CONSERVATION AND MANAGEMENT OF BIRDS IN AGROECOSYSTEMS IN EAST-CENTRAL ARGENTINA

by

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(Under the Direction of MICHAEL J. CONROY)

ABSTRACT

Bird conservation, and associated ecosystem services, is challenged by agricultural intensification and expansion. In Pampas grassland and Espinal forest ecoregions of east-central Argentina these processes have been ongoing and rapid, requiring the assessment of their impact on biodiversity so as to recommend management alternatives. The objective of this study was to gather evidence to inform decision-making for bird conservation in agroecosystems, focusing on foraging guilds and potential ecosystem services provided. I evaluated the effects of land use on birds at a regional scale in the Pampas and Espinal, using 10 years of a regional bird monitoring program, modeling occupancy with hierarchical multi-species dynamic models using a Bayesian approach. At a local scale, I evaluated factors influencing the use of soybean fields and borders by birds, using bird surveys and arthropod sampling in 78 borders and 20 soybean fields, in four crop stages for two years. I analyzed bird occupancy using multiple-groups single-season models, separating field interior and edges, and fitting Poisson mixed models for counts of the orders of arthropods consumed by birds. I used structured decision making (SDM) to find optimal management strategies to integrate bird conservation with soybean agriculture. I demonstrated how the regional scale results can be used as a tool for decision-making, mapping

species-based spatial distributions over time. Although potential ecosystem services offered by birds were distributed throughout the study area, few species could provide them in crop dominated areas. Most raptors, unlike other guilds, were associated with soybean. Most insectivore gleaners seemed unaffected by crops, suggesting their perception of landscape at smaller scales. Birds in soybean fields are mainly those common in agroecosystems, some likely providing pest control service, while most guilds benefited from native trees in borders. Counts of arthropods preyed by birds remained mostly constant throughout the soybean cycle. Finally, I identified the objectives of the SDM process: maximizing insectivorous birds and farmers' well-being, while minimizing management costs. Reducing insecticide applications in soybean, and either planting trees in borders or no management, were the best decisions dependent on constraints of cost allocation and percent of managed border.

INDEX WORDS: Birds, Foraging Guilds, Ecosystem Services, Monitoring, Occupancy
                Estimation, Bayesian Hierarchical Modeling, Structured Decision Making,
                Conservation, Agriculture, Soybean, Borders, Argentina

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DEDICATION

I dedicate this dissertation to all those people who have a dream...

…and are brave enough to go for it.

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INTRODUCTION AND LITERATURE REVIEW

RECONCILING MODERN AGRICULTURE AND BIODIVERSITY CONSERVATION

With the growth of the human population projected at 9 billion people by mid-century, one of the

biggest challenges presented today is to meet the growing demands of food supply while

ensuring environmental sustainability (Foley et al. 2005, Godfray et al. 2010, Foley et al. 2011,

Tscharntke et al. 2012, Cunningham et al. 2013). There is an increasing need for decision-

making strategies to focus on the tradeoffs between maximizing agricultural production in a

sustainable way, by minimizing biodiversity and habitat losses, conserving the associated

ecosystem services, and maintaining economic and social well-being. Modern agriculture has

been successful in increasing food production, by introducing new technologies which allow for

high-yielding cultivars, fertilizers, and pesticides, among others (Foley et al. 2005). However,

modern land use has also caused environmental damage, stemming from rapid intensification

and expansion, affecting not only biodiversity but also rural livelihoods (Gibbs et al. 2009, Foley

et al. 2011, Cunningham et al. 2013). Agricultural expansion consists of the replacement of

natural ecosystems by croplands or pastures, affecting associated biodiversity as one of the

consequences. Conversely, agricultural intensification refers to any way of making existing lands

more productive, for example, by the use of technologies such as irrigation or genetically

modified cultivars, or intensified use of fertilizers or pesticides (Foley et al. 2005, Foley et al.

2011, Cunningham et al. 2013). Some consequences of the intensification are the degradation of

water quality caused by runoff of agricultural chemicals – facilitated by the excessive use of fertilizers –, or the direct or indirect effects of fungicides, herbicides and insecticides on native biodiversity, among many others (Potts 1986).

How do we meet the production demands without causing more damage to the environment? Agricultural expansion is no longer the answer; recently, the solution has been focused on producing more food in the same land by closing the yield gap, which consists of improving crop yields in areas that may be underperforming (Godfray et al. 2010, Foley et al. 2011, Cunningham et al. 2013). For example, in Argentina, a yield gap of 60% has been found in rain-fed cereals (Godfray et al. 2010). At present, crop yields in developing countries are increasing at higher rates than in developed countries, and thus, the effect of agricultural change on natural ecosystems is greater in the former ones (Green et al. 2005). Increasing the yields would increase food supplies but could also have effects on the environment, then it becomes fundamental for these systems to be sustainable; some potentially sustainable tactics are the improvement of crop genetics, precision agriculture, buffer strips, target-specific pesticides, and enhancing ecosystem services, among many others (Godfray et al. 2010, Foley et al. 2011, Cunningham et al. 2013).

Faced with the challenge of producing more while minimizing the impact on natural ecosystems and associated biodiversity, two contrasting alternatives arise. Land sharing integrates biodiversity conservation and agricultural production on the same land, using wildlife friendly practices (i.e. reduced pesticides, habitat for biodiversity); while land sparing consists of segregating those elements producing high yields in some areas, and sparing others for biodiversity conservation (Green et al. 2005, Phalan et al. 2011, Tscharntke et al. 2012). There is ongoing debate about which of these is the best approach, and there are still many information

gaps to be filled. Some controversial points in this discussion are based on the disagreement over claims that wildlife friendly practices result in lower yields, disputed by some authors who argue that yields equivalent to those reached under intensified agriculture can be reached (Green et al. 2005, Vandermeer and Perfecto 2005). Furthermore, those who favor sparing land for intensive agriculture often ignore potential externalities, such as pollution, occasioned by high input pesticides and fertilizers, and also rely on the assumption that areas devoted for conservation will be protected from any unfavorable human use on biodiversity; other externalities may include changes in landscape configuration that result in the lack of connectivity, and the possible loss of ecosystem services (Green et al. 2005, Vandermeer and Perfecto 2005, Phalan et al. 2011, Tscharntke et al. 2012).

Although both approaches are in disagreement about the best solution, they agree on the necessity for well-informed regional solutions and the careful evaluation for either approach to be effective (Green et al. 2005, Gibbs et al. 2009, Phalan et al. 2011, Tscharntke et al. 2012). Although more research is being done to evaluate these strategies, my dissertation is premised on an approach that integrates ecosystem services to agriculture, sharing land for production and wildlife, while promoting the resilience of agroecosystems.

AGRICULTURE IN PAMPAS AND ESPINAL ECOREGIONS IN ARGENTINA

In Argentina, the total area of row crop planting has rapidly increased in recent decades, coinciding with the expansion of soybean cultivation introduced in the mid 1970's, which became the main crop instead of wheat after 1990 (Donald 2004, Aizen et al. 2008, SIIA 2013). This process was facilitated by deforestation and replacement of natural systems, land use intensification, crop rotation, homogenization of landscapes and changes in technologies,

particularly the development of soybean transgenic cultivars (resistant to Roundup®) and the introduction of direct seeding (a.k.a. no-till farming or zero tillage; Donald 2004, Aizen et al. 2008). For example, soybean implanted area increased more than 400% from 1985-2011, making Argentina the world's third largest producer of soybeans (Donald 2004, FAOSTAT 2013, SIIA 2013).

Land use change to annual crops, especially soybean, has been evident in the Pampas grassland and Espinal forest ecoregions, in east-central Argentina, replacing the original grasslands, forests, and also pastures for cattle grazing (Paruelo et al. 2005, SIIA 2013). Rio de la Plata grasslands can be classified into five subunits according to rainfall, soil and vegetation; the study area focus of this dissertation tookplace on sections of the Rolling, Inland and Mesopotamic Pampas (Soriano et al. 1991, Viglizzo et al. 2001), as well as in the northern part of the Espinal (Fig. 1.1). In general the Pampas have always been suitable for cattle, and arable soils limited for crop production to some regions. Specifically, Rolling Pampa has been used for cultivation, Inland Pampa for mixed agriculture and grazing, with some areas limitated for crop production by wind erosion; and Mesopotamic Pampa limited by salinity, drainage and water erosion in some areas (Viglizzo et al. 2001). In this region, the climate is mild, with a mean annual temperature of 10 to 20°C and a mean annual rainfall around 1,000 mm, which decreases to the south-west (Soriano et al. 1991, Baldi and Paruelo 2008).

Row crop agriculture expanded mainly from the Rolling Pampa in the late 1800's, although most of the Pampas had little human intervention and the majority of the land was utilized for cattle grazing (Viglizzo et al. 2001, Thompson 2007). By the1930's no areas were completely free from annual crops, although the conversion to row crops was not homoeneous, and Rolling Pampa experienced the highest conversion (Viglizzo et al. 2001). Till late 1980's,

the management of agricultural lands in the region remained with little or no use of inorganic fertilizers, pesticides, irrigation, concentrated cattle feed, or high yield crop varieties (Viglizzo et al. 2001, Thompson 2007).

Changes experienced in Argentina with the introduction of soybean cultivation and its associated intensification in the mid 1970's are reflected in this region. Most soybean cultivation in Argentina takes place here, with more than 80% of the planted area taking place in the provinces of Buenos Aires, Córdoba, Santa Fe, La Pampa and Entre Ríos. Jointly, the first three provinces are responsible for more than 70% of that area, in the originally Inland and Rolling Pampas (SIIA 2013). Soybean planted area, from 1990 to 2011 grew at a rate between 50% in Santa Fe to 2000% in Entre Ríos, evidencing the rapid land use change that occured in the area in only 10 years (SIIA 2013). Thompson (2007) documented for the the period of 1988-2002 an increased area of cultivated row crops, and a decresase in annual and perennial forage crops decreased; interestingly, cattle densities changed little over this period despite the increase in row crop area, at the same time, the number of land holdings with cattle decreased suggesting the intensification and concentration of cattle production.

Land use conversion to agriculture expanded beyond the preexisting limits of agricultural frontier, for example, in sections of Mesopotamic Pampa; in Entre Ríos, the area of row crop agriculture increased noticeably in the past decades (Bilenca et al. 2009). Entre Ríos occupies 2.8% of the country, and in 2007 was responsible for 8% of its soybean production, but the greatest agricultural expansion occurred in the last 20 years, over areas of native forests in the Espinal ecoregion, grasslands in Mesopotamic Pampas, and cattle grazing lands (Paruelo et al. 2005, SIIA 2013). For example, in Paraná department in Entre Ríos, located at the southern portion of the Espinal ecoregion, row crop agriculture is the main activity in the area, followed

by cattle production (Zaccagnini et al. 2008, unpublished); the main crop in Paraná department is soybean, with 184,500 ha sown in 2010/11, representing 37% of the department area; followed by wheat (10%) and corn (4.5%) (SIIA 2013).

Land use changes and agriculture intensification in Pampas and Espinal ecoregions have occurred at a rapid rate, affecting biodiversity of the original environment. Birds are an important group which could be indicators of other groups. Some species of birds adapted to those changes, and others were negatively affected. The effects on birds of land use and land conversion at a regional (Filloy and Bellocq 2007, Codesido et al. 2008, Schrag et al. 2009, Gavier-Pizarro et al. 2012), and local scales (Goijman and Zaccagnini 2008, Solari and Zaccagnini 2009, Di Giacomo and de Casenave 2010) have been well documented in the Pampas and Espinal ecoregions in Argentina, although long term studies at a large scale are lacking (Azpiroz et al. 2012), as well as studies evaluating specific management actions at local scales. Thus, there is a necessity of reliable data to inform decision-making in agroecosystems that reconcile the conservation of biodiversity with agricultural production in Pampas and Espinal ecoregions in Argentina.

BIRDS AND ECOSYSTEM SERVICES IN AGRICULTURAL LANDSCAPES

Birds are an important part of the agricultural landscape, having intrinsic value, and right of existence. Birds are also considered ecological indicators because of various features, such as their rapid response to habitat changes and disturbances, their conspicuity, and relative ease of identification. In addition, the diversity of bird species, with their different ecological requirements, are useful to assess a wide range of relationships between them and the habitat, the availability of feeding and nesting resources, and may also represent other ecological groups. In

addition, birds also provide a broad spectrum of ecosystem services to human welfare and the agricultural enterprise.

Ecosystem services can be defined as the benefits that people directly or indirectly obtain from ecosystems, and can be grouped into four categories: provisioning, cultural, regulating, and supporting (Millennium Ecosystem Assessment 2005). Provisioning services are products obtained from the ecosystems such as food, fiber, fuel, and clean water, among others. Cultural are nonmaterial benefits such as recreation, aesthetics, or spiritual. Regulating services are obtained from ecosystem processes, for example, pest control, pollination, climate and disease regulation. Supporting services are those necessary for the production of all the other ecosystem services, such as nutrient cycling, primary production, and soil formation. Agricultural landscapes deliver planned services like crop or fuel production, but also have other associated ecosystem services including those provided by biodiversity (Tscharntke et al. 2012).

Birds are involved in a wide variety of cultural and recreational activities like photography and bird watching, with some species sustainably hunted. In addition, many ecosystem services important to agriculture are related to bird feeding behavior. For example, insectivorous species have been documented as agricultural pest controllers (Kirk et al. 1996, Jones et al. 2005, Sekercioglu 2006, Whelan et al. 2008). Seed dispersal is considered the main ecosystem service provided by birds, while they can also contribute to agriculture by consuming weed seeds, and some serve as pollinators (Sekercioglu 2006, Whelan et al. 2008). Prey birds can contribute to control rodent populations either by direct consumption or by fear; and scavengers contribute to environmental sanitation by the elimination of dead animals (Sekercioglu 2006, Whelan et al. 2008). The economic value of most ecosystem services has yet

to be quantified; however, it is evident that birds are an important component of the agroecosystems.

Land use effects on birds have been amply documented, although there is not a single explanation for its impact across many species (Krebs et al. 1999, Newton 2004). Many components of agriculture intensification and expansion are correlated, for example, in areas with intensified agriculture, the size of the fields tends to be larger, the use of agrochemicals more intense, and the landscape is more fragmented when compared to less intensified areas. Therefore, pinpointing one factor is not easy without implementing experimental research. Some relevant factors negatively affecting some bird species at a large scale are the loss of high quality habitat and the increased proportion of croplands, agrochemicals, and cereal yield as a measure of intensification (Chamberlain et al. 2000, Donald et al. 2006, Filloy and Bellocq 2007, Schrag et al. 2009, Gavier-Pizarro et al. 2012). On the other hand, those species that perceive the landscape at finer scales could be benefited from local spatial heterogeneity provided by linear habitats such as field borders with natural vegetation, even in intensified agricultural landscapes (Jobin et al. 2001, Benton et al. 2003, Goijman and Zaccagnini 2008, Di Giacomo and de Casenave 2010).

Even though different factors are correlated, intensified agrochemical use has been documented as one of the most important factors affecting birds at diffrerent scales (Gibbs et al. 2009, Mineau and Whiteside 2013). Herbicides and insecticides could indirectly affect birds by changing the structure of the vegetation, affecting habitat suitability, or by reducing the availability of their feeding resource, either arthropods or seeds (Freemark and Boutin 1995, Beecher et al. 2002, Taylor et al. 2006, Geiger et al. 2010). In addition, toxic insecticides could directly affect birds either by direct contact or via contaminated food (Goldstein et al. 1999).

Then, even if birds are present in the landscape, it is important to acknowledge their potential susceptibility to toxic pesticides, or to the indirect effects of commonly used herbicides and insecticides, by evaluating their use of the habitat, and ultimately providing management recommendations to decision makers.

There are several scales at which land management can be done integrating biodiversity concerns in agricultural landscapes. Fundamentally, management actions should be oriented to provide spatial and temporal heterogeneity within the agricultural matrix, at small and regional scales, where even small patches of vegetation can make a big difference for birds and other organisms; as well as to provide connectivity between patches for their dispersion (Benton et al. 2003, Dardanelli et al. 2006, Cunningham et al. 2013). Provision of suitable habitat and connectivity for biodiversity have also to be accompanied by a correct use of agrochemicals. There is a wide range of specific management actions to achieve spatial and temporal heterogeneity in agricultural landscapes, and the implementation of some of those measures sometimes depends on the productive system in question. Specifically, at a regional or landscape scale, management alternatives should be oriented to conserve natural patches of vegetation, and could be achieved by many ways, such as cattle fed in natural grasslands, or forests, and conserving uncultivable lands untouched. Diversification of the productive landscape, by avoiding monocultures, is also fundamental. Conservation of linear habitats like field borders, hedgerows, or roadside vegetation and shoulders, provides habitat and connectivity across the landscape for a great number of species. At a local scale, actions mainly consist on maintaining spatial and temporal heterogeneity within fields, and can be achieved by conserving natural vegetation on borders, implementing terraces, or strip cover vegetation within fields.

DECISION ANALYSIS FOR CONSERVATION

Despite the fact that management actions that could reconcile conservation of biodiversity with agriculture are available and well known, many producers are unwilling to implement them, and few countries enforce or subsidize environmentally friendly alternatives. Indeed, the voluntary implementation of wildlife friendly practices is not always a straightforward decision, because it involves tradeoffs that not all producers are ready to accept. Similarly, to reach an agreement on policies to ensure biodiversity conservation and sustainable agriculture while providing social and economic welfare of producers is not an easy task. The difficulty is largely due to tradeoffs presented by conflicting objectives, where the achievement of one objective implies the failure to accomplish the other one. In cases like this, and in other natural resources problems where decisions are equivocal or easily accepted, and where stakeholders have multiple objectives, structured decision making (SDM) (Hammond et al. 1999, Clemen and Reilly 2001) becomes an extremely useful tool.

SDM is a formalized method of connecting decisions to objectives, where models are used to predict the effects of the decisions on achieving objectives, with the ultimate goal to make optimal decisions in light of multiple sources of uncertainty (Conroy and Peterson 2013). By organizing decisions in this formal way, SDM provides structure and guidance for thinking systematically about difficult, multiple-objective problems, provides clarity and transparency to the decision making process, fosters participation among stakeholders, and complements science with values, policies and laws (Clemen and Reilly 2001, Williams et al. 2009, Gregory et al. 2012, Conroy and Peterson 2013). The key is to divide the problem into its elements; identify those relevant to the decision; apply systematic thinking; and finally make the decision (Hammond et al. 1999). The key elements in this process are a problem statement, delineation of

objectives, comparison of decision alternatives, and the consideration of consequences and tradeoffs. The SDM process starts with the *problem statement*, which defines the context and the decision situation, including any relevant time horizons or spatial extents. *Objectives* define both the values that the decision maker is trying to achieve, as well as intermediate steps needed a to achieve these ultimate and fundamental goals. Once the problem and objectives have been identified*, decision alternatives* are delineated that define the actions available to achieve these goals. Evaluating the consequences is fundamental to understand how well each alternative will help achieve the objectives; modeling the consequences can integrate information from scientific studies, literature searches, and / or expert opinion. Objectives may conflict with one another, thus it is fundamental to evaluate the tradeoffs by prioritizing competing objectives. An important part of the SDM process is the identification of the different types of uncertainties involved in the decision making process, some that we can acknowledge and aim to reduce, and others that are irreducible. The different types of uncertainty can be summarized as statistical, observational, and structural (or model) uncertainties, which a can be incorporated into the decision process; environmental or demographic stochasticity, and partial controllability, are uncertainties that we cannot control, for example, in the latter, the decisions we make will not always turn out to be exactly as we expected.

SDM is a cyclical process (Figure 1.2, adapted from Clemen and Reilly 2001). As outlined, after choosing the best alternative is followed by the performance of analyses which will help identify the components with the greatest effects on the selection of the best decision alternative. For that purpose, sensitivity analyses will achieve that goal, and also will serve to evaluate the behavior of the decision model to make sure it performs as expected (Conroy and Peterson 2013). If the confidence in the model is low, then it can be reviewed before

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implementing a decision alternative; if there is agreement on the confidence of the model, the selected management alternative is ready to be implemented. Finally, if outcomes following the decisions are uncertain, the uncertainty may be incorporated via the use of alternative models; then if management decisions reoccur over space or time, model probabilities can be updated. This adaptive feedback provides learning through time and is known as Adaptive Resource Management (ARM) (Walters 1986, Williams et al. 2009, Conroy and Peterson 2013).

CHOOSE YOUR OWN ADVENTURE TOWARDS BIRD OCCUPANCY

Before continuing to the objectives of this dissertation, I wanted to make a "methodological parenthesis" and explain how I will approach the study of birds from a statistical point of view. I believe that in order to provide management recommendations that could ultimately inform conservation decision making, we need data based on sound scientific and statistical principles.

Although birds are usually thought of as being conspicuous group, in any given field monitoring effort there is a high probability of not detecting individuals that are present, and this probability varies among species, habitats, and over time (Kéry and Schmid 2004). Some species are more cryptic than others, or are more visible due to typical behaviors of the species, such as their mobility or their singing; furthermore, the same species may be more conspicuous depending on the season, for example, when performing courtship displays. Other factors that could affect detectability of a species or individual are the behavior associated with weather conditions, habitat, and the same observer knowledge and experience. For these reasons, it is fundamental to account for detection probabilities when conducting a study involving birds; the same statement could be made for other species, although these are not the focus of this dissertation (MacKenzie et al. 2002).

Occupancy estimation accounts for imperfect detection probabilities of a species, so that if a species is not observed at a certain point, it may be either truly absent, or present but not detected (MacKenzie et al. 2002, Tyre et al. 2003, MacKenzie et al. 2006). *Occupancy* is defined as the proportion of sample units that is occupied by a species, or as its expected value, the "probability of occupancy", which is the probability that a randomly selected site in an area of interest is occupied by a species. This method assumes that a location may be occupied or unoccupied, and it will remain at either state during repeated visits (i.e. closure assumption), however, in each visit the species may be detected or undetected. For example, if we visit a site three times, and detect the species only one time, the method assumes that the species was present but undetected in the two visits we did not detect it. To achieve the replication necessary to estimate detection probabilities at each location we may carry out repeated visits to the same location over time, over space, or carry out observations by multiple observers simultaneously. Occupancy estimation can be implemented more easily and less expensively than other methods used to estimate abundance, because it only requires presence/absence information (i.e. detection/ nondetection). For this reason, this method is particularly attractive for large-scale monitoring programs, where other methods for abundance estimation may be impractical.

In occupancy modeling the probability of detection and probability of occurrence are modeled simultaneously or hierarchically, and the incorporation of covariates and missing observations, and other modifications such as multi-species models is also allowed (MacKenzie et al. 2002, Tyre et al. 2003, MacKenzie et al. 2006). In the hierarchical formulation of these models, the data is represented in an explicit and formal way with an 'observation model', produced by the observed data, and a 'process model' which describes the variation in the ecological process and it is unobserved (Royle and Dorazio 2008). Basically, a site may be either

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occupied ($z = 1$) or unoccupied by a species ($z = 0$), following a Bernoulli distribution with probability of occupancy ψ (i.e. Bernoulli ~ (ψ)). If the site is unoccupied $z = 0$, the species cannot be detected. But if it is occupied, then at each repeated survey there is a probability of detecting the species, which means that this probability of detection is conditional to the site being occupied $z = 1$. The detection probability follows another Bernoulli distribution with probability of detection p in every repeated survey at the same location (i.e. Bernoulli ~ (p)). This model is also known as a zero-inflated binomial, which is basically an overdispersed model where there are more zeroes than expected. Model parameters can be obtained using a frequentist or Bayesian approach; the selection will depend on the user's objectives, being aware of the advantages and disadvantages of each, which I will briefly expound upon below.

Link and Barker (2010) and Kéry and Schaub (2012) synthesize the differences between the two approaches. One of the main differences is the way the two frameworks treat the model parameters, and the way they treat probability. While frequentists treat model parameters as fixed and unknown quantities, Bayesians treat them as random variables. Uncertainty for frequentists is evaluated over hypothetical replicates of the data sets, making probability statements about the data given the fixed parameters. Then, confidence intervals may be interpreted based on an infinite number of samples, where 95% of the calculated confidence intervals will contain the true parameters. Conversely, Bayesians do not require hypothetical replicates; probability is the measure of uncertainty about all unknown quantities, making probability statements about those quantities by the use of conditional probabilities. Under this framework, for example, a 95% credible interval is the 95% chance that the true value of the parameter lies in the interval, given the data and the model.

In Bayesian inference, prior knowledge is combined with new data into a model, to obtain posterior knowledge, which are the posterior distributions of the parameters of interest. The incorporation of prior information may be seen as extremely useful, or as a disadvantage, becoming the main critique to Bayesian inference. The use of prior information may be considered as subjective, however, it is has been suggested that it is possible to conduct objective Bayesian analyses by using uninformative or weakly informative priors that do not influence the posterior distribution of the parameters, although this is still being questioned in the literature (Gelman et al. 2004). A Bayesian framework is particularly appropriate for hierarchical models, allowing for the analysis of complex data and models, which is not possible, or difficult to implement, in the frequentist approach. Another advantage is its exactness for any sample size; an asymptotic approximation is not required, becoming also helpful for small data sets. One of the main drawbacks in the Bayesian approach is that the posterior distributions often have to be obtained via MCMC simulation, which can be computationally difficult. In addition, another disadvantage of this approach is the difficulty in performing multimodel inference (i.e. model selection and model averaging); there is still no consensus on the best way to do it, especially in hierarchical models where the number of parameters cannot be calculated. In order to incorporate model uncertainty, instead of conditioning our knowledge with a single model, we can evaluate a candidate set of models and determine which parameters are of interest, and which are unnecessary or nuisance parameters. Under the frequentist paradigm, the use of statistics to evaluate model acceptability, such as Akaike's Information Criterion (AIC) (Burnham and Anderson 2002), is widely accepted, although these have to be carefully used in models where the number of parameters is hard to calculate (e.g. mixed random models), and also assume that the set of models is plausible.

To summarize the main points, the Bayesian approach is particularly helpful to analyze bird occupancy – or any other parameter – if we intend to build a hierarchical model that is not possible to do under the frequentist framework, if we wish to incorporate prior information, or if we have a small sample size. It turns also to be very intuitive and transparent in accounting all sources of variation in hierarchical models; however, it is not always necessary to use this approach. In my opinion, multimodel inference is a very powerful tool to make inferences in ecology – or bird occupancy– and the fact that there is no agreement on how to implement them in a Bayesian approach, makes the frequentist framework still very helpful for the cases when the former is not necessarily required.

CHAPTER DESCRIPTION AND OBJECTIVES

Faced with a global scenario of rapid changes in land use, there is a critical need to reconcile agricultural production with conservation of biodiversity. The objective of this dissertation is to provide sound scientific evidence to fill knowledge gaps to ultimately inform decision making for the conservation and management of birds in agricultural lands in Argentina. I evaluated the effects of land use and land cover at different spatial and temporal scales on birds, and the potential provision of ecosystem services to agriculture, in sectors of the grassland Pampas, and forest Espinal ecoregions in central Argentina.

In Chapter 2, I used information of ten years from a large-scale bird monitoring program, which started in 2003, in the Pampas and portions of Espinal ecoregions in central Argentina which, to the best of our knowledge, is the first long-term regional bird monitoring program in southern Latin America (Zaccagnini et al. 2010, Azpiroz et al. 2012). I used hierarchical Bayesian multi-species dynamic occupancy models to estimate the influence of land use and land

cover, over time, on avian species. Based on the results, I made inferences on potential ecosystem services provision by evaluating the responses of species with similar ecological requirements. I also illustrated the potential application of the models developed in this section to conservation or management of specific species, which may help target future management efforts informing decision making in agricultural landscapes in central Argentina. In this chapter I provide evidence that suggests that many bird species, for example insectivore foliage gleaners, are not strongly affected by land use at a regional scale; in turn, they may perceive the landscape at smaller scales.

In Chapter 3, the objective was to identify the factors that influence the use of soybean fields and vegetated borders by bird species and foraging guilds in Entre Ríos, Argentina. In this chapter, the scale was smaller than in the previous chapter. Here, I accounted for temporal variation in the phenology of the crop, and the vegetation structure in borders; and I also explored the relationship between insectivorous birds and the availability of prey arthropods. By assessing the effects of vegetation structure and invertebrates on birds over time, I also intended to make inferences on management of borders in soybean fields, and the potential indirect effects of herbicides and insecticides on birds. The predictive models and monitoring framework identified here provide scientific information for the posterior elaboration of a conservation decision making framework.

In Chapter 4 I built a structured decision making framework to integrate bird conservation, with producers' values and needs, in Paraná department, in Entre Ríos, Argentina. The ultimate objective was to evaluate the tradeoffs between soybean agriculture, conservation of biodiversity focused on birds, and economic and social well-being of small and medium-scale producers, in order to provide management recommendations. To inform the decision making

process I used the empirical evidence obtained in Chapter 3, together with bibliographic searches, and interviews local farmers. I also used sensitivity analyses to identify key uncertainties, and information needs to be minimized before the optimal management alternatives found in this chapter could be recommended for implementation.

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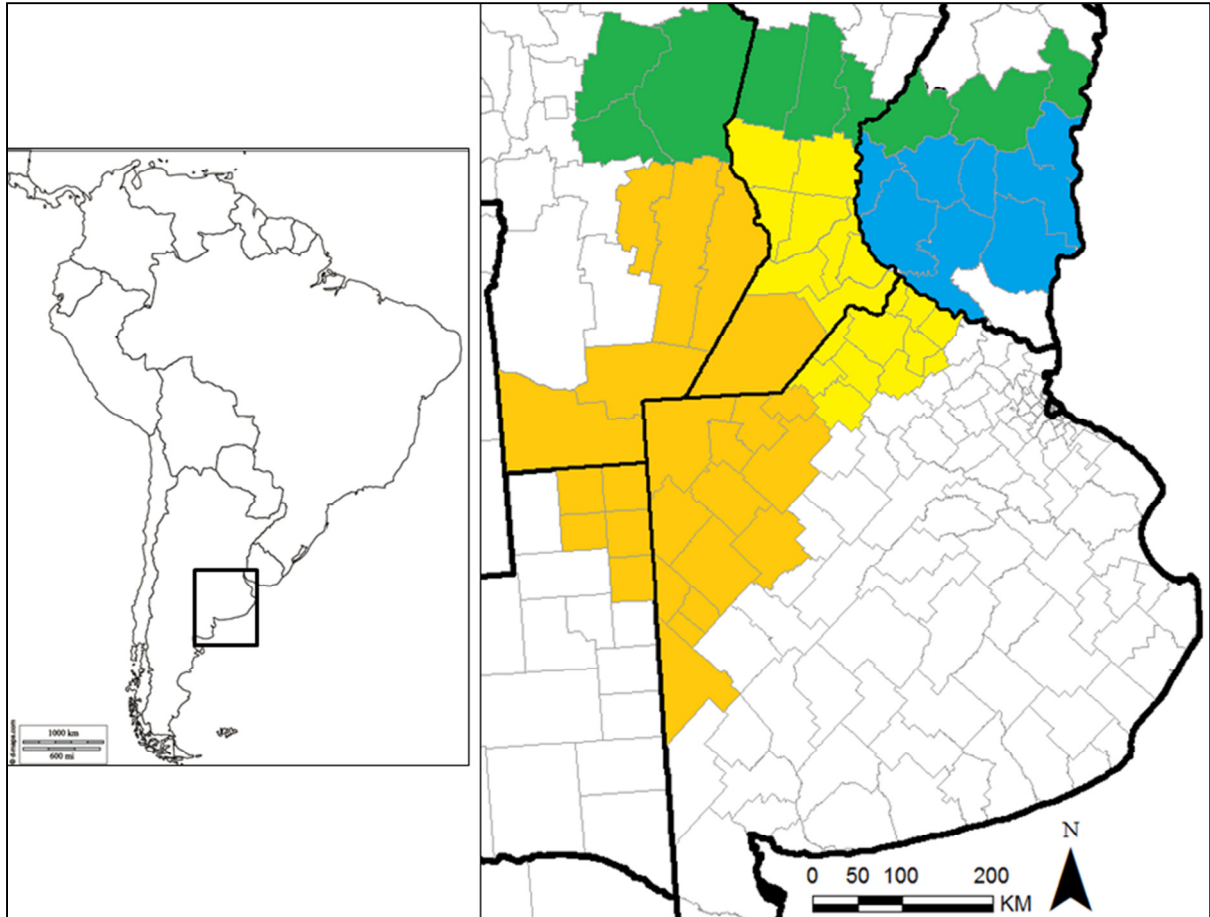
Figure 1.1. Study area by department in Argentina. Black lines delimite the provinces of Entre Ríos, Santa Fe, Córdoba, La Pampa and Buenos Aires (counterclock from the top). The original ecoregions are delineated approximately in the departments where the study was carried out only: Espinal (green), Mesopotamic Pampa (blue), Rolling Pampa (yellow), and Inland Pampa (orange).
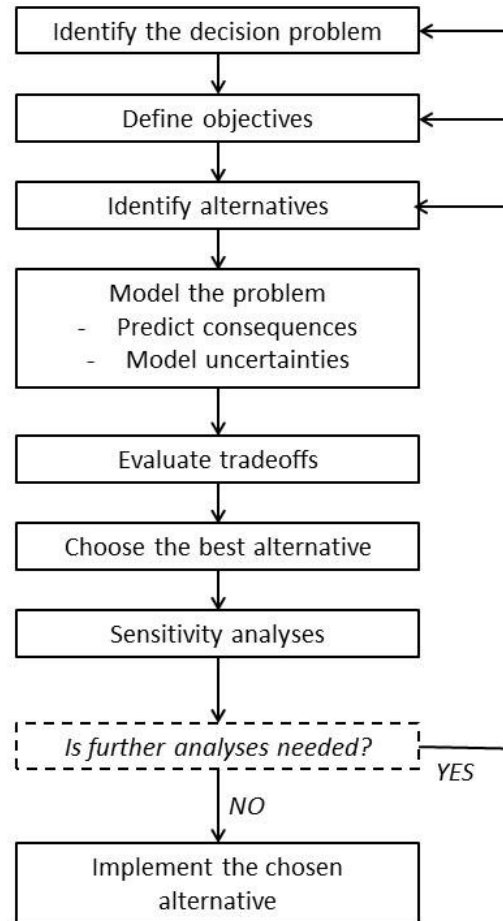
Figure 1.2. Structure decision making cycle. The key elements and steps for the selection of the best decision alternatives are identified to achieve the objectives of a natural resource problem. Adapted from Clemen and Reilly (2001)

CHAPTER 2

REGIONAL ANALYSIS OF MULTI-SPECIES OCCUPANCY DYNAMICS: IMPLICATIONS

FOR MANAGEMENT AND CONSERVATION OF BIRDS IN AGRICULTURAL LANDS IN

EAST-CENTRAL ARGENTINA[1]

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ABSTRACT

Conservation of biodiversity and associated ecosystem services in the face of rapid changes in land use requires well-informed regional solutions. Birds are affected by agricultural intensification, and are useful as indicators of changes in the environment and resource availability. Agricultural intensification has occurred in the Pampas grassland and Espinal forests ecoregions in Argentina, accelerating in recent decades. In 2003, we started a monitoring program in east-central portions of these ecoregions which, to our knowledge, is the first long-term and large-scale bird monitoring program in southern Latin America. We used a hierarchical Bayesian multi-species dynamic occupancy approach to evaluate land use and land cover on multiple species of birds over 2003-2012. Species responded differently to agricultural intensification, with some positively, others negatively affected, and others responding at a local rather than regional scale. A large number of species are distributed across the Mesopotamic Pampas and Espinal, with a gradient of species richness from the wettest and hottest to the driest and coolest areas. Several species associated with human impacted lands evidenced increasing trends, whereas some grassland birds seem to be declining. Insectivorous birds provide ecosystem service of invertebrate pest control throughout the area, but only a few species and guilds perform this function in soybean dominated areas. Our monitoring program provides valuable information to evaluate temporal and regional effects of land transformation on birds, and provides methodologically rigorous evidence to be incorporated into a decision-making framework for conservation or management of species, or group of species, in agroecosystems in central Argentina.

KEYWORDS: Birds, Monitoring, Land use, Ecosystem Services, Espinal, Pampas, Bayesian analysis, Hierarchical modeling,

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INTRODUCTION

An important challenge for conservation into the future is to conserve biodiversity, and the associated provision of ecosystem services, in the face of rapid changes in land use and agricultural intensification generated by growing demands for food supply (Benton et al. 2003, Foley et al. 2005, Foley et al. 2011, Tscharntke et al. 2012). Agricultural landscapes deliver planned services like crop or fuel production, but maintenance of biodiversity also provides associated ecosystem services. To achieve conservation of biodiversity, there is a necessity for well-informed regional solutions (Phalan et al. 2011, Tscharntke et al. 2012). However, the intensification of agriculture alters the suitability and extent of habitats for wild birds, and changes community assemblages (Krebs et al. 1999, Chamberlain et al. 2000, Newton 2004, Reif et al. 2008, Schrag et al. 2009).

Birds are comparatively easy to monitor and quantify, and responsive to environmental perturbations, thus comprising an ideal group to evaluate changes on the environment and resource availability. Because different species have specific habitat requirements, they respond differently to changes, and thus responses to environmental change can be anticipated to be heterogeneous. Some species provide relative ecosystem services to agriculture (e.g. pest control, pollination, seed dispersal) (Sekercioglu et al. 2004, Millennium Ecosystem Assessment 2005, Sekercioglu 2006, Whelan et al. 2008), while others could be considered pests; in either case, these attributes can be used to classify species into functional groups (Blondel 2003). Within these functional groups (i.e. species using the same resource), birds can be identified by guilds, sharing these resources in a similar way, therefore could be equally affected by habitat changes (Blondel 2003, Sekercioglu 2006, Codesido et al. 2008). Classifying birds in guilds could be useful in order to make inferences; however, grouping prior to any analysis could mask

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individual species responses, biasing conclusions (Philpott et al. 2009, Ruiz-Gutierrez et al. 2010, Pacifici et al. In prep.).

In Argentina, the total row crop planted area has rapidly increased in the past 15 years, coinciding with the expansion of soybean cultivation (Aizen et al. 2008, SIIA 2013). The expansion of soybean was facilitated by deforestation and replacement of natural systems, land use intensification, homogenization of landscapes and changes in technologies (Paruelo et al. 2005, Aizen et al. 2008, Baldi and Paruelo 2008). For example, soybean implanted area increased by more than 400% from 1985-2011, making Argentina one of the major soybean exporting countries; although this trend has slowed with <30% increase over the past ten years (Donald 2004, SIIA 2013). Agricultural intensification has been evident in the Pampas grassland and Espinal forests ecoregions, where croplands replaced the original grasslands, forests, and pastures for cattle grazing; and soybean area represents more than 50% of the whole country (Paruelo et al. 2005, Baldi and Paruelo 2008, SIIA 2013). Under this scenario of rapid changes, conservation of biodiversity becomes a main concern.

Long-term bird monitoring data exists in many different developed countries in Europe and North America, and has been used to evaluate trends and detect declines of birds (Schmid et al. 2001, Greenwood 2003, Risely et al. 2011, Sauer et al. 2012). These are usually carried out by volunteers, making them subject to some potential sources of error and variation, including inconsistent methodology, heterogeneity in quality of the observers, and variability in effort. These and other source of variation in turn affect the utility of resulting monitoring data, which must be either be reduced by design, or controlled for by appropriately statistical analysis to avoid erroneous inferences.

In 2003, we started a long-term, large-scale monitoring program (BMA) in the Pampas and a portion of the Espinal ecoregions in central Argentina which, to the best of our knowledge, is the first long-term regional bird monitoring program in southern Latin America (Zaccagnini et al. 2010, Azpiroz et al. 2012). To date, there is detection/non detection information for 263 bird species, and distance measurements for 20 focal species, over the course of 10 years. Unlike other regional bird monitoring programs, BMA is not carried out by volunteers; instead professional birdwatchers are hired to perform this task which follows a pre-established methodology, minimizing biases associated with misidentification of species or survey effort. Nevertheless, we acknowledge that, whereas some biases could be controlled by our design, others could not be. In particular, we recognize that our detection/ non-detection data is an incomplete sample of actual occurrence, with detection probabilities likely variable among species, and over space due to factors beyond the control of our design. We address these remaining issues by means of explanatory variables and statistical models discussed in detail below.

There are several studies reporting the responses of birds to land use in this ecoregion, but there are still gaps of information about the effects of agricultural intensification on native fauna in this area (Codesido et al. 2011, Medan et al. 2011, Azpiroz et al. 2012). Most studies are short term or involve only selected species (Filloy and Bellocq 2007a, Codesido et al. 2008, Schrag et al. 2009, Cerezo et al. 2011, Gavier-Pizarro et al. 2012). To understand the effects of this regional scale process on birds, it is also necessary to include longer term, regional studies. However, many of these studies assume perfect detection of the species of interest (Filloy and Bellocq 2007a, Codesido et al. 2008, Schrag et al. 2009, Cerezo et al. 2011, Codesido et al. 2011). In other studies using BMA data, Gavier-Pizarro et al. (2012) and Calamari et al. (In

prep.) accounted for imperfect detection using Distance Sampling techniques, on a selected group of species. Occupancy (detection/ non-detection) is a data structure that can be readily gathered under appropriate designs over large spatial extents for many species. It requires appropriate statistical models to account for imperfect and heterogeneous detection probabilities (MacKenzie et al. 2002, Tyre et al. 2003, MacKenzie et al. 2006).

In this study, we used dynamic occupancy models to evaluate the effect of land use and land cover extent on multiple species of birds, using the BMA over 10 years (2003-2012). Under this approach, multiple species are linked together within a hierarchical (or multi-level) model, allowing for a more efficient use of data, and increased precision of occupancy estimates (Sauer and Link 2002, Dorazio and Royle 2005, Royle and Dorazio 2008, Kéry et al. 2009, Zipkin et al. 2009). This is the first long-term and large-scale study in Argentina evaluating the impact of agriculture on birds over time, incorporating imperfect detection probabilities using a community hierarchical occupancy approach. By using regional information, we can also incorporate the variability in occupancy given the location, which could relate to factors acting at large scales, such as climate and the original geographic distribution of the species. We also aim to make inferences on potential ecosystem services provisioning by evaluating the responses of species with similar ecological requirements. Ultimately, our results will provide valuable information for decision making and implications for conservation of birds in agricultural landscapes in central Argentina.

METHODS

Study Area

The bird monitoring program in central Argentina (BMA) extends over sections of five provinces (Entre Ríos, Santa Fe, Córdoba La Pampa and Buenos Aires) characterized mostly by the Pampas grassland ecoregion, and a small proportion of Espinal forest ecoregion on the center of Santa Fe province, northeastern Córdoba and northern Entre Rios (Cabrera 1971, Cabrera 1994). It coincides with portions of Mesopotamic (northeast), Rolling (central), and Inland Pampas (western) (Soriano et al. 1992, Codesido et al. 2008, Medan et al. 2011, Azpiroz et al. 2012) between latitudes S37.375-30.783 and longitudes W64.819-58.267. The original vegetation in the Pampas was dominated by grasses, and in the Espinal by xerophytic forest with species like *Prosopis affinis*, *Acacia caven,* and *Geoffroea decorticans* (Cabrera 1971, León 1991); however, currently both regions have been highly modified by agricultural activities. The climate is mild, with a mean annual temperature of $10 - 20 \circ C$ and a mean annual rainfall around 1000 mm, which decreases to the south-west (Soriano et al. 1992, Viglizzo et al. 1997, Baldi and Paruelo 2008, Schrag et al. 2009). Initially, the BMA comprised 113,000 km^2 in 2003-2004, increasing to 150,800 km^2 in 2005, and 255,000 km^2 in 2006-2012 (Zaccagnini et al. 2010, Zaccagnini et al. 2011).

Data Collection

In January each year from 2003 to 2012 (austral bird breeding season) we surveyed 47, 64 and 90 transects (2003-04, 2005, 2006-12, respectively) located along unpaved secondary and tertiary roads (Fig. 2.1). Transect locations were chosen applying a 30 km × 30 km grid over a map of the study area with previously defined strata consisting of agro-production zones and

provincial boundaries. Grid cells were selected systematically alternating every other cell, and a 30 km route was randomly selected, with the starting point for the route within a cell randomly placed as well (Schrag et al. 2009, Zaccagnini et al. 2010, Zaccagnini et al. 2011, Gavier-Pizarro et al. 2012). Each route consisted of 30 points, spaced every 1 km, and at each point experienced surveyors recorded the presence of all bird species seen or heard, during 5 minutes. Surveys were conducted between 0600-1100 and 1500 – 2000, and every point was visited once per year in January, during the 10 years of monitoring.

Land use and cover was recorded within a circle of 200 m radius centered on each point, following bird sampling. We classified land uses in 7 categories: corn, soybean, and other annual crops (sunflower, wheat, sorghum, others), perennial pastures, other pastures (natural grasslands, fallow), native forest, and planted forests (e.g. Eucalyptus). Other uses, such as aquatic, plowed fields or urban were excluded from the analyses because of their low representation in the region, compared to other uses.

Modeling Framework

We used a hierarchical Bayesian multi-species dynamic occupancy model to estimate the influence of land use and land cover, over time, on avian species (Dorazio and Royle 2005, Royle and Dorazio 2008, Kéry et al. 2009, Zipkin et al. 2009). In this context, hierarchical models are valuable because they improve the precision of the estimators by sharing information across species regardless of their relationships, which becomes especially important for those species less frequently detected in the community (Sauer and Link 2002).

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, Tyre et al. 2003, MacKenzie et al. 2006). To achieve replication necessary to estimate detection probabilities at each location, we partitioned each 30 km route in segments (henceforth, 'site') of five points ($k=5$). We chose five spatial replicates within each route because we anticipated that detection probabilities for many species would be low, resulting in imprecise estimates of occupancy for fewer replicates (MacKenzie et al. 2002). We understand that using spatial replicates could potentially violate the closure assumption (i.e. occupancy status of each site is constant across replicates) by confounding temporary absence (availability) with nondetection; for example, if a species home-range is larger than the site, is possible that it was not available for detection at that moment. However, there is disagreement on the extent to which spatial replication introduces bias in occupancy estimation (Kendall and White 2009, Guillera-Arroita 2011). In addition, the use of spatial replicates could introduce bias in the estimates if the replicates were spatially autocorrelated; but not when samples are independent, as we assume in this study given the points were spaced by 1000 m to avoid double counting and dependence (Hines et al. 2010).

Site-occupancy (zero-inflated binomial) models can be formulated as a hierarchical state-space model, which basically links 2 binary regression models: a process model for occupancy of each species, and an observation model for detection conditional on occupancy (Royle and Dorazio 2008, Kery and Schaub 2012). Our process model assumes occupancy as a binary state $z(j,i,t)$ for each species $i = 1,2,…,N$ at site $j = 1,2,...,S$ and year $t = 1,2,…Y$; where $z(j,i,t) =1$ when the species is present and zero otherwise. This is a latent variable, since true occurrence it is imperfectly observed, modeled by a Bernoulli distribution with probability $\psi_{j,i,t}$ that the species i occurs at site j and year t, specified as $z(j,i,t) \sim$ Bern $(\psi_{j,i,t})$. What we observe instead is $y(j,k,i,t)$ at site j, point $k=1,2,…,K,$ for species i at year t. The observation model is also modeled by a

Bernoulli distribution as $y(j,k,i,t) \sim Bern\ (p_{j,k,i,t} \cdot z(j,i,t))$, with $p_{j,k,i,t}$ as the probability that species i at site j is detected at point k on year t, and $y(j,k,i,t)=1$ when the species is detected and zero otherwise. This formulation requires that the event of observing the species is conditional on the species being present (i.e. $z(j,i,t)=1$).

We used a Bayesian approach in the programs R and JAGS, through program R2jags, which uses Markov chain Monte Carlo (MCMC) to find the posterior distribution of the parameters of interest (Su and Yajima 2012).We assumed that occurrence and detection probabilities for each species can be influenced by covariates, and modeled the effects using the logit-link function, where for example $logit(\psi) = log(\psi/1 - \psi)$ (Kéry and Royle 2009, Kéry et al. 2009, Russell et al. 2009, Zipkin et al. 2009, Zipkin et al. 2010). We summed land uses of all points k within each site j and divided by the total proportion of each site j. We discarded correlated covariates (Pearson r >0.5) leaving those which were more represented in the landscape, and modeled occupancy with the following predictors: proportion of soybean, corn, perennial pastures (e.g. alfalfa), and native forests within a 200 m buffer of each point. We dealt with the non-independence between the proportions of land uses (i.e. sum to one) by discarding three out of the seven categories, because they were represented in low percentage in comparison to the others. Similarly, we assumed probability of detecting a species was subject to forest coverage in each point, each year. We also incorporated latitude and longitude of the middle point of each site (centered on zero) as potential factors influencing occurrence of each species in addition to land use and land cover. Since the monitoring program takes place in a large region, large scale drivers as climate, original distribution of the species, or other large-scale spatial patterns on land use could be affecting occupancy (Gavier-Pizarro et al. 2012). For example, Schrag et al. (2009) evidenced a climatic pattern in this portion of Argentina from the northeast

(wettest and hottest) to the southwest (driest and coolest). Our dynamic model assumed species

occurrence to be conditional on temporal covariates only, allowing for differences on occupancy

on different years and subject to land use changes (Royle and Dorazio 2008, Kéry et al. 2009).

We modified the model proposed by Kéry et al. (2009) to incorporate random time effects on the

baseline (i.e. intercept) occupancy and detection for each species on each site, as a means to

control for potential sources of variation on different years (e.g. climate, observers). Our global

occupancy model was:

$$\text{logit } \psi(j,i,t) = u(i,t) + a1(i) * lat(j,t) + a2(i) * long(j,t) + a3(i) * soy(j,t) + a4(i) * corn(j,t)$$
$$+ a5(i) * per\_past(j,t) + a6(i) * forest(j,t)$$

where the parameters denoting covariates effects *a1* through *a6* for each species *i*=1,2,…,*N* are

estimated; *u* is estimated for each species *i* and year *t*=1,2,…*Y* combination; and location (i.e.

latitude, longitude at the middle point of each site *j*), and land use/cover corresponds to each site

j=1,2,…,*S* and year *t*. We based our focus on estimation and prediction of the effects of

covariates on each species, and in this context, we believe the best approach was to construct a

global model and make the inferences based on 95% credible intervals (95%CRI), since different

species could respond differently to covariates, and average out the effects of covariates.

Similarly, we constructed the observation model as follows:

$$\text{logit } p(j,k,i,t) = v(i,t) + b1(i) * p\_fors(j,k,t)$$

where we estimated the effect of proportion of forest on detection *b1* for each species *i; v,* is

estimated for each species *i* and year *t*; and the proportion of forest corresponds to each site *j*,

year *t*, and point *k*.

The hierarchical multi-species occupancy approach allows the incorporation of a

community hierarchical component ('hyper-parameters') in the model, where the species-level

parameters are treated as random effects. In this way, each species can be analyzed

simultaneously rather than in separate models improving precision of the estimates (Sauer and

Link 2002). Unlike most aforementioned applications of multi-species site occupancy models

(Dorazio and Royle 2005, Kéry and Royle 2009, Kéry et al. 2009, Zipkin et al. 2009, Zipkin et

al. 2010), we did not incorporate unobserved species in the community. In this study, we were

only interested on the relationship of the observed species with land use and cover, and the

potential of suggesting management actions based on the observed species, without making

inferences about the 'true' community. We discarded species associated with aquatic habitats,

since our interest was on terrestrial birds; and we discarded species with less than 200

observations in the 10 year period, because inclusion of these sparsely-observed species created

computational difficulties and added little to model performance. Inclusion of species with <200

detections resulted in slow mixing, and some cases failure to converge even after 50,000

iterations ($\hat{R}$ diagnostics, see below). In addition, predicted occupancy and detection parameters

for rare or hard-to-observe species tend towards the means of the parameters of the entire

community, adding no information about the effects of covariates we were interested in this

study.

To increase computation efficiency, we classified the remaining species in six groups

with similar ecological requirements and taxonomic groups (see below) and performed separate

analyses for each group. We assumed species parameter estimates could be more closely related

between those species within a group, but still allowing for individual species effects. For

example, we assumed that *u(i,t)~N(μ.u (i), σ.u(i))* followed a normal distribution where *μ.u (i)*

and *σ.u(i)* are the mean and standard deviation across time, and similarly *μ.u (i) ~N(mu.μ.u ,*

sigma.σ.u) the 'hyper-parameters' across the group of species.

We used independent flat (uninformative) priors for the group level hyper-parameters (see Appendix A for the R and JAGS model code). We ran three chains of length 30,000 after a burn-in of 20,000, and thinned the posterior chains by 10 for economy of memory space and reduce autocorrelation to render 3000 iterations (9000 total for each parameter). We monitored convergence 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 1 (Gelman and Rubin 1992). We also assessed model fit using a Bayesian p-value, which estimates the probability that the simulated data could be more extreme than the observed data (Gelman et al. 2004).

RESULTS

We observed 263 species in the study area over the 10 years of monitoring, of which 205 are considered landbirds. We had sufficient data (>200 observations) to estimate occupancy of 74 species of landbirds We classified these as raptors, ground omnivores and herbivores, ground granivores, other granivores, insectivores associated with foliage and other insectivores. Within the species groups, we further identified 12 guilds according to their habitat preferences and foraging behavior (Table 2.1).

Soybean was distributed in the central portion of the study area, coinciding with the rolling Pampas, although it increased in the eastern section (i.e. mesopotamic Pampas and Espinal; Appendix B). Native forests were more abundant in the Espinal area, and perennial pastures were distributed towards the north. Corn was patchy throughout the region, and increased over time. We discarded other annual crops (sunflower, wheat, sorghum, others), and other pastures (natural grasslands, fallow), from the analyses since acreages of these were highly

and negatively correlated to soybean acreage. We also discarded planted forests (e.g. Eucalyptus) because we found inconsistencies in the criteria to assign percent land use by observers from different years, especially in the presence of windbreaks.

Detection probabilities

Mean detection probabilities were low in general, with most species < 0.5, and the response to percentage of forest cover was variable by species (Fig. 2.2, Appendix C). The species with higher probabilities of detection were the Eared Dove around 0.8, followed by Chimango Caracara, Southern Lapwing, and Rufous-collared Sparrow decreasing with forest cover; and Picui Ground Dove, Monk Parakeet, White-tipped Plantcutter, and Rufous Hornero. The probability of detecting the last three species increased rapidly with forest cover, a pattern observed for several other species; this response could be due to heterogeneity in abundance of the species, where the occurrence and detection probabilities of the species are likely to be correlated (Dorazio and Royle 2005, Zipkin et al. 2009). Finally, the incorporation of time as a random effect in the detection probabilities successfully absorbed variability brought by factors not controlled by the incorporation of forest cover in the detection of the species (Appendix C). For example, for unknown reasons, several species experience a notably lower detection in 2007.

Bird trends over time

We examined mean occupancy response of each species over time at the whole study area for the period 2003-2012, and found no clear pattern, where most species remain constant, and there is a great amount of variability, since we are pooling together the sites over a larger area (Appendix D). This lack of evidence of a clear tendency of the occupancy dynamics for most species is

consistent with the almost stable proportion of soybean, corn, native forests, and perennial pastures recorded at each point over the years in the study area (Fig. 2.3). There is a small decrease in the proportion of soybean over the years, although it is highly variable. Although we cannot relate occupancy trends with land uses specifically, the incorporation of time as a random effect on the intercept of the occupancy models, accounted for variation not explained by land use. Some species such as Monk Parakeet, Eared Dove, Grassland and House Sparrows, Screaming Cowbird, Bay-winged Cowbird, and Green-barred Woodpecker seem to increase over time (Fig. 2.4). Others, such as Dark-throated Seedeater, White-browed Blackbird, and Red-winged Tinamou, despite the variation, appear to be declining.

Occupancy at a regional scale

The effect of latitude and longitude on the occupancy of birds demonstrates that species occur along a geographic gradient in the study area (Fig. 2.5.a- f). When looking at the mean group responses, some of the 95% credible intervals of the coefficients of the latitude and longitude effects overlap zero; however, the granivores and insectivores mostly associated with foliage have strong positive latitude effects, which indicates there are higher occupancy probabilities to the north. It is still important to evaluate the effects at a species level since each has its own spatial distribution, which could be determined by different large scale drivers.

Species showing both negative effects of latitude and longitude, have higher occupancy probabilities to the southwest (i.e. inland Pampas), like Chimango Caracara; Grassland Yellow Finch, a granivore, and an aerial forager, the Forked-tailed flycatcher. Fewer species had higher occupancy probabilities to the southeast, like Roadside Hawk, Red-winged Tinamou, Spot-winged Pigeon, Hooded Siskin, and Tropical Kingbird, all from different guilds. Species with

higher occupancy probabilities to the northwest are such as Burrowing owl, Upland Sandpiper, White-tipped Plantcutter; omnivores such as White-browed Blackbird, Dark-billed-cuckoo; and insectivores from different guilds like Pale-breasted Spinetail, Firewood-gatherer, Cliff-sparrow, Great Kiskadee and Brown Cacholote. No raptors or omnivores had higher occupancy probabilities to the northeast, but species mostly associated with forests were distributed over this region; for example, the ground granivores such as Picui Ground Dove and White-tipped Dove; and half of the granivore foliage gleaners. Also in this region, insectivores from different guilds, but especially more than half of the insectivorous gleaners had higher occupancy probabilities to the northeast.

Land use and land cover effects on occupancy

In general, for all groups, species responded more strongly to the effects of the proportions of area in soybean and native forest, and less to the proportions of area in maize and perennial pastures (Fig. 2.6). Most raptor species were positively affected by soybean extent, and negatively to native forests, although the latter was a more variable response (Fig. 2.6.a). Although Swainson's Hawk and Southern Crested Caracara experienced negative effects of soybean, no raptor species responded positively to forests. Swainson's Hawk and Chimango Caracara were also negatively affected by corn; and the latter also negatively affected by perennial pastures. Finally, Roadside Hawk, Swainson's Hawk and Burrowing owl responded negatively to native forests.

All ground omnivore species showed negative responses to the proportion of soybean area, and all except Red-winged Tinamou also responded negatively to native forests and corn (Fig. 2.6.b). Only Spotted Nothura responded positively to perennial pastures. All Columbidae,

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responded positively to native forests, except Rock Dove and Picazuro Pigeon; the latter was the only species which was positively affected by soybean and corn (Fig. 2.6.c). Spot-winged pigeon, Picui Ground Dove responded negatively to soybean and corn. Finally, Monk Parakeet was strongly and positively affected by native forests and perennial pastures, but negatively by soybean and corn.

Granivore foliage gleaners did not have a single response to the extent of soybean crop; six species responded positively, and six negatively (Fig. 2.6.d). Three of the species positively associated with soybean are negatively affected by native forests. Most other species were positively affected by native forest extent. Only two species were positively associated with corn, and only five species were favored by perennial pastures. As a group, all insectivores mostly associated with foliage showed a positive response to native forests, as expected (Fig. 2.6.e). Most of the insectivore foliage gleaners were positively associated with native forests, and only three responded positively to soybean and perennial pastures. The common group response for the rest of the insectivores was negative for soybean and corn percent area (Fig. 2.6.f). However, no common response was found for insectivorous salliers; only one species favored by soybean, and only one by native forests, while three species were negatively associated with soybean (Fig. 2.6.f). There was variable responses for aerial foragers as well, while most ground insectivores where negatively affected by soybean. Only one species of each guild (ground, aerial, and salliers) were positively associated to forests.

Relevance to conservation decision making

In order to illustrate the potential application of our models to conservation, we detail the spatial response over time for one species of potential conservation concern (Vermilion flycatcher) and

another for potential pest management concerns (Picazuro pigeon) by evaluating their spatial

distribution of probability of occupancy over the region in relation to land use. The Picazuro

pigeon is positively affected by soybean and negatively by native forests; and the Vermilion

flycatcher is negatively affected by soybean. Occupancy probabilities of the Picazuro pigeon

increase especially over the north and west of its distribution over time, and seem to be evenly

distributed with high occupancy over the entire area (Fig. 2.7). Conversely, the Vermilion

flycatcher has a patchy distribution, with higher occupancy probabilities on the north and south

west of the study area. The response of this species over time does not seem to follow a clear

trend, with some areas increasing and other decreasing in different years (Fig. 2.8).

DISCUSSION

The effects on birds of different land uses and land conversion (Filloy and Bellocq 2007a,

Codesido et al. 2008, Schrag et al. 2009, Gavier-Pizarro et al. 2012), landscape structure (Cerezo

et al. 2011), and even large scale drivers (Schrag et al. 2009, Gavier-Pizarro et al. 2012), have

been well documented in the Pampas and Espinal ecoregions in Argentina, although long term

studies are lacking (Azpiroz et al. 2012). Our results generally agree with those of previous

studies, which have documented on the effects of agriculture in individual species. Overall, bird

responses to land use are difficult to generalize because each species responds differently, where

some species seem to be adapted or be tolerant to agricultural intensification, others are

negatively affected. In addition, some species seem to be responding at larger or smaller scales

not captured in our analyses. For example, by grouping land uses of five points into a single site

the differences of land use, and bird use of the habitat are not detected. And last, the greatest

number of species is distributed to the northeast (Mesopotamic Pampas and Espinal). However,

this is the first study in South America to evaluate trends and regional responses for more than 70 species of birds from a long-term large scale monitoring program, with a focus in potential ecosystem services provision. Our study uses current statistical approaches for modeling bird communities studies, explicitly modeling hierarchical community relationships models (Dorazio and Royle 2005, Kéry et al. 2009, Zipkin et al. 2009), accounting for detectability (MacKenzie et al. 2002, Tyre et al. 2003), avoiding *a priori* grouping of guilds when possible (Philpott et al. 2009, Ruiz-Gutierrez et al. 2010, Pacifici et al. In prep.); and presenting an application of the utility of occupancy multi-species.

The dynamic occupancy multi-species models allowed us to evaluate the effects of land use and land cover extent and geographic location, using information from BMA over 10 years. Our results support the importance of accounting for individual species detection probabilities, and occupancy responses, even for species with similar ecological requirements (i.e. guilds). We were able to account for individual responses, yet use our data in an efficient way by linking species' occupancy and detection probabilities in a hierarchical fashion, allowing for inferences on species which we would have not been able to model with single species approaches (Sauer and Link 2002, Dorazio and Royle 2005, Ruiz-Gutierrez et al. 2010, Zipkin et al. 2010). Most species evidenced low detection probabilities, and thus making inferences assuming perfect detection could lead to erroneous management decisions when addressing conservation of a single rare species or even a community assemblage (Gu and Swihart 2004, Kéry and Schmid 2004, Zipkin et al. 2010). We did not incorporate rare species since their responses would be drawn towards the group means; even if their incorporation is desired when estimating the size of the community, it was not the scope of our study (Dorazio and Royle 2005, Kéry and Royle 2009, Zipkin et al. 2009). Nevertheless, we believe that the results obtained from this study will

reveal valuable information that can be used for management and conservation purposes in agricultural lands, and could also translate into benefits to unaccounted species.

We could not relate community changes over time directly to land use, because the proportion of the land uses evaluated were mostly stable over the studied period; however, the observed trends can still be related to other factors such as agrochemicals, or the high breeding success of certain species, among others (Murton et al. 1974, Mineau and Whiteside 2013). A limitation of this study relies on land use observations subject to a 200 m radius in January only, thus land uses might not be well represented over the entire area. Nevertheless, official statistics for the period 2003-2012, report an increase in soybean and corn cultivated area only in some areas (SIIA 2013), although soybean cultivation is increasing over the entire country (Aizen et al. 2008, Baldi and Paruelo 2008). Despite some species that remained stable during the studied period could be adapting to agricultural landscapes, we suspect that the period of our study may not be long enough to document a clear pattern in the dynamics of all species, and thus document the potential time lag in their responses to habitat transformation (Chamberlain et al. 2000, Siriwardena et al. 2000, Gavier-Pizarro et al. 2012).

Several species common in the Pampas and associated with annual crops, pastures, scrublands or anthropogenic habitats, evidenced increasing trends (Leveau and Leveau 2005, Filloy and Bellocq 2007a, Schrag et al. 2009, Cerezo et al. 2011, Codesido et al. 2011). These results agree with several studies in Europe which found that some species may benefit from agriculture (Siriwardena et al. 2000, Robinson et al. 2001, Donald et al. 2006). Other species such as some grassland birds, seem to be declining; this phenomenon has also been well documented in Europe (Krebs et al. 1999, Siriwardena et al. 2000, Newton 2004, Donald et al.

2006), North America (Murphy 2003, Brennan and Kuvlesky 2005), and in southern South America (Codesido et al. 2011, Azpiroz et al. 2012).

A large number of species, many granivore and insectivore foliage gleaners, were concentrated in the mesopotamic Pampa and the Espinal ecoregions, where the greater proportion of native forest remnants and there is still intermediate levels of fragmentation (Baldi and Paruelo 2008, Azpiroz et al. 2012). This coincides with a gradient of species richness from the wettest and hottest areas (northeast) to the driest and coolest areas (southwest) in this portion of Argentina (Schrag et al. 2009). Species distributed throughout the region coincide with those associated with agricultural landscapes or anthropogenic environments (Filloy and Bellocq 2007a, Codesido et al. 2008, Schrag et al. 2009). Among those, most raptors show a positive association to soybean, as opposed to other findings (Filloy and Bellocq 2007b), where they still could find their prey items such as rodents and large insects, and do not have many competitors (Hector 1985, Medan et al. 2011, Fischer et al. 2012). Others, such as some ground or foliage omnivores, and most ground insectivores, were negatively affected by soybean and corn, but still widely distributed, indicating that they are finding other suitable habitats within the agricultural landscapes which allows them to disperse in the agricultural matrix; and also their possible association to other crops, natural grasslands or fallows. Some granivorous gleaners, coinciding with those described by Codesido et al. (2008) as border granivores in agroecosystems, evidenced tolerance to soybean. Most insectivore foliage gleaners seemed unaffected by crops, some of those are associated with native forests in the original Espinal ecoregion (Narosky and Yzurieta 2010).

The effects of land use on birds, and their population trends in agricultural landscapes, do not stem from a single explanation; usually a combination of factors affect each species in their

own way. For example, conversion of lands to agriculture, agriculture intensificarion via technological developments (e.g. intensified agrochemicals use), afforestation, deforestation, fragmentation and habitat deterioration, are some of the elements affecting birds negatively (Chamberlain et al. 2000, Siriwardena et al. 2000, Newton 2004, Brennan and Kuvlesky 2005, Donald et al. 2006, Medan et al. 2011, Azpiroz et al. 2012). Recently, Mineau and Whiteside (2013) found that pesticide toxicity to birds is a good correlate to explain grassland bird declines in the U.S. Some species respond to large scale drivers such as climate or their original distributions (Schrag et al. 2009); while others might be affected by habitat configuration, or smaller scales. Those that perceive the landscape at finer scales (e.g. gleaners) could benefit from local spatial heterogeneity within agricultural landscapes, directly or indirectly through different landscape processes (Peterson et al. 1998, Robinson et al. 2001), or persist in small patches of vegetation in the agricultural landscapes (Dardanelli et al. 2006) and linear habitats such as field borders or other features with natural vegetation (Jobin et al. 2001, Benton et al. 2003, Goijman and Zaccagnini 2008, Di Giacomo and de Casenave 2010).

The diet composition of several of the studied species has been previously documented, allowing us to assert the current consumption of invertebrate or vertebrate pest, carrion and seed dispersal by them (Canavelli et al. 2001, Rojas and Stappung 2004, Alessio et al. 2005, Biondi et al. 2005). However, we recognize that there are gaps in information about the real impact such consumption has as ecosystem service to agriculture in Argentina, although the same was studied to some extent in other regions (Sekercioglu 2006, Whelan et al. 2008, Capinera 2010). The main ecosystem services potentially provided by the subset of species in this study were seed dispersal, weed control, invertebrate and vertebrate pest control (which can be also generated by fear), carcass and waste disposal (Sekercioglu 2006, Whelan et al. 2008). Invertebrate pest

control was the most represented service accounting for 81% of the species, composed of some raptors, omnivores, all insectivores, and granivore foliage gleaners which consume invertebrates in their breeding season (Alessio et al. 2005). The next largely represented ecosystem services were seed dispersal and weed control, composed by granivore and omnivore guilds, with 46% of the species.

According to our results, the previously mentioned ecosystem services are distributed throughout the study area. Because many raptors seem unaffected by agriculture intensification, at least to a certain point, the ecosystem services they provide (i.e. pest control and carcass removal) are widely distributed; which might not be the case for the other services. For example, some species of granivorous gleaners consuming insects during their breeding season are positively associated with this crop, as well as other species such as the Burrowing Owl or Campo Flicker, while some insectivorous gleaners are unaffected and also potentially preying on pests. However, the diversity of birds associated or not affected by this crop is low, missing large number of representatives of other guilds that could perform pest control. So, although the ecosystem service of invertebrate pest control is present in the soybean crop in the Pampas and Espinal ecoregions, it is likely that only a few species and guilds might be responsible for it; and the same phenomenon could apply to seed dispersal and weed control services.

The conservation of bird diversity in agricultural landscapes is essential for its intrinsic value and because protected areas alone are insufficient, and also to maintain the diversity of guilds, and thus ecosystem functions and resilience (Perfecto et al. 2004, Sekercioglu et al. 2004, Fischer et al. 2006). Diversity of guilds is fundamental since they can share resources but exploit them in different ways, and for example, prey on different pests; but also diversity of species is fundamental given the variability of responses associated with their particular traits (Philpott et

al. 2009). In addition, to achieve a significant quantity on the service provided a numerical response is fundamental; for example, a pest controller should affect pest populations sufficiently to have a positive impact on the resource consumed by the pest (sensu Whelan et al. 2008); and this can be achieved by either maintaining species richness or abundance, although some studies evidenced that the former is more important (Perfecto et al. 2004, Sekercioglu 2006, Van Bael et al. 2008, Whelan et al. 2008).

Implications for Conservation Decision Making

The BMA program provides valuable information not only to evaluate temporal and regional effects of land transformation on birds, but also provides methodologically rigorous evidence, incorporating species-specific detection probabilities, to be incorporated into a decision making framework for conservation or management problems of species, or group of species, in agricultural landscapes in central Argentina. We illustrated how the temporal and spatial distribution of individual species can be mapped, and thus used to inform decision making on conservation of species of interest, a group of species, or management of pest species. Examples of a potential application of multi-species models in structured decision making can be found in Sauer et al. (2013), and other bird occupancy models to make management decisions at smaller scale can be found in Chapter 5.

Some examples that emerge from our results and that could be incorporated into management plans are the cases of two species considered pests, the Eared Dove and the Monk Parakeet, which show an apparent population growth and high occupancies (Murton et al. 1974, Bruggers et al. 1998, Canavelli et al. 2013). The Eared Dove is widely distributed with a tendency towards the north, and it is unaffected by soybean or corn, but positively associated

with native forests. This results agree with other studies showing a weak relationship of occupancy by this species and land use, and its association with agricultural lands with forest patches (Bucher 1990, Gavier-Pizarro et al. 2012), but contrary to those that associated this species negatively with agriculture (Filloy and Bellocq 2007a). The Monk Parakeet is distributed towards the mesopotamic Pampas and Espinal ecoregions, and in our study is associated with native forests but negatively to corn and soybean crops. This result is not surprising, since this species is considered a stronger pest in sunflower crop, although also in corn, and originally nests in natural forests (Bruggers et al. 1998).

Conversely, we also found evidence that raises awareness of some species of conservation concern. The Dark-throated Seedeater, the Red-winged Tinamou, and Swainson's Hawk had low occupancies and the first two could be declining. Besides, although they were not rare in the BMA region, all were negatively affected by soybean crop. The Red-winged Tinamou is globally listed as of least concern but might be experiencing a population decline (Azpiroz et al. 2012, BirdLife International 2013); and the Dark-throated Seedeater is considered a near threatened species, which is experiencing declines because of habitat loss and wild bird trade (Azpiroz et al. 2012, BirdLife International 2013). Finally Swaison's Hawk had suffered massive mortalities in the 1990s throughout the Pampas associated with insecticide poisoning and its foraging on grasshoppers and caterpillars (Goldstein et al. 1999, Canavelli et al. 2001).

Although the effects of land use on birds in Pampa and Espinal ecoregions in Argentina has been studied in the past, this is the first long-term study of these characteristics, evaluating many species (Medan et al. 2011, Azpiroz et al. 2012). In addition, although we already count with long term data, we believe that the continuation of the BMA is essential to explore even longer trends that can capture potential time lags in the response to agriculturization by birds. We

hope that this study will contribute to knowledge gaps, and encourage the implementation of conservation and management actions in Argentina.

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Table 2.1. Avian species observed in the regional monitoring program in Pampas and Espinal regions, in Argentina from 2003-2012 (registered >200 times). Groups for analyses purposes and guilds are indicated: raptors (RAP), ground omnivores and herbivores (OMN), ground granivores (GRA2), other granivores (GRA), insectivores mostly associated with folliage (INS1) and other insectivores (INS2) (Remsen and Scott 1990, Azpiroz 2003, Beltzer 2003)

Group (guild)[1]	Code	Family	Scientific name	Common name
RAP (1)	BUMA	Accipitridae	*Buteo magnirostris*	Roadside Hawk
RAP (1)	BUSW	Accipitridae	*Buteo swainsoni*	Swainson's Hawk
RAP (1)	ELLE	Accipitridae	*Elanus leucurus*	White-tailed Kite
RAP (1)	ROSO	Accipitridae	*Rostrhamus sociabilis*	Snail Kite
RAP (1)	FAFE	Falconidae	*Falco femoralis*	Aplomado Falcon
RAP (1)	FASP	Falconidae	*Falco sparverius*	American Kestrel
RAP (1)	MICH	Falconidae	*Milvago chimango*	Chimango Caracara
RAP (1)	POPL	Falconidae	*Caracara plancus*	Southern Crested Caracara
RAP (2)	SPCU	Strigidae	*Athene cunicularia*	Burrowing Owl
OMN (3)	CHTO	Anhimidae	*Chauna torquata*	Southern Screamer
OMN (3)	VACH	Charadriidae	*Vanellus chilensis*	Southern Lapwing
OMN (3)	BALO	Scolopacidae	*Bartramia longicauda*	Upland Sandpiper
OMN (3)	RHRU	Tinamidae	*Rhynchotus rufescens*	Red-winged Tinamou
OMN (3)	NOMA	Tinamidae	*Nothura maculosa*	Spotted Nothura
GRA2 (4)	COLI	Columbidae	*Columba livia*	Rock Dove
GRA2 (4)	COMA	Columbidae	*Columba maculosa*	Spot-winged Pigeon
GRA2 (4)	COPZ	Columbidae	*Columba picazuro*	Picazuro Pigeon
GRA2 (4)	COPI	Columbidae	*Columbina picui*	Picui Ground Dove
GRA2 (4)	LEVE	Columbidae	*Leptotila verreauxi*	White-tipped Dove
GRA2 (4)	ZEAU	Columbidae	*Zenaida auriculata*	Eared Dove
GRA2 (5)	MYMO	Psittacidae	*Myiopsitta monachus*	Monk Parakeet
GRA (5)	SAAU	Cardinalidae	*Saltator aurantiirostris*	Golden-billed Saltator
GRA (5)	SACO	Cardinalidae	*Saltator coerulescens*	Greyish Saltator
GRA (5)	PHRU	Cotingidae	*Phytotoma rutila*	White-tipped Plantcutter
GRA (5)	AMHU	Emberizidae	*Ammodramus humeralis*	Grassland Sparrow
GRA (5)	EMPL	Emberizidae	*Embernagra platensis*	Pampa Finch
GRA (5)	PACO	Emberizidae	*Paroaria coronata*	Red-crested Cardinal
GRA (5)	POME	Emberizidae	*Poospiza melanoleuca*	Black-capped Warbling Finch
GRA (5)	PONI	Emberizidae	*Poospiza nigrorufa*	Black-and-rufousWarbling Finch

GRA (5)	SIFL	Emberizidae	*Sicalis flaveola*	Saffron Finch
GRA (5)	SILU	Emberizidae	*Sicalis luteola*	Grassland Yellow Finch
GRA (5)	VOJA	Emberizidae	*Volatinia jacarina*	Blue-black Grassquit
GRA (5)	ZOCA	Emberizidae	*Zonotrichia capensis*	Rufous-collared Sparrow
GRA (5)	CAMA	Fringillidae	*Carduelis magellanica*	Hooded Siskin
GRA (5,9)	MOBA	Icteridae	*Agelaioides badius*	Bay-winged Cowbird
GRA (5,9)	STSU	Icteridae	*Sturnella supercilliaris*	White-browed Blackbird
GRA (5,9)	MOBO	Icteridae	*Molothrus bonariensis*	Shiny Cowbird
GRA (5,9)	MORU	Icteridae	*Molothrus rufoaxillaris*	Screaming Cowbird
GRA (5)	PADO	Passeridae	*Passer domesticus*	House Sparrow
GRA (5)	SPCA	Thraupidae	*Sporophila caerulecens*	Double-collared Seedeater
GRA (5)	SPRU	Thraupidae	*Sporophila ruficollis*	Dark-throated Seedeater
INS1 (6)	PHST	Furnariidae	*Phacellodomus striaticollis*	Freckle-breasted Thornbird
INS1 (6)	SCPH	Furnariidae	*Schoeniophylax phryganophila*	Chotoy Spinetail
INS1 (6)	SYAL	Furnariidae	*Synallaxis albescens*	Pale-breasted Spinetail
INS1 (6)	SYFR	Furnariidae	*Synallaxis frontalis*	Sooty-fronted Spinetail
INS1 (6)	GEAE	Parulidae	*Geothlypis aequinoctialis*	Masked Yellowthroat
INS1 (6)	PODU	Polioptilidae	*Polioptila dumicola*	Masked Gnatcatcher
INS1 (6)	TAMA	Thamnophilidae	*Taraba major*	Great Antshrike
INS1 (6)	TRAE	Troglodytidae	*Troglodytes aedon*	House Wren
INS1 (6)	SESU	Tyrannidae	*Serpophaga subcristata*	White-crested Tyrannulet
INS1 (7)	CHAU	Trochilidae	*Chlorostilbon aureoventris*	Glittering-bellied Emerald
INS1 (8)	LEAN	Dendrocolaptidae	*Lepidocolaptes angustirostris*	Narrow-billed Woodcreeper
INS1 (8)	COME	Picidae	*Colaptes melanochloros*	Green-barred Woodpecker
INS1 (6)	COLA	Cuculidae	*Coccyzus melacoryphus*	Dark-billed Cuckoo
INS1 (9)	GUGU	Cuculidae	*Guira guira*	Guira Cuckoo
INS1 (6)	TANA	Cuculidae	*Tapera naevia*	Striped Cuckoo
INS1 (9)	MISA	Mimidae	*Mimus saturninus*	Chalk-browed Mockingbird
INS1 (9)	TURU	Turdidae	*Turdus rufiventris*	Rufous-bellied Thrush
INS2 (10)	DRBR	Dendrocolaptidae	*Drymornis bridgesii*	Scimitar-billed Woodcreeper
INS2 (10)	ANAN	Furnariidae	*Anumbius annumbi*	Firewood-gatherer
INS2 (10)	FURU	Furnariidae	*Furnarius rufus*	Rufous Hornero
INS2 (10)	PSLO	Furnariidae	*Pseudoseisura lophotes*	Brown Cacholote
INS2 (10)	ANCH	Motacillidae	*Anthus lutescens*	Yellowish Pipit
INS2 (10)	HYPE	Tyrannidae	*Hymenops perspicillata*	Spectacled Tyrant
INS2 (10)	MARI	Tyrannidae	*Machetornis rixosus*	Cattle Tyrant
INS2 (11)	PISU	Tyrannidae	*Pitangus sulphuratus*	Great Kiskadee
INS2 (11)	PYRU	Tyrannidae	*Pyrocephalus rubinus*	Vermilion Flycatcher
INS2 (11)	TYME	Tyrannidae	*Tyrannus melancholicus*	Tropical Kingbird

INS2 (11)	TYSA	Tyrannidae	*Tyrannus savanna*	Fork-tailed Flycatcher
INS2 (11)	XOIR	Tyrannidae	*Xolmis irupero*	White Monjita
INS2 (12)	PEPY	Hirundinidae	*Petrochelidon pyrrhonota*	Cliff Swallow
INS2 (12)	PHTA	Hirundinidae	*Phaeoprogne tapera*	Brown-chested Martin
INS2 (12)	TALE	Hirundinidae	*Tachycineta leucorrhoa*	White-rumped Swallow
INS2 (8)	COCA	Picidae	*Colaptes campestris*	Campo Flicker

Guilds: 1= Diurnal raptors; 2= Nocturnal raptors; 3= Ground omnivores; 4= Ground granivores; 5= Granivore foliage gleaners; 6= Insectivorous foliage gleaners; 7= Insectivorous/nectarivorous; 8= Bark insectivores; 9= Ground and foliage omnivores; 10= Ground insectivores; 11 = Insectivorous salliers; 12 = Insectivorous Aerial foragers.
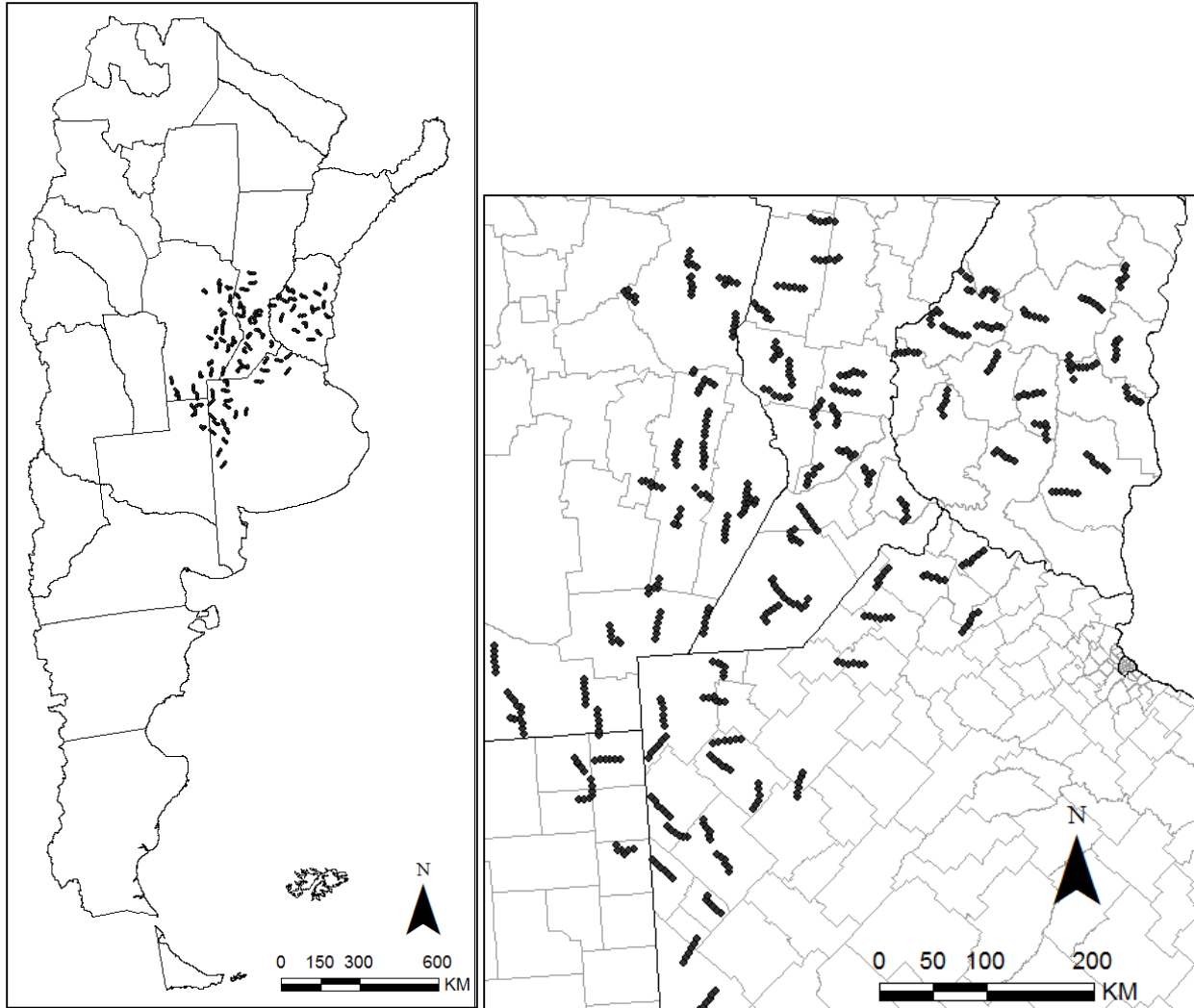
Figure 2.1. Regional bird monitoring program, indicating monitored routes, as of 2006-2012, covering an area 255,000 km^2, over parts of Entre Ríos, Santa Fe, Córdoba, La Pampa and Buenos Aires provinces, Argentina.
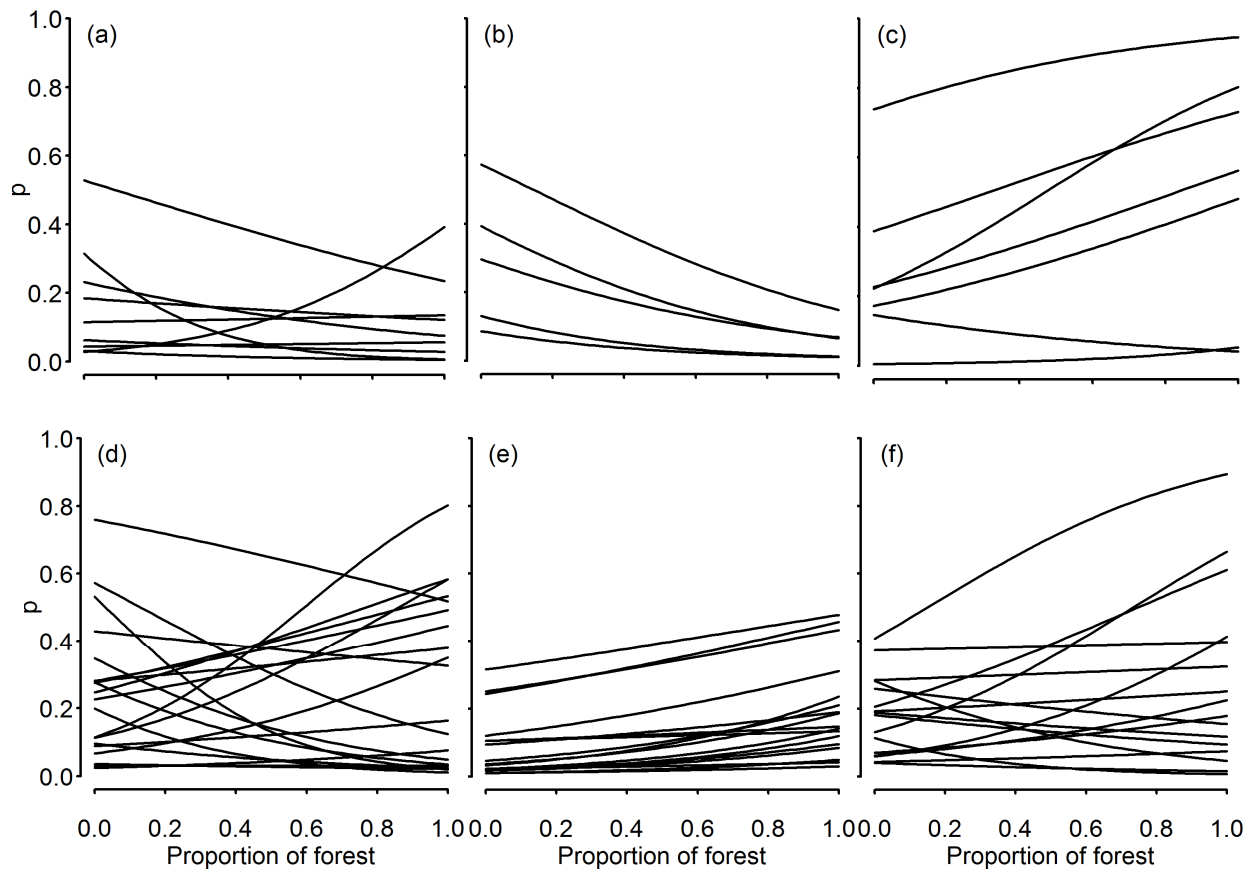
Figure 2.2. Detection probabilities ($\hat{p}$) as a function of proportion of forest cover in the regional bird monitoring program in Argentina, 2003-2012, for each bird species. (a) Raptors; (b) ground omnivores and herbivores; (c) ground granivores; (d) other granivores; (e) insectivores mostly associated with folliage; and (f) other insectivores.
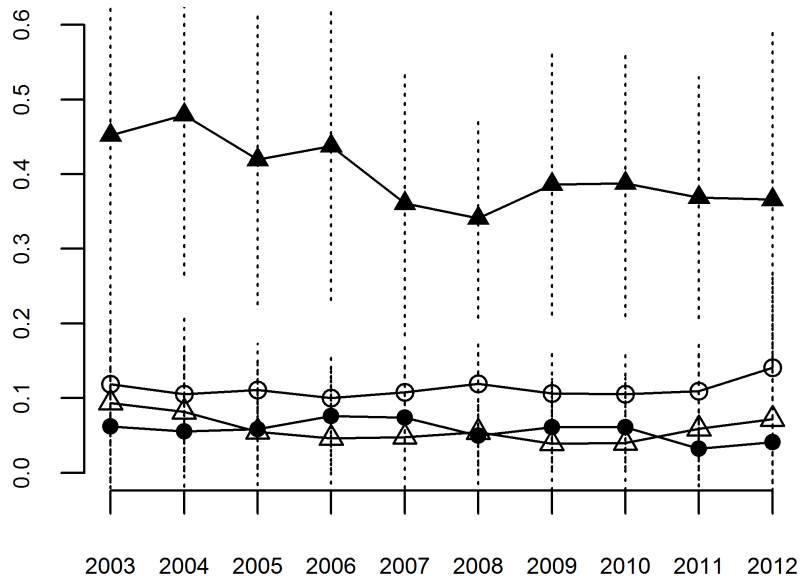
Figure 2.3. Proportion of land cover and land use (+SD) in 2003-2012, in the study area of the regional bird monitoring program in Argentina: soybean (full triangle); corn (open circle); perennial pastures (full circle); native forests (open triangle).

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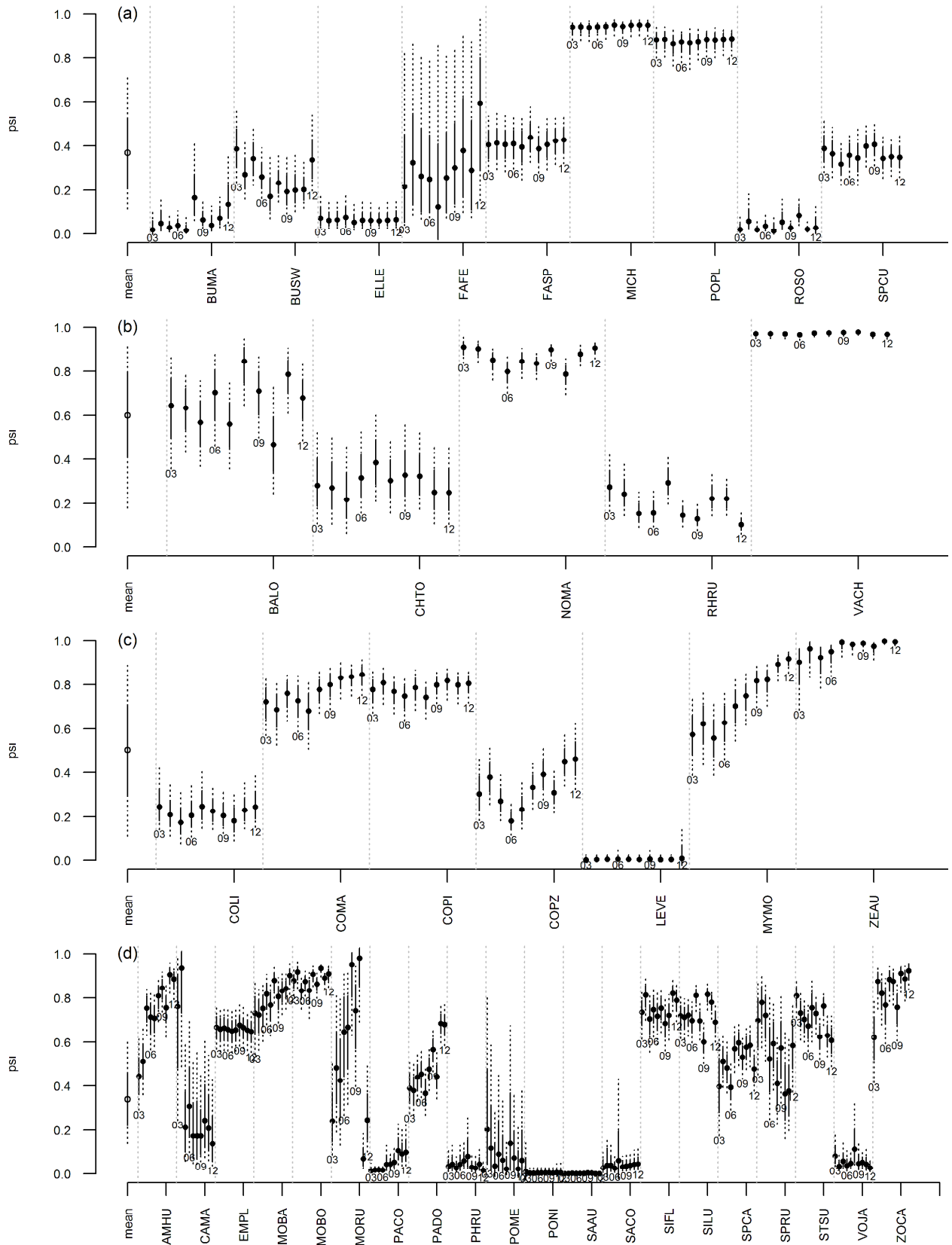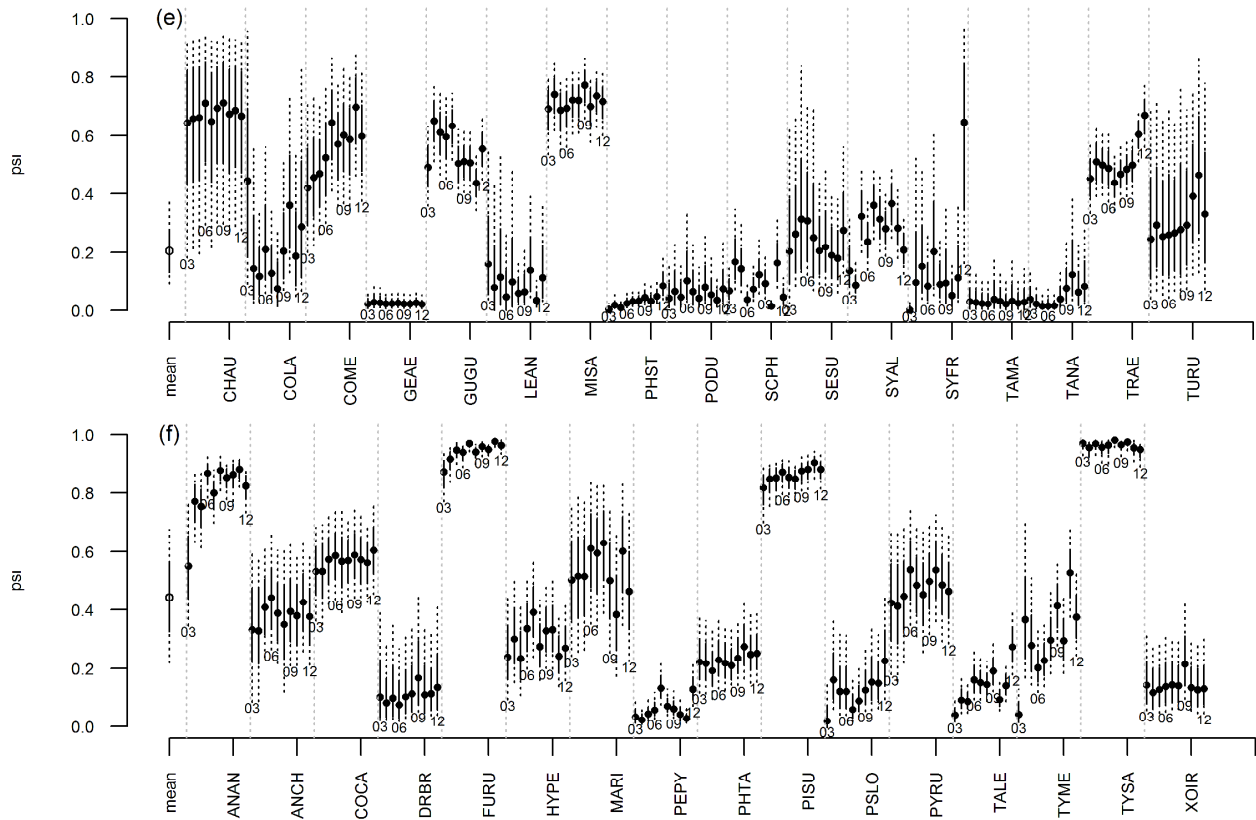
Figure 2.4. : Posterior occupancy model intercepts ($\hat{\psi}$ ±SD, 95% credible intervals), incorporating time as a random effect, during 2003-2012, in the regional bird monitoring program in Argentina for groups of species. (a) Raptors; (b) ground omnivores and herbivores; (c) ground granivores; (d) other granivores; (e) insectivores mostly associated with folliage; (f) other insectivores.
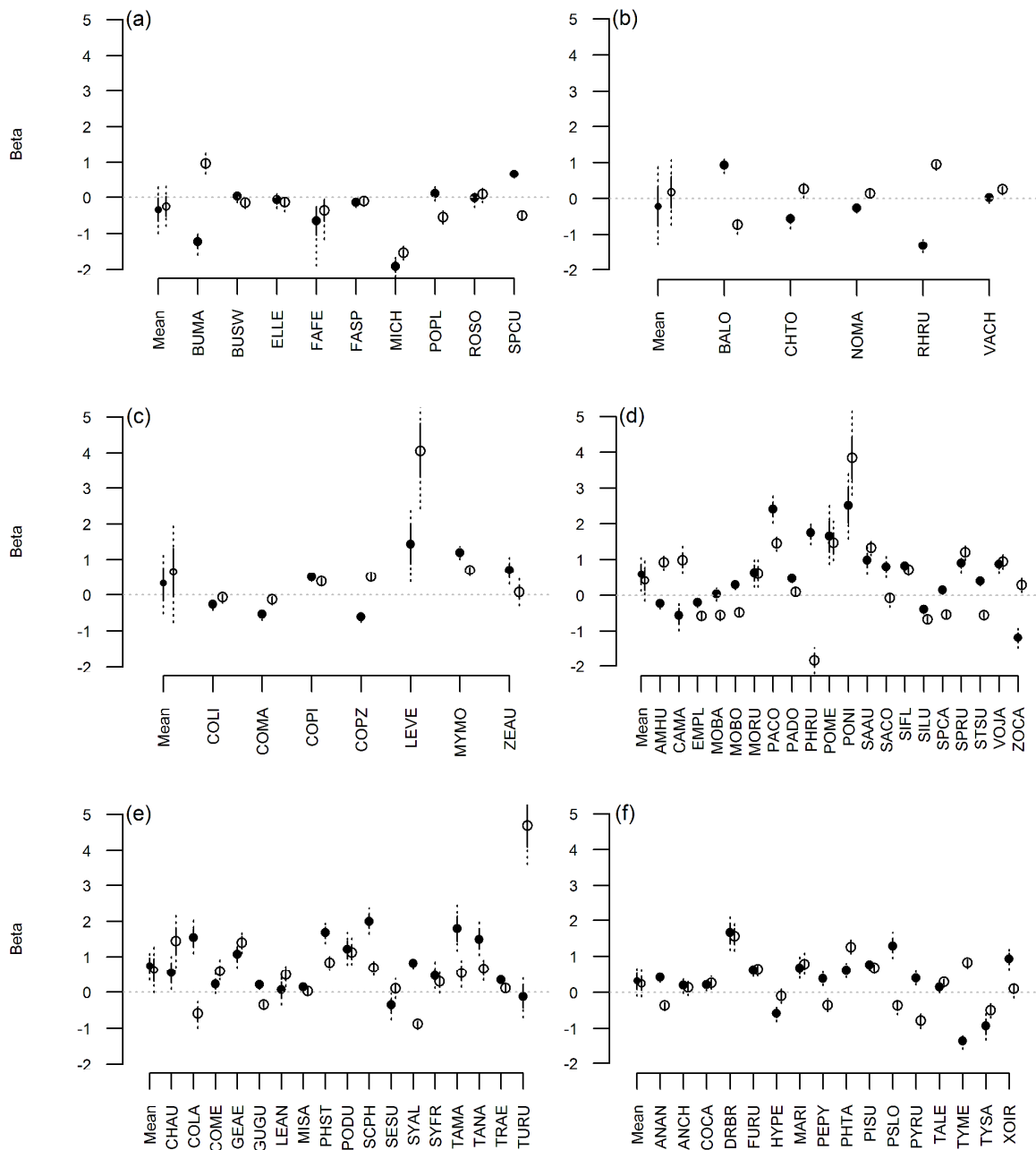
Figure 2.5. Latitude and longitude coefficients in the logit scale ($\hat{\beta} \pm$SD, 95% credible intervals) on logit occupancy (logit $\hat{\psi}$) of each bird species in the regional bird monitoring program in Argentina, 2003-2012. (a) Raptors; (b) ground omnivores and herbivores; (c) ground granivores; (d) other granivores; (e) insectivores mostly associated with folliage; (f) other insectivores. Latitude is indicated by full circle, and longitude by empty circle.

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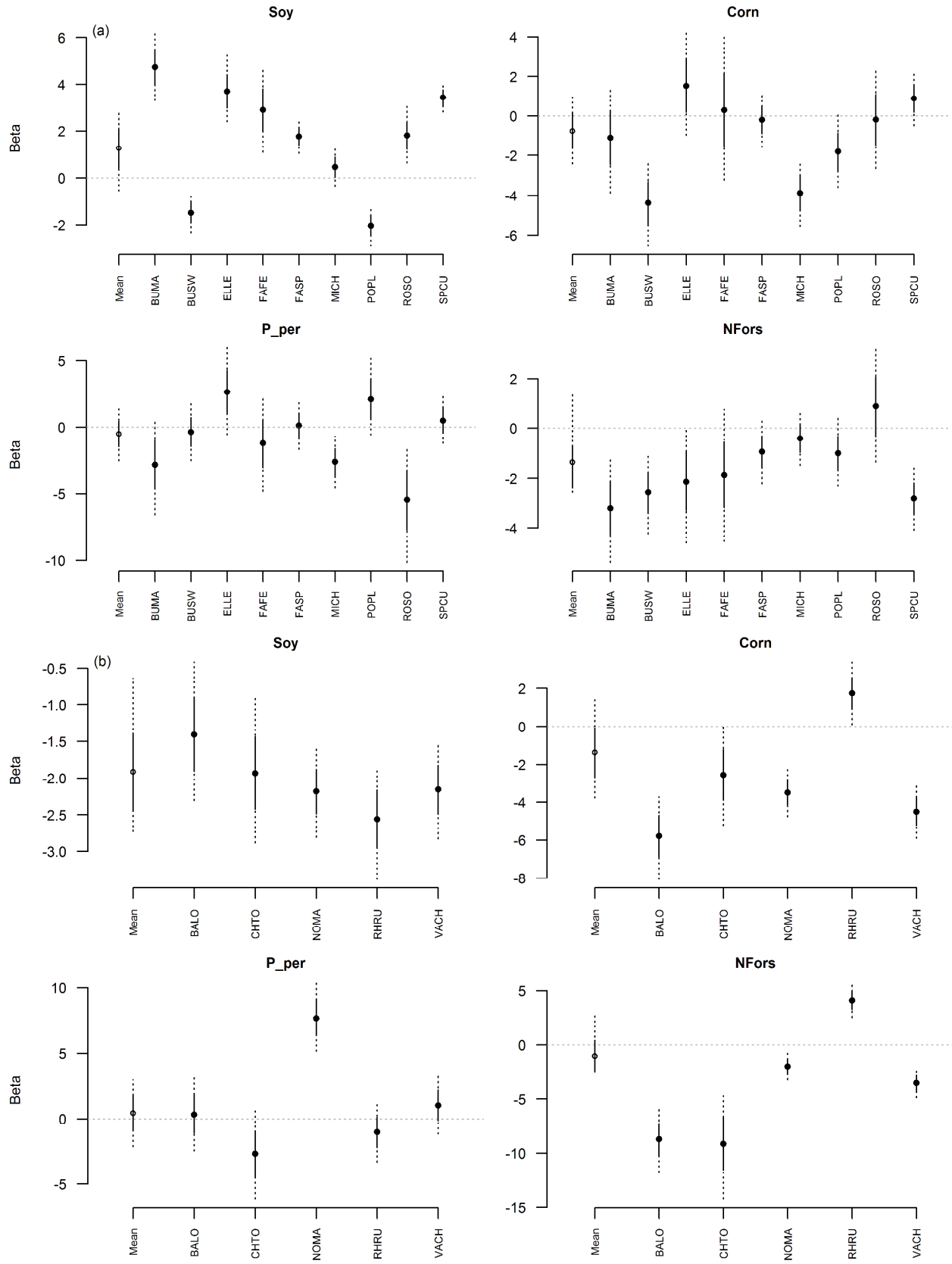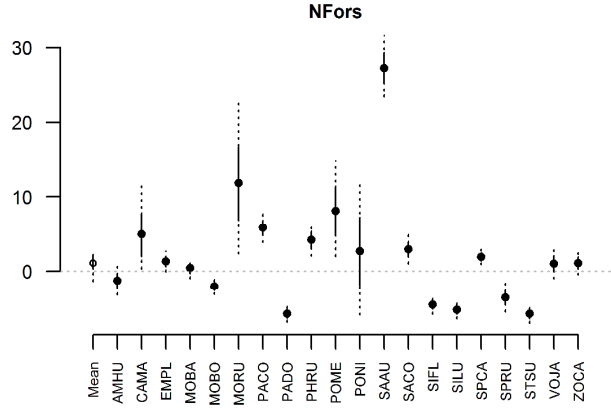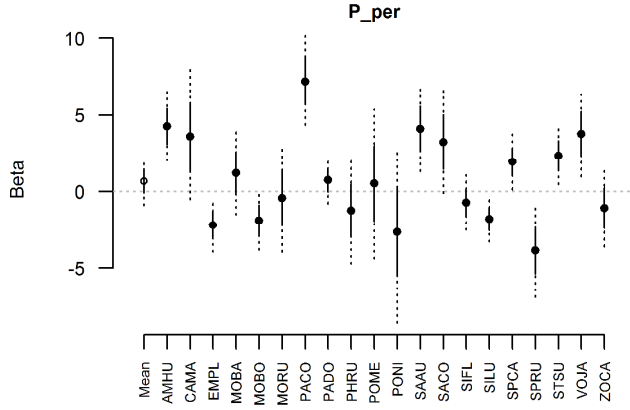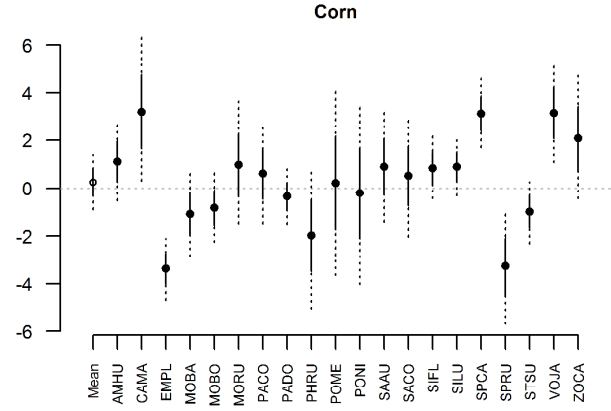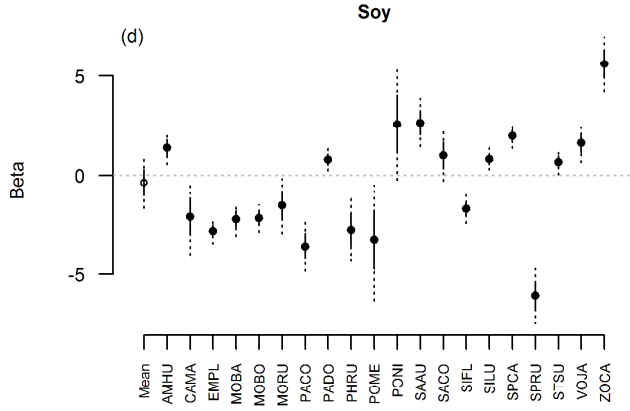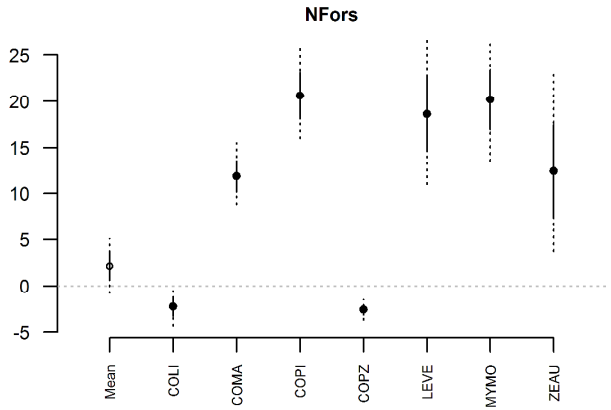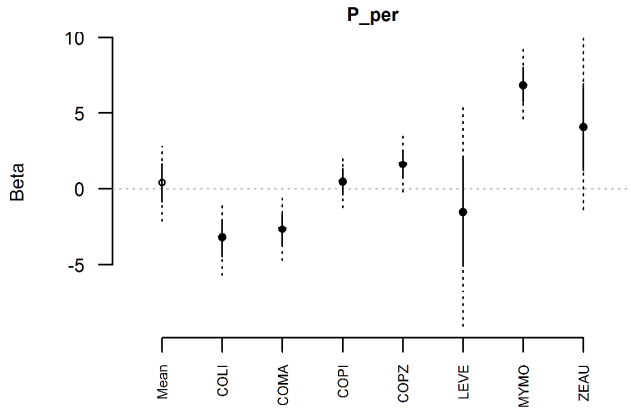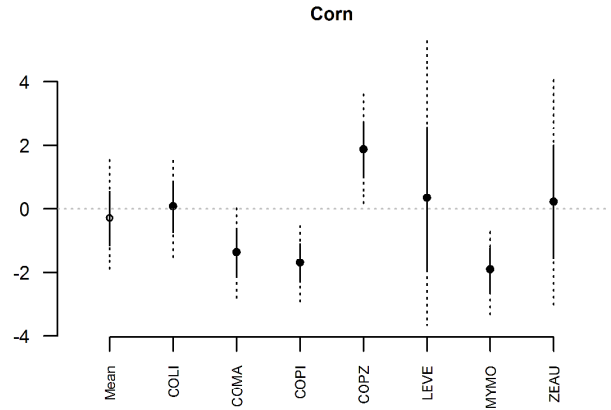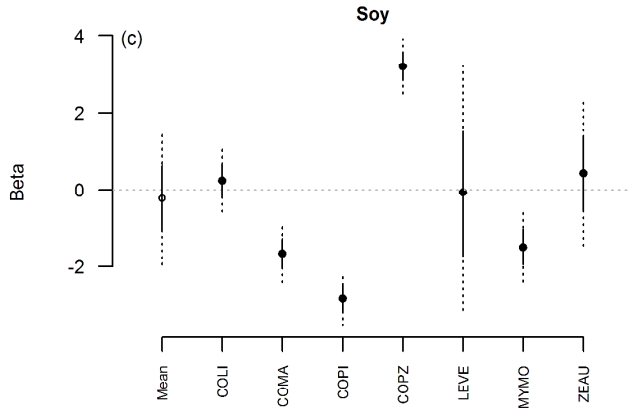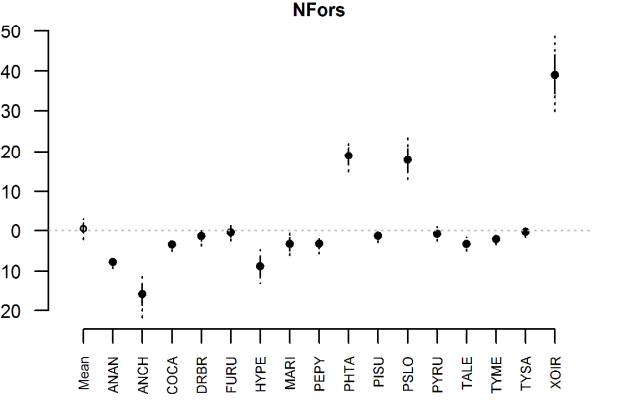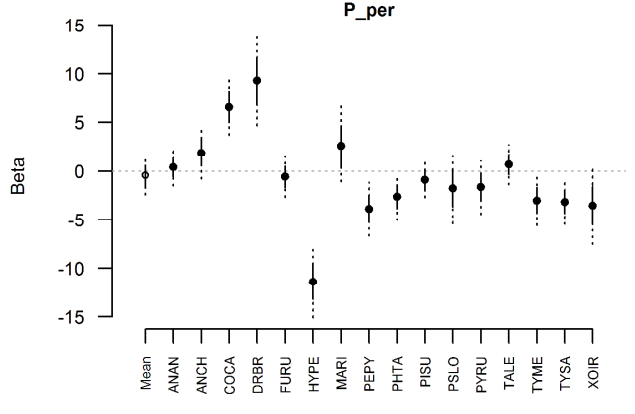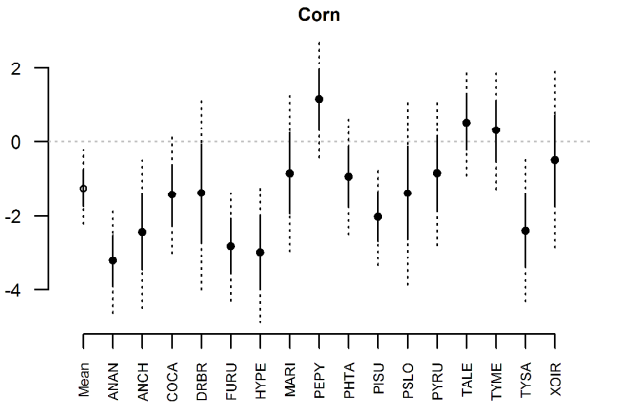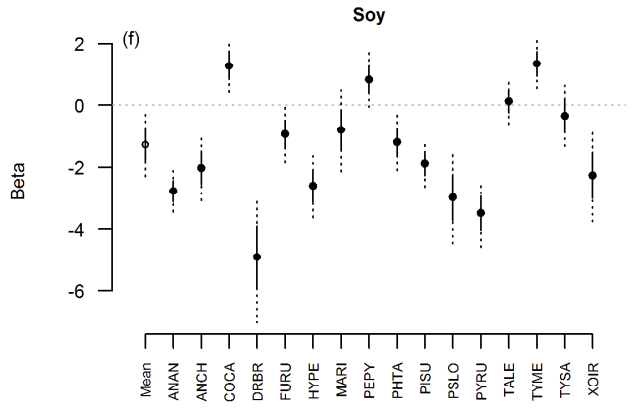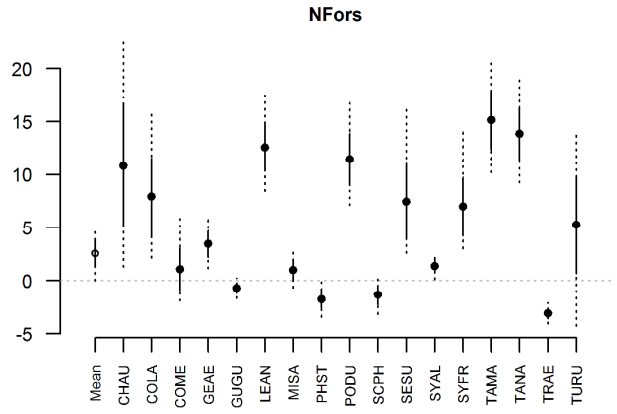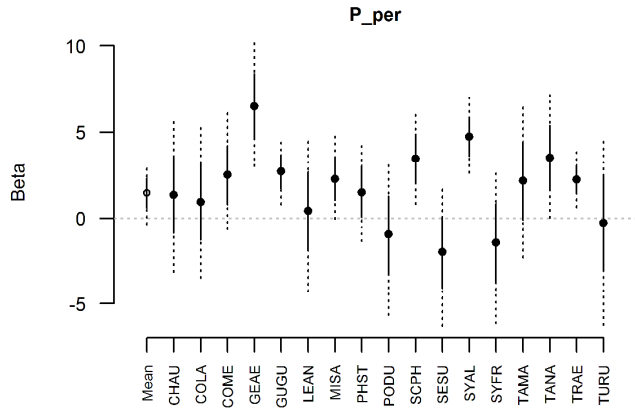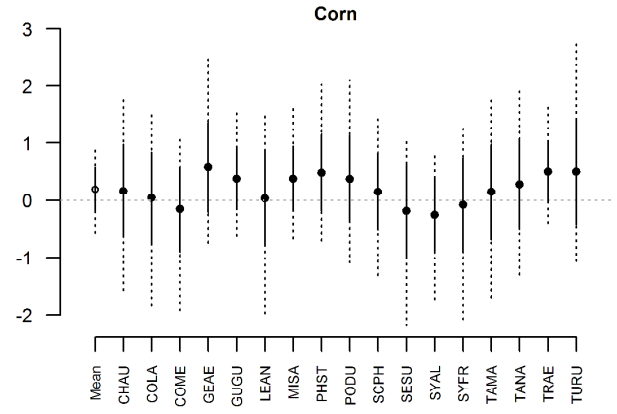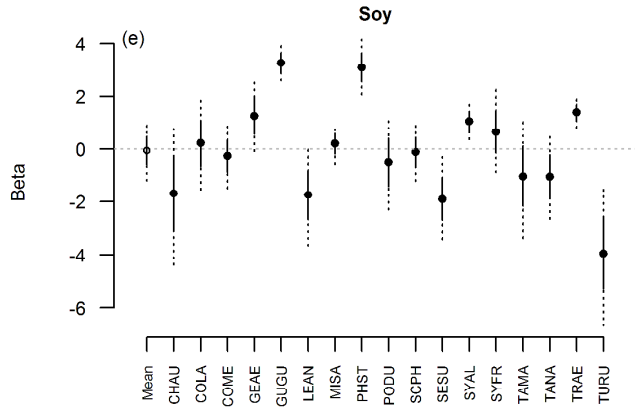
Figure 2.6. Land use and land cover coefficients in the logit scale ($\hat{\beta} \pm$SD, 95% credible intervals) on logit occupancy (logit $\hat{\psi}$) of each bird species in the regional bird monitoring program in Argentina, 2003-2012. (a) Raptors; (b) ground omnivores and herbivores; (c) ground granivores; (d) other granivores; (e) insectivores mostly associated with folliage; (f) other insectivores. Land use and cover are soybean (Soy), corn (Corn), perennial pastures (P_per), and native forests (NFors).
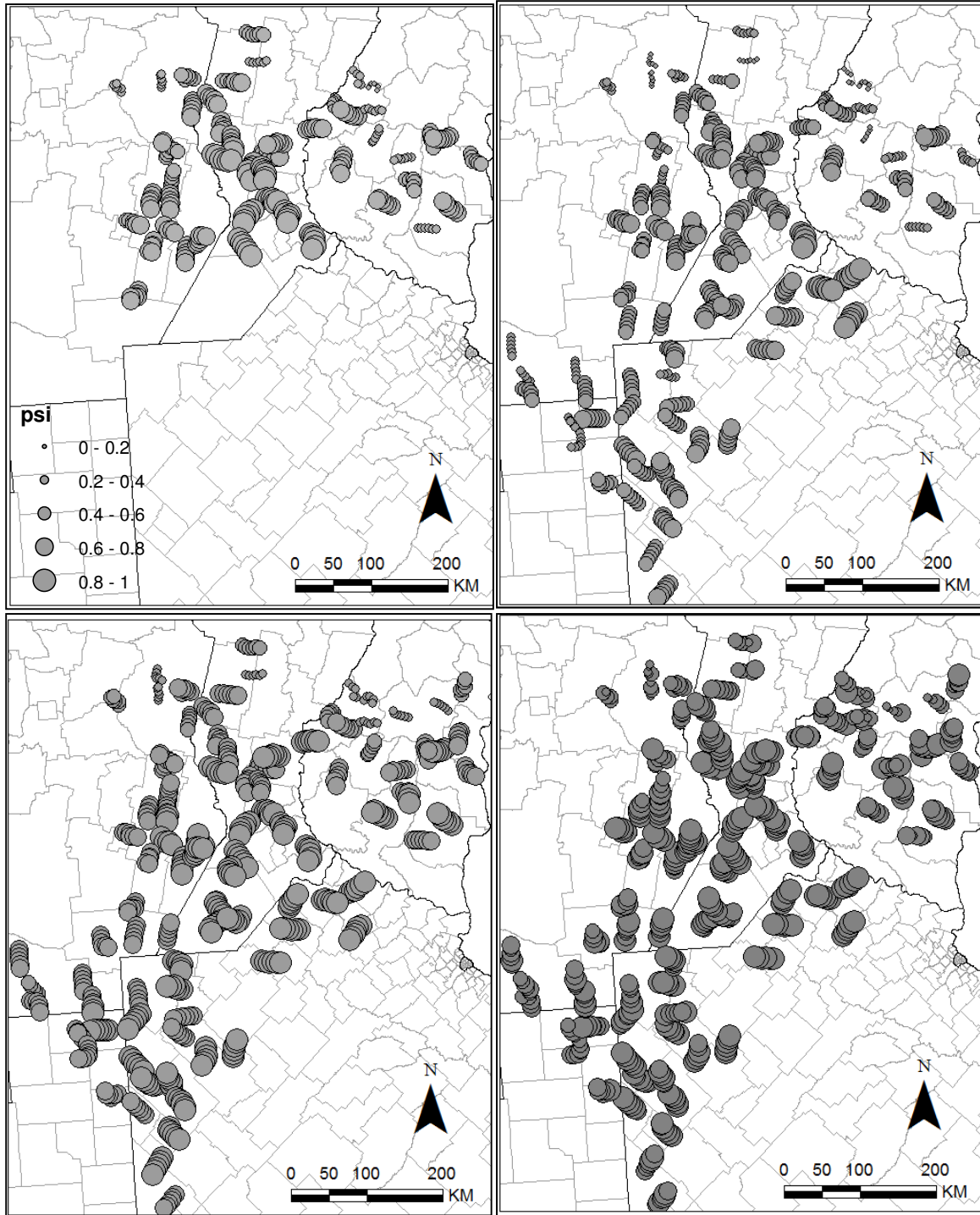
Figure 2.7. Picazuro Pigeon *Columba picazuro* occupancy ($\hat{\psi}$) in the regional bird monitoring area in Argentina for 2003, 2006, 2009, 2012 (left to right).
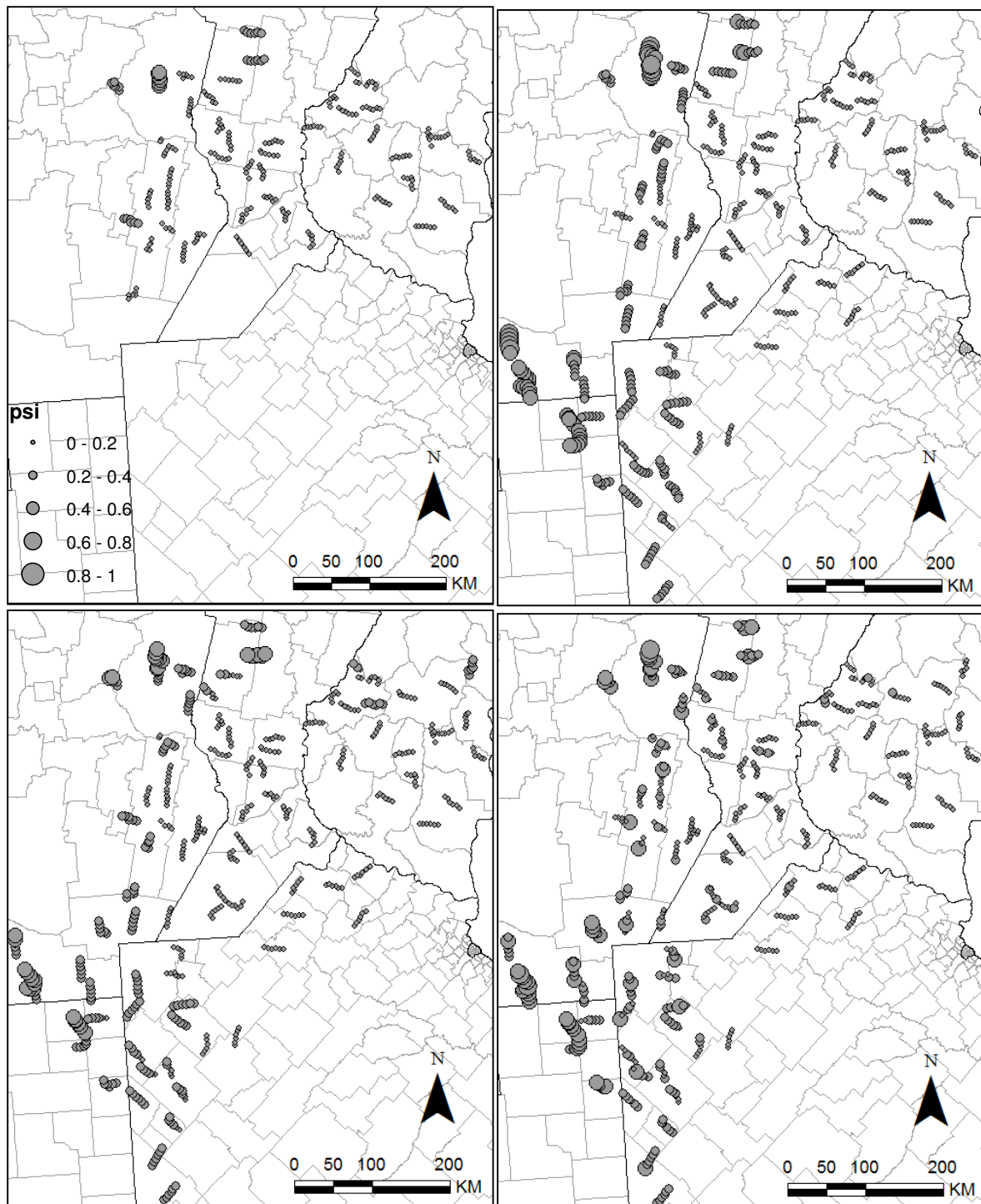
Figure 2.8. Vermilion Flycatcher *Pyrocephalus rubinus* occupancy ($\hat{\psi}$) in the regional bird

monitoring area in Argentina for 2003, 2006, 2009, 2012 (left to right).

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CHAPTER 3

TEMPORAL PATTERNS OF BIRD OCCUPANCY AND PREY INVERTEBRATES IN

SOYBEAN FIELDS AND EDGES: IMPLICATIONS FOR CONSERVATION DECISION

MAKING

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Ecological Applications.

ABSTRACT

Vegetated field borders in an agricultural landscape could benefit birds by providing suitable habitat, depending on their vegetation structure. However, borders may be subject to pesticide drift which could change the vegetation, or reduce the availability of arthropods for insectivorous birds. The objective of this study is to identify factors that influence the use of soybean fields and borders by bird species and foraging guilds in Entre Ríos province, Argentina, an area originally covered by native forests, to ultimately inform decision-making. We account for temporal variation in the phenology of the crop, vegetation structure, and the availability of prey arthropods for insectivore birds. We conducted bird surveys and sampled terrestrial arthropods in 78 borders and the interior of 20 soybean fields, in four different stages of the crop for two years starting in 2007. We analyzed occupancy of birds, accounting for imperfect detection probabilities, separately for field interior and edges, using multiple groups-single season models with covariates. We pooled the main orders of arthropods consumed by birds and fitted Poisson generalized linear mixed models for arthropod counts. Most bird species used borders, and only a small proportion of the interior of the fields; most consume insects potentially offering a valuable ecosystem service. Species in field interior are commonly those either nesting near the ground, granivores, or long flight insectivores. Arthropods remained constant througout the soybean cycle, even after insecticide applications, thus we could not evaluate them as a potential limiting factor for insectivorous birds. Our results suggest that increasing number of native trees and vegetation complexity on the edges of soybean fields benefits birds and may mitigate the deleterious effects of agricultural intensification.

KEYWORDS: Birds, insectivores, soybean, borders, occupancy, ecosystem services, arthropods, insecticides, conservation, Espinal, Argentina, mixed models.

INTRODUCTION

The reconciliation of agricultural production with conservation of biodiversity is a major challenge in the face of the growing demand for agricultural products, The impacts of agriculture on the environment stem from its intensification and expansion, which affect biodiversity and associated ecosystem services, as well as the maintenance of rural livelihoods (Gibbs et al. 2009, Foley et al. 2011, Tscharntke et al. 2012, Cunningham et al. 2013). Argentina is the world's third largest producer of soybeans, where agriculture intensification and expansion is an ongoing process (Paruelo et al. 2005, FAOSTAT 2013). Entre Ríos province occupies 2.8% of Argentina, and in 2007 was responsible for 8% of its soybean production, with most agricultural expansion occurring in the last 20 years and replacing the original forests in the Espinal ecoregion (Paruelo et al. 2005, SIIA 2013). In this context, there is a critical need of filling knowledge gaps, and provide management strategies to reconcile agricultural production with conservation of biodiversity.

Despite this ongoing agricultural intensification, the Espinal ecoregion in Entre Ríos retains a great diversity of birds. However, in the face of rapid and recent land use changes some species seem to be affected by agriculture at a landscape scale, while many others could be responding at smaller scales (Goijman and Zaccagnini 2008, Chapter 2). Some species in the area, like the Yellow Cardinal *Gubernatrix cristata*, are endangered, others are near threatened (but recovering), like Dark-throated Seedeater *Sporophila ruficollis* ; and other species, mainly raptors, have suffered massive mortalities in the past due to inappropriate use of agrochemicals (Goldstein et al. 1999, BirdLife International 2013). Birds are an important part of the agricultural landscape for their intrinsic value and own "right of existence", their role as ecological indicators, and their provision of a broad spectrum of ecosystem services for human

welfare including agricultural production (Kirk et al. 1996, Sekercioglu et al. 2004, Millennium Ecosystem Assessment 2005, Sekercioglu 2006, Whelan et al. 2008). Evaluating the effects of management practices at small scales can provide valuable information for the conservation of bird and their ecosystem services, or the management of those species considered pests (Bucher 1984, Kirk et al. 1996, Jones et al. 2005).

At a local scale, birds may be affected by food availability and/or habitat structure (Wiens and Rotenberry 1981); consequently birds can be affected by management actions such as tillage techniques (zero-tillage vs. tillage), removal of vegetation along the borders, and the use of agrochemicals, among others. Other landscape characteristics in agricultural settings such as field acreage, and distance to forest patches could also be determining bird use of the fields. Linear habitats such as vegetated field borders within an agricultural matrix may benefit birds by providing a suitable and important habitat, subject to their vegetation structure (Parish et al. 1994, Boutin et al. 1999, Jobin et al. 2001, Goijman and Zaccagnini 2008, Di Giacomo and de Casenave 2010), despite being subject to higher predation rates than in larger forest patches (Gates and Gysel 1978). These linear features might be subject to the drift of pesticides used in the adjacent fields, even though the use of pesticides is concentrated in the crop (Boutin and Jobin 1998, Boutin et al. 1999). Especially in genetically modified soybean crops, common in this region, herbicides consisting mainly of glyphosate, and a variety of insecticides, are widely used. Herbicides and insecticides could indirectly affect birds by changing the structure of the vegetation, which diminishes the availability of refuges, roosts and nest sites, or by reducing the availability of their feeding resource, either arthropods or seeds (Freemark and Boutin 1995, Boutin et al. 1999, Beecher et al. 2002, Geiger et al. 2010, Mineau and Whiteside 2013). However recent studies in the same region in Entre Rios showed that arthropod abundance

remained constant in soybean field margins after insecticide applications, suggesting that if these habitats are treated carefully and vegetated, they can still sustain bird insectivore populations (Weyland and Zaccagnini 2008, Varni 2010). On the other hand, there is evidence suggesting that reductions in prey arthropod abundance, although perhaps not influencing insectivorous birds' use of habitat over the short term and at local scales, nevertheless may affect bird abundance and/or their foraging activity patterns at broader scales and over longer time horizons, possibly because of impacts on breeding or post breeding success (Champlin et al. 2009, Benton et al. 2002).

Several published studies have examined the effects of agriculture on birds at a field scale, including the value of borders with natural vegetation. Some studies evaluated avian richness and density at linear habitats in soybean fields in a similar, or the same, agricultural landscape in Argentina (Goijman and Zaccagnini 2008, Solari and Zaccagnini 2009, Di Giacomo and de Casenave 2010). There are also several studies in North America (Best et al. 1990, Boutin et al. 1999, Best 2001, Jobin et al. 2001). However, there are very few studies in North America evaluating temporal patterns (Boutin et al. 1999), and two of them are in cornfields (Best 2001, Beecher et al. 2002). A few studies have evaluated the combined effects of linear habitats with spontaneous or sown vegetation on birds and invertebrate preys, and found positive relationships (Cederbaum et al. 2004, Douglas et al. 2009). To the best of our knowledge, there are no other studies evaluating the temporal effects of management practices in soybean crops and its edges on birds and prey arthropods. In addition, only a few of the aforementioned studies address issues of incomplete detectability, either by analysis or by acknowledging caveats in their results.

Our objective is to identify the factors that influence the use of soybean fields and borders by bird species and foraging guilds in Entre Ríos, Argentina, to ultimately inform

decision-making. We account for temporal variation in the phenology of the crop, and changes over time, either natural and/or due to herbicides and insecticides, on the vegetation structure, and the availability of prey arthropods for insectivore birds. In addition, we implement methods that account for imperfect detection, recognizing that if a bird species is not observed at a certain point, it might be either truly absent, or present but undetected (MacKenzie et al. 2002, Tyre et al. 2003). This study is further motivated by the need to provide reliable data to inform decision-making in agroecosystems that reconcile the conservation of biodiversity and ecosystem services with agricultural production in Entre Ríos. The predictive models and monitoring framework identified here will enhance the development and the reduction of uncertainty of a conservation decision making framework (Williams et al. 2009, Conroy and Peterson 2013).

METHODS

Study Area

The study area consisted of 21 randomly selected fields where farmers planned to plant soybean (*Glycine max* L. Merr.) in that season, near the towns of Cerrito, Palenque, María Grande, and El Pingo; at the north-central region of Paraná department, in Entre Ríos province, Argentina. This area is an agricultural landscape dominated by annual crops and grazing lands. Although there are areas with conservation value, the expansion of row crop agriculture, mainly soybean, is rapidly degrading the native forest. Originally, this area supported *Espinal* forest dominated by xerophytic woody species such as *Prosopis affinis, Acacia caven, Geoffroea decorticans, Celtis tala* and *Schinus longifolia,* and the weather is warm and humid with abundant precipitation during spring and summer (Cabrera 1971, Cabrera and Willink 1973, Burkart et al. 1999). The

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climate is temperate and humid, with a mean annual temperature of 18°C and a mean annual rainfall around 1100 mm.

Data Collection

Birds

We conducted bird surveys for two years in the center of soybean fields and their adjacent borders. All selected fields had corn in the previous season, and vegetation structure at each border was variable; to maintain similar conditions on all plots, we avoided fields next to native forests. Bird surveys were conducted during four consecutive seasons over two years, in ten fields each year, at different stages throughout the soybean cycle in 2007-08, and 2008-09. The first survey was completed in October, prior to the sowing, the next in December with soybean in the vegetative stage. The third survey was conducted at the end of January, in the flowering season, and the last was prior to harvest in the beginning of March. The second year of this study, some fields were planted late or not planted due to drought. The survey period is spring and summer in South America and coincides with the bird breeding season and the presence of migrants.

We surveyed birds in randomly allocated line transects of 200 m in field interiors (n=60, 4 seasons) and borders (n=78, 4 seasons). Observations were carried out in the morning (0600 – 0900) when birds were more active, and all birds seen or heard actively using field border or interior were registered; and we avoided days with rainfall or strong winds (Bibby et al. 2000).

Arthropods

We sampled terrestrial arthropods in the above bird sampling transects, both at the edges and interior of soybean fields. The surveys were conducted between 1400 – 1600, when temperatures were high and the arthropod activity level was the highest (Hill 1980). During the first year of study, we used a sweep net to collect aerial arthropods sweeping 25 times at four points in each transect, and then pooling the samples per transect. The second year of study we collected arthropod samples using a vacuum (STIHL BR 420), during 40 sec intervals in three points per transect, pooling the samples per transect.

All samples were transferred to 70% alcohol for the posterior taxonomical classification. We identified arthropods to the order level, and in this study we accounted only for those which mainly compose insectivorous bird diets, such as Araneae, Coleoptera, Diptera, Hemiptera, Hymenoptera, Lepidoptera, and Orthoptera (Wilson et al. 1999, Alessio et al. 2005, Moorman et al. 2007, Capinera 2010).

Field Characteristics and Vegetation

We recorded various characteristics of the fields and edges, such as field size, distance to nearest forest patch, phenological stage of the crop, and pesticide applications on each field; in addition for each border we measured its width, and the total number of trees and native tree species numbers. In two randomized points in each edge transect we recorded the herbaceous vegetation height and percent cover in a square meter of each strata of vegetation, classifying into six categories the herbaceous and dead vegetation, shrubs, trees, and bare soil. In the interior of the fields we repeated the procedure, measuring coverage of herbaceous vegetation, stubble, and bare soil.

Statistical Analyses

We analyzed avian occupancy using occupancy models, where imperfect detection probabilities of each species are accounted for, 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, Tyre et al. 2003, MacKenzie et al. 2006). We implemented occupancy models using the package RMark (Laake 2013), which provides a formula-based interface for MARK (White and Burnham 1999), in R version 3.0.1 (R Core Team 2013). We performed separate analysis for field interior and edges, using multiple groups - single season models with covariates (MacKenzie et al. 2006). Because the number of detections was low for many species, we pooled species with similar habitat requirements that reached less than 10% detections. Although we pooled species because of their low detections, we are aware of the caveats presented by a priori grouping, which could mask individual species responses associated with their particular traits (Philpott et al. 2009, Ruiz-Gutierrez et al. 2010).

At field interior we modeled occupancy for each season separately, because we did not achieve a sufficient number of replicates to implement multi-season models that estimate local turnover (colonization and extinction) probabilities. We assumed that occupancy of birds could differ by season given crop phenology, and possibly in relation to insecticide applications that could have affected arthropod availability. We estimated detection probabilities using transects within each field (n=20) as spatial replicates (k=3), and groups, arthropod counts, minimum temperature, and wind as covariates. We modeled occupancy at the field scale, using field size, distance to the nearest forest, arthropod, and its collection method (only for those groups of birds with insectivorous diets), bare soil, or herbaceous coverage as individual covariates.

At the borders (*n*=78) we assumed closed populations for the period of study, thus we estimated occupancy considering seasons as temporal replicates (*k*=4) to model detectability. The assumption that bird occupancy remains constant at each border during the studied period, but with possible changes in probability of detection, was based on the fact that herbicides and pesticides are not directly applied on borders, especially because applications in this area are not aerial; hence the potential drift effect does not generate dramatic changes in vegetation and arthropod abundance (Weyland and Zaccagnini 2008, Varni 2010). To support this statement, we evaluated arthropod counts throughout the studied period as well, as described below. A reduction of arthropod counts, however, could affect bird detectability via lowering their abundance and/or affecting their conspicuousness by reducing their foraging attack rates (Champlin et al. 2009). In addition the study was carried out mostly during the breeding season and therefore many birds remained in the edge for the entire period; we also found little difference in frequency of observations between seasons, except some migrant aerial foragers and salliers for which we eliminated the last season (pers. obs.). We discarded correlated covariates (Pearson's r > 0.4) to model detectability and occupancy of birds in borders. We modeled detection using arthropod counts, herbaceous height, wind, and minimum temperature as time specific covariates, and total number of trees, which was constant over time. Occupancy models were modeled with individual covariates, which are not expected to vary over time. Those covariates used were the number of native trees (highly correlated to border width, total trees, and shrub coverage), dead vegetation coverage, herbaceous height, distance to nearest forest, arthropod counts averaged over four seasons for each site, and the collection method incorporated as a dummy variable.

All explanatory variables were standardized (mean subtracted from all values and the result divided by standard deviation), to ensure that the numerical optimization algorithm finds the correct parameter estimates (Cooch and White 2013). We made inferences regarding the strength of the effect of covariates on the response variable of interest by looking at the 95% upper and lower confidence limits (CI), where if zero is contained in the CI the effect of the covariate is considered weak. We evaluated relative plausibility of candidate models following an information theoretic approach using Akaike's Information Criterion with the correction for small samples and overdispersion (QAICc) (Burnham and Anderson 2002). We corrected for possible overdispersion of the data using the c-hat ($\hat{c}$) variance inflation factor, calculated as the saturated model deviance over the bootstrapped deviance after 1000 simulations calculated in MARK (Cooch and White 2013). We performed model averaging over real parameter values, which consists on weighting the estimates from each model in the candidate model set by its normalized QAICc model weight of evidence relative to the best model (w_i) and accounting for model uncertainty (Burnham and Anderson 2002), after eliminating those with poorly estimated parameters (e.g. standard errors two orders of magnitude greater than the parameter).

We analysed arthropod counts by pooling counts of the main orders consumed by birds (i.e. Araneae, Coleoptera, Diptera, Hemiptera, Hymenoptera, Lepidoptera, and Orthoptera), fitting generalized linear mixed models (GLMM) using the glmer function implemented in the lme4 package (Bates et al. 2013) in R (R Core Team 2013). We fitted the models assuming that arthropod counts followed a Poisson distribution, and we considered the possible effects of seasons, nested as repeated measures in each border, and the latter nested in fields. We also included standardized covariates such as herbaceous vegetation height, dead vegetation and shrub coverage, number of native trees, the application of insecticides, grasses or bare soil

coverage, soybean stage and insecticide applications in the center of the fields, to model the arthropod abundance in borders. Because our count data of arthropods were over-dispersed (i.e. variance/mean ratio >1) we added an observation-level, normally distributed random effect, equivalent to a lognormal-Poisson model (Elston et al. 2001). By doing this, we reduced underestimation of standard errors that arises with over-dispersed data. For example, the number of arthropods n_{ijk} counted on border i of field j at season k followed a Poisson distribution n_{ijk} ~$Poisson(u_{ijk})$, then the model was specified as follows:

$$\log(\mu_{ijk}) = \alpha_k + \beta x_{ijk} + \varepsilon_j + \varepsilon_{ij} + \varepsilon_{ijk}$$

where α_k is a categorical fixed effect of the seasons, a continuous β fixed effects of a covariate x_{ijk} (e.g. herbaceous vegetation height), and random effects of the field ε_j, border ε_{ij}, and observation ε_{ijk} with Normal distributions. We then compared models with and without the "field" random effect, and different covariates, and selected the best model using Akaike's Information Criterion with a correction for small samples (AICc) (Burnham and Anderson 2002).

RESULTS

We observed 82 bird species from 27 families actively using field borders and soybean fields during the period of study. We were able to analyze occupancy of those species that, individually or pooled with other species of the same guild, had sufficient observations (at least in 10% of the surveys the species/group was detected); resulting in 52 species of 18 families, classified in nine guilds according to their foraging behavior and in some cases considering their nesting requirements (i.e. ground nesters) (Table 3.1; Remsen and Scott 1990, Azpiroz 2003, Beltzer 2003, De la Peña 2005). In the interior of the fields we detected fewer species than at the

borders, and only 13 species had sufficient detections to be incorporated in the occupancy analyses, either pooled into a group or individually. Only species belonging to the ground omnivores and granivores, ground granivore foliage gleaners, and some aerial foragers were observed using the center of the fields, but not in all seasons.

Field sizes averaged 35 ha, ranging from 17-140 ha., and the nearest native forest patch was located at an average of 500 m (range 137-1100 m). All fields followed the expected soybean phoenological stages during the first year, but due to drought the the second year one field was planted later, and four fields were not sown; two of these were discarded from the analysis because of the presence of cattle. Most insecticide applications started in the third season, coinciding with the soybean flowering stage.

Bird occupancy

Detection and occupancy probabilities of birds using the interior of the fields were highly variable; this variability in addition to the low detection probabilities for some groups, made occupancy estimates highly variable (Fig. 3.2). Ground omnivores such as Upland Sandpiper, Spotted Nothura, and Southern Lapwing used the center of the fields in the pre sowing season and at soybean leaf stages, where height averaged 15 ± 7 cm. On the other hand, Columbiformes used fields only before planting of soybean. The only groups detected throughout the soybean cycle in the interior of the fields were the ground granivore foliage gleaners, and Spotted Nothura. Aerial foragers used the interior of the fields mostly when soybean was in the vegetative and the flowering stages.

High variability in the estimates indicates that our sample size in the interior of the fields was not large enough, and this fact hindered our ability to detect the influence of covariates

affecting detection or occupancy of the groups. The best model explaining occupancy of those groups detected using the interior of the fields was usually the null model, ranging between 20-60% of support. Models with covariates explaining occupancy evidenced weak effects (i.e. all overlapping zero; Appendix E, Table E.3.1). Detection probabilities for ground omnivores in the first season increased with an increase in minimum temperatures (Best QAICc model, $\hat{\beta}$=0.79, CI (0.211 - 1.36). At this same season detection of ground granivores (i.e. Columbiformes) was negatively affected by wind (2nd model, $\hat{\beta}$=-1.31 (-2.23 - -0.39). Probability of detection of aerial foragers in the 3rd season was positively affected by an increase in minimum temperatures (best model, $\hat{\beta}$=2.11 (0.51 - 3.70).

Detection probabilities at field borders for granivore foliage gleaners, excluding ground nesters, insectivore foliage gleaners, ground and foliage omnivores, and Rufous Hornero were positively related to total number of trees (Fig.3.3.c, e). Detection of Chalk-browed Mockingbird was negatively affected by trees, while the effect was weak or null for the rest of the groups; and the best models explaining ground nester granivores and aerial foragers were grasses height and minimum temperatures, respectively, although the effects were also weak (Fig.3.3.a-f; Appendix E, Table E.3.2). Overall, detection probabilities at field borders were the highest ($\hat{p}$>0.5) for Picui Ground Dove, and Grassland and Rufous-collared Sparrows, and the lowest for ground omnivores.

Occupancy of most species at field borders was positively related to number of native trees, with the exception of ground omnivores and ground nesting granivore gleaners (Appendix E, Table E.3.2). Generally, Grassland and Rufous-collared Sparrows evidenced the highest occupancy probabilities. On the other hand, the effect of native trees was strong for most groups were the maximum occupancy was reached with only more than ten trees (Fig. 3.4.a-c).

Granivore foliage gleaners were also positively affected with grasses height, although the effect was variable (Fig. 3.5). Ground omnivores and ground nesting gleaners were not strongly associated with any covariates.

Arthropods

In the interior of soybean fields, arthropod counts available for birds was lower in the first season (pre-sow) compared to the other seasons and it was higher with increasing grass coverage. Overall, abundance was unrelated to insecticide applications and the stage of the crop.

In this habitat, grass cover had strong positive relationship to arthropod counts when the collection method utilized was the sweep net, and this effect was supported by all best models (Fig. 3.6.a, and Appendix E, Table E.3.3). In addition, there was an interaction between seasons and grass cover on arthropod counts; where the slope of the first season differed from the other seasons (Fig. 3.6.a). When arthropods were collected with vacuum, the effect of bare soil, grasses, insecticide or stage on arthropod abundance was supported 43, 36, 34, 13% of the total model weights respectively, but all these effects were weak, and the null model was supported, with 19% of the model weights.

Similarly to the interior of the fields, arthropod counts in borders were higher in the last three seasons when compared to the first one, independently of the collection method. There was a weak positive effect of herbaceous vegetation height on arthropods collected with sweep net, and a negative effect when collected with vacuum. However, the overall count of arthropods is always higher when collected with sweep net, at all seasons except the first one, meaning that the negative effect found with vacuum collection is of minor importance (Fig.3.7.a, b). Dead vegetation coverage has a strong negative effect on the arthropod counts collected with sweep

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net, while the effect was weak for arthropods collected with vacuum with <50% support (Fig. 3.8.a, b, and Appendix E, Table E.3.3). Insecticide applications had no effect for those arthropods collected with sweep net, while this effect was positive but weak with vacuum. Finally the number of native trees affected negatively counts collected with vacuum and did not affect those collected with sweep net, thus the overall effect of native trees is negligible, as seen with herbaceous vegetation height.

DISCUSSION

In the two years of our study, we recorded 82 bird species using soybean fields or borders, which represents around 45% of landbirds potentially distributed in the study area in Entre Ríos (De la Peña 2006, Narosky and Yzurieta 2010, Dardanelli *per. comm.*). Most species used borders, and only a small proportion the interior of the fields. The most common species in our study coincide with those common in agricultural and/or human dominated environments (Goijman and Zaccagnini 2008, Solari and Zaccagnini 2009, Di Giacomo and de Casenave 2010). Moreover, most species in our study are exclusively insectivorous, omnivorous, or consume insects in the breeding season, potentially offering a valuable ecosystem service to agriculture.

Our data were insufficient to directly relate detection or occupancy probabilities of birds in the interior of soybean fields to field characteristics or prey arthropod availability. Here, the species are those either nesting near the ground, granivores, or long flight insectivores, with some exceptions; and all are common in agricultural landscapes. These species are of great concern, as they may be directly or indirectly affected by agrochemicals, plus some species are considered pests, and others invertebrate pest controllers (Kirk et al. 1996, Tremblay et al. 2001, Philpott et al. 2009). For example, ground granivores were observed prior to soybean was

planted, likely feeding on corn grains remaining from the previous season, and only Eared Dove and Spotted-winged Pigeon are considered problem species (Bruggers et al. 1998). Likewise, prior to planting, and when soybean plants were short, ground insectivore species occuring were the Upland Sandpiper – a non-breeding migrant from North America (BirdLife International 2013) – and the Southern Lapwing. Since only herbicides are applied in these seasons (Saluso et al. 2007), these species are likely not vulnerable to direct toxicity, since herbicides such as glyphosate have no reported ecotoxicolgic effects on birds to date (Freemark and Boutin 1995, Bernardos et al. 2007). Further, even though arthropod counts in the first season were low, these insectivores are still selecting this habitat and apparently finding sufficient food. These observations supports Champlin et al. (2009) who suggested that prey abundance does not affect habitat use in the immediate term, although this could affect future selection of habitat (Benton et al. 2002), or reproductive success for those breeding.

Grassland Yellow Finch, Rufous-collared and Grassland Sparrows, which are ground nesting granivore gleaners, as well as Spotted Nothura, used the interior of the fields throughout the soybean cycle. Behavioral observations of these birds lead us to assume that they nest and feed in this habitat, and therefore may be vulnerable to direct insecticide toxicity. Some of the insecticides used in the area, have low toxicities such as cypermethrin, but others such as endosulfan have high toxicity levels for birds (Mineau 2002, Bernardos et al. 2007). Last, insectivores such as Brown-chested Martin, and Fork-tailed flycatcher, were found mainly in the vegetative and flowering stages of the crop, where arthropod counts seemed to be the highest, despite insecticide applications. This group is likely providing a valuable ecosystem service by preying on arthropods in the seasons when pests usually emerge (Saluso et al. 2007), and could also be vulnerable to direct effects of toxic insecticides both by direct contact or by contaminated

food (Goldstein et al. 1999, Mineau 2002). Besides, Paquette et al. (2013) suggested the possibility that the effect of insecticides could lead to aerial foragers to an "ecological trap", and the same can be assumed for other guilds. They can be faced with the possibility of prey abundance to be reduced by insecticides, after they already selected nesting sites under different conditions.

Our study suggests that increasing complexity in vegetation structure and the number of trees on the edges of soybean fields benefit birds, consistent with previous studies (Parish et al. 1994, Jobin et al. 2001, Goijman and Zaccagnini 2008, Di Giacomo and de Casenave 2010). This result highlights the importance of these elements of the landscape, which has even been documented as habitat for birds despite adjacent crop (Di Giacomo and de Casenave 2010), although birds may be exposed to high predation rates (Gates and Gysel 1978). Most guilds benefited from density of native trees, which was positively correlated with border width, and shrub coverage, where even 10-20 trees sufficed to achieve maximum occupancy probabilities. Granivore foliage gleaners in general benefited as well from herbaceous vegetation height, although with weaker effects. These results were expected, especially in this study area, where vegetation was originally composed of shrubs and trees, and most observed species nest on woody vegetation, and/or are dependent on woody vegetation for foraging activities like the insectivore and granivore foliage gleaners. Ground omnivores and ground nesting granivores where unaffected by native trees, which is also expected given that these species are associated with open habitats – and were also common in the interior of the fields – and most nest near the ground. Similarly, insectivorous salliers and aerial foragers were not strongly associated with native trees which is expected since these are long-flight species, able to move at a larger scale.

The strong relationship between the number of native trees and some species adds support to the hypothesis that these species respond to habitat characteristics at a finer scale than the landscape one, and in our study this seemed particulary evident for small size passerines like insectivore and granivore gleaners (Peterson et al. 1998, Robinson et al. 2001, Filloy and Bellocq 2007). In addition, similarly to Jobin et al. (2001), we did not find a strong relationship between bird occupancy and distance to the nearest forest patch, indicating that border features were more important than landscape characteristics. Our results also support the inferences made in Chapter 2, where we found that several species could be responding to a smaller scale than the regional in agricultural landscapes.

Our data did not indicate a negative effect of herbicides and insecticides on birds. However, a plausible mechanism for indirect effect still exists, in that bird occupancy is related to the complexity of vegetation structure, which in turn can be affected by herbicides. Counts of arthropods preyed on by birds remained constant throughout the soybean cycle, in the interior and border of the fields, even after insecticide applications; thus we could not evaluate them as a potential limiting factor for insectivorous birds. Arthropod counts were the lowest in the first season and it was not associated with herbicides, given only 15% of the fields by that time were treated, and none was treated with insecticides; thus, it was probably related to vegetation structure in the borders and the ansence of crop in the first season, and seasonality intrinsec to invertebrate populations, especially in temperate areas (Wolda 1988). Vegetated borders host a greater number of arthropods than the interior of the fields, and have been documented to serve as refuge in the presence of insecticides (Lee et al. 2001, Weyland and Zaccagnini 2008, Varni 2010). This fact, combined with the great power of dispersion of several species of arthropods, helps mantain total abundance in vegetated edges at a certain level, despite insecticide

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applications (Lee et al. 2001). Similarly, arthropod abundance in the interior of the fields remained constant after the first season, which could be the consequence of a rapid recovery of some arthropod populations dispersing from other fields or borders (Duelli et al. 1990, Weyland and Zaccagnini 2008). Finally, it is important to note that we did not analyze the order composition of arthropods, which probably varied in the edges, fields, and with pesticide applications, unlike the total abundance.

Implications for Conservation Decision Making

Managing bird populations is important given their intrinsic value, and the ecosystem services they provide to agricultural production. In this region in Entre Rios, bird conservation is still possible, given the large number of species persisting, the presence of a good amount of native forest patches, vegetated borders of fields, and small and medium-scale farmers, some who still live in rural areas and value the elements of the natural environment.

Our results can be used as input to inform a conservation decision making process, to provide management recommendations for farmers in this area to reconcile soybean production and bird conservation. For example, most birds in the area have an exclusively or partly insectivorous diet, represented by many guilds; this means that not only birds could contribute to invertebrate pest control, but can also involve invertebrates of different habits, such as terrestrial or aereal among others. Yet, many birds important for conservation, or those likely feeding on pests, use the interior of the fields and could be vulnerable to direct toxicity of some of the pesticides, as has also been inferenced in other studies, although further work is needed to establish the actual risk (Boutin et al. 1999). In addition, by regulating the use of agrochemicals and conserving habitat for birds, other taxa may be benefited as well. Therefore it is important to

inform farmers about the risks of some pesticides, and advise them on the use of the least toxic ones; also highlight the importance of performing pest threshold monitoring before applying pesticides, instead of doing it preventively. Although pesticides are used in the interior of the fields, the borders can be subject to drift effects. Alhough we had no evidence of such effects on birds and even found that borders could be refuge for invertebrates from insecticides, we found how vegetation structure is important for most birds, and thus we advise avoiding the application of herbicides in borders.

Additionally, we found that the number of native trees in borders, associated with shrubs and grasses coverage too, greatly benefit birds. The small reduction in soybean yield resulting from conserving vegetation in borders may be an acceptable tradeoff if it results in a great benefit to birds and invertebrate natural enemies (Lee et al. 2001, Stamps et al. 2008, Varni 2010). Although this alternative seems simple to implement, it is not, because farmers in this area are accustomed to removing the vegetation in borders. To advise agricultural producers about the benefits of keeping vegetated borders will require a participatory, transparent and well informed process that can integrate their values, needs and beliefs with reliable empirical science, such as a structured decision making approach (Williams et al. 2009, Conroy and Peterson 2013).

Last, the use of the occupancy estimation approach to evaluate bird responses to habitat characteristics has proved to be a useful tool. Most bird species or bird guilds had imperfect detection probabilities, even under less than 50%, and some were affected by either weather conditions or vegetation structure. This demonstrates the importance of acknowledging and incorporatring imperfect detection probabilities, especially if we intend to provide reliable scientific results to inform decision makers.

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Table 3.1. Birds observed actively using edges and/or soybean fields in 2007-2009, Paraná department, Entre Ríos, Argentina, incorporated in the occupancy analyses. Separate analyses were performed for each group, indicated in the first column along with their guilds: ground omnivores and insectivores (GRD), Columbiformes (COL), passerine granivores (GRAN), ground nesters passerine granivores (GRGR), insectivores (INS), and insectivorous aerial foragers and salliers (AER). Subgroups (g1, g2, g3, g4, g5) considered for field borders (Edge) and interior (Crop) analyses are also indicated

Group (guild) [1]	Edge	Crop	Family	Scientific name	Common name
GRD (1)	g2	g1	Charadriidae	*Vanellus chilensis*	Southern Lapwing
GRD (1)	g2	g2	Scolopacidae	*Bartramia longicauda*	Upland Sandpiper
GRD (1)	g2	g3	Tinamidae	*Nothura maculosa*	Spotted Nothura
COL (2)	g1	g1	Columbidae	*Columba maculosa*	Spot-winged Pigeon
COL (2)	g1	-	Columbidae	*Columba picazuro*	Picazuro Pigeon
COL (2)	g1	-	Columbidae	*Leptotila verreauxi*	White-tipped Dove
COL (2)	g2	g1	Columbidae	*Columbina picui*	Picui Ground Dove
COL (2)	g3	g1	Columbidae	*Zenaida auriculata*	Eared Dove
GRAN (3)	g1	-	Cardinalidae	*Saltator aurantiirostris*	Golden-billed Saltator
GRAN (3)	g2	-	Emberizidae	*Paroaria coronata*	Red-crested Cardinal
GRAN (3)	g2	-	Emberizidae	*Poospiza melanoleuca*	Black-capped Warbling Finch
GRAN (3)	g2	-	Emberizidae	*Poospiza nigrorufa*	Black-and-rufousWarbling Finch
GRAN (3)	g2	-	Thraupidae	*Saltaltricula multicolor*	Many-colored Chaco Finch
GRAN (3)	g2	-	Emberizidae	*Sicalis flaveola*	Saffron Finch
GRGR (4)	g1	g1	Emberizidae	*Ammodramus humeralis*	Grassland Sparrow
GRGR (4)	g2	g1	Emberizidae	*Zonotrichia capensis*	Rufous-collared Sparrow
GRGR (4)	g3	-	Emberizidae	*Embernagra platensis*	Pampa Finch
GRGR (4)	g3	g1	Emberizidae	*Sicalis luteola*	Grassland Yellow Finch
GRGR (4)	g3	-	Emberizidae	*Volatinia jacarina*	Blue-black Grassquit
GRGR (4)	g3	-	Thraupidae	*Sporophila caerulecens*	Double-collared Seedeater
GRGR (4)	g3	-	Thraupidae	*Sporophila ruficollis*	Dark-throated Seedeater
INS (5)	g1	-	Troglodytidae	*Troglodytes aedon*	House wren
INS (6)	g2	-	Mimidae	*Mimus saturninus*	Chalk-browed Mockingbird
INS (7)	g3	-	Furnariidae	*Furnarius rufus*	Rufous Hornero
INS (5)	g4	-	Cuculidae	*Coccyzus melacoryphus*	Dark-billed Cuckoo

INS (5)	g4	-	Tyrannidae	*Elaenia parvirostris*	Small-billed Elaenia
INS (5)	g4	-	Tyrannidae	*Lathrotriccus euleri*	Euler's Flycatcher
INS (5)	g4	-	Tyrannidae	*Euscarthmus meloryphus*	Tawny-crowned Pygmy-tyrant
INS (5)	g4	-	Parulidae	*Geothlypis aequinoctialis*	Masked Yellowthroat
INS (5)	g4	-	Tyrannidae	*Myiophobus fasciatus*	Bran-coloured Flycatcher
INS (5)	g4	-	Furnariidae	*Phacellodomus striaticollis*	Freckle-breasted Thornbird
INS (5)	g4	-	Furnariidae	*Phacellodomus sibilatrix*	Little Thornbird
INS (5)	g4	-	Polioptilidae	*Polioptila dumicola*	Masked Gnatcatcher
INS (5)	g4	-	Furnariidae	*Schoeniophylax phryganophila*	Chotoy Spinetail
INS (5)	g4	-	Tyrannidae	*Serpophaga subcristata*	White-crested Tyrannulet
INS (5)	g4	-	Furnariidae	*Synallaxis albescens*	Pale-breasted Spinetail
INS (5)	g4	-	Furnariidae	*Synallaxis frontalis*	Sooty-fronted Spinetail
INS (5)	g4	-	Cuculidae	*Tapera naevia*	Striped Cuckoo
INS (5)	g4	-	Thamnophilidae	*Taraba major*	Great Antshrike
INS (6)	g5	-	Icteridae	*Agelaioides badius*	Bay-winged Cowbird
INS (6)	g5	-	Icteridae	*Molothrus bonariensis*	Shiny Cowbird
INS (6)	g5	-	Icteridae	*Molothrus rufoaxillaris*	Screaming Cowbird
INS (6)	g5	-	Turdidae	*Turdus rufiventris*	Rufous-bellied Thrush
INS (6)	g5	-	Turdidae	*Turdus amaurochalinus*	Creamy-bellied Thrush
AER (8)	g1	g1	Tyrannidae	*Tyrannus savanna*	Fork-tailed Flycatcher
AER (8)	g2	-	Tyrannidae	*Tyrannus melancholicus*	Tropical Kingbird
AER (8)	g2	-	Tyrannidae	*Pitangus sulphuratus*	Great Kiskadee
AER (8)	g2	-	Tyrannidae	*Pyrocephalus rubinus*	Vermilion Flycatcher
AER (8)	g2	-	Tyrannidae	*Xolmis irupero*	White Monjita
AER (9)	g3	g1	Hirundinidae	*Petrochelidon pyrrhonota*	Cliff Swallow
AER (9)	g3	g1	Hirundinidae	*Phaeoprogne tapera*	Brown-chested Martin
AER (9)	g3	g1	Hirundinidae	*Tachycineta leucorrhoa*	White-rumped Swallow

[1] *Guilds:* 1 = Ground omnivores; 2 = Ground granivores; 3 = Granivore foliage gleaners; 4 = Ground nesting granivore foliage gleaners; 5 = Insectivorous foliage gleaners; 6 = Ground and foliage omnivores; 7 = Ground insectivores; 8 = Insectivorous salliers; 9 = Insectivorous Aerial foragers.
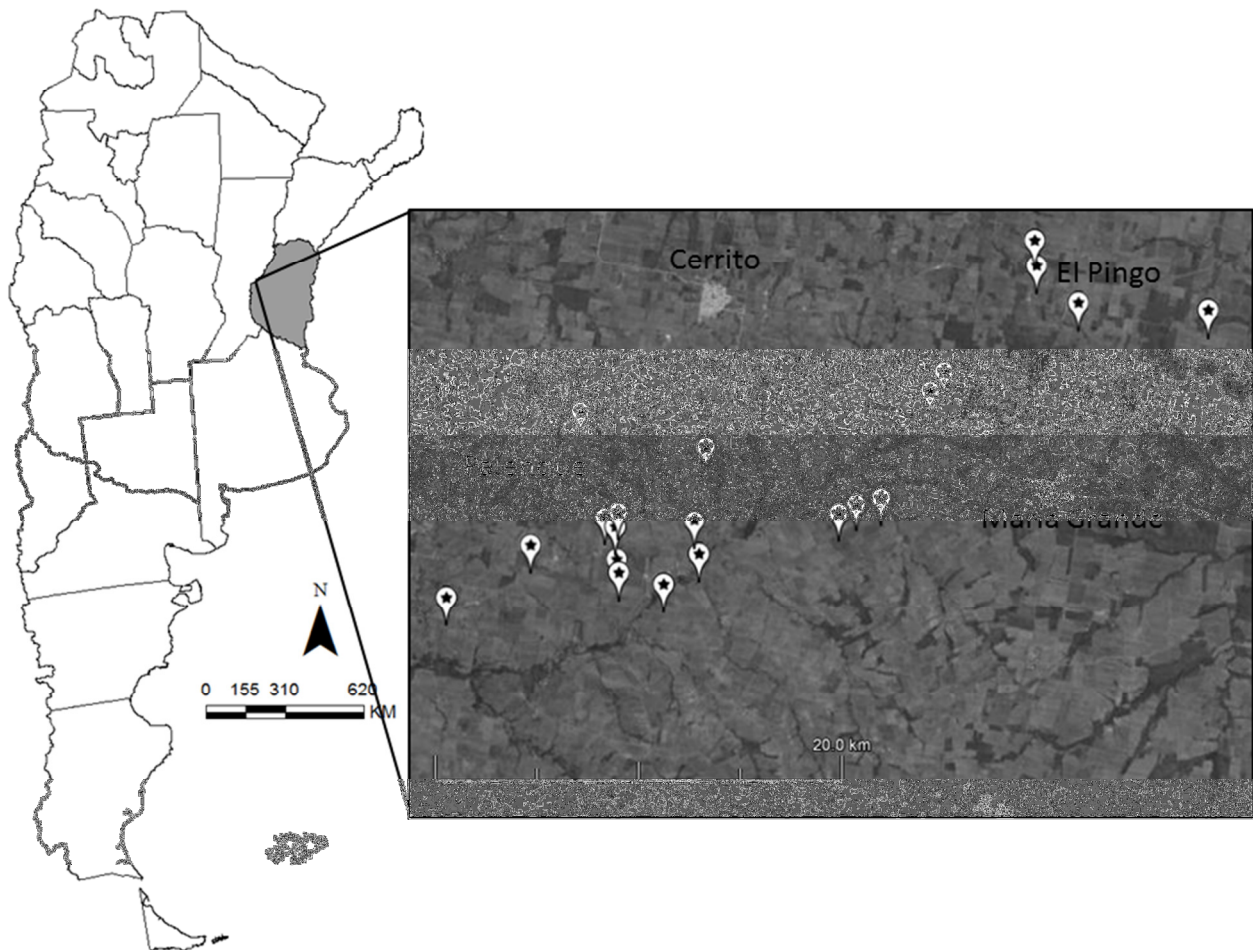
Figure 3.1. Location of soybean fields and borders where bird and arthropod sampling was conducted in 2007-2009, Paraná department, Entre Ríos, Argentina; near the towns of Cerrito, El Pingo, María Grande, and Palenque (GOOGLE EARTH 2013).
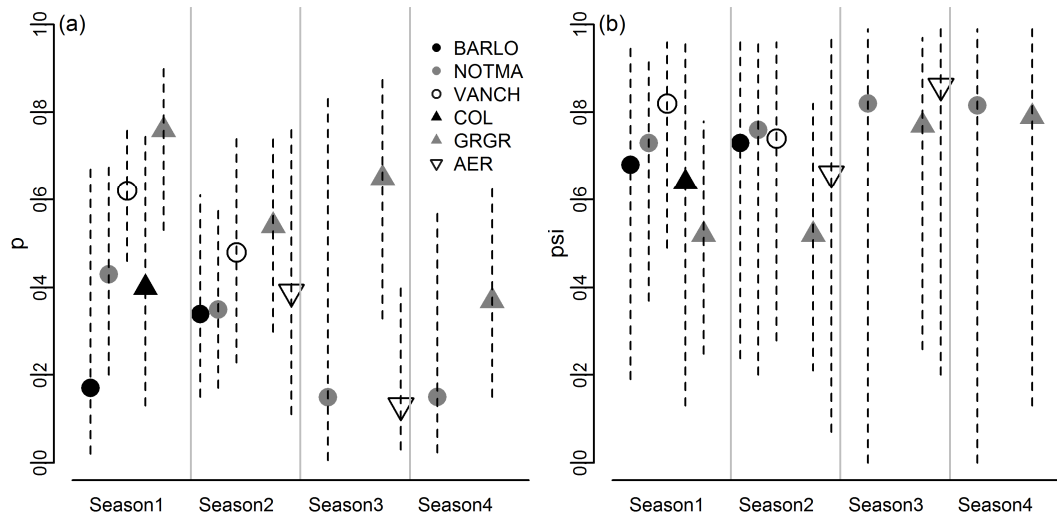
Figure 3.2. Probability of (a) detection ($\hat{p}\pm$95%CI) and (b) occupancy ($\hat{\psi}\pm$95%CI) of different groups, in the interior of soybean fields at different seasons (S1=Season 1, S3=Season 3), 2007-2009, Paraná department, Entre Ríos, Argentina. Details for species in each group can be found in Table 3.1 (BARLO= Upland Sandpiper, NOTMA= Spotted Nothura, VANCH= Southern Lapwing, COL=Columbiformes, GRGR= Ground nesting granivore gleaners, AER= Aerial foragers).
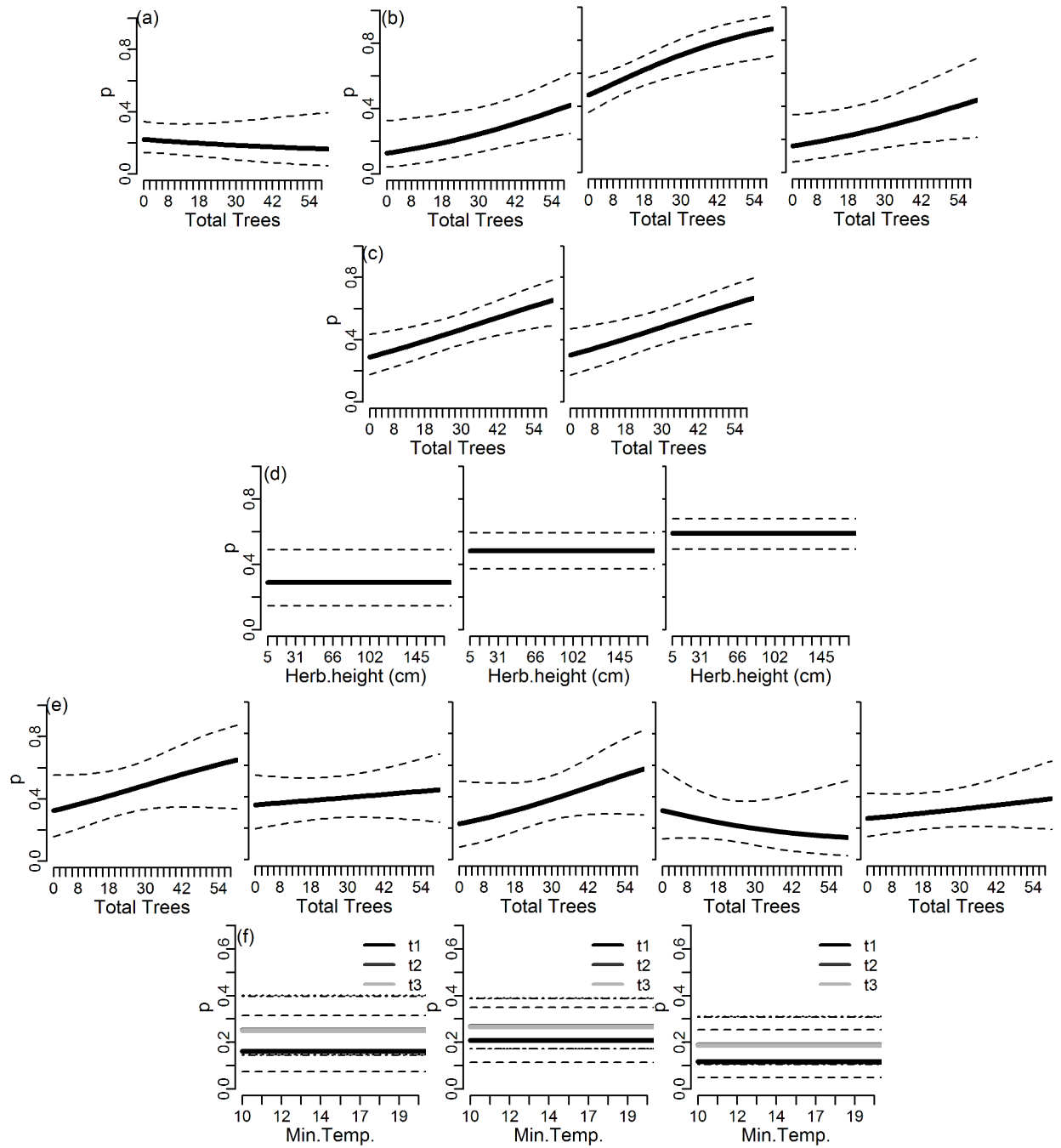
Figure 3.3. Probability of detection ($\hat{p}\pm$95%CI) of birds at borders of soybean fields by the most influential covariate for each group, in Paraná department, Entre Ríos, Argentina, 2007-2009. (a) Ground omnivores; (b) ground granivores: Columbiformes, Picui Ground Dove, Eared Dove (left to right); (c) granivore foliage gleaners, and Golden-billed Saltator (left to right); (d) ground nesting granivore foliage gleaners, Grassland Sparrow, Rufous-collared Sparrow (left to right); (e) insectivore foliage gleaners, House Wren, ground and foliage omnivores, Chalk-browed Mockingbird, Rufous Hornero; (f) insectivorous salliers, Fork-tailed Flycatcher, aerial foragers (left to right), in three seasons separately: t1 (black, dash CI), t2 (grey, dot CI), t3 (light-grey, dash-dot CI). More details for species in each group can be found in Table 3.1
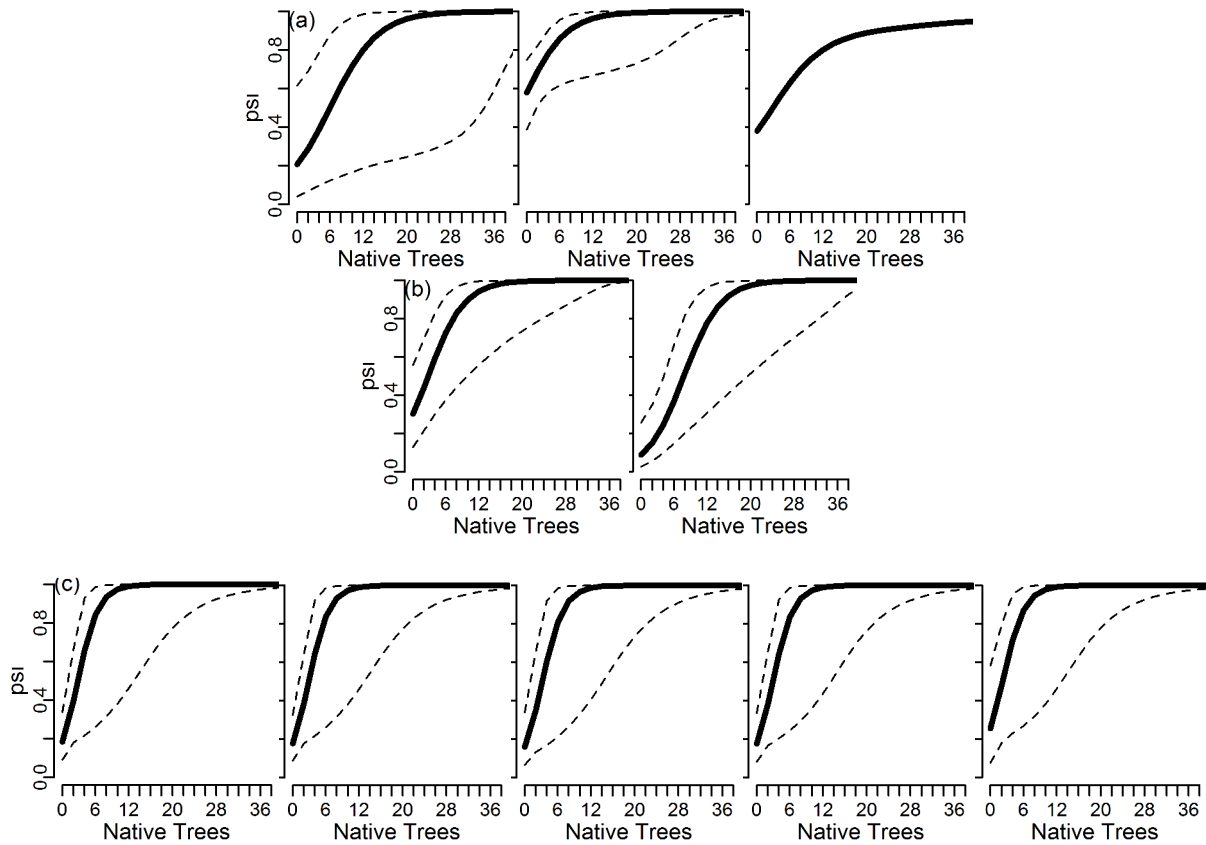
Figure 3.4. Probability of bird occupancy ($\hat{\psi}\pm95\%$CI) by the number of native trees at borders of soybean fields, in Paraná department, Entre Ríos, Argentina, 2007-2009. (a) Ground granivores: Columbiformes, Picui Ground Dove, Eared Dove (left to right); (b) granivore foliage gleaners, and Golden-billed Saltator (left to right); (c) insectivore foliage gleaners, House Wren, ground and foliage omnivores, Chalk-browed Mockingbird, Rufous Hornero (left to right). More details for species in each group can be found in Table 3.1
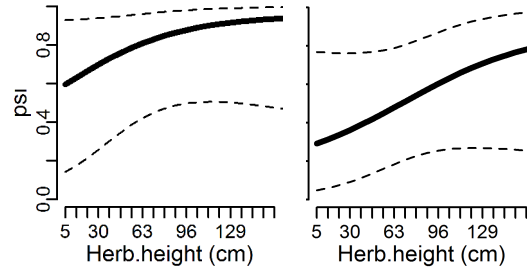
Figure 3.5. Probability of bird occupancy ($\hat{\psi}\pm$95%CI) by the grasses height at borders of soybean fields in Paraná department, Entre Ríos, Argentina, 2007-2009. More details for species in each group can be found in Table 3.1
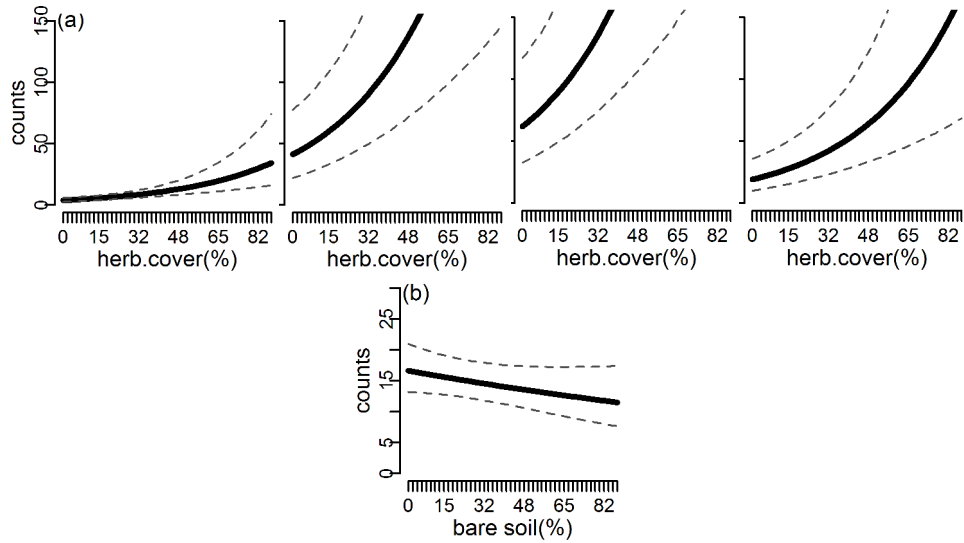
Figure 3.6. Counts ± 95%CI of main arthropod orders consumed by birds at the interior of soybean fields, in Paraná department, Entre Ríos, Argentina, 2007-2009. (a) Collected with sweep net, by herbaceous vegetation coverage, seasons 1 to 4 (left to right); (b) Collected with vacuum, by bare soil coverage, all seasons equal.
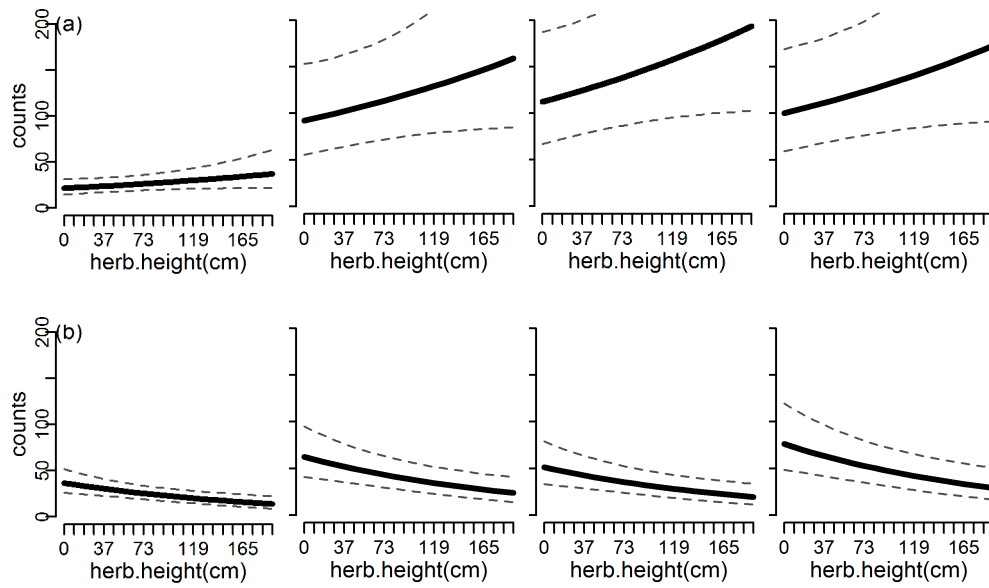
Figure 3.7. Counts ± 95%CI of main arthropod orders consumed by birds by height of herbaceous vegetation at the borders of soybean fields, at seasons 1 to 4 (left to right), in Paraná department, Entre Ríos, Argentina, 2007-2009. (a) Collected with sweep net; (b) collected with vacuum.
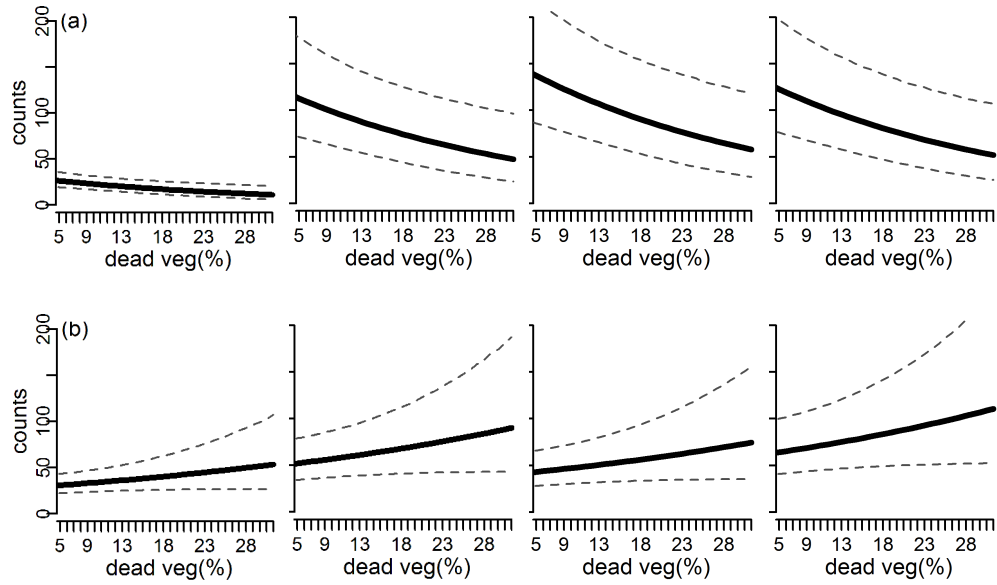
Figure 3.8. Counts $\pm$ 95%CI of main arthropod orders consumed by birds by percent of dead vegetation at the borders of soybean fields, at seasons 1 to 4 (left to right), in Paraná department, Entre Ríos, Argentina, 2007-2009. (a) Collected with sweep net; (b) collected with vacuum.

CHAPTER 4

A CONSERVATION DECISION MAKING APPROACH TO INTEGRATE BIRD

DIVERSITY WITH VALUES OF AGRICULTURAL PRODUCERS IN ENTRE RÍOS,

ARGENTINA [1]

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[1] A. P. Goijman, M. J. Conroy, and M. E. Zaccagnini. To be submitted to Biological Conservation

ABSTRACT

In the face of human population growth one of the biggest challenges of modern agriculture is to reconcile food demands with environmental sustainability and social well-being. At a local scale, some of the actions to integrate biodiversity concerns in agricultural landscapes consist of maintaining the heterogeneity within fields offering suitable habitat for biodiversity, and also managing the amount of agrochemicals used in crops and their toxicity. Specifically, birds can offer ecosystem services to agriculture such as pest control or pollination, and vegetated linear habitat can greatly benefit insectivorous birds in an agricultural matrix. Using decision analysis to integrate natural resources problems with social values and needs can be a powerful tool to reach consensus on the different interested parties. This approach, also known as structured decision making (SDM) makes the selection of optimal management alternatives on complex problems a transparent and well informed process. We worked on an agricultural landscape dominated by annual crops and grazing lands, still holding native forests patches, in Entre Ríos Argentina, to integrate bird conservation with soybean agriculture. We delineated a set of 12 management alternatives to achieve the maximization of insectivorous birds and farmers' well-being, while minimizing the cost of the management actions. The optimal management alternative varied according to the considered constraints, mainly depending on how much of the cost was going to be assumed by the producers. Both optimal decisions shared the reduction of insecticide applications in soybean production, and the use of least toxic products as an alternative, and varied between planted native trees or leaving the borders unmanaged.

KEYWORDS: Structured decision making, adaptive management, soybean, agriculture, birds, ecosystem services, insecticides

INTRODUCTION

With the growth of the human population projected at 9 billion people by mid-century, one of the biggest challenges presented today is to meet the food demands while ensuring environmental sustainability, human health, and economic and social well-being (Godfray et al. 2010, Sachs et al. 2010). There is a demand for science evaluating the trade-offs between optimizing agriculture in a sustainable way, while reducing biodiversity and habitat losses, conserving ecosystem services, and minimizing water pollution from agricultural chemicals, among other challenges (Godfray et al. 2010, Foley et al. 2011). For example, conventional agricultural intensification could result in contamination by agrochemicals affecting not only functional agrobiodiversity but also human health and environmental quality, especially in landscapes dominated by medium and small-scale agricultural producers.

There are different scales at which land management can be done integrating biodiversity concerns in agricultural landscapes in the design of agri-environment schemes. At a local scale, actions mainly consist of maintaining spatial and temporal heterogeneity within fields, offering nesting ground, food resources and refuge for biodiversity. Other actions consist of conserving natural vegetation on borders, implementing terraces, or strip cover vegetation within fields, and also managing agrochemical applications. Maintaining bird diversity in agricultural landscapes could be beneficial through the ecosystem services they provide, such as consumption of insect pests and pollination (Millennium Ecosystem Assessment 2005, Sekercioglu 2006, Whelan et al. 2008). Vegetated linear habitats such as borders and terraces could provide habitat for birds (Goijman and Zaccagnini 2008, Di Giacomo and de Casenave 2010, Chapter 3). Although herbicides and insecticides use is concentrated in crops, edges could be subject to drift, diminishing available refuges for birds by changing vegetation structure, and reducing the

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availability of food resources (Boutin et al. 1999, Jobin et al. 2001, Cederbaum et al. 2004, Jones et al. 2005, Taylor et al. 2006). However, studies in soybean fields showed that predator and total arthropod abundance remained constant in borders through time, despite pesticide applications; this suggests that if these habitats are treated carefully, they can still sustain bird insectivore populations (Weyland and Zaccagnini 2008, Varni 2010, Chapter 3). In addition, in some areas there is a tendency to replace toxic insecticides (e.g. for birds), to less harmful ones (Mineau and Whiteside 2006).

Conservation decision making can be achieved by using a structured decision making framework (SDM) to solve natural resource problems, which consists of organizing a problem using the tools provided by the decision analysis. Decision analysis offers structure and guidance for thinking systematically about difficult problems to solve; it does not recommended a single management alternative that should be blindly accepted, but rather assists the decision maker in understanding the situation in depth, making the process transparent and participatory (Clemen and Reilly 2001, Gregory et al. 2012, Conroy and Peterson 2013). Once a decision has been selected and accepted, an additional step, adaptive resource management (ARM) process may be implemented. ARM requires prediction of outcomes under alternative structural models, application of the best management alternative selected by weighting over the uncertainty between models, and monitoring of the results of their application, incorporating those into a new cycle of decision analysis in order to learn about the system and reduce structural uncertainty (Clemen and Reilly 2001, Williams et al. 2009, Gregory et al. 2012, Conroy and Peterson 2013). The SDM process addresses natural resource management problems complementing science with values and needs of stakeholders, and policies and laws, in an

informed, rigorous, and multidisciplinary fashion, facilitating decision making in conservation with multiple or competing objectives, and under the presence of uncertainty.

In this chapter, we will use SDM to integrate biodiversity conservation, focused on birds, with producers' values and needs, in order to evaluate the tradeoffs between soybean cultivation, environmental sustainability, and economic and social well-being of small and medium-scale producers. We aim to identify key uncertainties and suggest the ways that those can be reduced, proposing an ARM framework for future monitoring of the system. The ultimate goal is to develop agricultural management systems that promote sustainable agriculture compatible with conservation of biodiversity, and producers' rural lifestyles. We place emphasis on those components of biodiversity that provide ecosystem services, adding a fundamental link between agricultural production and conservation of biodiversity.

Decision Context

The area of interest consists in an agricultural landscape dominated by annual crops and grazing lands at the north-central region of Paraná department, in Entre Ríos province, Argentina. Originally, this area supported *Espinal* forest which is also known as *"Selva de Montiel"*, dominated by *Prosopis affinis, Acacia caven, Geoffroea decorticans, Celtis tala,* and *Schinus longifolia* (Cabrera 1971, Cabrera and Willink 1973, Burkart et al. 1999). Although there are areas with good conservation value, the expansion of agriculture is rapidly degrading the native forest, and some sectors are currently dominated by *A. caven* (Sabattini et al. 2002). Row agriculture is the main productive activity in the area, followed by cattle production (Zaccagnini et al. 2008, unpublished). The main crop in Paraná department is soybean, with 184,500 ha sown in 2010/11, representing 37% of the department area; followed by wheat (10%) and corn (4.5%)

(SIIA 2013). Many farmers in the area consider that agricultural activities generates risks to the people and environment, and those risks are associated with the improper use of agrochemicals, followed by biodiversity loss, and health problems; and also recognize that pollution could affect their future rural lifestyle (Appendix G). They also acknowledge the value of biodiversity, especially insectivorous birds, as necessary for the ecosystem.

METHODS

Formulating objectives

We based the objective formulation on expert opinion and our knowledge of producers' needs and opinions in the study area, together with information about producers' economic, social, and ecologic-environmental values and needs gathered through interviews in the study area (Appendix G). The *fundamental objective* was identified as the achievement of sustainable agriculture compatible with conservation of biodiversity. To accomplish this objective it was decomposed it in three lower-level fundamental objectives (Clemen and Reilly 2001): (1) maximize insectivorous bird presence in the landscape; (2) maximize agricultural producers' well-being; (3) minimize cost of management actions.

The first lower-level objective was based on the fact that birds are ecological indicators as they are easy to monitor and quantify, responsive to changes, among other characteristics (Gregory et al. 2003). Insectivorous birds are indicators of habitat and resource availability; besides, while considering different guilds with specific habitat requirements, many patterns can be evaluated (Niemi et al. 1997). This group offers a valuable ecosystem service in agricultural landscapes, and is valued by producers (Appendix G). The consumption of insect pest by birds has been documented, but its economic value has not been successfully quantified in most cases

(Kirk et al. 1996, Tremblay et al. 2001, Whelan et al. 2008, Philpott et al. 2009); only in some studies in organic farms and coffee plantations (Jones et al. 2005, Kellermann et al. 2008).

We incorporated farmers' well-being as another fundamental objective to achieve sustainable agriculture because, in this area, there are no government incentives or regulations promoting conservation of native habitats for biodiversity, and ecosystems services, with the exception of soil conservation measures to avoid erosion (i.e. Provincial law n. 8318). Hence, to persuade producers to implement management practices that promote sustainable agriculture, we must maximize their well-being, and minimize management costs.

An objective network helped us visualize the ways of achieving the fundamental objectives, via the *means objectives* (Fig. 4.1). We limited the context of this decision problem to soybean production, because it is the main crop in the area. Then, to maximize insectivorous birds' presence, we want to maximize their occupancy in soybean fields and borders. In addition, soybean yield is another means objective, affecting farmers' well-being, and so is pollution which affects farmers who have rural lifestyles, or live in cities near the country (Appendix G). Pollution is an externality which could affect farmers' returns in the long run by affecting soil fertility, residues on food, water quality and health; it can also contribute to loss of biodiversity affecting returns in the future (Norris and Kogan 2000, Jergentz et al. 2004, 2005, Arregui et al. 2010, Birch et al. 2011).

Decisions

We built an influence diagram delineating key management and environmental variables with effects on the means and fundamental objectives, and identified possible decisions (Fig. 4.1) (Clemen and Reilly 2001). We addressed management alternatives decisions at within-field scale

to maximize bird occupancy and farmers' well-being, while minimizing costs. Bird occupancy in soybean borders could be affected by the number of native trees (see Chapter 3); and bird occupancy in center could be altered by insecticide applications, and field size, and distance to forest (Best et al. 1990, Boutin et al. 1999, Jones et al. 2005, Taylor et al. 2006, Di Giacomo and de Casenave 2010). Pollution could be affected by field size, insecticide applications, and toxicity; and soybean yield can be subject to insecticide applications and birds in the crop.

We generated a set of 12 decisions by combining the management alternatives (Table 4.1, Fig. 4.2). The first management alternative consisted on leaving a non-sown strip of natural vegetation within the field, which could provide habitat for some birds and increase the edge to area ratio of the soybean fields. This could as well be a terrace, practice commonly used in Entre Ríos to avoid soil erosion from water runoff. Terraces have been shown to increase bird diversity of some species when the vegetation is not eliminated as well as arthropod richness (Goijman and Zaccagnini 2008, Weyland and Zaccagnini 2008, Solari and Zaccagnini 2009). Increasing edge availability and decreasing field size could also increase bird occupancy per soybean hectare and benefit species that feed closer to the borders and grassland species (Best et al. 1990, Puckett et al. 2009, Di Giacomo and de Casenave 2010). Despite the vast number of studies of birds on linear habitats in agricultural landscapes (Macdonald and Johnson 1995, Goijman and Zaccagnini 2008, Puckett et al. 2009, Solari and Zaccagnini 2009, Di Giacomo and de Casenave 2010, Blank et al. 2011), there is no agreement on a single type of border suitable for all bird species regarding width, vegetation structure, and composition (Hinsley and Bellamy 2000, Vickery et al. 2009). For example, a program within the USDA Continuous Conservation Reserve Program was designed to provide incentives for substituting herbaceous buffers for crop field margins to promote habitat for upland birds. It was found that buffers with a minimum of 9

m wide could be used without compromising crop production objectives in soybean (Stamps et al. 2008). Other studies showed that vegetation height and structure are fundamental for bird use in narrow borders (2-5 m; Macdonald and Johnson 1995, Harvey et al. 2005). Width is critical in borders with less complex vegetation structure, and the minimum evaluated in some studies is 9-10 meters (Stamps et al. 2008, Blank et al. 2011). To illustrate our decision situation, we will assume 9 m as the optimal strip width to benefit birds without compromising soybean yield (Stamps et al. 2008).

Managing native trees in the borders is another component of the decisions; as evidenced in Chapter 3 the number of native trees highly benefits birds in this area. This decision could be classified in three levels: plant trees, keep only the pre-established trees, or leave borders unmanaged. Common tree species in the borders, characteristic from *Espinal* forests in this region in Entre Rios, are *Acacia caven, Prosopis affinis,* and *Celtis tala,* among others (Cabrera 1971, Cabrera and Willink 1973). *A. caven* is abundant and usually farmers dislike it for that reason and for their perception of unplanted vegetation as 'trash' (pers. obs.), also it lacks economic value (Sabattini et al. 2002). *P. affinis*, and *Prosopis nigra*, are native and have a potential economic value; thus those species would be the suggested for planting (Burkart 1976, Sabattini et al. 2002). There is evidence that 2-5 month old seedlings of *Prosopis sp.*, had a survival of 50-80% with no artificial irrigation, with the maximum survival in plots protected from predators (Catalan et al. 1994, Arredondo et al. 1998).We suggested 30 trees per 500 m of border – average in the study area – as the number of trees to plant, with 50% survival rate there will be 15 trees per border by the next time step. Most farmers agreed they would allow 10-15 trees in a border (Appendix G), and this number of trees benefits birds as well (Chapter 3). The next level of border management is to keep the trees that are already established, but eliminate

new seedlings, which is common because many times herbicides are applied in the borders. The last action in borders is to leave the border unmanaged, allowing new seedlings to establish naturally. We predict that by leaving the border unmanaged, this will mainly result in the establishment of *A. caven*, which is the most common tree species in the borders (pers. obs.), and a fast colonizer with high survival rate (Arredondo et al. 1998). We have no information on how many new trees would establish and survive to the next time-step; hence, we assume for illustration purposes rule that there would be one more tree per pre-established tree.

The last alternative consists of controlling how many and which insecticides are applied to control pests in soybean. Here, we recommend a maximum two insecticide applications, and to use insecticides with low toxicity for humans and birds (Bohmont 2007, Arregui et al. 2010). Two applications is the average that farmers reported using (Appendix G), supported by our findings at the study fields during the bird sampling period. Most applications in this area are intended to control the velvetbean caterpillar, stink bugs, and a lower percentage for other pests like thrips or borers (Saluso et al. 2005, Saluso et al. 2007). Controls for velvetbean caterpillars are performed before the flowering stage, and after for stink bugs. Therefore, if correct threshold monitoring is performed, two applications could be sufficient to control pests. It is also essential the use of insecticides with low toxicity for birds, since bird conservation is one of the goals. We estimated the risk of acute mortality for birds exposed to the insecticides used in the same soybean fields where the bird monitoring took place (Chapter 3), and found that approximately half were highly toxic for birds (Table 4.2) (Mineau 2002, Bernardos et al. 2007).

We incorporated flexibility in the search of the optimal decisions by considering different constraints of different percentages of borders to be managed, and percentages of costs taken by

farmers for planting trees considering some stated they would agree on assuming half the cost of tree planting (henceforth, 'constraints'; Appendix G).

Birds

We parameterized the decision network based on empirical occupancy estimates of insectivorous birds (see Chapter 3). We chose four guilds of insectivorous birds likely to prey on insect pests, each with different habitat preferences and foraging behavior, allowing the representation of a wider spectrum of ecological groups. The guilds represented were insectivorous salliers (Tyrannidae), aerial foragers (Hirundinidae); also insectivorous foliage gleaners and ground insectivores, which prefer borders (Appendix F) (Remsen and Scott 1990). We incorporated bird occupancy model weights for edges and interior of soybean fields in the decision network, accounting for structural uncertainty (Williams et al. 2002, Conroy et al. 2008, Conroy and Peterson 2013). At the interior of the fields model incorporated the covariates: null, distance to forest, flowers present, size of the fields, and insecticide applications. At the border: number of native trees, and distance to forest.

In order to parametrerize the bird conservation node, we pooled occupancy estimates from borders and center, and combined them with the effects of insecticide toxicity used in soybeans. Since borders can provide suitable habitat for birds, for illustration purposes, we assumed occupancy in this habitat may have twice the conservation value than in the center of the fields. We considered the risk of acute mortality for birds by the insecticides (Table 4.2) by multiplying the 'bird' node by the average probability of survival estimated using the "Bird eco-toxicologic risk calculator" (Bernardos et al. 2007) (Table 4.2). For example:

$$Birds(center = yes; \ border = yes; toxicity = high) = (1 + 2) * (1 - 0.84) \qquad (1)$$

Cost

Each decision has a cost associated according to the combination of components from border management, strip of vegetation, and insecticides management. The cost is subject to the percentage of border to be managed, and the percentage farmers would pay for planting native trees. Leaving unmanaged borders has no associated cost. Keeping only pre-established trees has the cost of applying herbicide in the borders; for which we roughly assumed the average dose of a common herbicide is 2 l/ha, and a maximum cost of $10/liter. The cost of planting new trees is subject to the preexisting number of trees, and the cost of new ones. We estimated from expert opinion and web searches, that the cost of planting a tree would be approximately $4 per tree.

We calculated cost of planting a strip of vegetation as the monetary loss in soybean yield per hectare in the area devoted to the new strip of vegetation. Gross margin per hectare in Entre Rios in 2011 ranged from $384-$655, varying if the land was owned or rented (Rodriguez et al. 2011). Finally, the cost of insecticide applications was estimated as an average of the prices of different insecticides products per hectare ($3.15) multiplied by the size of the field. The decision to reduce insecticide was set to two applications, and under the no-management alternative the applications remained as the original number in each field.

Well-being

We determined pollution control and soybean yield as key elements to maximize farmers' wellness. Quantifying pollution by insecticides would require the consideration of several variables such as product, dose, number of applications; also effects on soils, water, and biodiversity, and how it could affect returns in the long run (Norris and Kogan 2000, Wilson and Tisdell 2001, Jergentz et al. 2005, Arregui et al. 2010). We quantified pollution accounting for

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toxicity at each dose used, the number of applications, and size of the fields. We considered toxicity as the risk of acute mortality for birds (low or high), which could eventually affect humans (Table 4.2), and assumed that insecticides with the lowest toxicities could affect pollution by 50%, and 100% for those highly toxic. Based on mixtures toxicology, we assumed a linear relationship of the number of applications of insecticides, the size of the fields, and the amount of residues on the environment.

Soybean yield is affected by the number of insecticide applications, and could be affected by bird occupancy in the interior of the field. Evidence suggests that yield is affect very little or not at all by vegetated borders – contrary to farmers' thoughts (Stamps et al. 2008). We measured yield as kg ha^{-1}, and assuming the lack of effect of field size. An experimental study in small fields, found that the gross return per hectare in soybean plots with three insecticide applications was three times larger than those with none (Tung and Fernandez 2007). Other study, under similar conditions to ours, compared yields under control and one insecticide application (lambda-cyhalothrin) and fungicide, showing a 8% increase in yield when insecticide was applied (Henry et al. 2011). Soybean yield in Entre Rios is 2300 kg ha^{-1} (Rodriguez de Rodriguez and Cancio 2011), with 2.5 pulverizations as an average in the region, and a linear relationship, we assumed the following relationship:

$$Yield = 1840 + Ins * 184 \tag{2}$$

There is some evidence on bird predation on insect pests in orchards and coffee plantations (Jones et al. 2005, Kellermann et al. 2008); however, there is no evidence on the economic impact that they could have on soybean. Then, as an illustration, we considered that birds could benefit soybean yield in 1% of its yield.

Yield and pollution measurements were re-scaled using proportional scoring (eq. 5); and assigned weights to the importance that each one has for local farmers. 65% of the farmers having a rural lifestyle (63%) think pollution could affect their future lifestyle. In general, 47% think improper use of pesticides, fungicides and pollution in general are a threat for people and the environment. We estimated the importance of pollution as a weighted average, and incorporated it in the "well-being" node:

$$Wellbeing_i = pollution(x_i) * 0.58 + yield(x_i) * (1 - 0.58) \tag{3}$$

Optimal decisions

We parameterized the various components and modeled their relationships via probabilistic networks, or Bayes networks (e.g. Conroy et al. 2008, Conroy and Peterson 2013). To perform calculations we used Netica (www.norsys.com), which provides a convenient graphical representation of decision problems and also readily allows for Bayesian updating.

We built the utility function combining information from each of the fundamental objectives: bird conservation, well-being, and cost of management alternatives. To calculate the expected value of each decision, Netica provides calculations by the uncertainty-weighted outcome values (Conroy and Peterson 2013):

$$E(D_j) = \sum_x U(x_i) * p(x_i) \tag{4}$$

where D_j represents each alternative decision, $U(x_i)$ is the utility value of the combination x_i of each level of the components of the utility node, and $p(x_i)$ is the probability associated with each outcome x_i. Thus, the expected value of a decision is the sum of the utility values of each outcome, weighted by its probability of occuring.

Since each of the objectives are measured in different units, we used proportional scoring to provide unitless measures (Clemen and Reilly 2001). For each component of the utility function c (i.e. cost, well-being, birds) with its levels i the utility is calculated as:

$$U(c_i) = \frac{[y_i - worst(y_i)]}{best(y_i) - worst(y_i)} \tag{5}$$

where y_i is the measurement on the original attribute scale and *worst* and *best* and are the least and most desired values of the attribute over the anticipated range.

In order to assign the weight (i.e. probabilities) each component has in the utility function we used the indifferent scoring, also known as the pricing out approach (Hammond et al. 1999). The approach is based on estimating how much are we willing to give up on the utility value of some component to achieve a particular gain on another one. Here, we asked professionals how much loss of birds they would be willing to accept to reduce costs in 0.1 value, and increase well-being the same quantity.

We also assigned random weights (sum of weights constrained to 1) to each component in the utility function as an exercise to evaluate the influence of cost, birds and well-being on the selection on the optimal decision alternative, and their expected values. We simulated 50 combinations under two constraints: farmers paying 0% and 100% for tree planting and managing 50% of the borders.

Sensitivity analysis and reduction of uncertainty

A critical step on decision making is to examine the sensitivity of the model output and decisions to variation in parameters. We developed one-way sensitivity analysis to identify which variables have the greatest influence towards achieving the fundamental objectives and the optimal decision, and identify key uncertainties for future allocation of monitoring effort. We assessed

the variation in the probabilities of obtaining high well-being, high bird conservation, and lower costs, evaluating them at the maximum and minimum values of the different model components. We built tornado diagrams which are a handy way to evaluate how sensitive the fundamental objectives are to variations on different variables. Tornado diagrams allow comparison of several variables at once, and represent the extent to which the probabilities of obtaining the desired outcomes are sensitive to variation between high and low values of each input (Clemen and Reilly 2001, Conroy and Peterson 2013). We also performed one-way sensitivity analysis on the optimal decision to evaluate how the model components influences its expected value.

We incorporated model (a.k.a. structural) uncertainty using several bird occupancy models with their respective weights, and we illustrate how uncertainty could be reduced through an updating process using Bayes theorem, using a hypothetical simulated example. We simulated a new new data set assuming surveys on 78 new borders (n= 78) over four repeated times (k=4). We also simulated the covariates of number of native trees and distance to forest. We updated the posterior weights of each occupancy model in the decision network as follows:

$$p(m_i|x) = \frac{p(x|m_i) \times p(m_i)}{\sum_1^i p(x|m_i) \times p(m_i)} \qquad (7)$$

$p(m_i|x)$ is the posterior probability of "model$_i$" (model weight), where i=1 to 5 are models in the case of borders. "x" is the data to estimate occupancy; and $p(x|m_i)$ is the likelihood of the new data collected under model$_i$. Last, $p(m_i)$ is the prior weight of model$_i$. Through this updating process, the re-parameterization of the network is not required, it is sufficient to incorporate the new model weights to reduce uncertainty.

Finally, as an updating process, we recommend to consider a time step of three years, which is the minimum time needed for new seedling to become young trees, even if they need more time to reach full maturity (Burkart 1976, Ortiz Silva 1990).

RESULTS

Optimal decisions

Indifferent scoring.- Under this approach, the model weights in the utility function were assigned considering we were indifferent to lose 0.18 on the value of birds (in a 0-1 scale) in order to reduce cost, or increase well-being on 0.1. The resulting model weights in the utility function were:

$Utility\ (Birds_i, Cost_j, Well\_being_k)$

$$= 0.217 * U(birds_i) + 0.391 * U(cost_j) + 0.392 * U(well\_being_k)$$

Assuming uniform probabilities on the percentage to pay for planting trees, and on the percentage of border to manage, the best decision alternative was to reduce insecticide applications and leaving the borders unmanaged (D4), followed closely by the decision of reducing insecticides and planting trees (D5) (Fig. 4.3). On the other hand, under the constraint where the farmers pay zero to plant trees, independently from the percentage of border to manage, the optimal is D5 (Fig. 4.4). However, if the farmers were to pay 50% of the trees planted, and manage more than 50% of the border the best is D4. This decision is also the optimal when producers are to pay 100% of the cost, except the situation where 25% of the border is managed.

Random weighing.- Randomly assigning weights to cost, bird conservation, and well-being in the utility function represented, under different constraints, the influence of each fundamental objective on the expected values of the decisions. As a result, cost was the most influential variable, and the expected value of the decisions increased with cost weights (Fig. 4.5 a-b). Overall, the best decision was to plant trees and reduce insecticide applications (D5), followed by leaving borders unmanaged and reducing insecticides (D4). When the percentage to pay was set to zero the optimal decision was D5, 78% of the times, and 2% for D4. However, D5

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was chosen 36% of the times when it payment was set to 100% and 50% D4. With weights greater than 0.3, the best decision is D4 (100% pay; Fig. 4.5 b). Introducing a strip of natural vegetation, planting trees and reducing insecticides (D2) was the best decision 20% and 14% (pay 0% and 100%, respectively) only when the weight on costs was low. The expected values of the decisions tend to decrease with increasing weights on bird conservation and well-being; however, the relationship with the best decisions was weak (Fig. 4.5 c-f).

Reducing uncertainty

Sensitivity analysis.- The most influential variable at reducing cost was the size of the fields (Fig. 4.6 a), while the toxicity of insecticides, and number of trees in the borders affected high bird conservation value, in addition to model uncertainties (Fig. 4.6 b). Achieving high well-being was most sensitive to several model components such as pollution and yield; also size, and cost (Fig. 4.6 c). On the other hand, the decision most frequently selected as the optimum, planting trees and reducing insecticides (D5), was mostly affected by soybean yield (Fig. 4.7).

      Structural uncertainty update.- We illustrated how, using Bayes rule (equation 7), we can update posterior model weights reducing model (i.e.structural) uncertainty. To update the information we used the prior model weights from the empirical study, and the likelihood of a newly simulated dataset under each model. In this illustration, the new weights supporting the model that incorporates native trees effects and different species (group) are the highest, reducing uncertainty of models, which prior to the update, had similar weights for the native trees effects and bird groups, and the model with only native trees effects (Table 4.3).

DISCUSSION

We identified the management of field borders, insecticides, and field sizes as the components in the decision alternatives to maximize insectivorous birds conservation, and producer welfare, while minimizing management costs. The combination of the different levels of each component was evaluated under different constraints of cost allocation and percentage of border management. All three elements of the decision are important while planning an agri-environment scheme to mitigate impacts of agriculture and increase biodiversity conservation; not only by providing habitat for birds in borders, but also by improving the environmental value of the cropped area, controlling insecticide use. Under all the alternative constraints, reducing the number of insecticide applications in the cultivation of soybeans and using less toxic products, was the best decision. In most cases, the implementation of a strip of vegetation in the middle of the field, was not selected as a good alternative because of the high cost of its implementation. Finally, the most variable component was managing the borders of soybean fields, where the alternative of no-management, or plant native trees were selected depending on the constraints. In general, the optimal decision was subject to the percentage of costs that producers were expected to pay to plant trees, and the percentage of edge to be managed.

The optimal decision regarding insecticide management, was the reduction of applications to a maximum of two, and choosing less toxic products. As a rule, the decision to apply insecticide must be made after a threshold monitoring, and expert advice; and it is essential to follow the recommended dosage on the product label. In this study, we recommend a maximum of two applications for the control of major soybean pests in Paraná, Entre Ríos; velvetbean caterpillar is controlled before flowering, and the stink bugs then (Saluso et al. 2005, Saluso et al. 2007). Other pests as thrips are controlled in the vegetative and reproductive stages,

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but they are usually of less concern. It is essential to use insecticides with low toxicity levels for

humans and birds, however, we found that almost 50% of the insecticides used were higly toxic

to birds (Bernardos et al. 2007). Endosulfan is a very toxic organochlorine that is banned or

slowly phased out from most countries, it has the highest toxicity level which is estimated to be

fatal to humans at a dose of less than 5 grams (EPA 2012). This product is in a step process of

being eliminated from Argentina by 2015 (SENASA 2010), but it was used a 30% of the time in

the studied fields. 63% of the insecticides were moderately toxic and only 37% were slightly

toxic (Bohmont 2007, EPA 2012). Although pesticides' toxicity depends on the dosage, there are

products with the lowest toxicities for birds and humans (e.g. Cypermethrin, Table 4.2;Wilson

and Tisdell 2001). A low percentage of producers claim to perform pest threshold monitoring in

the study area, and some follow expert advice from agronomists and unions, and even the

agrochemical selling companies (Appendix G, pers. obs.); hence, it is critical in the future to

involve those advisors in the the decision making process.

In the face of a growing necessity to optimize agriculture while reducing biodiversity and

habitat losses, in this region, expecting the elimination of insecticides seems an unlikely option.

However, the minimum use of least toxic products can be an alternative to avoid pollution by

agrochemicals affecting functional agrobiodiversity, human health, and environmental quality. In

this study, we are not considering alternative agriculture practices (e.g. organic farming),

although there are many studies affirming that organic agriculture benefits biodiversity (Wilson

and Tisdell 2001, Bengtsson et al. 2005, Crowder et al. 2010), the ecological effects of organic

agriculture per unit is not always better than conventional agriculture, and does not always match

conventional yields, resulting in habitat loss to match yields (Green et al. 2005, Sachs et al. 2010,

Seufert et al. 2012).

The implementation of a natural strip of vegetation within the crop could benefit birds; however, the high cost associated with its implementation prevented it from being selected as an optimal management alternative. In addition, the possible cost of implanting a strip of vegetation its not accounted for here, only the monetary loss in yield per hectare devoted to this habitat. Neither have we incorporated the relative cost to the size of the fields, where loosing cultivated area might be more important for smaller fields. The implementation of a natural strip of vegetation was only favored under the constraint that considers the costs as not important to achieve sustainable agriculture, which would be an unlikely situation. Nevertheless, the implementation of non-cultivated terraces is a common practice in Entre Ríos to prevent soil erosion from water runoff, which when vegetated benefit for birds in the area (Goijman and Zaccagnini 2008). It might be necessary to include this management alternative in future model updates, where the cost of its implementation has already been accounted as necessary by the producers. In addition, different strip width can be incorporated as another management alternative for future model updates, especially since there is no agreement on a given edge width as a better alternative (Macdonald and Johnson 1995, Harvey et al. 2005, Stamps et al. 2008, Blank et al. 2011).

The decision whether to plant native trees, leave the borders unmanaged, or keep only pre-established trees, was sensitive to the different constraints of cost and amount of managed borders. Whether producers assume the cost of planting trees or not was key to determine the best alternative. In the events that they should not take on that cost, or if the cost was unimportant, the optimal decision was to plant native trees. Also, for example, if producers were to be requested to pay half of the cost, the best decision was to plant trees in 50% of the border. This result is encouraging, as some producers agreed to pay half of the cost for planting trees

(Appendix G). There are many factors to consider regarding the planting of native trees in the borders, and we restricted ours to the study area in Entre Ríos. We suggested planting 6 trees per 100 m of Ñandubay *Prosopis affinis*, and Black Carob tree *Prosopis nigra*, both natives to the region and with potential economic value (Burkart 1976, Sabattini et al. 2002). These species need at least three years to become adult trees, and even more time to reach full maturity, which will determine the time step needed to monitor the system and implement the adaptive management process (Burkart 1976, Ortiz Silva 1990).

Although this was a first approach to a structured decision making process to achieve sustainable agriculture compatible with conservation of biodiversity in Entre Ríos, we were able to identify the most influential variables on the decisions, as well as some uncertainties and information needs, and thus recognize where to allocate future monitoring or expert elicitation efforts. Learning through monitoring, and updating, is important as a follow up in the decision making process; although information about the sources of uncertainty only has value if it leads to more informed decisions (Clemen and Reilly 2001, Williams et al. 2009, Gregory et al. 2012, Conroy and Peterson 2013). Soybean yield was the most influential element on the decision to plant native trees and reduce insecticide applications, as well as influential on well-being. Yield per hectare in soybean fields was directly related to the number of insecticide applications and the pest control service provided by birds in the interior of the fields. The relationship between insecticide applications and yield was determined using existing literature for similar areas, although in future updates of the model we think it would be a priority to convene an expert elicitation process on this relationship specific to Paraná department in Entre Ríos. Regarding the second component of yield, we recognize there is great uncertainty associated and thus

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quantifying the degree of pest control performed by insectivorous birds in soybean – which has not been previously evaluated – becomes a future research priority.

Pollution, size of the fields and cost of the management actions also influenced the fundamental objective of farmer´s well-being. Among those components, pollution was difficult to measure and uncertain therefore should be focus of learning. We used as proxy for pollution the toxicity of products according to the "Bird eco-toxicologic risk calculator" (Bernardos et al. 2007), combined with the number of aplications and the size of the fields, though we believe this component should be subject to discussion with experts for future updates of the model, convening an expert elicitation process to determine how much those products pollute and affect human health, and how to measure it. With respect to the other fundamental objectives, cost, was higly influenced by the size of the fields which was without difficulty parameterized as its monetary value, whereas bird conservation was highly influenced by toxicity, number of trees, and the models used to estimate occupancy in borders and interior of the fields. The evaluation of the effect of toxicity on bird conservation was facilitated by the "Bird eco-toxicologic risk calculator" (Bernardos et al. 2007), and we exemplified how model (a.k.a. structural) uncertainty on bird occupancy can be reduced updating the models with new information, and thus monitoring efforts could be allocated with that purpose although in our model this is not a highly influential variable.

Although stakeholders did not participate on the definition of the decision situation and the objectives, we used information from interviews in the study area to recognize producers' interests and values, as well as the willingness and limitations on implementing some management alternatives. Interviews allowed us to learn how producers make decisions, and thus to determine other actors (e.g. agronomists, unions) that should also be involved in this process;

and also revealed the need to reinforce environmental education in the area, since there is some lack of knowledge about the role of biodiversity in agricultural lands. Future efforts should be focused in determining the real importance allocated by stakeholders on the cost, well-being, and bird conservation to reach the fundamental objective. To the best of our knowledge, this was one of the first examples of delineating a SDM approach to address a problem regarding biodiversity conservation on agricultural lands, where the decision makers are the landowners; and another example of the use of bird occupancy models to inform a decision making process (Sauer et al. 2013). Although we focused the study on a particular landscape in Argentina, it provides an illustration that can be followed and adapted to other regions in the world, to offer solutions and towards reducing the environmental impacts of agriculture while meeting food demands.

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Table 4.1. Decision alternatives to achieve maximum bird occupancy and farmers' well-being in soybean fields in Parana department, Entre Ríos, Argentina. The alternatives are a combination of strip vegetation in the middle of the field, insecticide applications, and how native trees in the borders are managed. (*) Indicates the "do nothing" alternative.

Decision	Strip of vegetation	Trees in borders	Insecticides
D1	Yes	Leave unmanaged	Reduce, and less toxic
D2	Yes	Plant new trees	Reduce, and less toxic
D3	Yes	Keep established	Reduce, and less toxic
D4	No	Leave unmanaged	Reduce, and less toxic
D5	No	Plant new trees	Reduce, and less toxic
D6	No	Keep established	Reduce, and less toxic
D7	Yes	Leave unmanaged	Same
D8	Yes	Plant new trees	Same
D9	Yes	Keep established	Same
D10*	No	Leave unmanaged	Same
D11	No	Plant new trees	Same
D12	No	Keep established	Same

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Table 4.2. Insecticides applied in Parana department, Entre Ríos, on soybean in 2007-2009 periods. I indicate the common names and toxicity level according to EPA (I: most toxic; II: moderately; III: slightly toxic) (Bohmont 2007, EPA 2012). I also estimated the risk of acute mortality for birds, and the percentage of use based on the overall number of applications.

Common name	Risk (*)	%
High toxicty to birds	*0.840*	*47.37*
Endosulfan (I)	0.939	28.95
Chlorpyrifos (II)	0.992	7.89
Deltamethrin + endosulfan (II)	0.994	7.89
Fenitrothion (II)	0.434	2.63
Low toxicty to birds	*0.043*	*52.63*
Bifenthrin (II)	0.004	7.89
Thiametoxam + lambda-cyhalotrin (II)	0.126	5.26
Gamma-cyhalotrin (II)	-	2.63
Cypermethrin (III)	0.020	28.95
Lambda-cyhalotrin (III)	0.023	7.89

Note: (*) Risk of acute mortality for birds estimated with the "Bird eco-toxicologic risk calculator" (Bernardos et al. 2007)

Table 4.3. Posterior model weight updating to reduce structural uncertainty using Bayes rule.

Example using a new set of simulated data for n=78 sites, and k=4 replicates, and calculating the

likelihoods of observing those data under each different models in borders of soybean fields.

Model i	Prior weight $p(m_i)$	Likelihood[1] $p(x\|m_i)$	Posterior weight $p(m_i\|x)$
No effect	0.00001	0.00257	0.00000
Bird group	0.00001	0.02219	0.00000
Tree	0.57830	0.58101	0.16351
Tree + Bird group	0.42160	4.07702	0.83649
Forest	0.00001	0.18890	0.00000

[1]Likelihood values where multiplied by 1E+237 to eliminate excessive zeroes
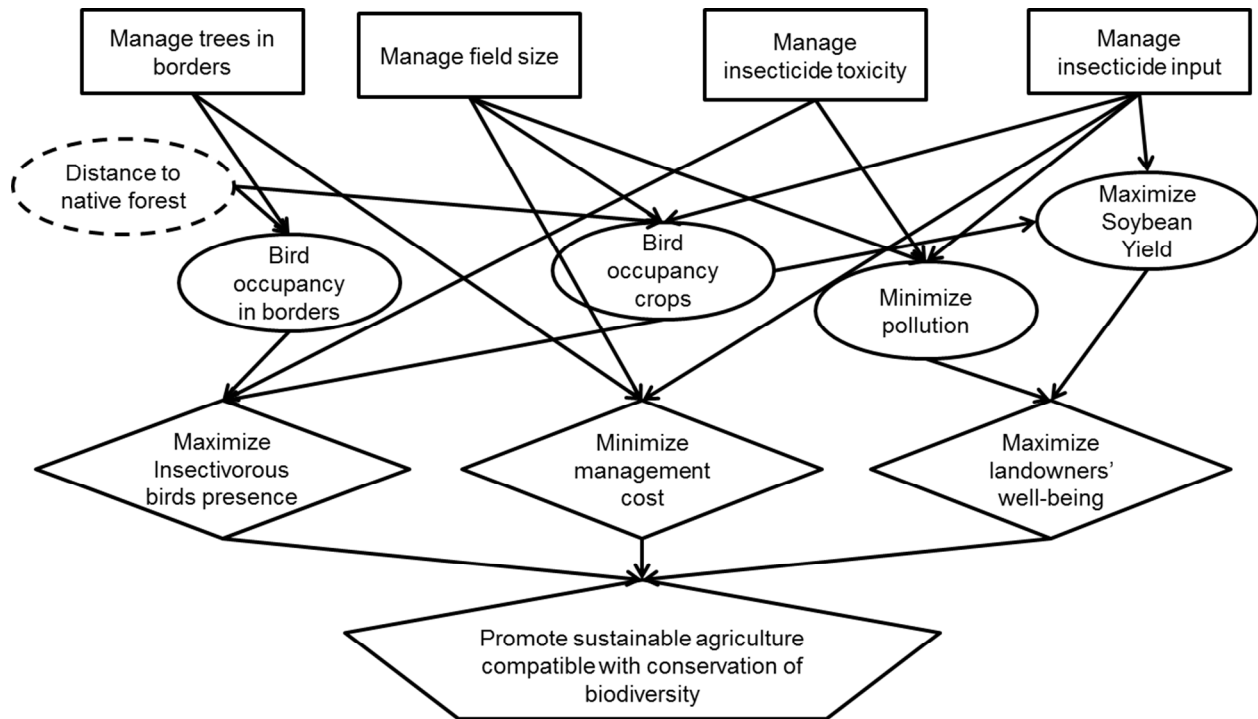
Figure 4.1. Objective network and Influence diagram for achieving sustainable agriculture compatible with conservation of biodiversity in soybean fields, Entre Rios, Argentina. Fundamental objective (pentagon), lower-level fundamental objectives (diamonds), means objectives (ovals) and management decisions (rectangles). Objective with dashed borders are those that are not under management control.
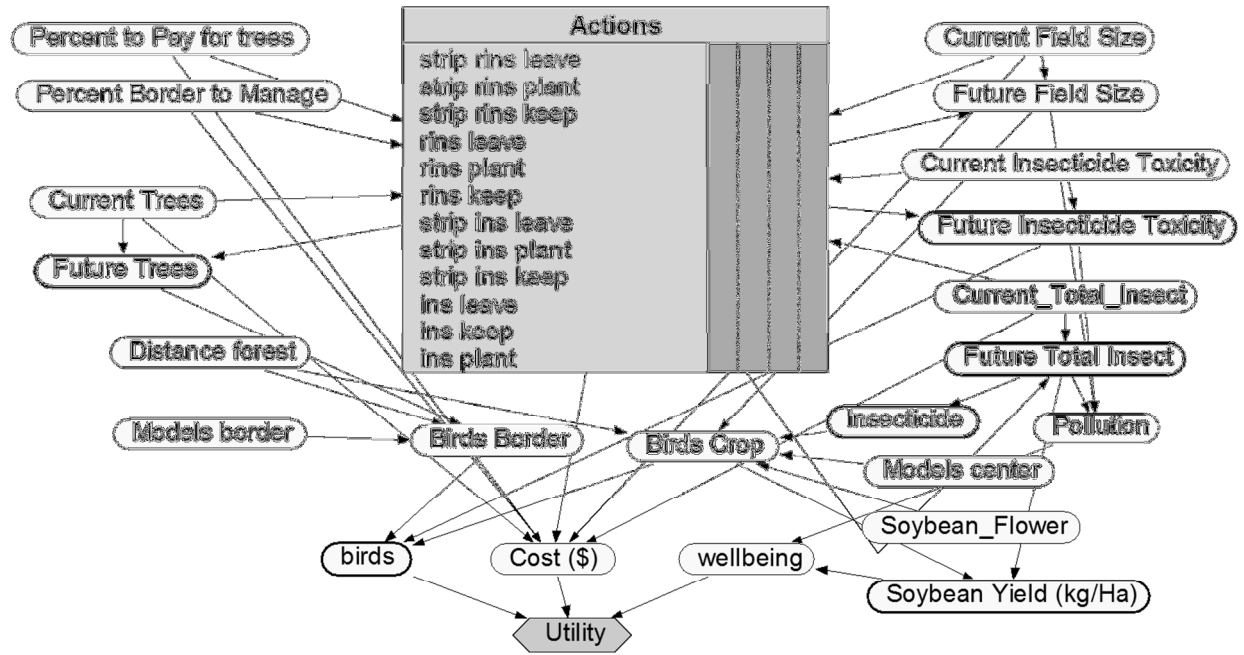
Figure 4.2. Decision network incorporating alternative management decisions to achieve sustainable agriculture compatible with conservation of biodiversity in soybean fields in Entre Rios, Argentina. Current and future states after applying decisions are also incorporated in this diagram.
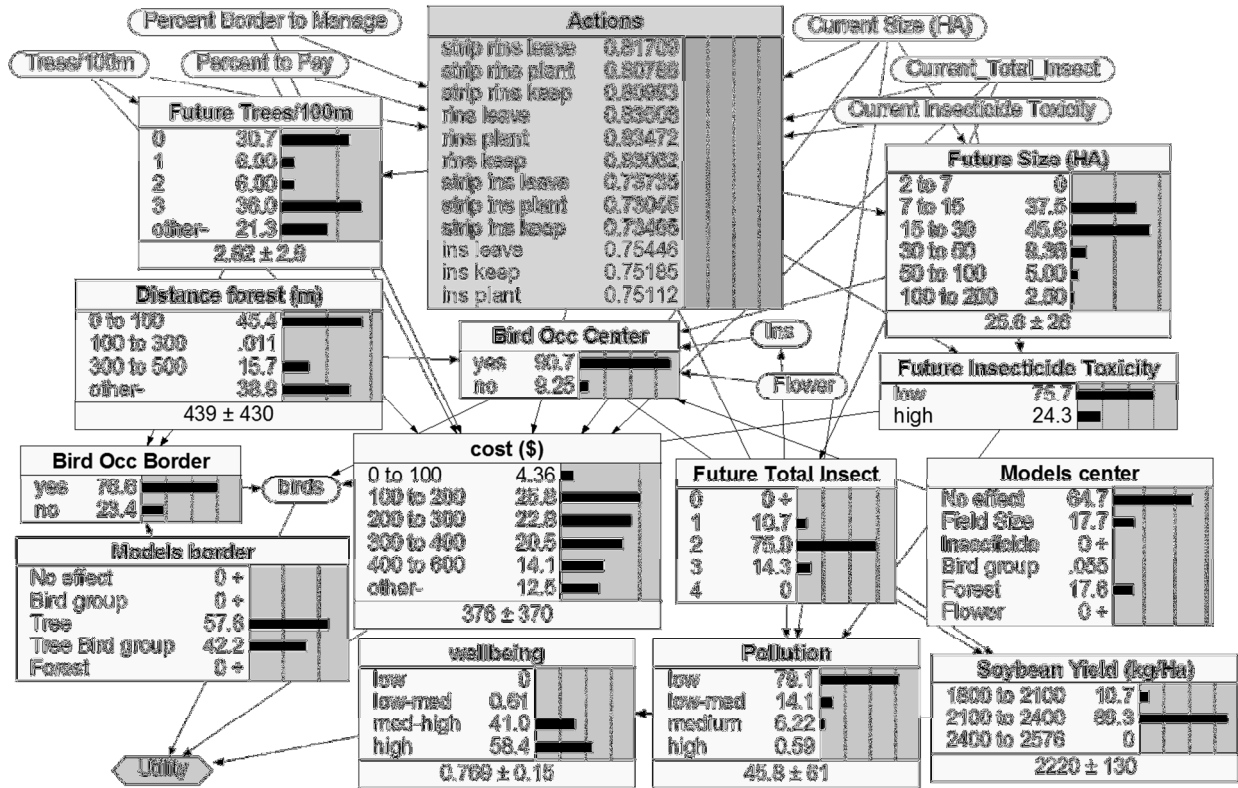
Figure 4.3. Parameterized Bayesian belief network using the indifference scoring approach for maximizing bird occupancy, and farmers' well-being, while minimizing costs in soybean fields in Entre Rios, Argentina.
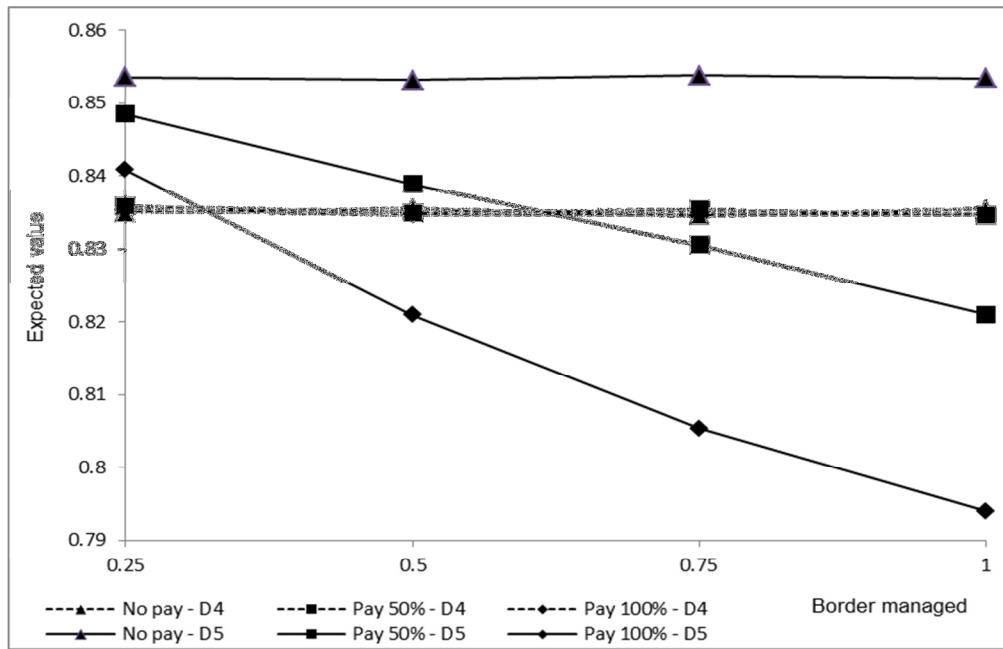
Figure 4.4. Expected value of the two best decision alternatives with varying percentages of

managed border and to pay by farmers, using the indifference scoring approach for maximizing

bird occupancy, and farmers' well-being, while minimizing costs in soybean fields in Entre Rios,

Argentina. D4: reduce insecticide applications and leave borders unmanaged; D5: reduce

insecticide applications and plant native trees.
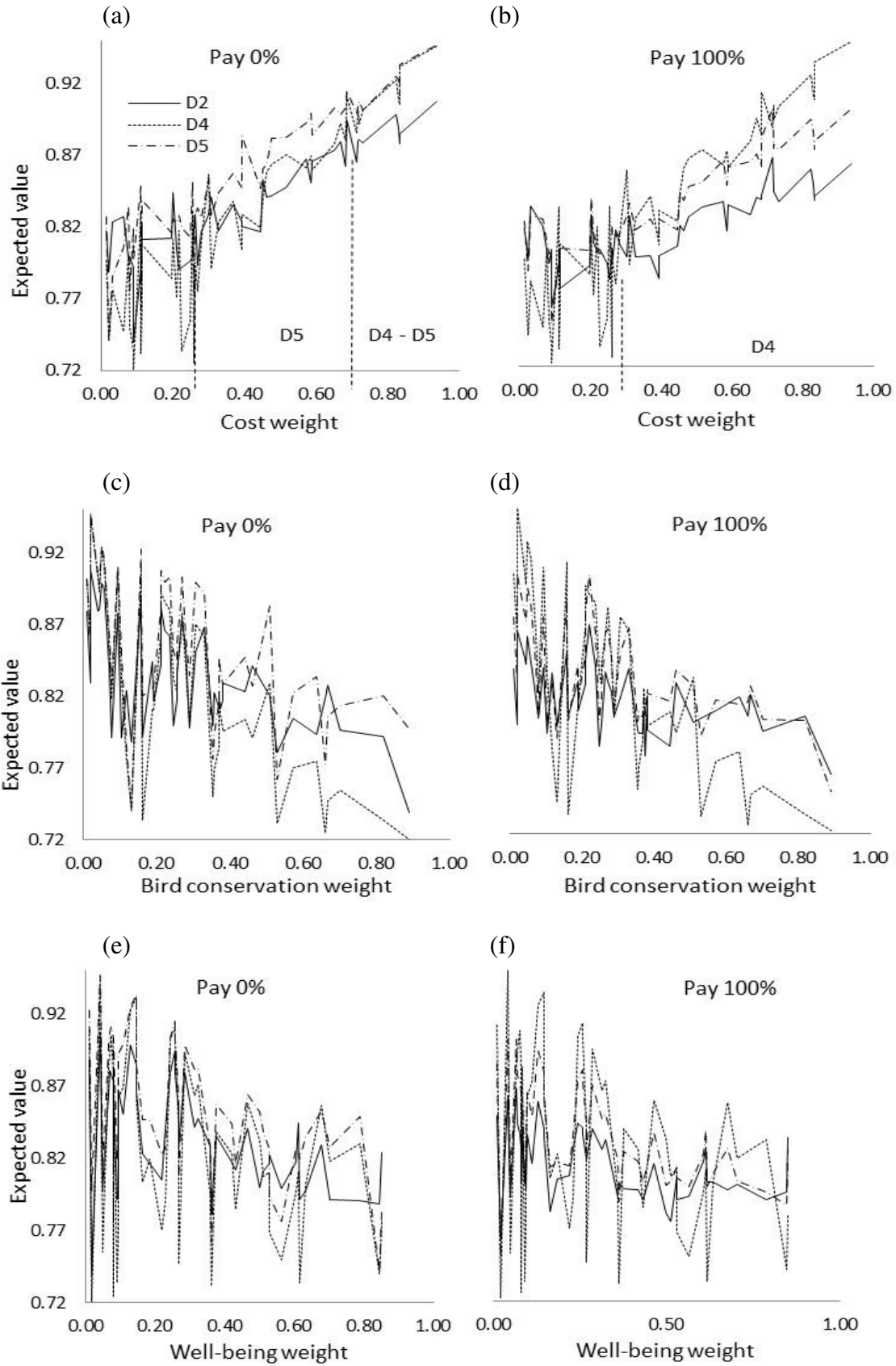
Figure 4.5. Response profiles of expected value of best decisions for management of soybean fields and border, in Entre Ríos Argentina, using random weights. Relationship with a) cost weights and no payment for planting trees; b) 100% payment; c) bird conservation weights and no payment for planting trees; d) 100% payment; e) well-being weights and no payment for planting trees; f) 100% payment. D2: strip cover vegetation, reduce insecticide applications and plant native trees; D4: reduce insecticide applications and leave borders unmanaged; D5: reduce insecticide applications and plant native trees.
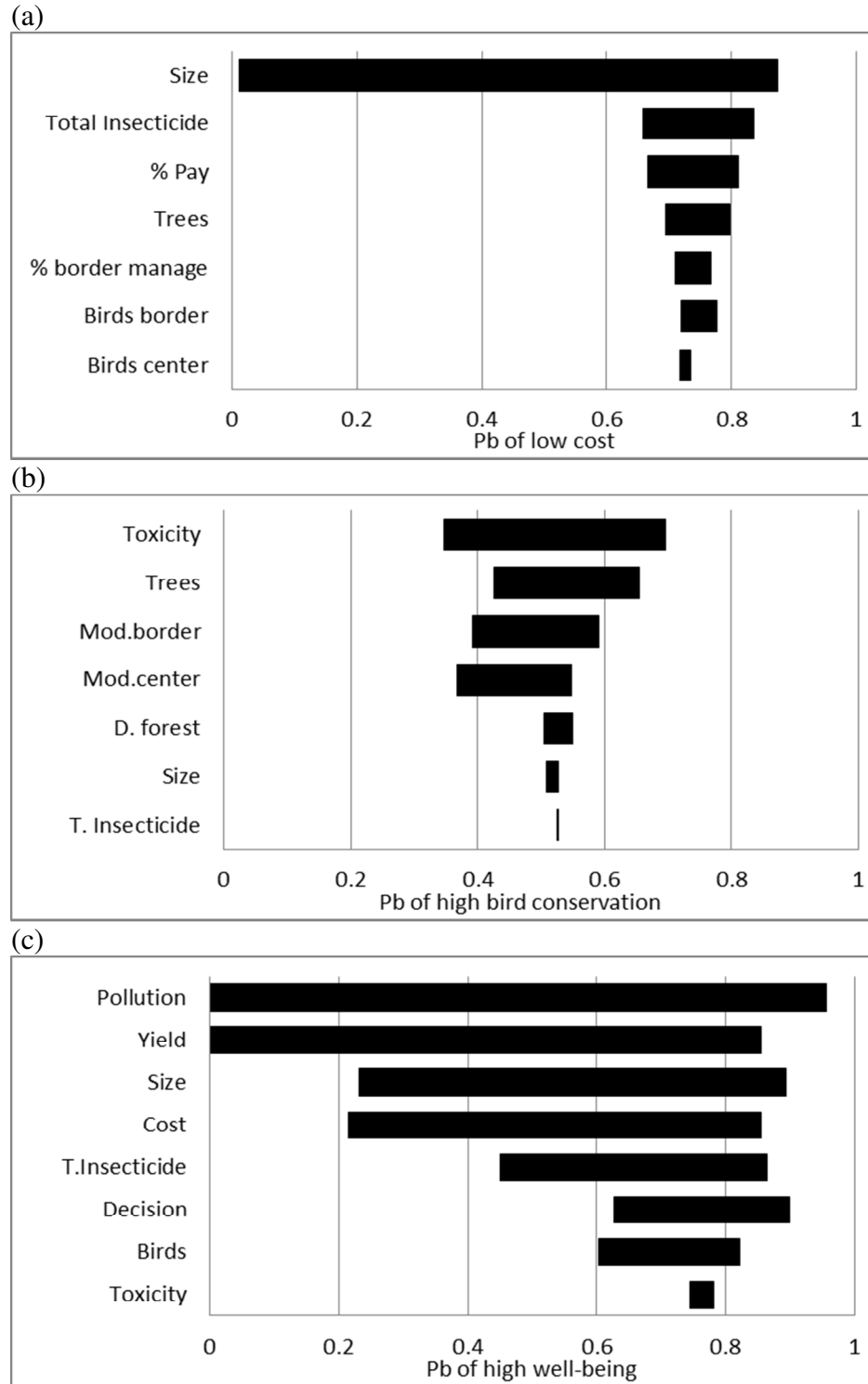
Figure 4.6. One-way sensitivity analysis to examine the sensitivity to variation of model components (*y*-axis) on the probabilities of obtaining different objectives (*x*-axis), in Entre Ríos, Argentina. (a) low cost; (b) high bird value; (c) high well-being.
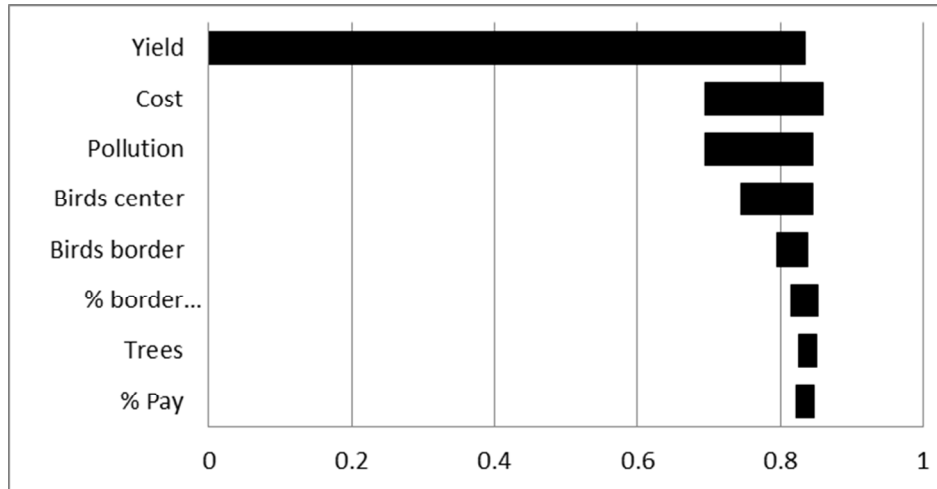
Figure 4.7. One-way sensitivity analysis to examine the sensitivity of the expected value of the decision to reduce insecticide applications and plant native trees (D5, *x*-axis), in soybean fields in entre Ríos, Argentina, to changes in models components (*y*-axis).

CHAPTER 5

CONCLUSION

The expansion and agricultural intensification in Central Argentina has been, and continues to be, ongoing and rapid processes. This fact drives the need to assess the impact on biodiversity and the environment, and to provide agricultural management and land planning recommendations to prevent the deterioration of natural resources and the decline of wildlife species, facilitating the resilience of productive ecosystems. Decision-making strategies should then focus on the tradeoffs between optimizing agriculture, while reducing biodiversity and habitat losses, conserving the associated ecosystem services, and maintaining economic and social well-being. One strategy is to integrate biodiversity into productive ecosystems, taking advantage of the potential benefits provided by organisms, seeking alternatives that promote such integration while maintaining agricultural productivity. Although there have already been studies on the effects of land use and agricultural intensification in different parts of the world, each region has a history of land use, their own technological advances, climate, it original vegetation and wildlife, which makes it unique and requires targeted research.

The Pampas and Espinal ecoregions in Argentina have a relatively recent history of agriculture, less than 200 years, which really experienced its expansion in the mid-1900s. However, with the introduction of soybeans, around 1970, the expansion was accelerated and agriculture was intensified (Viglizzo et al. 2001, Thompson 2007, Aizen et al. 2008, Bilenca et

al. 2009, SIIA 2013). To evaluate these recent and rapid changes on land use and land cover over time, in Chapter 2, we used ten years of information from the first long-term regional bird monitoring program in South America (Zaccagnini et al. 2010), and modeled occupancy under a hierarchical multi-species dynamic model using a Bayesian approach (Dorazio and Royle 2005, Royle and Dorazio 2008, Kéry et al. 2009, Zipkin et al. 2009). Bird responses to land use were difficult to generalize because each species responded differently, where some species seem to be adapted or be tolerant to agricultural intensification, others are negatively affected. Some species, usually those associated with human dominated environments, evidenced increasing trends although these could not be related to a specific land use; while other species, mostly those associated with grasslands, appeared to be declining. It is important to mention that rare species were excluded from the analyses in this chapter because of the difficulty of making inferences about them with the methods used. Such species might have been affected by land use and exhibited different trends than those found for more common species. Although generalizations for all or groups of species are problematic and in cases incorrect, I demonstrated how the results from this chapter could be used as a tool for decision-making, for example, by mapping species-based spatial distributions over time.

The lack of a marked tendency over time can have several explanations; first, it could be the result of a percentage of land use constant when averaged over the entire region of study over time. It could also be the consequence of shorter period than the necessary to document patterns in the dynamics of all species, and to document a potential time lag in their responses to habitat transformation (Chamberlain et al. 2000, Siriwardena et al. 2000, Gavier-Pizarro et al. 2012). Lastly, it could mean that some species are adapting to agricultural landscapes, or find patches of habitat to persist (Siriwardena et al. 2000, Robinson et al. 2001, Donald et al. 2006). In future

attempts to explore the effects over time of different lands uses on birds using the regional bird monitoring program, the analysis may be separated into differente areas with more homogeneous land uses. In addition, to eliminate the possibility of lagged responses of birds to land transformation, it will be necessary to count with more years of data. Although the effects of land transformation over time has not provided clear results, the large area represented in the monitoring program, presents a spatial variation of land uses, which I also evaluated to shed some light on its effects on birds.

Land use affected each species differently, and in some cases even on species sharing similar ecological requirements. Yet, we can point out some observations. The greatest number of species was distributed to the northeast – Mesopotamic Pampas and Espinal areas – coinciding with the greater proportion of native forest remnants and intermediate levels of fragmentation, and a decreasing gradient of temperature and humidity to the south-west (Schrag et al. 2009). On the other hand, most raptors showed a positive association with soybean; as opposed to some ground or foliage omnivores, and most ground insectivores, which were negatively affected by soybean and corn, but still widely distributed. Most insectivore foliage gleaners seemed unaffected by crops. This latter result could be determined by the adaptation of some birds to crops, or may suggest that these species perceive the landscape on a smaller scale, where small patches of vegetation could be providing a suitable habitat immersed in the agricultural matrix; which can also explain why species negatively affected by crops are still broadly distributed (Jobin et al. 2001, Dardanelli et al. 2006, Goijman and Zaccagnini 2008, Di Giacomo and de Casenave 2010).

The ecosystem services potentially provided by the species in this study were seed dispersal, weed control, invertebrate and vertebrate pest control, carcass and waste disposal; and

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were distributed throughout the study area (Sekercioglu 2006, Whelan et al. 2008). From the subset of species, 81% of the species prey on invertebrates, and potentially providing pest control; including some raptors, omnivores, insectivores, and granivore foliage gleaners which consume invertebrates in their breeding season. The next largely represented ecosystem services were seed dispersal and weed control, composed by granivore and omnivore guilds, with 46% of the species. However, the diversity of birds that could provide these ecosystem services in areas dominated by crops was low. This could affect the amount of the service, which according to previous studies, increases with species richness, or the likely presence of a highly efficient species (Perfecto et al. 2004, Van Bael et al. 2008).

In Chapter 3 I changed the scale of analysis to evaluate effects of agriculture on birds, and instead of a regional scale, I studied the factors that influence occupancy of birds in soybean fields and vegetated borders in Entre Ríos, Argentina. Results indicate that species using the interior of soybean fields are mostly those either nesting near the ground, granivores, or long flight insectivores; all common in agricultural landscapes. The low detection rates of some species in the interior of the fields suggest that for future studies evaluating the occupancy of birds in this habitat, more sites would need to be sampled, since our data were insufficient (Mackenzie and Royle 2005). Nevertheless, there are some interesting facts, for example, despite arthropod abundance in the first season was low, Upland Sandpiper and the Southern Lapwing, both insectivore species, were common in the interior of the fields; this indicates that prey abundance does not always affect habitat use in the immediate term (Champlin et al. 2009). In addition, insectivores such as Brown-chested Martin and Fork-tailed flycatcher were found mainly in the vegetative and flowering stages of the crop where arthropod counts seemed to be

the highest, despite insecticide applications, and likely providing a valuable ecosystem service by preying on arthropods in the seasons when pests usually emerge.

The results of this chapter indicate that a complex structure of vegetation on edges favors birds, coinciding with previous studies (Parish et al. 1994, Jobin et al. 2001, Di Giacomo and de Casenave 2010). Most foraging guilds benefited from increased native trees density, which was also positively correlated with border width, and shrub coverage, where even 10-20 trees in a 200 m border sufficed to achieve maximum occupancy probabilities. This strong relationship adds support to the hypothesis that some species respond to habitat characteristics at a finer scale than the landscape one, as suggested in Chapter 2; and in this chapter this habitat relationship is particulary evident for small size passerines, such as insectivore and granivore gleaners (Peterson et al. 1998, Robinson et al. 2001). Last, counts of arthropods preyed on by birds remained constant througout the soybean cycle, in the interior and border of the fields, even after insecticide applications; thus we could not evaluate them as a potential limiting factor for insectivorous birds. Vegetated borders hosted a greater number of arthropods than the interior of the fields, and have been documented to serve as refuge in the presence of insecticides (Lee et al. 2001, Weyland and Zaccagnini 2008, Varni 2010)

Conclusions from Chapter 3 provide scientific information for the posterior development of a conservation decision making framework, as I illustrated in Chapter 4. For example, most birds in the study area in Entre Ríos, have an exclusively or partly insectivorous diet, and are represented by many guilds; then, birds could contribute to invertebrate pest control covering a variety of invertebrates with different habits, such as terrestrial or aereal. Yet, several species use the interior of the fields and could be vulnerable to direct or indirect effects of pesticides. Based on those results, recommendations could be oriented to generate awareness on the use of

pesticides, and to prevent their drift to the borders in order to conserve a rich vegetation structure that benefits birds.

The implementation of occupancy analyses both in Chapters 2 and 3 proved to be a fundamental tool for the study of birds in agricultural landscapes in Argentina. Detection probabilities of birds were imperfect and sometimes affected by weather conditions or habitat. In many cases, the presence of elements of woody vegetation affected detection of birds. This demonstrates the importance of acknowledging and incorporatring imperfect detection probabilities, especially if the intention is to provide reliable scientific results to inform decision-making for conservation.

In Chapter 4 I used results from Chapter 3 to inform a decision making process, combined with bibliographic searches, and interviews local farmers. The goal was to integrate bird conservation, with producers' values and needs, in Paraná department, in Entre Ríos, Argentina, into a structured decision making framework (SDM). I was able to delineate the fundamental objective as the achievement of sustainable agriculture compatible with conservation of biodiversity, which is comprised and can be achieved by three lower-level fundamental objectives: the maximization of insectivorous bird presence in the landscape; maximization of agricultural producers' well-being; and minimization of the cost of management. I identified different components of the alternative management decisions to reach the fundamental objectives, consisting of managing the borders of fields, the application of insecticides, and changing the size of the fields by changing the edge to interior area; all evaluated against different constraints of cost allocation and percentage of border management.

Under all constraints, reducing insecticide applications in the cultivation of soybeans and using less toxic products, was the best decision. Based on the treatment of soybean pests in this

particular study area, the reccomendation consisted of a maximum of two sprays for the control

of major pests in the area, velvetbean caterpillar is controlled before flowering, and the stink

bugs then (Saluso et al. 2007). In most cases, the implementation of a strip of vegetation in the

middle of the field to reduce its size, was not selected as a good alternative because of the high

cost associated. Finally, the most variable component consisted on managing the borders of

soybean fields, where the alternative of no management of the borders, or plant native trees were

selected depending on the amount of border to manage, and the percentage of the costs that the

producers should take. In case the decision was to plant trees, the suggestion involved planting 6

trees per 100 m of Ñandubay *Prosopis affinis*, and Black Carob tree *Prosopis nigra*, both natives

to the region and with potential economic value (Burkart 1976, Sabattini et al. 2002).

I recognize information gaps exist in the model, and through sensitivity analyses we

identified several key uncertainties and information needs which need to be the focus of future

learning. For example, the decision of planting trees and reducing insecticides, which was the

optimal under several alternative constraints, was sensitive to yield; this was expected because of

the close relationship between yield and pesticide applications, and thus future efforts could be

focused on reducing uncertainty on those components. Future efforts should also focus in

determining the real importance allocated by stakeholders on the cost, well-being, and bird

conservation to build the utility function and reach the fundamental objective.

In this first SDM approach to achieve sustainable agriculture compatible with

conservation of biodiversity in Entre Ríos, I determined information needs that must be

minimized before applying a management alternative, and start and adaptive management

process (ARM) of learning through monitoring and updating the process (Clemen and Reilly

2001, Williams et al. 2009, Gregory et al. 2012). The next step will be to review and reach a

consensus regarding the process with stakeholders, and review the objectives, if necessary. This is one of the first examples of delineating a SDM approach to address a problem regarding biodiversity conservation on agricultural lands. Although we focused the study on a particular landscape in Argentina, it provides an illustration that can be followed and adapted to other regions in the world, to offer solutions and towards reducing the environmental impacts of agriculture while meeting food demands.

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APPENDIX A

CHAPTER 2 R CODE FOR ANALYSES

R and JAGS model code and specifications, for our hierarchical Bayesian multi-species dynamic

occupancy model to estimate the influence of land use and land cover, over time, on avian

species in Argentina, in the regional bird monitoring area, 2003-2012. We implemented the

model using flat priors using program R2Jags. We ran three chains of the model for 30,000

iterations each after a burn-in of length 20,000 and thinned the model by 10. We assessed

convergence of the model using $\hat{R}$.

```
sink("global.jags")
cat("

#JAGS code starts
model
{

## Model missing covariates (some examples as an illustration)
for (j in 1:6) {
  for (k in 1:5) {
pfor[j,k,1]~dunif(0,1)
}}
#...
for (j in 1:6){
soy[j,1]~dunif(0,0.5)
}
#...
for (j in 1:6){
corn[j,1]~dunif(0,0.5)
}
#...
```

```
for (j in 314:318){
forest[j,8]~dunif(0,0.5)
}
#…
for (j in 1:6){
per_past[j,1]~dunif(0,0.5)
}
#…

## Prior distributions
mean.mu.u~ dunif(0, 1)
mean.u <- log(mean.mu.u)- log(1-mean.mu.u)

mean.mu.v ~ dunif(0, 1)
mean.v <- log(mean.mu.v)- log(1-mean.mu.v)

mu.a1 ~ dnorm(0, 0.37)
mu.a2 ~ dnorm(0, 0.37)
mu.a3 ~ dnorm(0, 0.37)
mu.a4 ~ dnorm(0, 0.37)
mu.a5 ~ dnorm(0, 0.37)
mu.a6 ~ dnorm(0, 0.37)
mu.b1 ~ dnorm(0, 0.37)

 sd.a1~dunif(0,10)
 tau.a1<-pow(sd.a1,-2)

 sd.a2~dunif(0,10)
 tau.a2<-pow(sd.a2,-2)

 sd.a3~dunif(0,10)
 tau.a3<-pow(sd.a3,-2)

 sd.a4~dunif(0,10)
 tau.a4<-pow(sd.a4,-2)

 sd.a5~dunif(0,10)
 tau.a5<-pow(sd.a5,-2)

 sd.a6~dunif(0,10)
 tau.a6<-pow(sd.a6,-2)

 sd.b1~dunif(0,10)
 tau.b1<-pow(sd.b1,-2)
```

```
  sd.mu.u~dunif(0,10)
  tau.mu.u<-pow(sd.mu.u,-2)

  sd.mu.v~dunif(0,10)
  tau.mu.v<-pow(sd.mu.v,-2)

 for (i in 1:n){
   mu.u[i] ~ dnorm(mean.u, tau.mu.u)
   mu.v[i] ~ dnorm(mean.v, tau.mu.v)
   a1[i] ~ dnorm (mu.a1, tau.a1)
   a2[i] ~ dnorm (mu.a2, tau.a2)
   a3[i] ~ dnorm (mu.a3, tau.a3)
   a4[i] ~ dnorm (mu.a4, tau.a4)
   a5[i] ~ dnorm (mu.a5, tau.a5)
   a6[i] ~ dnorm (mu.a6, tau.a6)
   b1[i]~dnorm(mu.b1,tau.b1)
   sd.u[i]~dunif(0,10)
   sd.v[i]~dunif(0,10)
   tau.u[i]<-pow(sd.u[i],-2)
   tau.v[i]<-pow(sd.v[i],-2)

  for (t in 1:Y){
    u[i,t] ~ dnorm(mu.u[i], tau.u[i])
    v[i,t] ~ dnorm(mu.v[i], tau.v[i])
    }

## Process model
    for (j in 1:site) {
     for (t in 1:Y){
        z[j,i,t]~dbern(psi[j,i,t])
        logit(psi[j,i,t]) <- u[i,t] + a1[i]*lat[j,t]+ a2[i]*long[j,t]+ a3[i]*soy[j,t]+ a4[i]*corn[j,t]+
a5[i]*per_past[j,t]+ a6[i]*forest[j,t]

## Observation model
        for (k in 1:R) {
          y[j,k,i,t] ~ dbern(mu.y[j,k,i,t])
          mu.y[j,k,i,t] <- p[j,k,i,t]*z[j,i,t]
          logit(p[j,k,i,t]) <-  v[i,t] + b1[i]*pfor[j,k,t]

## Observed deviance
          dev[j,k,i,t]<-y[j,k,i,t]*log(mu.y[j,k,i,t])+(1-y[j,k,i,t])*log(1-mu.y[j,k,i,t])

### Predict new observation and compute deviance
```

```r
            y.new[j,k,i,t] ~ dbern(mu.y[j,k,i,t])
            dev.sim[j,k,i,t]<- y.new[j,k,i,t]*log(mu.y[j,k,i,t])+(1-y.new[j,k,i,t])*log(1-mu.y[j,k,i,t])

         } #R
         }#year
         }#site
      }#species

      sum.dev<-sum(dev[,,,])
      sum.dev.sim<-sum(dev.sim[,,,])

      test<-step(sum.dev.sim - sum.dev)
      bpvalue<-mean(test)

} #model
",fill=TRUE)
sink()

## Create the necessary arguments to run the jags() command in R
### Load all the data
sp.data = list(y=y, R=R, site=site , n=n ,Y=Y, pfor=pfor, lat=lat, long=long, soy=soy, corn=corn,
per_past=per_past, forest=forest)

## Initialize z to be sites where at least 1 detection
zst<-array(0,dim=c(site,n,Y))
y<-y

for (i in 1:site)  {
   for (s in 1:n)  {
      for (t in 1:Y) {
       zst[i,s,t]<-(sum(y[i,,s,t])>0)*1
         }}}
zst[is.na(zst)]<-1

## Specify the initial values for the chains
inits1<- list(z=zst,mean.mu.u=runif(1,0,1),mean.mu.v=runif(1,0,1),mu.a1=runif(1,-
2,2),mu.a2=runif(1,-2,2),mu.a3=runif(1,-2,2), mu.a4=runif(1,-2,2), mu.a5=runif(1,-
2,2),mu.a6=runif(1,-2,2),mu.b1=runif(1,-2,2),sd.mu.u=runif(1,0.1,5),sd.mu.v=runif(1,0.1,5),
sd.a1=runif(1,0.1,5),sd.a2=runif(1,0.1,5),sd.a3=runif(1,0.1,5),sd.a4=runif(1,0.1,5),
sd.a5=runif(1,0.1,5),sd.a6=runif(1,0.1,5),sd.b1=runif(1,0.1,5),sd.u=runif(n,0.1,5),sd.v=runif(n,0.1
,5))
inits2<- list(z=zst,mean.mu.u=runif(1,0,1),mean.mu.v=runif(1,0,1),mu.a1=runif(1,-
2,2),mu.a2=runif(1,-2,2),mu.a3=runif(1,-2,2), mu.a4=runif(1,-2,2),mu.a5=runif(1,-2,2),
mu.a6=runif(1,-2,2),mu.b1=runif(1,-2,2),sd.mu.u=runif(1,0.1,5),sd.mu.v=runif(1,0.1,5),
```

```
sd.a1=runif(1,0.1,5), sd.a2=runif(1,0.1,5), sd.a3=runif(1,0.1,5),sd.a4=runif(1,0.1,5),
sd.a5=runif(1,0.1,5),sd.a6=runif(1,0.1,5), sd.b1=runif(1,0.1,5), sd.u=runif(n,0.1,5),
sd.v=runif(n,0.1,5))
inits3<- list(z=zst,mean.mu.u=runif(1,0,1),mean.mu.v=runif(1,0,1),mu.a1=runif(1,-
2,2),mu.a2=runif(1,-2,2),mu.a3=runif(1,-2,2), mu.a4=runif(1,-2,2),mu.a5=runif(1,-2,2),
mu.a6=runif(1,-2,2),mu.b1=runif(1,-2,2),sd.mu.u=runif(1,0.1,5),sd.mu.v=runif(1,0.1,5),
sd.a1=runif(1,0.1,5),sd.a2=runif(1,0.1,5),sd.a3=runif(1,0.1,5),sd.a4=runif(1,0.1,5),
sd.a5=runif(1,0.1,5),sd.a6=runif(1,0.1,5),sd.b1=runif(1,0.1,5),sd.u=runif(n,0.1,5),sd.v=runif(n,0.1
,5))


inits<-list(inits1,inits2,inits3)


## Specify the parameters to be monitored
sp.params <- c("mean.mu.u","mean.mu.v", "mu.a1", "mu.a2", "mu.a3", "mu.a4", "mu.a5",
"mu.a6","mu.b1", "sd.mu.u", "sd.u", "sd.v","sd.a1","sd.a2", "sd.a3","sd.a4", "sd.a5", "sd.a6",
"sd.b1","u","v","a1","a2","a3","a4","a5","a6","b1","test","bpvalue")


##Run the model and call the results "fit"
fit = jags(sp.data, inits, sp.params, "global.jags", n.chains=3, n.iter=50000,  n.burnin=30000,
n.thin=10)
```
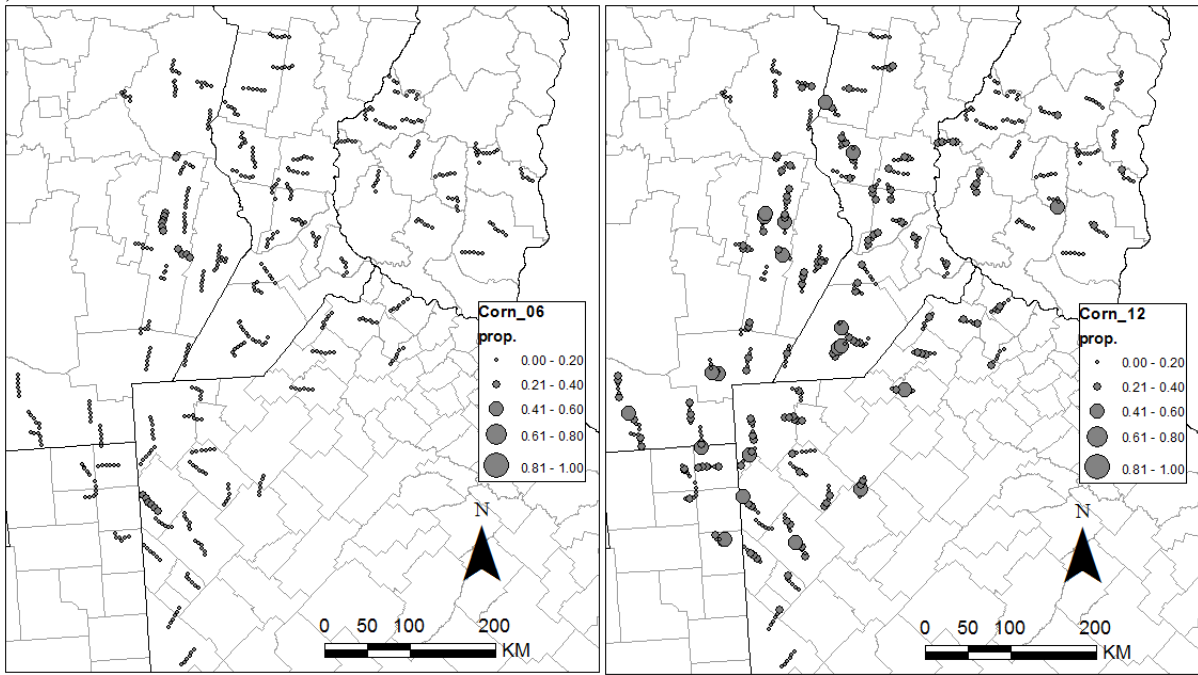
APPENDIX B

CHAPTER 2 ADDITIONAL FIGURES OF LAND USES IN THE REGIONAL BIRD

MONITORING PROGRAM AREA

(a)
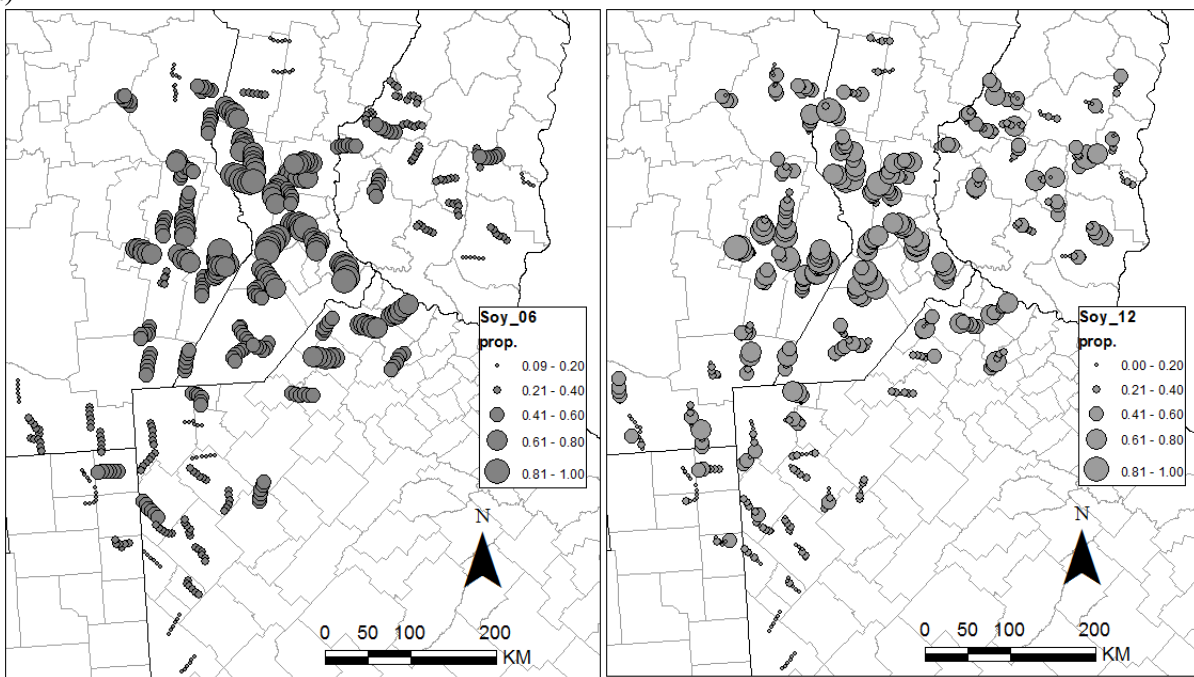
(b)

(c)
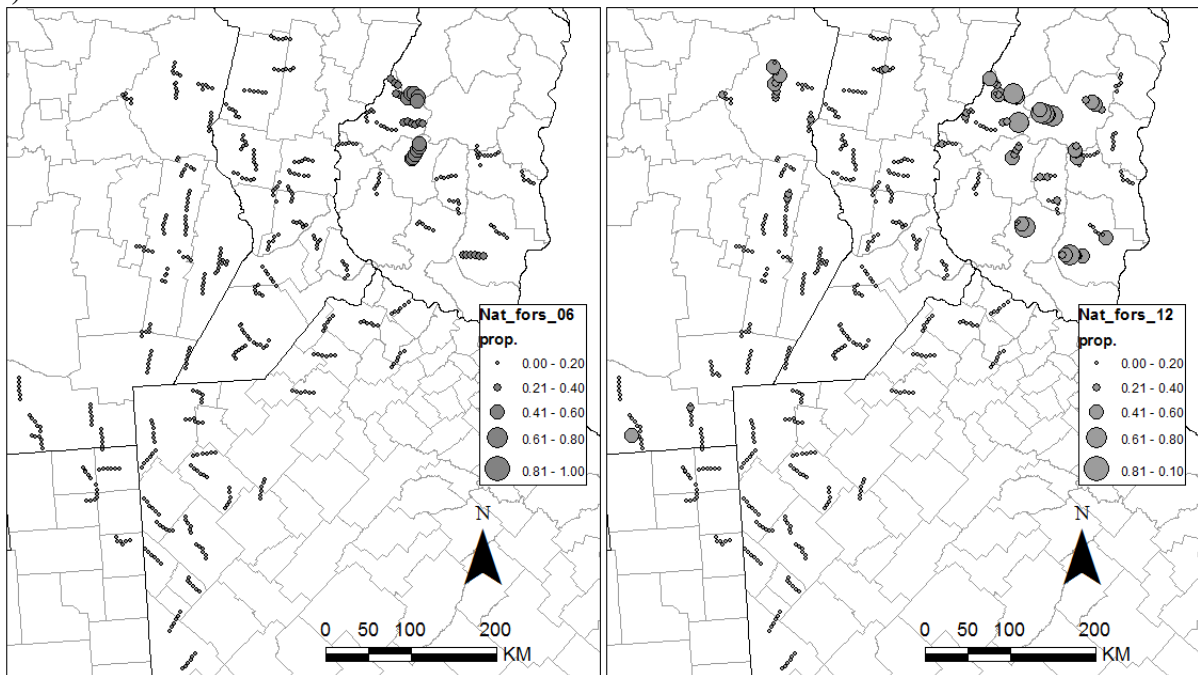
(d)
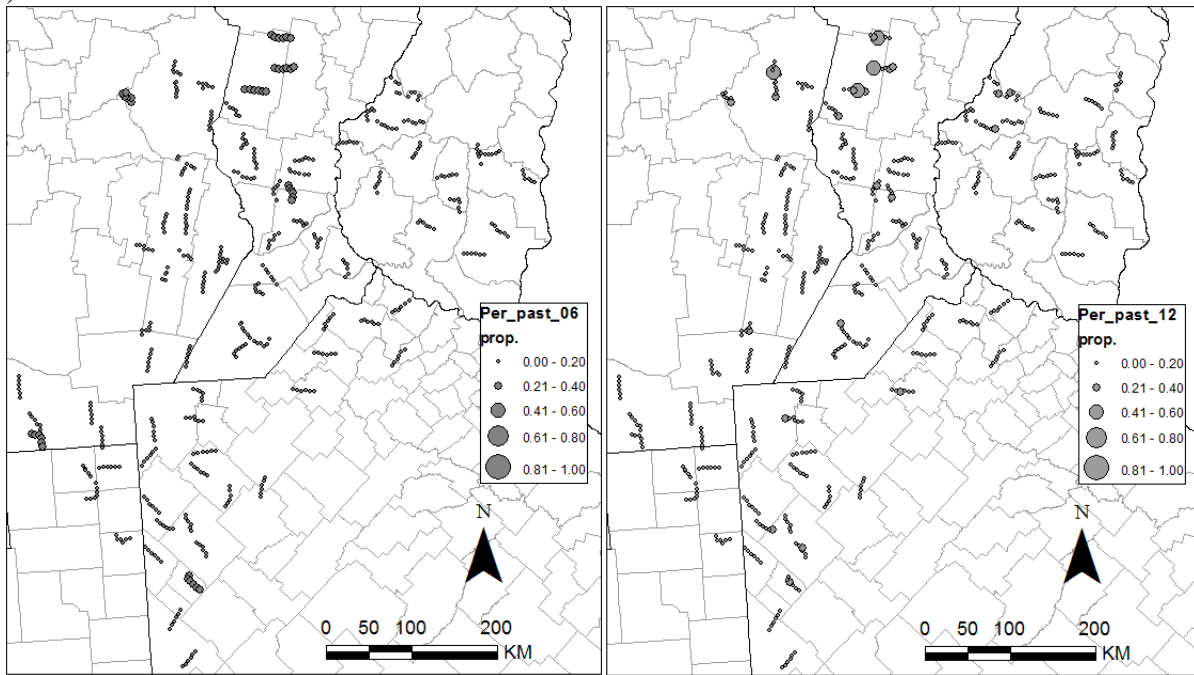
Figure B.2.1. Proportion land uses in 2006 and 2012, in the area of regional bird monitoring program in Pampas and Espinal ecoregions in Argentina. (a) soybean, (b) corn, (c) perennial pastures, and (d) native forests.

APPENDIX C

CHAPTER 2 ADDITIONAL FIGURES AND TABLES OF DETECTION PROBABILITIES

Table. C.2.1. Posterior proportion of forest effects in the logit scale (logit($\hat{\beta}_{forest}$)), with upper and lower 95% credibility intervals (LI, UCI) for each species in the regional bird monitoring program in Argentina, 2003-2012.

Group	Species	logit(β_{forest})	LCRI	UCRI
RAP	BUMA	3.123	2.451	3.795
RAP	BUSW	0.189	-0.508	0.888
RAP	ELLE	0.266	-0.604	1.078
RAP	FAFE	-1.967	-3.071	-0.967
RAP	FASP	-1.322	-1.778	-0.889
RAP	MICH	-1.298	-1.556	-1.041
RAP	POPL	-0.497	-0.785	-0.224
RAP	ROSO	-0.839	-1.966	0.184
RAP	SPCU	-4.324	-4.784	-3.874
OMN	BALO	-2.523	-3.433	-1.877
OMN	CHTO	-2.227	-3.314	-1.323
OMN	NOMA	-2.225	-2.509	-1.936
OMN	RHRU	-1.749	-2.206	-1.212
OMN	VACH	-2.045	-2.227	-1.858
GRA2	COLI	-1.388	-2.275	-0.505
GRA2	COMA	1.477	1.281	1.679
GRA2	COPI	2.658	2.455	2.869
GRA2	COPZ	1.494	1.239	1.755
GRA2	LEVE	2.555	2.170	2.935
GRA2	MYMO	1.461	1.287	1.639
GRA2	ZEAU	1.851	1.643	2.059
GRA	AMHU	-4.052	-4.326	-3.783
GRA	CAMA	-0.538	-1.293	0.176
GRA	EMPL	-2.359	-2.878	-1.851
GRA	MOBA	2.405	2.130	2.672

GRA	MOBO	1.008	0.784	1.231
GRA	MORU	0.696	0.261	1.106
GRA	PACO	0.445	0.155	0.727
GRA	PADO	0.912	0.593	1.230
GRA	PHRU	3.450	2.628	4.308
GRA	POME	1.210	0.620	1.776
GRA	PONI	0.000	-0.630	0.599
GRA	SAAU	1.458	1.208	1.712
GRA	SACO	2.019	1.242	2.789
GRA	SIFL	1.107	0.871	1.345
GRA	SILU	-2.255	-2.529	-1.977
GRA	SPCA	-0.442	-0.639	-0.251
GRA	SPRU	-3.172	-3.892	-2.484
GRA	STSU	-2.383	-2.728	-2.047
GRA	VOJA	-1.488	-2.104	-0.881
GRA	ZOCA	-1.078	-1.240	-0.920
INS1	CHAU	0.637	0.033	1.207
INS1	COLA	1.699	1.167	2.279
INS1	COME	1.705	1.352	2.067
INS1	GEAE	0.813	0.354	1.254
INS1	GUGU	0.682	0.492	0.880
INS1	LEAN	1.997	1.431	2.569
INS1	MISA	0.821	0.588	1.047
INS1	PHST	0.387	-0.114	0.863
INS1	PODU	1.810	1.408	2.210
INS1	SCPH	0.271	-0.187	0.712
INS1	SESU	2.205	1.553	2.931
INS1	SYAL	1.201	0.858	1.535
INS1	SYFR	1.689	1.115	2.245
INS1	TAMA	2.024	1.592	2.476
INS1	TANA	1.618	1.238	2.004
INS1	TRAE	0.957	0.736	1.181
INS1	TURU	1.146	0.671	1.622
INS2	ANAN	0.346	-0.072	0.761
INS2	ANCH	-3.009	-4.686	-1.516
INS2	COCA	-0.565	-0.891	-0.245
INS2	DRBR	1.061	0.526	1.594
INS2	FURU	2.514	2.326	2.702
INS2	HYPE	-2.095	-3.424	-0.804
INS2	MARI	0.593	0.063	1.111
INS2	PEPY	-0.648	-1.529	0.177
INS2	PHTA	0.194	-0.047	0.425
INS2	PISU	1.799	1.598	2.007

INS2	PSLO	1.486	1.120	1.845
INS2	PYRU	2.423	1.706	3.125
INS2	TALE	-0.760	-1.508	-0.070
INS2	TYME	2.584	2.171	3.003
INS2	TYSA	0.092	-0.095	0.282
INS2	XOIR	-0.995	-1.445	-0.561

Notes: raptors (RAP), ground omnivores and herbivores (OMN), ground granivores (GRA2), other granivores (GRA), insectivores mostly associated with folliage (INS1) and other insectivores (INS2). For details of species names, see Table 3.1. in the manuscript.
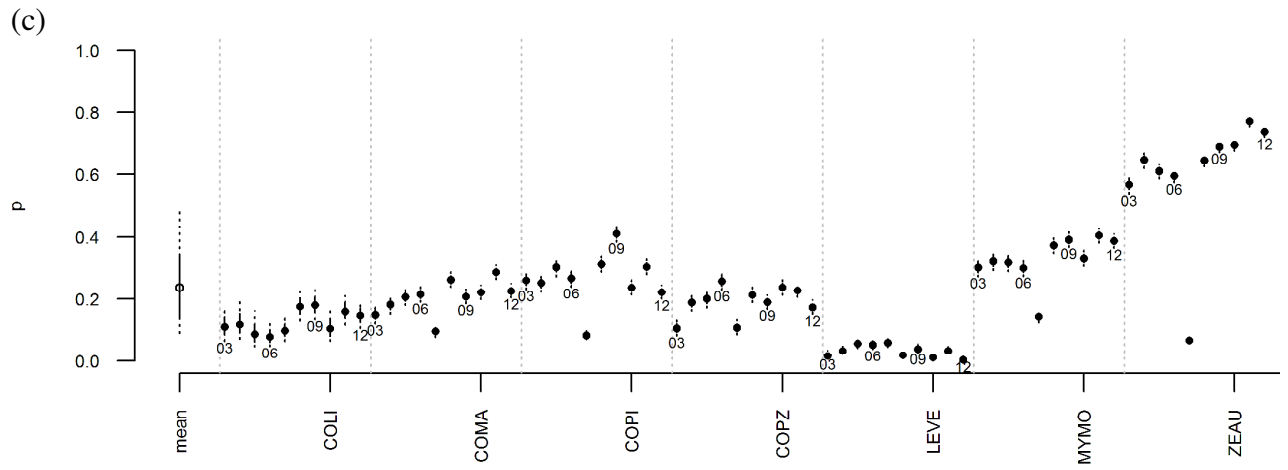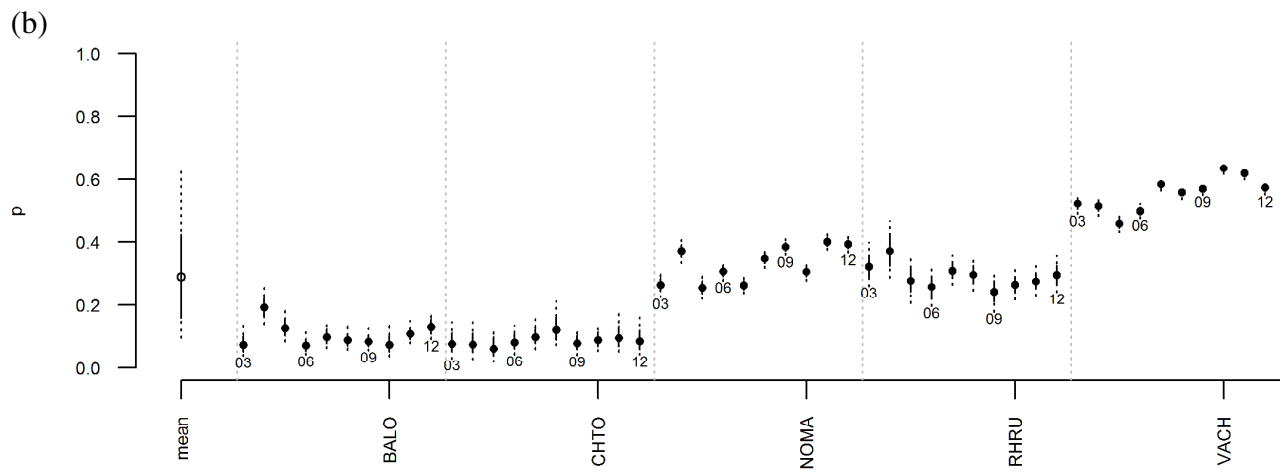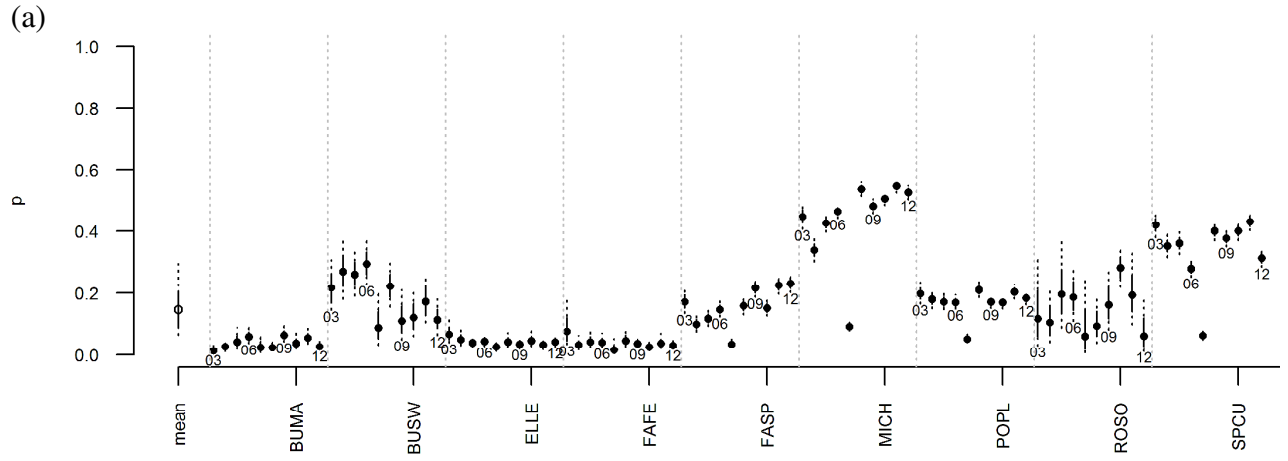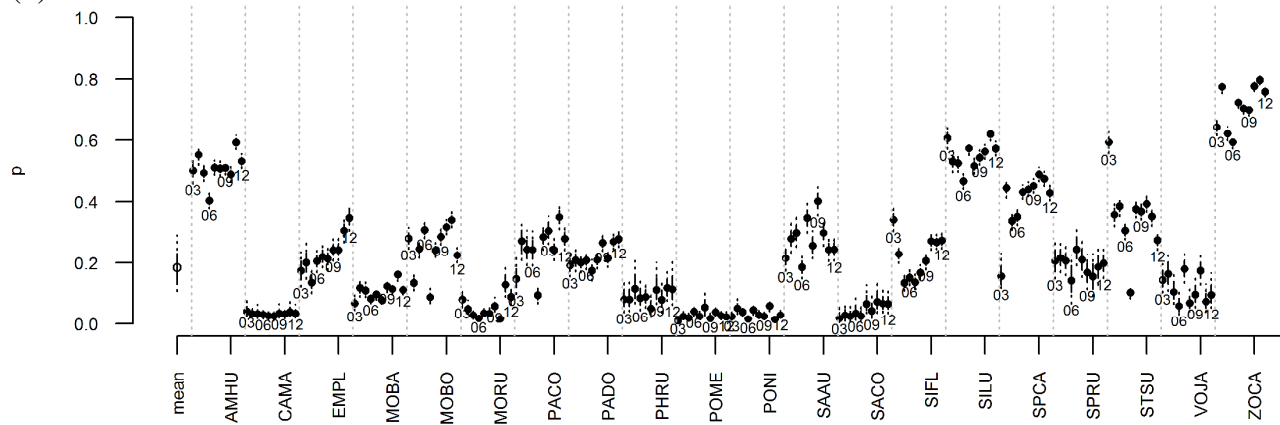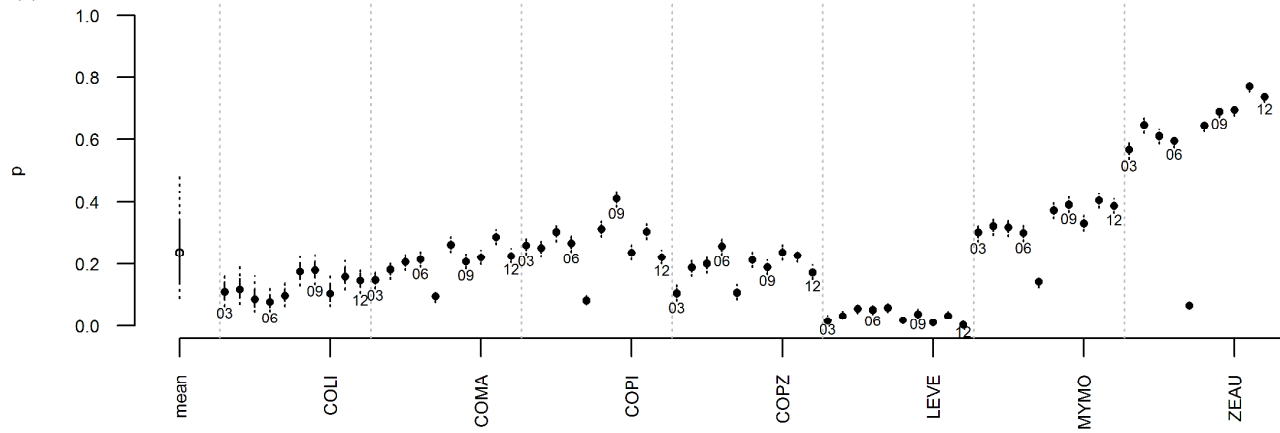
Figure C.2.1. Posterior detection model intercepts ($\hat{p}\pm$SD, 95%CI), in probability scale, incorporating time as a random effect, during 2003-2012, in the regional bird monitoring program in Argentina for all species. (a) Raptors; (b) ground omnivores and herbivores; (c) ground granivores; (d) other granivores; (e) insectivores mostly associated with folliage; (f) other insectivores.

APPENDIX D

CHAPTER 2 ADDITIONAL FIGURES OF OCCUPANCY PROBABILITIES

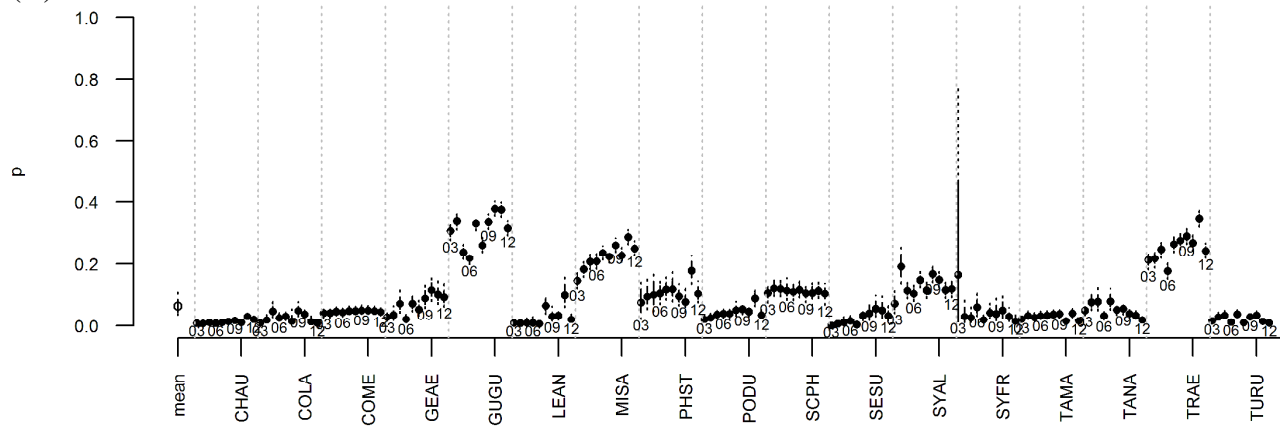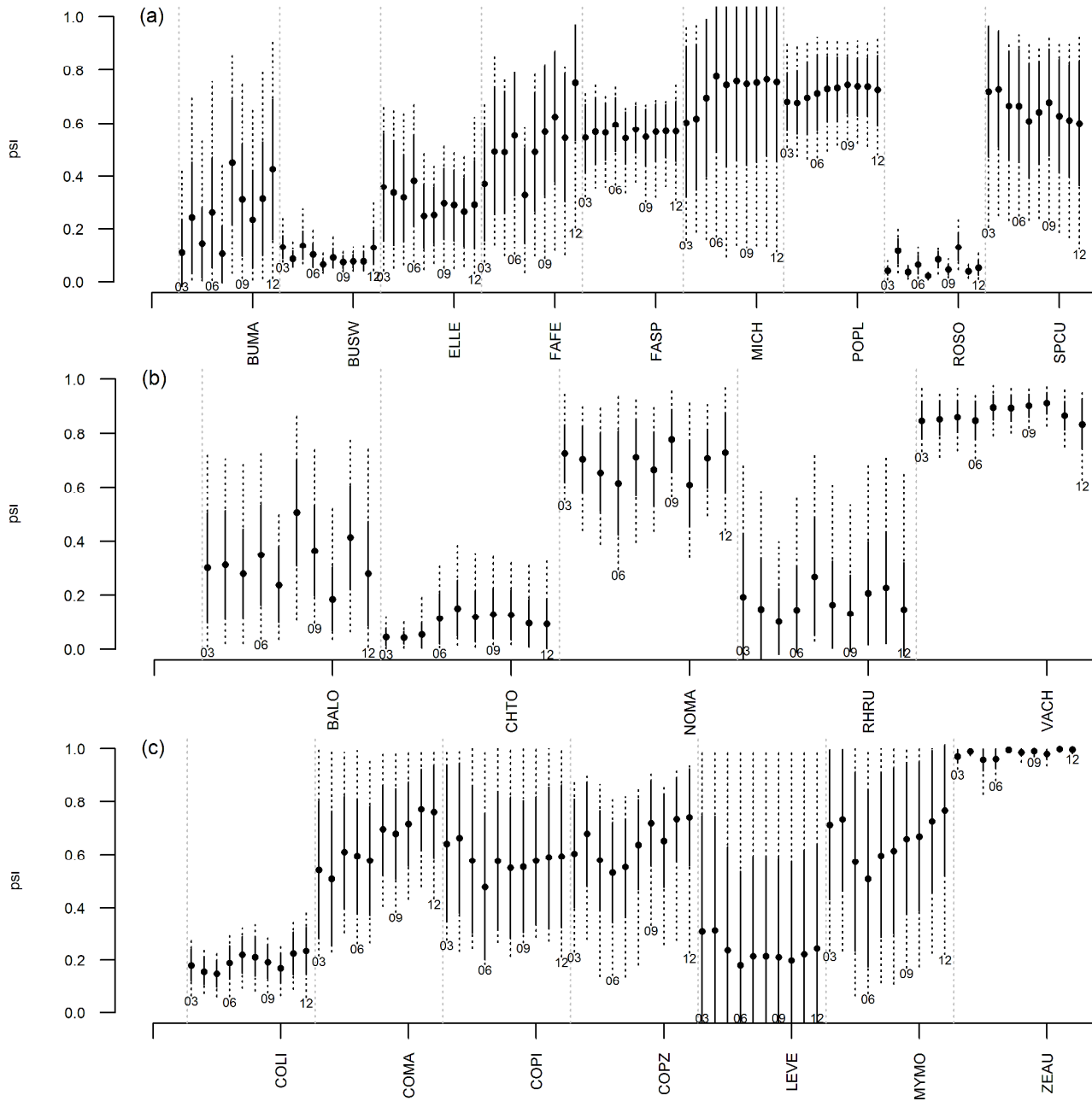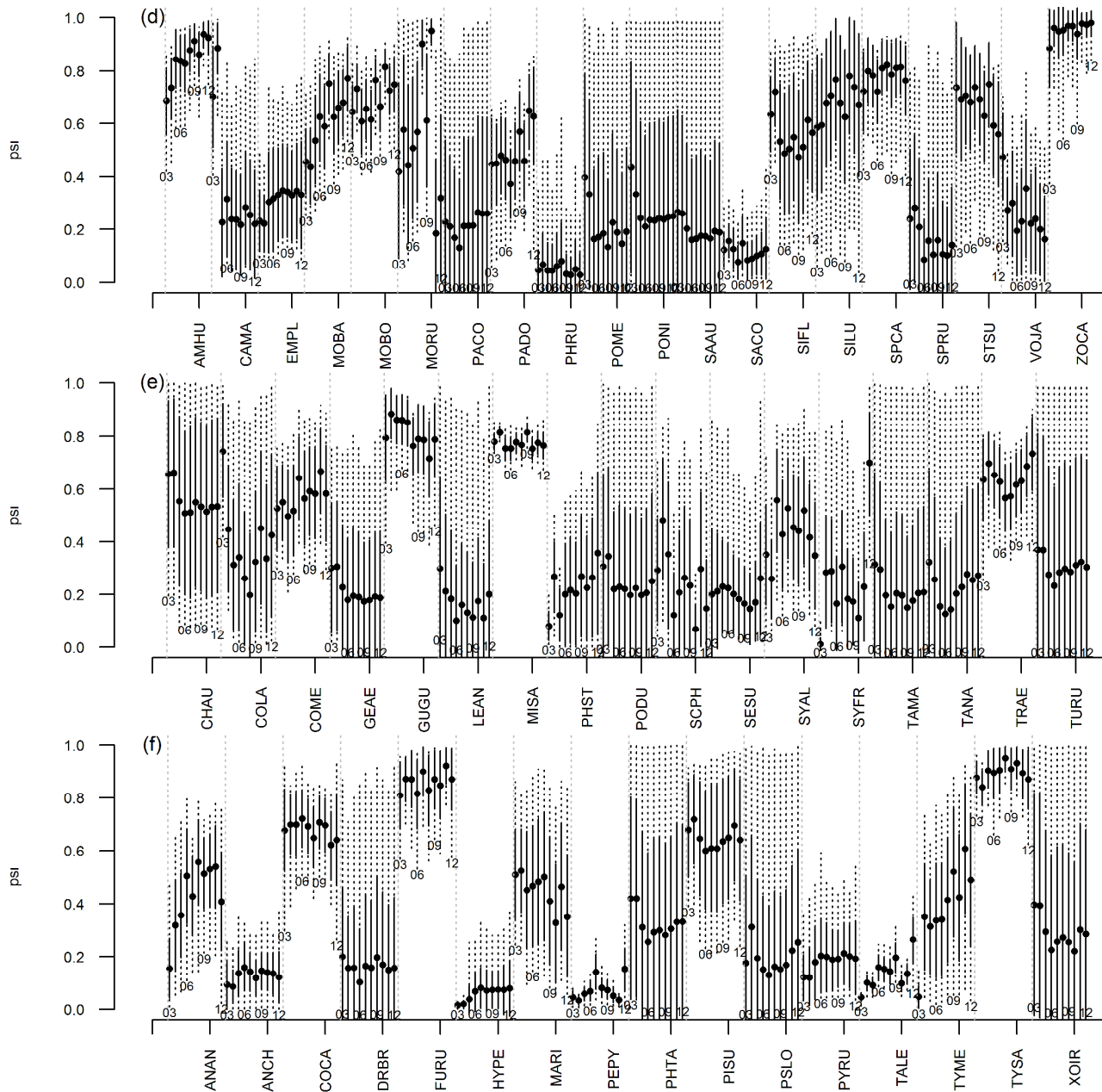Figure D.2.1. Mean occupancy ($\hat{\psi}\pm$SD, 95%CI) for the complete study area during 2003-2012, in the regional bird monitoring program in Argentina for each species. (a) Raptors; (b) ground omnivores and herbivores; (c) ground granivores; (d) other granivores; (e) insectivores mostly associated with folliage; (f) other insectivores.

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APPENDIX E

CHAPTER 3 ADDITIONAL TABLES OF MODEL RESULTS

Table E.3.1. Best models (ΔQAICc <2) predicting bird occupancy ($\hat{\psi}$) and probability of detection ($\hat{p}$), indicating number of parameters (K), ΔQAICc values, weights, and $\hat{c}$ in the interior of soybean fields in, Entre Ríos, Argentina, 2007-2009. Separate analyses for seasons and different groups are indicated, but see Table 3.1. for details of group composition: ground omnivores and herbivores (GROUND), Columbiformes (COLUM), ground nesters passerine granivores (GRGR), and insectivorous aerial foragers and salliers (AER).

	Model	K	ΔQAICc	*weight*	$\hat{c}$
GROUND, Season 1	p(groups + tmin)Psi(.)	5	0.000	0.229	1.000
	p(groups * tmin)Psi(.)	7	1.995	0.084	1.000
GROUND, Season 2	p(.)Psi(.)	2	0.000	0.090	1.903
	p(groups)Psi(.)	4	0.272	0.079	1.903
	p(.)Psi(fors)	3	0.875	0.058	1.903
	p(groups)Psi(bare)	5	1.786	0.037	1.903
	p(.)Psi(leaf)	3	1.924	0.035	1.903
	p(wind)Psi(.)	3	1.975	0.034	1.903
GROUND, Season 3	p(.)Psi(.)	2	0.000	0.538	2.801
GROUND, Season 4	p(.)Psi(.)	2	0.000	0.675	1.000
COLUM, Season 1	p(.)Psi(.)	2	0.000	0.166	1.647
	p(wind)Psi(fors)	4	0.361	0.139	1.647
	p(wind)Psi(stub)	4	0.600	0.123	1.647
	p(wind)Psi(bare)	4	0.755	0.114	1.647
	p(.)Psi(stub)	3	1.585	0.075	1.647
	p(.)Psi(fors)	3	1.861	0.066	1.647
GRGR,	p(.)Psi(size)	3	0.000	0.189	1.286

Season 1	p(.)Psi(.)	2	0.017	0.188	1.286
	p(.)Psi(fors)	3	1.435	0.092	1.286
	p(tmin)Psi(.)	3	1.672	0.082	1.286
GRGR, Season 2	p(.)Psi(.)	2	0.000	0.188	1.000
	p(tmin)Psi(.)	3	0.968	0.116	1.000
	p(.)Psi(fors)	3	1.350	0.096	1.000
	p(wind)Psi(.)	3	1.963	0.071	1.000
GRGR, Season 3	p(.)Psi(.)	2	0.000	0.172	2.082
	p(tmin)Psi(.)	3	0.481	0.135	2.082
	p(.)Psi(fors)	3	1.426	0.084	2.082
	p(wind)Psi(.)	3	1.429	0.084	2.082
	p(tmin)Psi(fors)	4	1.480	0.082	2.082
	p(.)Psi(herb)	3	1.832	0.069	2.082
GRGR, Season 4	p(.)Psi(.)	2	0.000	0.220	1.696
	p(.)Psi(size)	3	1.025	0.132	1.696
	p(tmin)Psi(.)	3	1.950	0.083	1.696
	p(wind)Psi(.)	3	1.986	0.082	1.696
AER Season 2	p(.)Psi(.)	2	0.000	0.307	2.590
	p(.)Psi(size)	3	1.358	0.156	2.590
	p(.)Psi(fors)	3	1.753	0.128	2.590
AER Season 3	p(tmin)Psi(.)	3	0.000	0.652	1.252
	p(tmin)Psi(fors)	4	2.808	0.160	1.252

Notes: p(.) and Psi(.) are constant detection and occupancy probabilities on a logit scale.

p(groups), p(tmin), p(wind) are bird groups, minimum temperatures, and wind, respectively, as coefficients affecting detection probabilities on a logit scale.

Psi(fors), Psi(bare), Psi(leaf), Psi(stub), Psi(size), Psi(herb) are forest distance, bare soil cover,

leaves cover, stubble cover, field size, and herbaceous vegetation cover, respectively, as

coefficients affecting occupancy probabilities on a logit scale.

Table E.3.2. Best models (ΔQAICc <3) predicting bird occupancy ($\hat{\psi}$) and probability of detection ($\hat{p}$), indicating number of parameters (K), ΔQAICc values, weights, and $\hat{c}$ at the border of soybean fields in Entre Ríos, Argentina, 2007-2009. Different groups are indicated, but see Table 3.1. for details of group composition: ground omnivores and herbivores (GROUND), Columbiformes (COLUM), granivore foliage gleaners (GRAN), ground nesters granivore foliage gleaners (GRGR), insectivores (INS), and insectivorous aerial foragers and salliers (AER).

	Model	K	ΔQAICc	*weight*	$\hat{c}$
	p(ttre)Psi(.)	3	0.000	0.168	1.007
	p(.)Psi(.)	2	1.748	0.070	1.007
	p(wind)Psi(.)	3	1.898	0.065	1.007
	p(ttre)Psi(hh)	4	2.096	0.059	1.007
GROUND	p(ttre)Psi(dvcv)	4	2.148	0.058	1.007
	p(ttre)Psi(ntree)	4	2.191	0.056	1.007
	p(ttre)Psi(fors)	4	2.203	0.056	1.007
	p(hheigh)Psi(.)	3	2.446	0.050	1.007
	p(.)Psi(ntree)	3	2.951	0.038	1.007
	p(groups * ttre)Psi(groups + ntree)	10	0.000	0.141	1.226
	p(groups + ttre)Psi(groups + ntree)	8	0.095	0.135	1.226
	p(groups + ttre)Psi(groups * ntree)	10	0.135	0.132	1.226
	p(groups + ttre)Psi(groups + ntree + hh)	9	0.398	0.116	1.226
	p(groups * ttre)Psi(groups + ntree + hh)	11	0.478	0.111	1.226
COLUM	p(groups + ttre)Psi(ntree)	6	1.132	0.080	1.226
	p(groups + ttre + wind)Psi(groups + ntree)	9	2.084	0.050	1.226
	p(groups + ttre + wind)Psi(groups * ntree)	11	2.194	0.047	1.226
	p(groups + ttre + wind)Psi(groups + ntree + hh)	10	2.401	0.043	1.226
	p(groups * ttre)Psi(ntree)	8	2.421	0.042	1.226
	p(groups * ttre)Psi(groups * ntree)	12	2.691	0.037	1.226
	p(ttre)Psi(groups + ntree + hh)	6	0.000	0.395	1.256
GRAN	p(groups + ttre)Psi(groups + ntree + hh)	7	1.996	0.145	1.256
	p(ttre)Psi(groups + ntree)	5	2.864	0.094	1.256
	p(groups * hheigh)Psi(groups * ntree)	12	0.000	0.167	1.436
	p(groups * hheigh)Psi(groups + hh)	10	2.378	0.051	1.436
GRGR	p(groups * hheigh)Psi(groups)	9	2.526	0.047	1.436
	p(groups * hheigh)Psi(.)	7	2.544	0.047	1.436
	p(groups * hheigh)Psi(groups + dvcv)	10	2.746	0.042	1.436
	p(groups * hheigh)Psi(groups * hh)	12	2.832	0.041	1.436

	p(groups + ttre + wind)Psi(groups * hh)	11	2.900	0.039	1.436
INS	p(groups * ttre)Psi(ntree)	12	0.000	0.499	2.000
AER	p(TMIN)Psi(ntree)	4	0.000	0.219	1.313
	p(ttre)Psi(fors)	4	1.367	0.110	1.313
	p(groups + tmin)Psi(ntree)	6	1.836	0.087	1.313
	p(TMIN)Psi(groups + ntree)	6	2.279	0.070	1.313
	p(.)Psi(ntree)	3	2.884	0.052	1.313

Notes: p(.) and Psi(.) are constant detection and occupancy probabilities on a logit scale.

p(ttre), p(wind), p(hheigh) , p(groups) are total number of trees in border, wind, herbaceous vegetation height, and bird groups, respectively, as coefficients affecting detection probabilities on a logit scale.

Psi(hh), Psi(dvcv), Psi(ntree), Psi(fors), Psi(groups) are herbaceous height, dead vegetation cover, number of native trees in border, forest distance, and bird groups, respectively, as coefficients affecting occupancy probabilities on a logit scale.

Table E.3.3. Poisson generalized linear mixed models (ΔAICc <3) predicting abundance of main arthropod orders consumed by birds, with two collecting methods (net and vacuum), indicating fixed and random effects, number of parameters (*K*), ΔAICc values, and weights at the interior and edges of soybean fields in Entre Ríos, Argentina, 2007-2009, along the crop cycle.

	Fixed effect	Random effect	*K*	ΔAICc	*weight*
Center (net)	grass, season	transect, obs	7	0.000	0.506
	grass, season	field, transect, obs	8	0.877	0.327
	grass, bare, season	transect, obs	8	2.217	0.167
Center (vacuum)	Bare	transect, obs	4	0.000	0.130
	Null	transect, obs	3	0.109	0.123
	bare, ins	transect, obs	5	0.898	0.083
	Grass	transect, obs	4	0.920	0.082
	Ins	transect, obs	4	1.220	0.070
	Null	field, transect, obs	4	1.270	0.069
	grass, bare	transect, obs	5	1.558	0.060
	Bare	field, transect, obs	5	1.818	0.052
	Grass	field, transect, obs	5	1.888	0.050
	stage, grass	transect, obs	7	1.972	0.048
	Stage	transect, obs	6	2.036	0.047
	grass, ins	transect, obs	5	2.068	0.046
	grass, stage, ins	transect, obs	8	2.479	0.038
	grass, bare, ins	transect, obs	6	2.526	0.037
	Ins	field, transect,obs	5	2.598	0.035
	bare, ins	field, transect,obs	6	2.916	0.030
Border (net)	h.height, dead veg, season	border, obs	8	0.000	0.273
	dead veg, season	border, obs	7	0.384	0.225
	h.height, dead veg, season	border, obs	9	2.199	0.091
	h.height, ntree, dead veg, season	border, obs	9	2.309	0.086
	ntree, dead veg, season	border, obs	8	2.310	0.086
	h.height, dead veg, ins, season	border, obs	9	2.349	0.084
	dead veg, season	border, obs	8	2.440	0.081
	dead veg, ins, season	border, obs	8	2.600	0.074
Border (vacuum)	h.height, ins, season	field, border, obs	9	0.000	0.189
	h.height, ntree, dead veg, ins, season	field, border, obs	11	0.043	0.185
	h.height, season	field, border, obs	8	0.697	0.133
	h.height, ntree, ins, season	field, border, obs	10	0.699	0.133

h.height, ntree, dead veg, season	field, border, obs	10	1.039	0.112
h.height, ins, dead veg, season	field, border, obs	10	1.419	0.093
h.height, ntree, season	field, border, obs	9	1.430	0.092
h.height, dead veg, season	field, border, obs	9	2.200	0.063

Notes: Abbreviations are: grass, herbaceous vegetation cover; season, sampling season; bare, bare soil cover; null, constant; ins, insecticide applications; h.height, herbaceous vegetation height; dead veg, dead vegetation cover; ntree, number of native trees in border; obs, observation.

APPENDIX F

CHAPTER 4 TABLE OF BIRD SPECIES

Table F.4.1. Insectivorous bird guilds, with the component species, selected to model occupancy in soybean and border fields in Paraná department, Entre Rios, Argentina, 2007-2009.

Guild	Common name – *Scientific name*
Insectivorous salliers	Great Kiskadee - *Pitangus sulphuratus*
	Vermilion Flycatcher - *Pyrocephalus rubinus*
	Tropical Kingbird - *Tyrannus melancholicus*
	Fork-tailed Flycatcher - *Tyrannus savanna*
	White Monjita - *Xolmis irupero*
Aerial foragers	Cliff Sparrow - *Petrochelidon pyrrhonota*
	Brown-chested Martin - *Phaeoprogne tapera*
	White-rumped Swallow - *Tachycineta leucorrhoa*
Insectivorous foliage gleaners	Great Antshrike - *Taraba major*
	Little Thornbird - *Phacellodomus sibilatrix*
	Freckle-breasted Thornbird - *Phacellodomus striaticollis*
	Chotoy Spinetail - *Schoeniophylax phryganophila*
	Pale-breasted Spinetail - *Synallaxis albescens*
	Sooty-fronted Spinetail - *Synallaxis frontalis*

Masked Yellowthroat - *Geothlypis aequinoctialis*

Masked gnatcatcher - *Polioptila dumicola*

House Wren - *Troglodytes aedon*

Small-billed Elaenia - *Elaenia parvirostris*

Euler's Flycatcher - *Empidonax euleri*

Tawny-crowned Pygmy Tyrant - *Euscarthmus meloryphus*

Bran-colored Flycatcher - *Myiophobus fasciatus*

White-crested Tyrannulet - *Serpophaga subcristata*

	Firewood-gatherer - *Anumbius annumbi*
Ground	Rufous Hornero - *Furnarius rufus*
insectivores	Short-billed Pipit - *Anthus furcatus*
	Cattle Tyrant - *Machetornis rixosus*

APPENDIX G

INTERVIEWS TO AGRICULTURAL PRODUCERS IN ENTRE RÍOS, ARGENTINA

INTRODUCTION

This appendix presents additional information collected to capture the background of agricultural

producers in Entre Ríos, how they make decisions concerning application of pesticides, and their

perceptions on natural vegetation and birds in the rural environment. I also present interviews

responses revealing producers' economic, social, and ecologic-environmental information.

METHODS

To understand how farmers make management decisions and what changes are they willing to

accept, we conducted interviews (n=24) in Cerrito, El Pingo, Maria Grande and Palenque in

Entre Rios, during July 2011. Given the interviews were exploratory, we counted with a

convenient and random sample from the contacts made in the past, those made by the local

extension agency from Instituto Nacional de Tecnología Agropecuaria (INTA), and by the local

agrarian federation (Federación Agraria). We oriented the questions to capture their background,

how decisions on agrochemical applications are made, and their understanding on ecosystems

services. Next, I asked questions involving their willingness to accept alternative management

decisions regarding pesticide applications, and border management by planting different

numbers of native trees. Finally, we used a stated preference approach to capture their

willingness to pay (WTP) for planting native trees to achieve bird conservation. Last, triangulated the responses regarding decisions on pesticide applications with professionals, and score them according an environmentally friendly perspective, averaging the different opinions.

We also used the information collected in 40 interviews conducted in 2007 at Paraná department, in the cities of María Grande, Aldea Santa María, Cerrito y Crespo, revealing producers' economic, social, and ecologic-environmental information (Zaccagnini et al. 2008, unpublished).

RESULTS AND DISCUSSION

Most farmers in the area have worked there since they were 10-20 years old, and live in rural communities, or in nearby cities (Zaccagnini et al. 2008, unpublished). Of the interviewees, only 58% completed primary school, 33% secondary and 3% university. Most farmers (80%) think that agricultural activities generates risks to the people and environment, and those risks are associated with the improper use of agrochemicals, followed by biodiversity loss, and health problems. Most farmers revealed they think pollution could affect their future rural lifestyle. Moreover, most producers in the area, recognize the value of biodiversity, especially insectivorous birds; in general, they understand organisms like prey and insectivorous birds, frogs and toads, snakes, and some insects to be necessary for the ecosystem.

In 2011, we interviewed 15 producers living in urban and 9 in rural areas, and as we will show, there were no evident differences in any of the responses from farmers leaving in rural or urban areas. Soybean represented 65% of their lands; and on average they grew 455.4$\pm$ 94.9 Ha (median=240 Ha) of this crop in summer 2010-1011 (Fig. G.4.1). Most farmers own the lands they cultivate soybean, and some rent the lands (Fig. G.4.2). We found that producers apply

herbicides 2.5±0.6 times, and 2±0.7 times insecticides, per soybean cycle (Fig.G.4.3, G.4.4) .

Most farmers acknowledged that some insects perform pest control; however, three of 24

interviewees did not know some birds feed on insects, representing a potential source of pest

control. All interviewees agreed with the statement that native forests could function as habitat

for native bird diversity.

We considered responses with respect the timing of herbicides application as

environmentally friendly those who based decisions on monitoring for the presence of weeds

(39%), followed by those who only rely on expert advice (48%), and last those who reported

decisions by choice, without any justification (9%). With respect to decisions about which

herbicides used, we classify those who responded that it depends on the weeds, and those who

use the least harmful as the most environmentally friendly practices (22%), followed by those

who rely on expert advice (30%); expert advice, cost and choice (22%); and last, those who use

glyphosate only (26%). When we pooled both questions regarding herbicides decisions, on a 0-1

scale, 1 being the most environmentally friendly and 0 the least, 44% of the farmers made

decisions scored below 0.5, 35% between 0.5-0.75, and 22% made the best decisions (Table

G.4.1).

We also looked at the insecticide applications decisions, and regarding the timing of

applications, the most environmentally friendly decisions were those who stated performing pest

threshold monitoring (22%), followed by expert advice only (57%), 'bugs' presence (13%), and

again 9% had no explanation. As to which insecticides the use, 17% expressed the decisions

where subject to the pest, which we classified as the best responses, followed by those following

expert advice only (61%), and then those who depend on the price only or use whatever product

is available, including Endosulfan (17%). When we pooled the responses with respect to

insecticides decision-making, only 18% of the farmers made decisions scored below 0.5, and the remaining 70% made the most environmentally friendly decisions (Table G.4.1). Decision-making regarding application of insecticides would seem to be carried out in a more careful fashion, when compared to decisions on herbicides. Overall, 61% of the decisions farmers make concerning insecticides and herbicides scored closer to what we considered environmentally friendly; however, we should be aware of the 17% that were classified with the lowest scores (Table G.4.1).

More than 90% of the interviewees stated they would let someone plant between 10-20 native trees in the field border, provided no cost associated (Fig. G.4.5). The percentage rapidly decreased to 60% when asked if they would accept 20-30 trees; and from those who didn't accept, half said that it depends on how much space is needed. Most agreed on conserving the trees that they already have in their borders, but from those who did not, some said they would conserve the trees but eliminate the rest of natural vegetation. Less 20% of producers were willing to pay half of the price of planting trees in their borders, while the remaining split between those who were unwilling to pay, and those whose willingness depended on cost and tree species. Some who expressed unwillingness to pay, offered labor to plant and take care of the trees.

In our attempt to use a stated preference approach to capture agricultural producers' willingness to pay for planting native trees to achieve bird conservation, we did not succeed in assigning monetary value, because interviewees were mostly unsure on their responses (Fig. G.4.5). This technique is the only one used to associate values to natural resources (non-market resources) like bird conservation, and their ecosystem services, although it is still controversial (Liu et al. 2010). The negative results on this approach I experienced in this study is not

uncommon given the difficulty people have on assigning a monetary value to something not exchanged in the market (Gregory et al. 2012). In general, valuation of biodiversity and ecosystem services remains a challenging area among environmental economics, and some other techniques have been gaining popularity in the last decades to solve this problem (Adamowicz 2004, Liu et al. 2010). For example, the Multiple Criterion Decision Analysis (MCDA), this integrates subjective and qualitative variables incorporating uncertainty.

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Gregory, R., L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson. 2012. Structured decision making: a practical guide to environmental management choices. Wiley-Blackwell.

Liu, S., R. Costanza, S. Farber, and A. Troy. 2010. Valuing ecosystem services Theory, practice, and the need for a transdisciplinary synthesis. Ecological Economics Reviews 1185:54-78.

Table G.4.1. Percentage (%) of pooled responses to how the interviewee makes decisions concerning when and which herbicides, and insecticides are applied, in 2011, Paraná Entre Ríos, Argentina. Responses were scored and scaled between zero and one, from the least to the most environmentally friendly management decisions.

Scores	Responses (%)
Herbicides	
Low (0-0.25)	22
Mid-low (0.25-0.5)	22
Mid-high (0.5-0.75)	35
High (0.75-1)	22
Insecticides	
Low (0-0.25)	9
Mid-low (0.25-0.5)	9
Mid-high (0.5-0.75)	13
High (0.75-1)	70
Overall	
Low (0-0.25)	17
Mid-low (0.25-0.5)	22
Mid-high (0.5-0.75)	39
High (0.75-1)	22
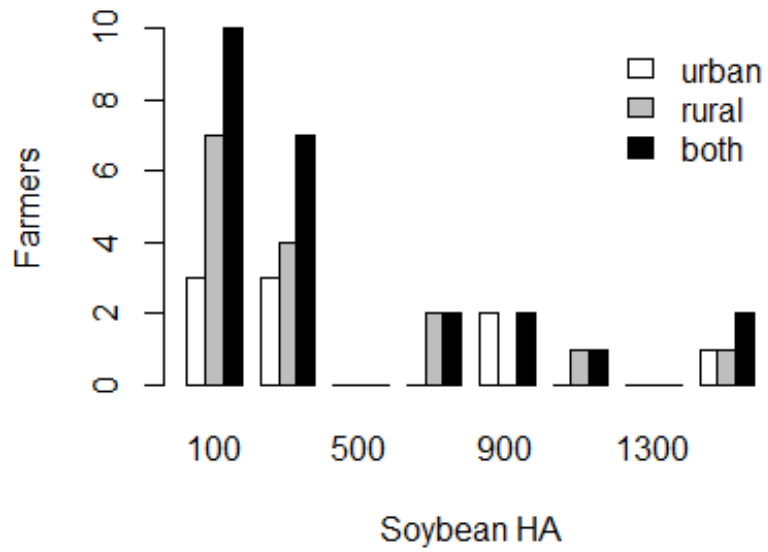
Figure G.4.1. Hectares of soybean sown in 2010/11 by producers living in urban and rural areas in Cerrito, El Pingo, Maria Grande and Palenque, in Entre Rios.
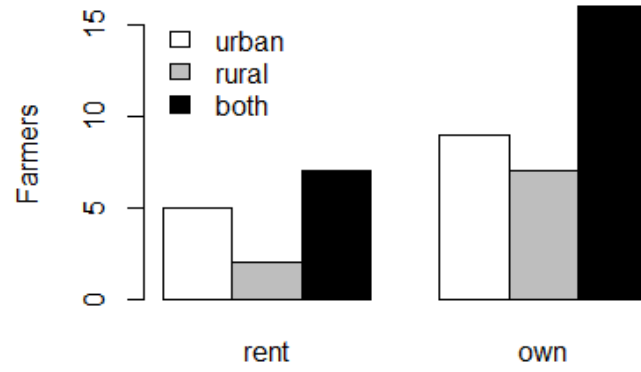
Figure G.4.2. Farmers who cultivate soybean on own or leased lands 2010/11 in Cerrito, El
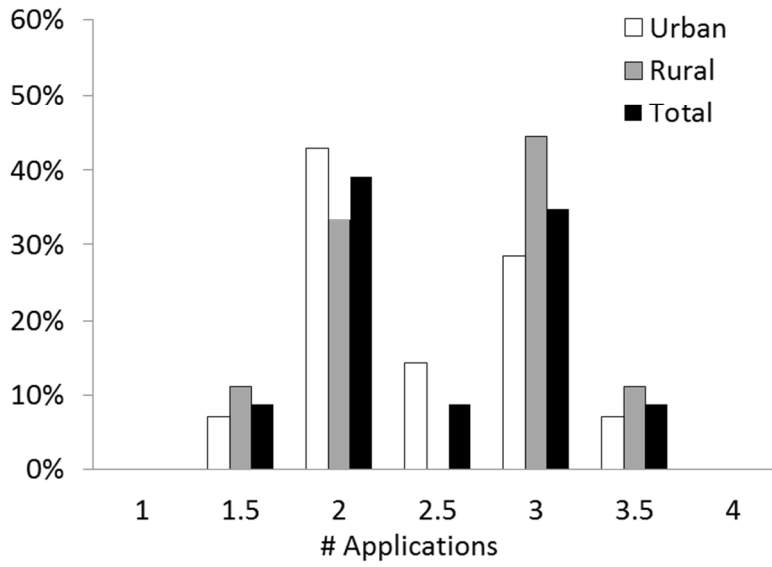
Pingo, Maria Grande, and Palenque, in Entre Rios.

Figure G.4.3. Responses on average number of herbicide applications per soybean cycle 2010/11

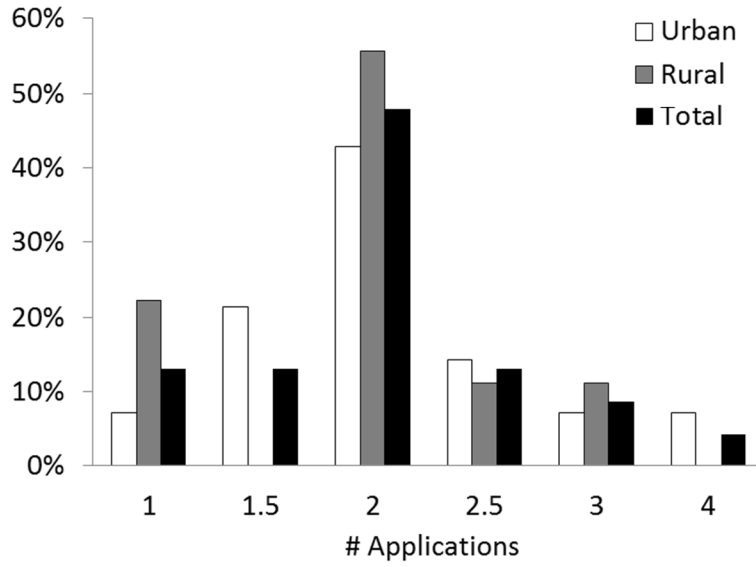in Cerrito, El Pingo, Maria Grande and Palenque, in Entre Rios.

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Figure G.4.4. Responses on average number of insecticide applications per soybean cycle

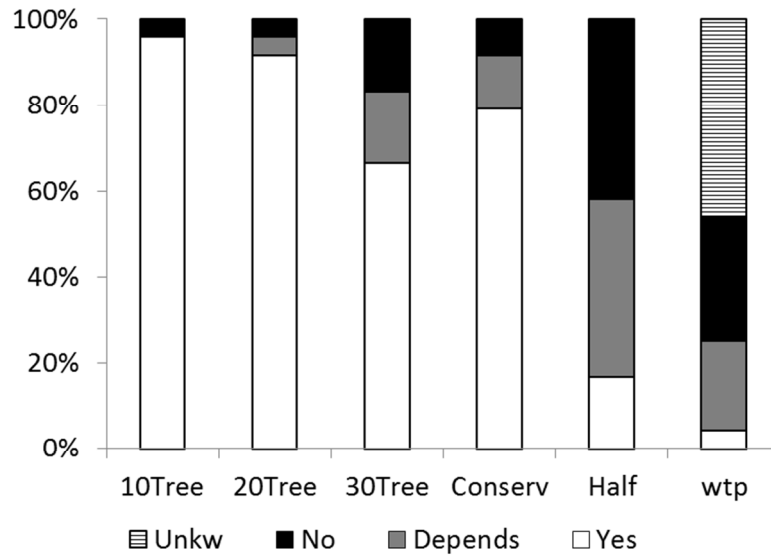2010/11 in Cerrito, El Pingo, Maria Grande and Palenque, in Entre Rios.

Figure G.4.5. Percentage (%) of responses to border management actions: plant 5-10 (10Tree),

10-20 (20Tree), or 20-30 (30Tree) native trees at no cost; conserve original borders; pay half the

cost; and willingness to pay for planting (wtp), in 2011 by producers in Cerrito, El Pingo, Maria

Grande, and Palenque, in Entre Rios.