

Small mammal responses to farming practices in central Argentinian agroecosystems: The use of hierarchical occupancy models

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Abstract Organic farming is more environmentally friendly than conventional agriculture, promoting greater levels of habitat heterogeneity. Field borders could be more suitable for biodiversity in agricultural anthromes. Small mammals are crucial in these anthromes due to their contribution to food webs and seed consumption. We used hierarchical multi-season occupancy models to assess the effect of organic *versus* conventional farming on multiple small mammal species in agricultural anthromes of central Argentina. We modelled detectability and increased precision of estimates, overcoming deficiencies of previous studies. Small mammals were seasonally surveyed in 70 field borders (conventional) and 63 (organic) during two years. We were able to include less frequent specialist species, detecting a positive relationship with organic management possibly because of higher habitat quality of borders. Vegetation volume was the most important explanatory variable in both managements. Species' richness was greater under organic management mainly in spring when the habitat quality differences with conventional management were the greatest. Spring is key for the rodent assemblage because of the beginning of reproductive period, when resource demand is important. We suggest that maintaining high quality border habitats, as those supported by organic management, could allow farmers to obtain economic profit while also contributing to biodiversity conservation. Considering the positive role that native rodents may have in some agricultural anthromes, the maintenance of high population numbers may be important for biodiversity conservation. The approach used in this study shows the importance of modelling imperfect detection, reducing bias in parameter estimates, and it should be implemented in similar studies.

Key words: agricultural anthrome, border habitats, conventional management, multi-species, organic management.

INTRODUCTION

Human populations have modified biodiversity and ecosystems processes through changes in land use (Smith 2007; Ellis & Ramankutty 2008). Therefore, there is an increasing need for conservation outside protected areas, and many landscapes outside those areas offer an opportunity to conservation of biodiversity (Quinn *et al.* 2014). Anthromes refer to a classification of the range of land cover patterns that have been generated and sustained by humans but incorporating the relationship with biotic communities (Ellis & Ramankutty 2008; Quinn *et al.* 2014). Cropland anthromes contain an agricultural matrix and linear habitats and patches with natural vegetation. This kind of anthrome offers a key opportunity

to biodiversity conservation in agroecosystems (Quinn *et al.* 2014).

Land transformation for intensification and expansion of agriculture has degraded ecological systems converting natural landscapes into croplands and pastures, constituting one of the most worldwide land-use activities (Foley *et al.* 2005). This conversion results into habitat loss and fragmentation, altering habitat quality and suitability, becoming a key cause of native biodiversity declines. Species responses to these alterations depend on their biology, behaviour and habitat requirements. For example, the responses of small mammals could vary with the degree of habitat specialization. Specialist species are more dependent on habitat quality and they suffer more from habitat disturbance than generalists (Fischer *et al.* 2011, 2013; Coda *et al.* 2015; Gomez *et al.* 2015; Schlinkert *et al.* 2016). Therefore, the abundance of generalist or specialist species could be an indicator of the degree of habitat disturbance.

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Balancing agricultural land use and biodiversity conservation has become a challenge in the last years. Recent discussions about agriculture and biodiversity pose the debate between land sharing and land sparing approaches (Perfecto & Vandermeer 2012; Tscharntke *et al.* 2012). Land sharing argues that the application of wildlife-friendly farming methods, including the retention of patches of natural habitat and extensively farmed seminatural habitats within the countryside, and farming in ways that minimize the negative effects of fertilizers and pesticides on non-target organisms, would reduce the impact of agriculture on biodiversity (Green *et al.* 2005; Perfecto & Vandermeer 2012; Tscharntke *et al.* 2012). Organic farming meets the goals of land sharing arguments since it involves practices that result in greater levels of habitat heterogeneity, and contains greater densities of uncropped habitats compared to conventional farming (Fuller *et al.* 2005). Besides, insecticides, herbicides and inorganic fertilizers are entirely or largely avoided, promoting well maintained and more suitable border habitats (Norton *et al.* 2009). Consequently, this practice is more environmentally friendly than conventional agriculture, which is mainly dependent on external inputs of agrochemicals for the productions of crops and animals (Bengtsson *et al.* 2005; Tuck *et al.* 2014). Among the taxa that coexist in agricultural landscapes, small mammals are common and can have positive or negative impacts. This group of animals has received a great interest for its role as pest, causing damage to agricultural products (Michel *et al.* 2006), and as reservoir hosts for several rodent-borne diseases (Vorou *et al.* 2007; Fischer & Schröder 2014). However, small mammals can be crucial in agroecosystems because of their contribution to well-structured food webs (Salamolard *et al.* 2000; Butet & Leroux 2001; Arlettaz *et al.* 2010), the consumption and dispersal of plant material (Kiviniemi & Telenius 1998; Kollmann & Bassin 2001; Kollmann & Buschor 2002; Baraibar *et al.* 2009; Fischer *et al.* 2011) and mycorrhizal fungi (Schickmann *et al.* 2012), as well as the consumption and control of invertebrates (Gliwicz & Taylor 2002). As they rapidly respond to environmental change, due to their short live cycles and restricted spatial areas, they can serve as model organisms for better understanding agricultural anthromes (Barrett & Peles 1999).

While in Europe, land-use changes due to agricultural intensification have been recognized by scientists for almost five decades, in Latin America these research topics are still underrepresented and became of public interest not before the 1980s. However, the rates of agricultural expansion and intensification have also increased considerably in Latin America due to technological changes in the last decades (e.g. no-tillage techniques, genetically modified crops) and

market conditions (e.g. global increase in soybean demand) (Baldi & Paruelo 2008). Particularly in Argentina, the structural complexity of agroecosystems differs deeply from the well-studied European systems, since they are characterized by an extensive and homogeneous cropland mosaic, which comprise large arable fields and sparse linear habitat networks (Baldi *et al.* 2006; Poggio *et al.* 2010). In this country, the farming area dedicated to no-tillage cropping system increased from 2 Mha in 1992–1993 to 27 Mha in 2010–2011 (Aapresid 2012; Álvarez *et al.* 2015) and during this process many field borders were removed to enlarge crop areas (Aizen *et al.* 2009). On the other hand, the area dedicated to organic farmland is small; currently, there are 3.6 Mha under this practice, only 7% of this surface is intended to crop production, whereas the rest is dedicated to pastures for cattle production (SENASA 2014). Córdoba province is located in the centre of Argentina and it is one of the main productive regions of the country with almost 9 Mha under crop production in years 2014/2015 (MAGyA 2013). This province has undergone a marked transformation due to the expansion and intensification of agriculture. However, these important changes in agricultural landscapes are insufficiently documented in studies involving farmland biodiversity with conservation goals (see Coda *et al.* 2014, 2015; Gomez *et al.* 2015). Thus, better data are needed to identify how extension and arrangement of novel elements embedded in agricultural anthromes contribute to biodiversity conservation.

In previous studies, which took place in Córdoba province, the relationship between environmental characteristics and small mammals was assessed using generalized linear models (GLM) (Coda *et al.* 2015; Gomez *et al.* 2015). Although it was considered as an appropriate tool, this approach assumes perfect detection of the species of interest possibly biasing parameter estimates by not incorporating non-detection error (Mackenzie *et al.* 2002; Gu & Swihart 2004; Urban & Swihart 2011). Occurrence (detection/non-detection) is a data structure that can be readily gathered for many species using an appropriate design. It requires suitable statistical models to account for imperfect and heterogeneous detection probabilities, and estimate occupancy probabilities. An important benefit of occupancy modelling is that only data on species detection or non-detection are needed (Mackenzie *et al.* 2006; Nicholson & Van Manen 2009).

The aim of this study was to use hierarchical multi-season occupancy models to assess the effect of farming management, organic *versus* conventional, on multiple species of small mammals in agricultural anthromes of central Argentina. Under this approach, multiple species are linked together under a

community-level distribution, allowing for a more efficient use of data, and increased precision of occupancy for individual species, even those less frequently detected. Previous research in the study area elucidated effects of farming management on some species (Coda *et al.* 2015), but they have only considered more abundant species including only one specialist species, and without considering detection probabilities, so our understanding of such effects is incomplete. The scope of this study enabled us more adequately to evaluate hypotheses concerning farming management. By explicitly modelling detectability, occupancy models should yield less-biased conclusions than previous models. We predicted that if the effects of farming practices on small mammal populations vary with the degree of specialization of the species, higher occupancy probabilities of specialist species would be registered in border habitats of organic than of conventional farms. Besides, we predicted that species richness would be greater in border habitats of organic than of conventional farms.

METHODS

Study area and trapping design

Study area was located in the south-east of Córdoba province, Argentina (Fig. 1). Rodent sampling was conducted seasonally from spring 2011 to autumn 2013 (with the exception of winters) in an agricultural landscape. This region is a land mosaic where part of the original flora is restricted to uncultivated border habitats. These linear habitats support a mixed vegetation type dominated by native and invasive herbaceous species. The most frequent crop sequences are wheat-soybean or soybean-maize (as alternate single summer crops per year with a winter fallow), even though the soybean monoculture as a single summer crop per year is also a common practice (Puricelli & Tunesca 2005; Satorre 2005).

In this region, the small mammal assemblage is mainly represented by native Cricetidae rodents (Simone *et al.* 2010). These are ranked, considering species-specific habitat specialization, from generalists (species occur in almost all habitats within the agriculture landscape) to specialists

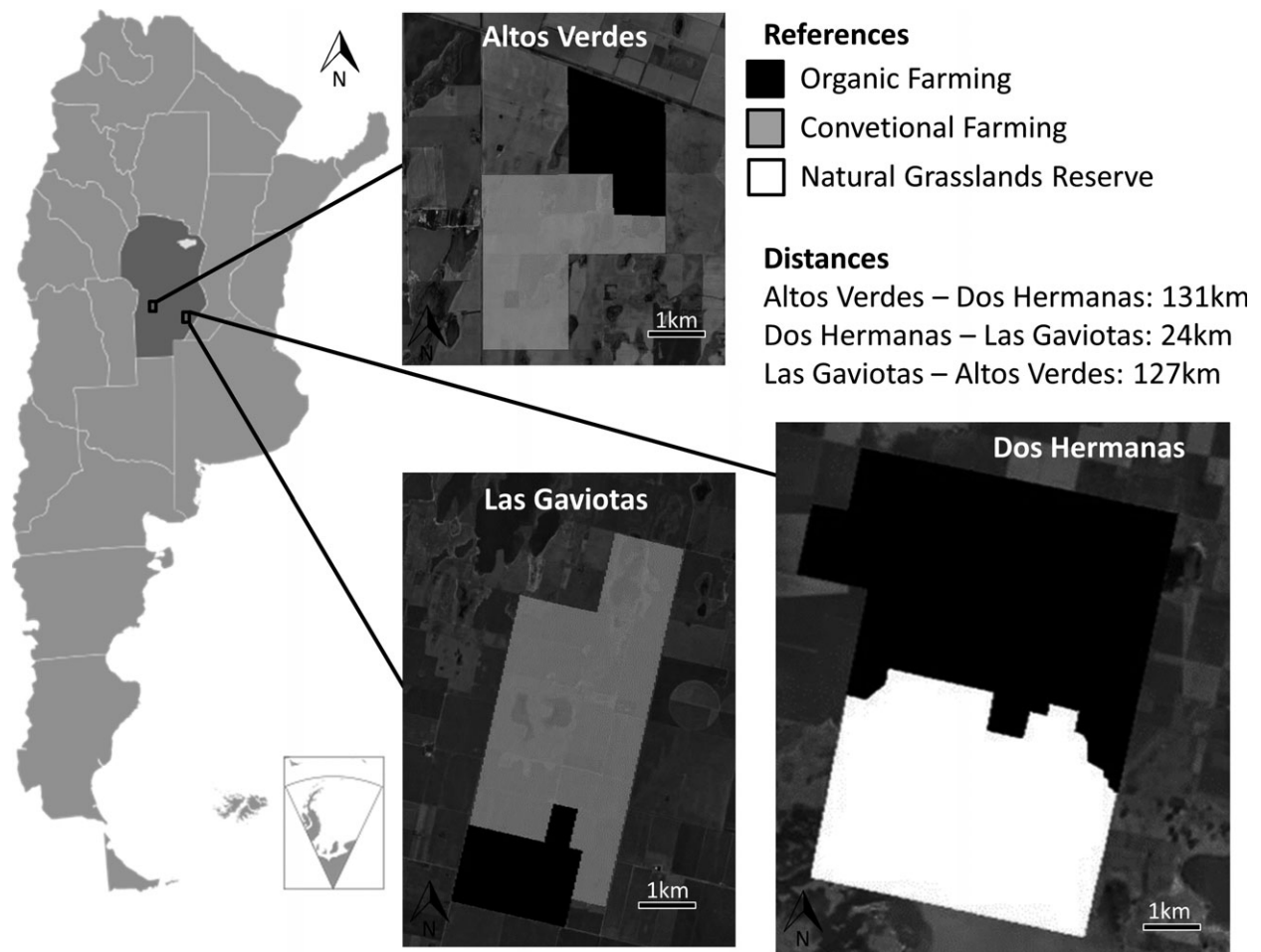


Fig. 1. Study area, agricultural systems of south-eastern Córdoba province with the three farms (Dos Hermanas, Las Gaviotas and Altos Verdes) that include organic and conventional managements, with distances between them.

(species occur in habitats with high vegetation cover): *Calomys musculinus*, *C. laucha*, *Akodon azarae*, *Oligoryzomys flavescens*, *C. venustus*, *A. dolores* and *Oxymycterus rufus* (Martínez *et al.* 2014). The commensal and generalist rodent *Mus musculus* is also registered. *Calomys* sps as well as *O. flavescens* have been classified as granivorous for their consumption of large quantities of seeds (50–73% of stomach volume). Although *Akodon* sps are generally considered omnivorous, *A. azarae* is largely entomophagous, with 41–62% of arthropods in their diet at all seasons (Dellafiore & Polop 1994; Castellarini *et al.* 1998, 2003; Ellis *et al.* 1998). Population numbers of native species show both seasonal and interannual variation, but rodent outbreaks of these species are not common in the region (Jaksic & Lima 2003; Andreo *et al.* 2009; Simone *et al.* 2010).

Given the area under organic farming is small in Argentina, particularly in south-eastern Córdoba province, we were able to survey the entire surface under this management. We sampled three farms: Las Gaviotas (Postel S.A.) (33°50'S, 62°39'W) (1689 ha), Dos Hermanas (Foundation Rachel and Pamela Schiele) (33°39'S, 62°30'W) (4023 ha) and Altos Verdes (Huanqui S.A.) (33°18'S, 63°51'W) (1010 ha) (Fig. 1). Dos Hermanas farm includes a natural grassland reserve of 1922 ha and a productive area (2101 ha) which has been under organic management since 1992. Las Gaviotas and Altos Verdes farms have both organic and conventional managements; organic plots of these two farms have been under this management for 10 years. During the study period, the main crops were soybean and maize, both in organic and conventional farms. Private companies, OIA (Organización Internacional Agropecuaria, 2014) for Las Gaviotas, and Argencert (Argencert 2014) for Altos Verdes and Dos Hermanas, certify their organic crop and livestock production.

Our study was conducted in field border habitats of organic and conventional farms. We selected these linear habitats because they reflect the effect of management practices. Under organic management, field borders maintain high plant cover throughout the year. In contrast, under conventional management these habitats are subject to drift of broad-spectrum herbicides from the neighbouring crops (Coda *et al.* 2014, 2015). Besides, rodents could use field margins as a permanent habitat while foraging in crop fields (Baraibar *et al.* 2009; Gomez *et al.* 2011; Coda *et al.* 2015). We considered the field border as a 1.5–2.5 m wide vegetation strip located in the inner margin of fields. Land use on both sides of the border was classified into crop (fields cultivated with soybean or maize on both sides), or pasture – pasture/crop (fields used for cattle next to either a pasture or a crop).

Capture, mark and recapture (CMR) trapping sessions were conducted for four consecutive nights during spring, summer and fall. CMR methods take account for variation in capture probabilities and provide unbiased estimates. A total of 133 sites were surveyed during two years, 70 in conventional and 63 in organic farms, respectively. Each line had 20 traps similar to Sherman live-traps, with a trap every 10 m in the middle of a border. The minimum distance between lines was 300 m to avoid correlation between neighbouring lines and the influence of neighbouring farms (Sommaro *et al.* 2010; Gomez *et al.* 2011). Traps were baited with a mixture of peanut butter and cow fat.

Animals caught through live trapping were identified and marked with numbered ear-tags and released at their site of capture in order to allow subsequent recaptures. In each line, vegetation measurements were made using a quadrat of 1 m² centred in a trap, 10 traps were surveyed. Vegetation volume (m³) was estimated in each quadrat unit as shelter × height. Shelter was the combination of the surface of green cover and plant litter (m²) and height was estimated as the mean value of ten measurements of green cover and plant litter randomly registered in the 1 × 1 m quadrat. Values from the ten quadrats were averaged to obtain a unique value of vegetation volume per line.

Statistical analyses

We used multi-species occupancy models accounting for multiple seasons (MS-MSOM) with known species richness, using a Bayesian approach to estimate the influence of farming management, land use and vegetation volume on small mammal species over seasons (Dorazio *et al.* 2005; Royle & Dorazio 2008; Zipkin *et al.* 2009; Kéry & Royle 2016). Occupancy estimation accounts for imperfect detection probabilities of each species ($P < 1$), so that if a species is not observed at a certain point, it can be either truly absent, or present but undetected (Mackenzie *et al.* 2002, 2006; Tyre *et al.* 2003).

Occurrence (z) for each species (k) at each site (i) and season (l) is specified as a Bernoulli random variable, $z_{i,k,l} \sim \text{Bern}(\psi_{i,k,l})$, where $\psi_{i,k,l}$ is the probability that species k occurs at site i and season l . True occurrence is imperfectly observed, where $z_{i,k,l} = 1$ when the species is present and zero otherwise. We differentiated between species absence and non-detection $z_{i,k,l} = 0$, by estimating species detection probabilities from the detections/non-detections history from repeat surveys (K = number of nights) at each site i . We assumed detection to be species- and season-specific, then $y_{i,k,l}$ is the detection frequency, i.e. the number of times over K nights a species is detected. We modelled detection (y) as a Binomial distribution as $y_{i,k,l} \sim \text{Bin}(K, p_{k,l} \times z_{i,k,l})$, where $p_{k,l}$ is the probability that species i is detected on season l , and is conditional on the species being present (i.e. $z_{i,k,l} = 1$).

We allowed individual species occurrence probabilities to vary by seasons (spring, summer and fall) and subject to temporal-site-level covariates. We assumed that vegetation volume, land use and management could influence occurrence probabilities for each species. We modified the model proposed by Kéry *et al.* (2009) (see Goijman *et al.* 2015) to incorporate random time effects on the baseline occupancy and covariate effects for each species, as a mean of controlling for potential sources of variation in different seasons. As an example, one of our occupancy models was:

$$\text{logit}(\Psi_{[i,k,l]}) = \beta_{0[k,l]} + \beta_{1[k,l]} \text{VegVol}_{[i,l]} + \beta_{2[k,l]} \text{Mgmt}_{[i,l]} + \beta_{2[k,l]} \text{Use}_{[i,l]}$$

where both the parameters denoting covariates effects and the intercept β for each species $k = 1, 2, \dots, N$ and season $l = 1, 2, 3$ were estimated. Vegetation volume was

standardized, and both management and use where binary variables (organic = 1, conventional = 0; pastures = 1, crop = 0, respectively). We based our focus on estimation and prediction of the effects of covariates on each species and compared several *a priori* competing models testing a combination of the aforementioned covariates and interactions. We used WAIC to compare models (Watanabe-Akaike information criterion; Watanabe 2010), adapted to MS-MSOMs models based on Broms *et al.* (2016) (formulas provided in Appendix S1). We made inferences based on 95% Bayesian credible intervals (95% BCI), assuming a strong effect when BCI does not overlap zero, and an important effect when the interval overlapped zero less than 25% (i.e. 75% of the interval had the same sign of the mean effect).

We used a Bayesian approach in the programs R and JAGS, through package jagsUI, which uses Markov chain Monte Carlo (MCMC) to find the posterior distribution of the parameters of interest (Kellner 2017). We used independent uninformative priors for the group level hyperparameters, testing sensitivity to priors through comparison of model results and convergence under various parameterizations (Broms *et al.* 2016). We ran three chains of length 90 000 and retained 4990 values per chain, after discarding 40 100 for adaptation and burn in, thinning by 10 for economy of memory space and to reduce autocorrelation (14 970 total samples of the posterior for each parameter). We assessed model fit using Bayesian *P*-value and convergence through visual examination of the trace and density plots, and using the Gelman and Rubin diagnostic (\hat{R}), which includes the variance between the means from the parallel chains and the average of the within-chain variances, and convergence is reached when \hat{R} is near one (Gelman & Rubin 1992). We calculated site-specific species richness by summing the estimated number of species (e.g. the latent occurrences, z values) for each season.

RESULTS

We trapped a total of 647 individuals corresponding to six native rodent species, 323 in conventional and 321 in organic farms. The numbers (and % of captured individuals) by species and management were: 206 (64%) and 173 (54%) *C. musculinus* individuals were captured in conventional and organic farms respectively, 76 (23.53%) and 82 (25.55%) *C. laucha*, 32 (9.91%) and 44 (13.71%) *A. azarae*, 5 (1.55%) and 21 (6.54%) *O. flavescens*, 2 (0.615%) and 1(0.31%) *O. rufus* and finally 2 (0.615%) *C. venustus* captured in conventional farms. We also captured the exotic species *M. musculus*, 6 and 13 in conventional and organic farms, respectively. Unlike previous studies, we had data to estimate occupancy for all species although inferences about *O. rufus* and *C. venustus* are cautious, based on their low number of captures. The model including vegetation volume, farming management and land use, with no interactions, ranked as the best predicting model to explain occupancy probabilities of rodent species (Table 1).

Table 1. Models indicating variables predicting rodent occupancy, and detection probabilities accounting for multiple seasons (MS-MSOM). Best fitting model is in bold-face. M: management; Vv: Vegetation volume; U: land use on both sides of the border; WAIC: Watanabe-Akaike information criterion

Model	Deviance	WAIC	<i>P</i> -value
Vv + M + U	1619.05	1691.63	0.554
Vv + M	1653.99	1716.49	0.583
Vv + M + U + Vv × M	1709.42	1910.95	0.551
Vv + M + Vv × M	1705.09	1952.09	0.559
Vv + M + U + U × M	1783.68	2118.53	0.550

Detection probabilities varied by species but were generally low (i.e. $\hat{p} < 0.5$), varying by season in some cases as well (Appendix S2). The highest detection probabilities were for the generalist *C. musculinus* across all seasons, and the lowest for the specialist *O. flavescens* in spring. Overall, vegetation volume had a positive effect on the occupancy of both generalist and specialist rodent species (Community mean: $\hat{\beta} = 1.765 \pm 0.849$, 95% BCI: 0.013–3.456; Fig. 2A; Appendix S2). The generalist *C. laucha* (in spring) and *C. musculinus* (in summer) were the only species that responded negatively to organic management (with more than 75% of the interval had the same sign of the mean effect), while there was a positive effect of this management in the specialist species *A. azarae* in spring and fall and *O. flavescens* in summer and fall (Fig. 2B). Both the generalist species *C. musculinus* and *C. laucha*, and the specialist species *A. azarae* and *O. flavescens* responded negatively to borders surrounded by pastures (i.e. positive effect of crops on both sides; community mean: $\hat{\beta} = -0.727$, 95%BCI: $-2.362, 0.671$; Appendix S2), all species in summer, both specialist in summer and fall (Fig. 2C). To describe occupancy pattern, we selected results from the specialist *A. azarae* and the generalist *C. laucha* in spring since they were the clearest, results for the remaining species are shown in Appendices S2 and S3. Considering the results of *A. azarae* and *C. laucha* in spring, the former had higher occupancy probabilities in organic than in conventional farms at similar vegetation volume values (Fig. 3A) whereas the latter showed the opposite pattern (Fig. 3B). Land-use effect was greater in *A. azarae* than in *C. laucha*. Mean occupancy probabilities for the former were higher in borders surrounded by crops ($\widehat{psi} = 0.322$, 95% BCI: 0.033–0.873; $\widehat{psi} = 0.209$, 95% BCI: 0.001–0.666, organic and conventional respectively), than pasture ($\widehat{psi} = 0.0170$, 95% BCI: 0.009–0.672; $\widehat{psi} = 0.109$, 95% BCI: 0.001–0.520), independently of farming management; whereas occupancy probabilities of *C. laucha* were higher under conventional ($\widehat{psi} = 0.783$, 95%BCI: 0.431–0.998; $\widehat{psi} = 0.730$,

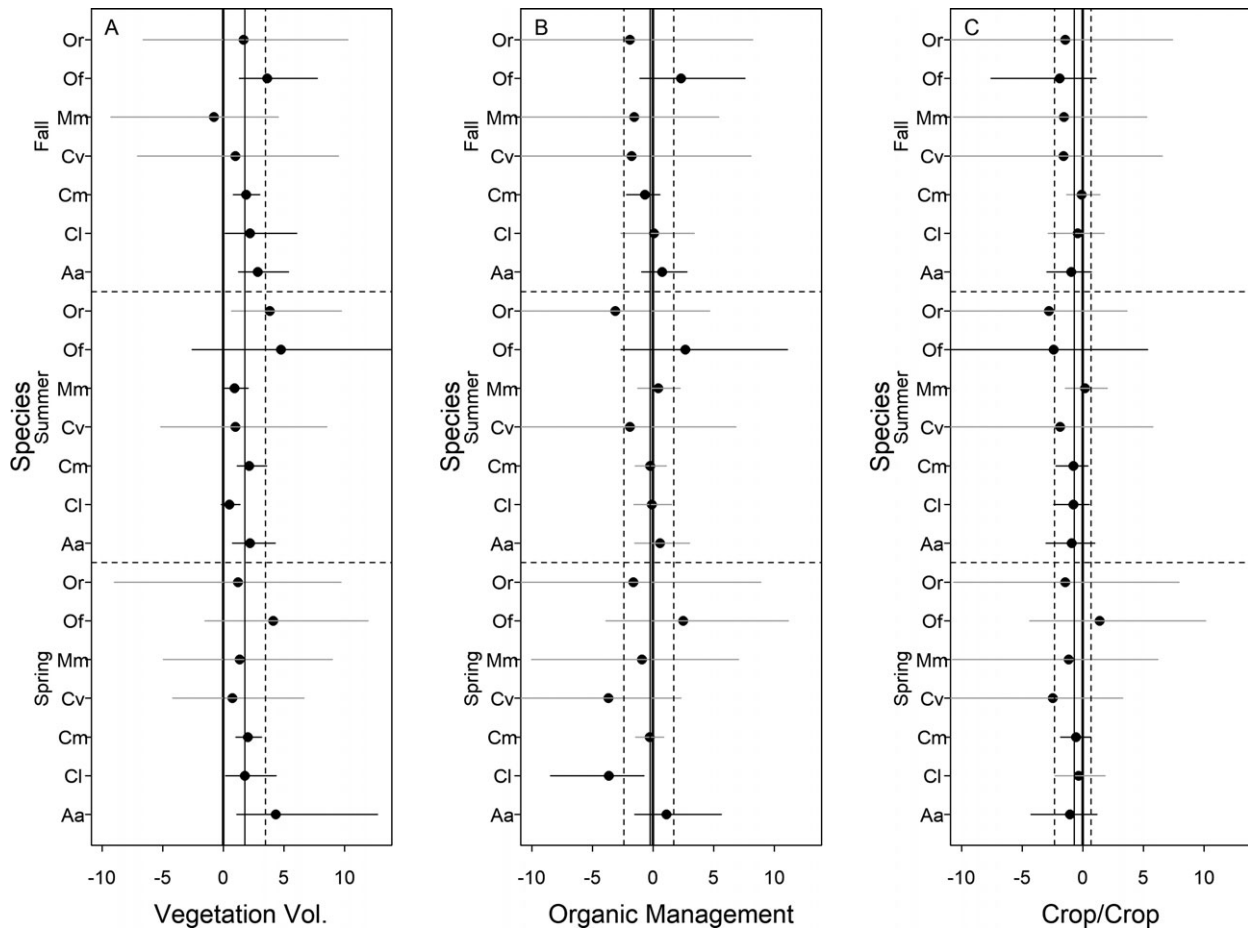


Fig. 2. Vegetation volume (A), organic management (B) and crop/crop (C) coefficients in the logit scale ($\hat{\beta} \pm \text{SD}$, 95% BCI) on logit occupancy (logit (\widehat{psi})) of each small mammal species by season for the best fitting model. In black, lines where 75% of the interval had the same sign of the mean effect. Aa: *Akodon azarae*; Cl: *Calomys laucha*; Cm: *C. musculus*; Cv: *C. venustus*; Mm: *Mus musculus*; Of: *Oligoryzomys flavescens*; Or: *Oxymycterus rufus*.

95%BCI: 0.292–0.996, crop and pasture, respectively) than organic management ($\widehat{psi} = 0.186$, 95% BCI: 0.014–0.609; $\widehat{psi} = 0.138$, 95% BCI: 0.012–0.454) independently of land use.

The number of species increased with vegetation volume, but there were a higher number of species in sites under organic than under conventional management in spring starting from a vegetation value around 0.3 m^3 (Fig. 4A,C). In summer and fall, the number of estimated species increased with vegetation volume but with a similar pattern in both managements (4B).

DISCUSSION

Previous studies on small mammals have documented important effects of attributes of habitats, farming management and land-use intensity. However, they failed to incorporate imperfect detection probabilities of the species, and assumed constant

detection was among sites and species. Unfortunately, assuming perfect or homogeneous detection could lead to substantial bias (e.g. Nichols 1992; Mackenzie *et al.* 2006). In addition, prior studies have often focused on the most common species since sample sizes associated with rarer species were too small to warrant conventional analyses (e.g. Coda *et al.* 2015; Gomez *et al.* 2015). Our study addressed each of these issues by explicitly modelling non-detection error and using a hierarchical model allowing for a more efficient use of data, and increased precision of occupancy estimates, overcoming deficiencies associated with prior studies. To our knowledge, this is the first study that incorporates detection bias into analyses of small mammal responses to farming management. Previous studies often considered number of individuals (or species) captured, as population and community response variables, ignoring rare species. Consequently, previous conclusions on small mammal community responses to farming management likely provided less

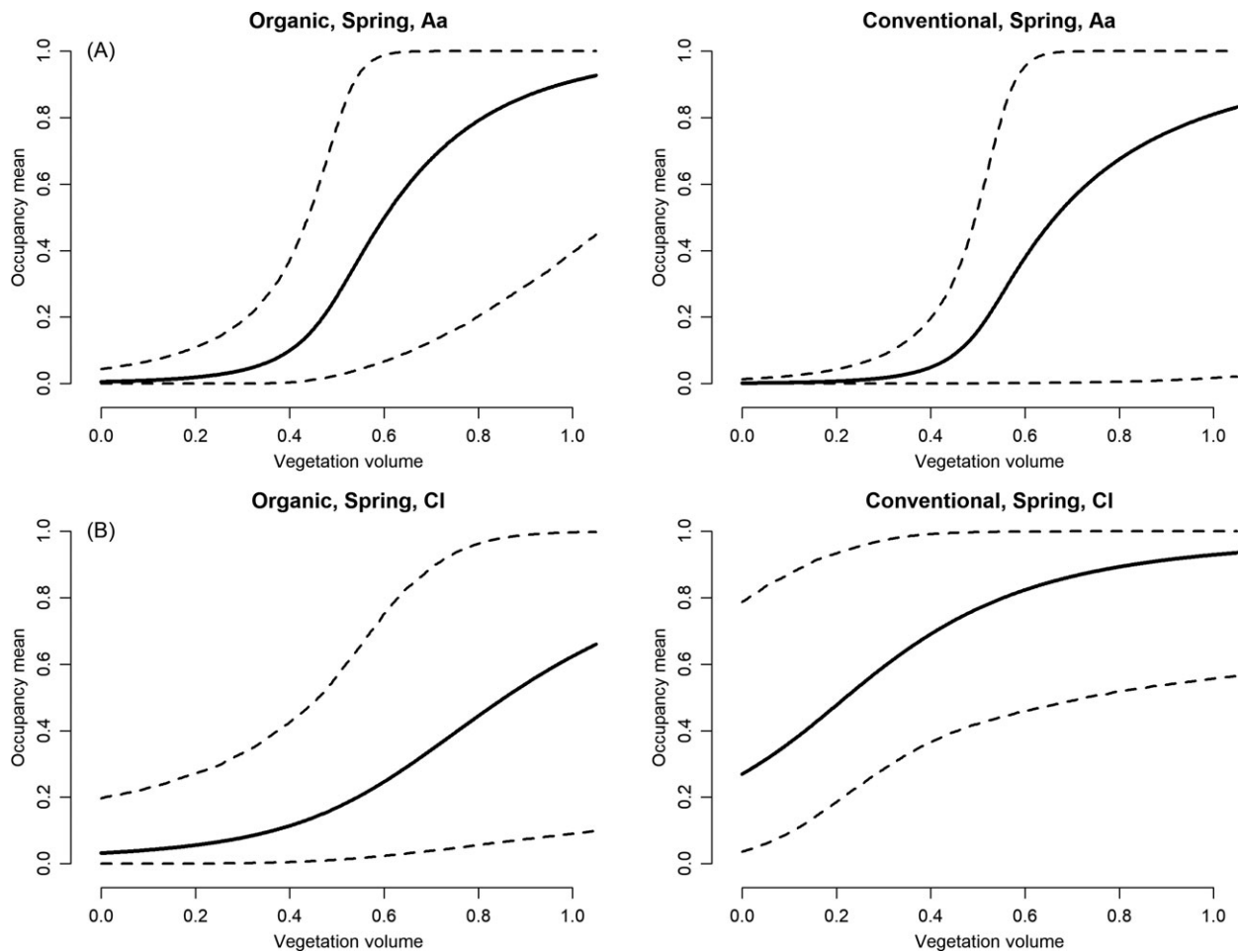


Fig. 3. Probabilities of occupancy ($\psi \pm \text{SD}$, 95% BCI) for (A) *Akodon azarae* (Aa) and (B) *Calomys laucha* (Cl) by management and vegetation volume in spring for the best fitting model.

accurate estimates than those that incorporate imperfect detection.

Unlike Coda *et al.* (2015), who did not consider imperfect detection of small mammals using part of this same data, this study allowed modelling the effect of the explanatory variables not only on abundant species, but also in less frequently captured species. Although incorporating detectability on small mammal responses to land use could be difficult for direct comparison with earlier studies, some important results could be highlighted. Considering species responses to environmental covariates, this study broadly agrees with the previous one. Both agree that vegetation volume is a key for explaining abundance and occupancy probabilities of specialist and generalist species. In addition to the generalist species *C. laucha* and *C. musculus*, and the specialist *A. azarae*, this study also allowed us to establish a positive relationship between vegetation volume and the specialist *O. flavescens*. The effect of vegetation volume and small mammals is strong and is detected despite the type of statistical analyses. On the other

hand, we found some differences in relationship with farm management. Since in the present study we included less frequent species such as the specialist *O. flavescens*, we were able to detect its positive relationship with organic management. As it was previously suggested for other small mammal species, this relationship would be caused by the high habitat quality of organic farm borders (Coda *et al.* 2014, 2015). This positive relationship was also observed for the specialist *A. azarae*, which has also been suggested in the previous study (Coda *et al.* 2015) but not confirmed due to its low frequency. Opposite to what was observed in the previous study (Coda *et al.* 2015), we did not find a positive effect of organic management on *C. musculus* or *C. laucha* in any season. What is more, conventional management could positively affect occupancy probabilities of these species, which was more evident for *C. laucha* in spring. Both generalist species are opportunistic, good settlers of disturbed habitats (Busch *et al.* 2000; Sommaro *et al.* 2010; Gomez *et al.* 2015) and may benefit from competition relaxation in more altered

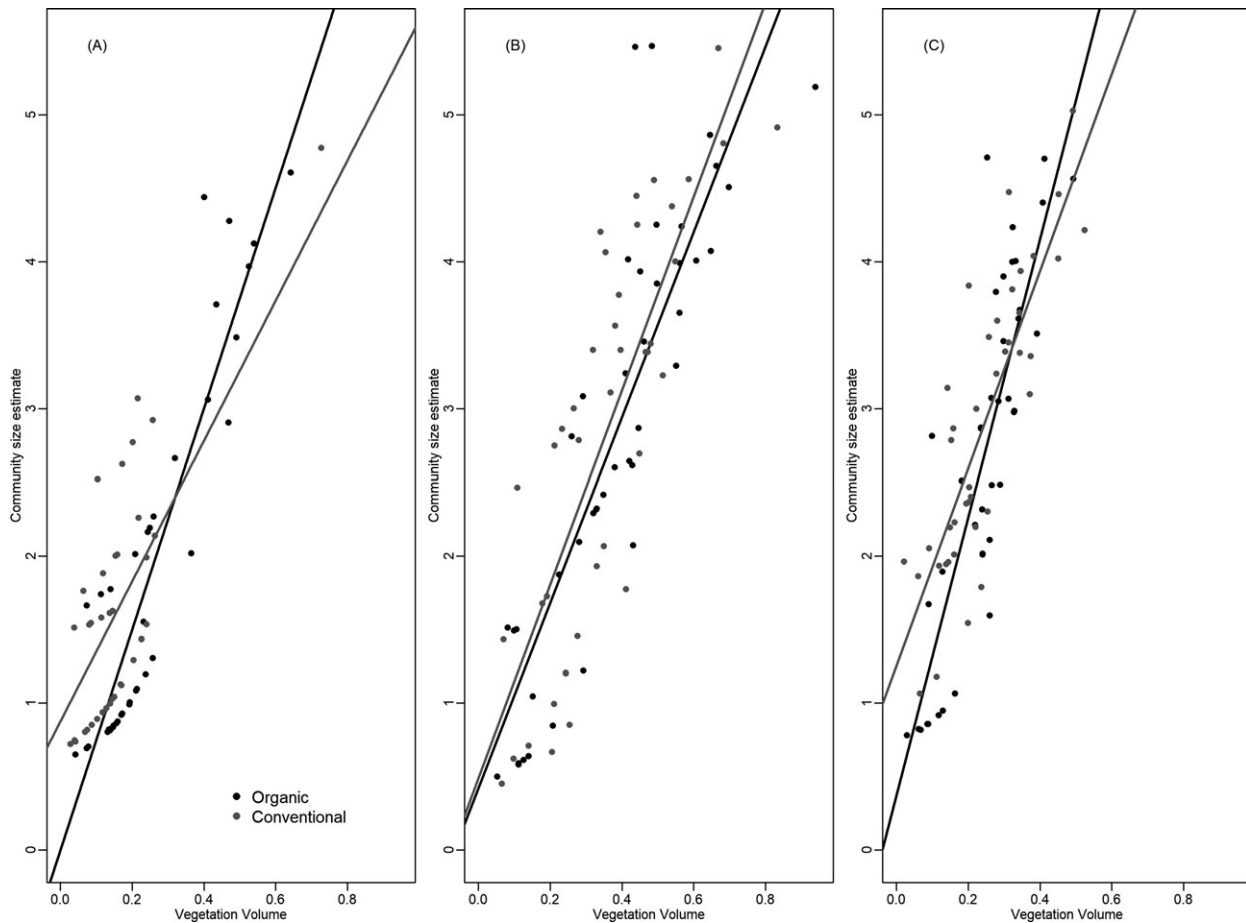


Fig. 4. Derived total number of species at each of 133 sites under the best fitting occupancy model considering management and vegetation volume; (A) spring, (B) summer and (C) fall.

systems as conventional farms. The population number of these native species does not reach outbreak densities and does not cause serious harvest losses in the common crops in the study region. However, due to the role of some species as reservoir of zoonotic diseases (Mills *et al.* 1992), an increase in the population density of *C. musculus* recorded in conventional farms could lead to a greater Junin virus maintenance and an expansion of the population (Mills *et al.* 1992).

The effect of farming management on species richness could not be evaluated on the previous study (Coda *et al.* 2015). With the approach used in this study, we found that the number of species is greater in sites under organic management mainly in spring, when the greatest habitat quality differences between border habitats of organic and conventional managements were observed. In the former, better quality consisted of increased vegetation volume, mainly comprised of green cover, when compared to conventional border habitats (Coda *et al.* 2014, 2015). Spring is a key season for all species of the rodent assemblage because of the beginning of reproductive period, where

resource demand is important. During summer, when quality of both conventional and organic border habitats was similar (Coda *et al.* 2014), the number of species was also similar. Finally, although there are habitat quality differences between borders during fall, they are less noticeable as well as the differences between species richness.

Similarly to other agricultural systems around the world (Quinn *et al.* 2014), the central region of Argentina was originally dominated by grasslands, but now agriculture is predominant, better described as a cropland anthrome. Novel ecosystems embedded in anthromes are an opportunity for biodiversity conservation since they provide nest sites, foraging space and cover for many species (Quinn *et al.* 2013, 2014). In cropland anthromes, these habitats exist because of social structures, soil and water conservation practices or landscapes features that prevented the land from being farmed (Quinn *et al.* 2014). Border habitats with remnant grassland vegetation, adjacent to the wiring of the fields, are good examples of the latter in Argentinean agroecosystems. Borders of organic farms, with high green vegetation cover that remain constant

throughout the seasons, offer suitable habitats for supporting small mammal populations, particularly habitat specialist species, which are more dependent on habitat quality. These results suggest that conservation of linear habitats with high quality (measured as green cover) allows farmers to continue to obtain profit while also contributing to conservation of small mammal biodiversity. Moreover, the results of this study extend the understanding of the value of using anthromes as a conservation framework (Martin *et al.* 2014; Quinn *et al.* 2014), particularly for species that are not historically of conservation concern in agroecosystems of Central Argentina.

Finally, we showed the importance of modelling detection probabilities, especially for species with low frequencies. This approach was important for reducing bias in parameter estimates and model selection, and should be used in other studies (Moore & Swihart 2005). We support the idea that habitat heterogeneity is a key element of wildlife-friendly farming (Quinn *et al.* 2014), suggesting that land sharing, where farming and conservation goals are met on the same land, benefit biodiversity. Currently, factors that involve landscape variables together with local variables are being assessed using the same methodology to reliably predict the impact of farming practices on small mammal biodiversity in Córdoba agroecosystem.

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RESEARCH ETHICS

The research on live animals was performed in a humane manner and followed guidelines for the care and use of animals (www.sarem.org).

DATA ACCESSIBILITY

Data will be available from the Dryad Digital Repository

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article:

Appendix S1. R and JAGS model code, specifications and WAIC, for our best fitting model of multi-species occupancy accounting for multiple seasons (MS-MSOM) with known species richness, to estimate the influence of farming management, land use and vegetation volume on small mammal species over seasons.

Appendix S2. Supplemental table.

Appendix S3. Supplemental figures.