., Legendre P., Butet A., 2007. Role of habitat and landscape in structuring small mammal assemblages in hedgerow networks of contrasted farming landscapes in Brittany, France. Landsc. Ecol. 22: 1241–1253.

Millán de la Peña N., Butet A., Delettre Y., Paillat G., Morant P., Le Du L., Burel F., 2003. Response of the small mammal community to changes in western French agricultural landscapes. Landsc. Ecol. 18: 265–278.

Moore N.P., Askew N., Bishop J.D., 2003. Small mammals in new farm woodlands. Mammal Rev. 33: 101–104.

Mortelliti A., Boitani L., 2009. Distribution and coexistence of shrews in patchy landscapes: A field test of multiple hypotheses. Acta Oecol. 35: 797–804.

New T.R., 1997. Are Lepidoptera an effective "umbrella group" for biodiversity conservation? J. Insect Conserv. 1: 5–12.

Ouin A., Paillat G., Butet A., Burel F., 2000. Spatial dynamics of wood mouse (*Apodemus sylvaticus*) in an agricultural landscape under intensive use in the Mont Saint Michel Bay (France). Agr. Ecosyst. Environ. 78: 159–165.

Pankakoski E., 1979. The cone trap - a useful tool for index trapping of small mammals. Ann. Zool. Fenn. 16: 144–150.

Pizzolotto R., Brandmayr P., 1996. An index to evaluate landscape conservation state based on land-use pattern analysis and geographic informations system techniques. Coenoses 11: 37–44.

Recher H.F., Lunney D., Matthews A., 2009. Small mammal populations in a eucalypt forest affected by fire and drought. I. Long-term patterns in an era of climate change. Wildl. Res. 36: 143–158.

Reidsma P., Tekelenburg T., Van den Berg M., Alkemade R., 2006. Impacts of land-use change on biodiversity: An assessment of agricultural biodiversity in the European Union. Agr. Ecosyst. Environ. 114: 86–102.

Robinson R.A., Sutherland W.J., 2002. Post-war changes in arable farming and biodiversity in Great Britain. J. Appl. Ecol. 39: 157–176.

Schoener T.W., 1983. Field experiments on interspecific competition. Am. Nat. 122: 240–284.

Silva M., Hartling L., Opps S.B., 2005. Small mammals in agricultural landscapes of Prince Edward Island (Canada): effects of habitat characteristics at three different spatial scales. Biol. Conserv. 126: 556–568.

Simeonovska-Nikolova D., 2004. Seasonal changes in social behaviour and spatial structure of *Crocidura leucodon* in north-western Bulgaria. Acta Theriol. 49(2): 167–179.

Stoate C., Boatman N.D., Borralho R.J., Rio Carvalho C., De Snoo G.R., Eden P., 2001. Ecological impacts of arable intensification in Europe. J. Environ. Manage.63: 337–365.

Szacki J., Liro A., 1991. Movements of small mammals in the heterogeneous landscape. Landsc. Ecol. 5: 219–224.

Szpunar G., Aloise G., Mazzotti S., Nieder L., Cristaldi M., 2008. Effect of global climate change on terrestrial small mammal communities in Italy. Fresenius Environ. Bull. 17: 1526–1533.

Umetsu F., Naxara L., Pardini R., 2006. Evalueting the efficiency of pitfall traps for sampling small mammals in the Neotropics. J. Mammal. 87: 757–765.

Vickery J., Carter N., Fuller R.J., 2002. The potential value of managed cereal field margins as foraging habitats for farmland birds in the UK. Agr. Ecosyst. Environ. 89: 41–52.

Wang G., Getzb L.L., Linzeyc A.V., Sladed N.A., Kesnerc M.H., 2012. Fast and slow dynamics of northern small mammal populations. Ecol. Model. 225: 95–102.

Yahner R., 1992. Dynamics of a small mammal community in a fragmented forest. Am. Midl. Nat. 127: 381–391.

Zanghellini S., Salvadori C., Ambrosi P., 2004. Study of a community of small mammals in subalpine spruce woods at Lavazè Pass (Trentino, Italy). Studi Trent. Sci. Nat., Acta Biol. 81: 219–223.

Zukal J., Gaisler J., 1992. Testing of a new method of sampling small mammal communities. Folia zool. 41: 299–310.

*Associate Editor:* F. Cagnacci

## Supplemental information

Additional Supplemental Information may be found in the online version of this article:

**Appendix S1.**  Supporting statistical analysis.