Agriculture and Gamebirds: A Global Synthesis with and emphasis on the Tinamiformes in Argentina

by

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(Under the direction of John P. Carroll)

Abstract

I conducted a synthesis of agricultural changes during the mid-18th century until the present in the principal temperate agricultural regions of the world (western Europe, the United States of America, the Republic of South Africa, and austral South America, with a concentration on the Republic of Argentina) from the perspective of the impacts that intensification in agriculture has on wildlife in agroecosystems. Additionally, I produced a review and synthesis of the effects of agricultural land use on birds, with an emphasis on the Galliformes. Based upon these analysis, and field research on the spotted tinamou (*Nothura maculosa*) in agroecosystems in the Pampas of Argentina, I discuss the past, present, and future implications for wildlife conservation and management in agroecosystems in the Pampas and Chaco and Yungas forest of Argentina.

In the regions analyzed, agricultural production has become increasingly intensified since the mid-20th century; typified by increased mechanization, irrigation, agrochemical use, farm consolidation, regional specialization, area of cultivated pastures, and livestock densities. Combined, these reduce the amount, quality, and heterogeneity in habitats across scales from the region to within fields, which the analysis revealed were the factors most influential in determining distribution and abundance of avian species dependent upon these systems. Specifically, the loss of fallow or idle land, woody encroachment,

homogeneity in cover types and vegetation structure and composition, indirect effects of pesticides on food availability, and earlier and more frequent mowing were key in explaining reductions in avian diversity and abundance in temperate agroecosystems.

For the Galliformes the loss and/or degradation of preferred habitats, during both the breeding and non-breeding seasons, due to changes in agroecosystem management was related to observed decreases in populations and survival. Over-winter mortality increased during extremes in minimum temperature and snow cover, and was exacerbated where sufficient wintering cover was limited. Moreover, increased over-winter mortality was associated with the proximity of woody areas, which facilitates higher predation. The loss of preferred nesting habitat not only decreases the number of nesting individuals, but increases nest loss and mortality of incubating adults through increased predation and losses to agricultural activities. Of particular importance were decreases in the abundance of preferred arthropod prey for foraging chicks due to direct and indirect effects of pesticides, which were responsible for increased chick mortality.

Although there exists a large body research into the effects of land use and birds, particularly gamebirds, little research exists for these species in Argentina or Latin America in general. In Argentina the most important gamebird species is the spotted tinamou (*Nothura maculosa*). This species has become increasingly scarce in a significant portion of its range, possibly due to agricultural intensification over the last 15 years. Using radio telemetry, I examined habitat use, movements, and survival of spotted tinamous in 2 landscapes in the province of Buenos Aires, Argentina; one dominated by annual row crops and the other used for annual crops and grazing. During winter, individuals used in order of preference: fallow fields and areas with short herbaceous vegetation, followed by wetlands. Areas in winter wheat and field edges were used least in relation to their availability. Although birds generally maintained small home ranges, in some cases changes in cattle density and the structure of row crops caused birds to move considerable distances. Survival mid-winter to early spring was more than double in the mixed

landscape ($\hat{s}$ = 0.73, SE = 0.19) compared with the landscape dedicated to row crops ($\hat{s}$ = 0.33, SE = 0.19). Given the general trends documented for the Galliformes in relation to agricultural intensification, and considering the Tinamiformes as ecological equivalents to the Galliformes in agroecosystems, these results are not unexpected and suggest a precarious future for the conservation of grassland and agroecosystem species in Argentina in light of present agricultural trends.

The intensification and expansion of row crop agriculture and grazing in Argentina has negative implications for wildlife as habitat is converted and degraded as witnessed in the spotted tinamou. In the Pampas, the biggest threat for wildlife conservation is the conversion of remnant grassland and residual areas in row crop regions and the expansion of row crops and perennial forage crops into former extensive grazing areas. In northern Argentina the deforestation of Chaco and Yungas forest for soybean cultivation has been extensive and is accelerating, threatening the relatively high biodiversity of these areas. The lack of sufficient funding and infrastructure, and the decentralized nature of wildlife exploitation in Argentina, hinders effective management. Recent success, however, in managing commercially exploited species (parrots) suggest that innovative specie-specific management actions maybe viable.

INDEX WORDS: Agriculture, Agroecosystem, Argentina, austral South America, Birds, Chaco, Europe, Galliformes, Gamebirds, Grasslands, Grazing, Intensification, North America, Pampas, South Africa, Tinamiformes, Yungas

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Dedication

As always to my parents

And to the residents of San Miguel del Monte and Victorica who took me into their homes and their hearts

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You care for nothing but shooting, dogs, and rat-catching, and you will be a disgrace to yourself and all your family.

— Robert Darwin to his son Charles, 1825

The true biologist deals with life, with teeming boisterous life, and learns something from it, learns that the first rule of life is living.

— John Steinbeck, *The Log from the Sea of Cortez*

Its been a good month of Sundays and a guitar ago, had a tall drink of yesterday's wine, left a long string of friends, some sheets in the wind, and some satisfied women behind

— Billy Joe Shaver *Ride Me Down Easy*

I do not believe that any one could have shown more zeal for the most holy cause than I did for shooting birds.

— Charles R. Darwin

We have read much about the great and rapid changes now going on in the plants and animals of all the temperate regions colonised by Europeans. In this connexion we hear most frequently of North America, New Zealand and Australia; but nowhere on the globe has civilization 'written strange defeatures' more markedly than on that great level area called La Pampa.

— William Henry Hudson, *The Naturalist in La Plata*

.....few countries have undergone more remarkable changes, since the year 1535, when the first colonist of La Plata landed with seventy-two horses. The countless herds of horses, cattle, and sheep, not only have altered the whole aspect of the vegetation, but they have almost banished the guanaco, deer, and ostrich.

— Charles R. Darwin, *Voyages of the Adventure and Beagle*

Chapter 1

Introduction

Globally, the distribution and abundance of terrestrial gamebirds in temperate grazing and agricultural systems have been decreasing over the last 60 years, with these changes occurring across continents (Africa, Europe, North America, South America) and taxa (Galliformes, Gruiiformes, Pteroclidiformes, Tinamiformes). These changes, which have also been evident in songbirds, insects, and plants are largely attributable to the intensification in management of agroecosystems (Krebs et al., 1999; Robinson and Sutherland, 2002; Benton et al., 2003).

Agricultural intensification is the process that maximizes the amount of primary production available for human consumption directly or through first order consumers (*i.e.* livestock). This entails the conversion of low-energy input agricultural systems to high-input systems by using external energy to increase yields. Yields are increased through the use of cultivars that dedicate a greater proportion of their biomass to the production of harvestable products, while petroleum-based fertilizers and pesticides substitute for arable land and natural pest control respectively, machinery substitutes for labor, and irrigation substitutes for precipitation (Naylor, 1996; Matson et al., 1997; Tilman, 1999). Collectively these inputs facilitate the maximizing of yields by augmenting ecosystem functions and reducing the limitations imposed by those functions on production.

The intensification of agriculture, including grazing, affects wildlife by determining the amount, type, configuration, and juxtaposition of habitats. This process leads to decreased habitat heterogeneity across scales ranging from the field, farm, landscape and region, while often leading to a decrease in quality of habitats by reducing food resources and habitat structure and facilitating higher rates of predation (Benton et al., 2003; Freemark and Boutin, 1995; Robinson and Sutherland, 2002). The implications for wildlife of these alterations in the amount, configuration, and quality of habitats has, and is increasingly, a focus of conservation research.

Gamebirds, particularly the Galliformes of temperate regions (Europe, North America, southern Africa), are globally some of the most intensively studied birds due to their economic and cultural significance, with the majority of the most intensively studied species being those associated with row crop and grazing systems. Because of this relationship with agricultural systems much of the research into the population dynamics of Galliformes pertains to land management decisions and practices in agroecosystems. Moreover, research has shown that these species are particularly sensitive to changes stemming from agricultural intensification and serve as indicators of agroecosystem health in temperate regions (Rands, 1988).

Compared to other temperate regions, there has been little research in austral South America into the effects of agricultural management on wildlife or gamebirds in particular. This is of significance since there have been large-scale and rapid changes in agroecosystem management in austral South America during the last 15 years, with the proportionally greatest changes occurring in Argentina. Although there has been some research into the broad ecological effects of the recent changes in agricultural management in Argentina there has been little that has related the potential implications of these changes on wildlife.

Because of the observed responses of terrestrial gamebirds to intensified management of temperate agroecosystems in Europe, North America, and South Africa it can be expected that similar species would be effected in a similar manner in temperate South American agroecosystems. In Argentina, and austral South America, terrestrial Galliformes are absent and are replaced by the ecologically equivalent Tinamiformes (tinamous). Although in Argentina there are 16 specie of tinamous, 5 species (Brushland tinamou *Nothoprocta cinerascens*, Darwin's tinamou *Nothura darwinii*, Elegant-crested tinamou *Eudromia* *elegans*, Red-winged tinamou *Rhynchotus rufescens*, Spotted tinamou *Nothura maculosa*) are the most commonly associated with row crop and grazing systems.

If the Tinamiformes are ecologically equivalent to the Galliformes in Argentine agroecosystems than it would be expected that impacts from the intensification of agricultural management upon the ecology and abundance across both orders would be similar. If this is the case than inferences can be made based upon Galliformes in relation to the ecology of the Tinamiformes (Thompson, 2004) as well as the general effects of agricultural intensification on wildlife in Argentine agroecosystems.

Although there exists a plethora of studies on the ecology of Galliformes of temperate regions in relation to agricultural effects there has not been a cross-continent synthesis of the commonalities of these effects. Nor has there been an evaluation and synthesis of the pattern and process of agricultural intensification amongst global temperate regions from the perspective of its potential impact on wildlife. To understand the commonalities and differences in agricultural differences across continents, and its impacts on gamebirds, requires an analysis and synthesis of both agricultural changes that are relevant to gamebirds and the ecological implications of those changes on gamebird ecology.

Here I present a review and synthesis of the process of agricultural intensification in the prinicipal agricultural regions of North America, Europe, South Africa, and austral South America, in the context of the potential impacts of the process on wildlife, with particular emphasis on Argentina. Additionally, I conduct a meta-analysis of research into the effects on avian ecology in temperate regions stemming from agricultural intensification, including a specific meta-analysis on Galliformes. Using these synthesis and analysis I evaluate the implications for wildlife conservation of the large-scale changes in agriculture and grazing that have occurred in Argentina in recent years, including specific research on the Spotted Tinamou (*Northura maculosa*) in agricultural systems in Argentina.

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Chapter 2

The agricultural and ecological characteristics of agricultural intensification

2.1 Components of agricultural intensification

Globally, the intensification of agriculture has 4 major components in common: the mechanization of farm processes in lieu of draft animals and associated decreases in inputs of human labor; introduction of higher yielding, hybridized cultivars; increased use of chemical inputs in the form of fertilizers and pesticides, and the improvement of management of farm operation (i.e. farm specialization and amalgamation) (Cochrane, 1993; Fuller, 2000; Shrubb, 2003; Pain and Pienkowski, 1997; Solbrig and Vera, 2001). Additional changes in management, such as irrigation are also important, but generally follow the previously mentioned improvements and is found only in some areas.

2.1.1 Farm mechanization

Labor saving mechanization of agriculture has a history as long as that of agriculture, however, the process increased rapidly in the later half of the 19^{th} century and further accelerated with the introduction of the internal combustion engine during the 1930s (Rasmussen, 1982; Cochrane, 1993; O'Connor and Shrubb, 1986; Shrubb, 2003). Formerly, the amount of land in production was dependent upon time constraints imposed by how much land could be prepared, planted, and harvested within a sufficient time frame, the result being that not all land could be put into production and was often rotated among crops, fallow, and pasture.

As farm machinery has increased in size over the last 60 years it has allowed for faster tilling, planting, and harvesting, leading to increases in the area in row crop production, as well as consolidation of fields (Primdahl, 1990; Meeus et al., 1990; Fuller, 2000). Moreover, advances in the technology of farm machinery, particularly the internal combustion engine, has greatly reduced the need for farm labor. For example, the largest animal drawn combines required 5 laborers to complete all the same harvesting operations that a small sized mechanized combine can complete with a single operator.

2.1.2 Improved cultivars

Increases in crop yields have mainly come from the development and introduction of crop strains that mature quickly, more efficiently utilized inorganic fertilizers, and dedicate a higher proportion of their biomass to the production of edible parts. There has been no net gain in biomass of cultivars; instead energy allocation has been diverted from combating pest and competitors, this being substituted with pesticides, and from root development with water and nutrient uptake substituted by irrigation and inorganic fertilizers (Evans, 1993; Naylor, 1996).

The improvement in cultivars has culminated since the late 20th century in the introduction of genetically modified (GM) crops, principally cotton, corn, and soybean. Gentically modified cotton and corn include insect resistant varieties incorporating ecotoxins from the bacterium *Bacillus thuringiensis* (Bt) and the more common herbicide resistant varieties of soybeans, corn, and cotton that are resistant to glyphosate herbicide, often marketed as "Roundup Ready®". These crops, allow though highly regulated in Europe, have globally gained in importance and have become dominant in many regions (Gould, 1998; James, 2006; Traxler, 2006).

2.1.3 Agrochemicals

An important component of agricultural intensification is the use of inorganic fertilizers and chemical pesticides (Matson et al., 1997; Naylor, 1996; Pimentel et al., 1993). As noted above the use of agrochemicals, combined with high-yield cultivars dependent upon agrochemicals, is an integral part of agricultural intensification. Globally, fertilizer and pesticide use increased starting in the mid-20th century, with this trend continuing to the present. It some areas total pesticide use has decreased with increasing pesticide efficacy, however, the number of application have greatly increased (Ewald and Aebischer, 2000; Freemark and Boutin, 1995; Pimentel et al., 1993).

The large-scale adoption of inorganic fertilizers in agriculture has its roots in the late 1940's and 1950's when munition plants in North America and Europe, built for the war effort during World War II, were converted for fertilizer production. The growing importance of inorganic fertilizers in agriculture is illustrated by its exponentially increasing use in the United States up to about 1980 (Fig. 2.1). Whereas in Europe and North and Central America fertilizer consumption has leveled off or decreased, in other regions, particularly Asia and South America its use has continued to increase (Fig. 2.2).

Traditionally the control of arthropod pests was dependent upon natural controls (*e.g.* predators) and through crop rotations and fallows which inhibited the growth of pest populations beyond detrimental levels. Similarly, crop rotations and fallow periods combined with soil disturbance were the standard practices for weed control. Although the development and use of pesticides (insecticides and fungicides) began in the 1800's their widespread adoption did not begin until the 1940's with the introduction of the insecticide DDT and growth-regulating herbicides, followed by fungicides in the 1950's. The toxicity of pesticides have decreased due to environmental concerns, however, they have greatly increased in effectiveness, specificity, and efficacy.

As mentioned above the increasing effectiveness of pesticides was, and is, an important factor in the development of high productivity cultivars since plants do not need to devote resources in defense against pests or in competition with other plants. Meanwhile farm machinery has been developed to more effectively and economically apply herbicides, which combined with their efficacy has also allowed for the development of no-till management, and in turn improved harvest efficiencies by reducing weed abundance.

2.1.4 Farm management

With farm labor not dependent upon draft animals, labor savings stemming from mechanization, and well developed transportation infrastructures there has been a trend towards the simplification of farm practices, regional specialization in production, and amalgamation of fields and farms (Robinson and Sutherland, 2002). Increased mechanization has eliminated the need to produce fodder and maintain pastures for draft animals and has made the amalgamation of fields economically feasible and more profitable. Furthermore, based upon economies of scale it is also more profitable for farms to be larger to maximize the return of investment on farm machinery and other expenditures (Fuller, 2000).

Eliminating the need to produce forage for farm animals, combined with economies of scale, has driven the shift towards the increased area under cultivation; a function if the greatly increased efficiency and size of farm machinery that has eliminated limitations to the area that can be under cultivation. Whereas formerly land was in essence waiting to be put into production, limited by the amount of land that could be worked, farm machinery has now become so efficient that effectively tractors now wait for more land to work.

Due to the ability to maximize the area under cultivation and the economic advantage stemming from economies of scale has been integral to regional specialization in production since land can be dedicated to its most productive and profitable use. Furthermore regional specialization leads to, and is also faciltated by, improving transportation networks while developing a specialized infrastructure devoted to the production of those crops which further drives the process of specialization.

2.2 Effects of agricultural intensification on the diversity and quality of wildlife habitat

Agricultural land use determines the amount, type, quality, and distribution of habitats and the ecological processes occurring in those habitats. Both the spatial and temporal effects of agricultural management on habitats are hierarchically stratified, acting synergistically, to impact habitats at the regional, landscape, among farm, within farm, and within field scales (Benton et al., 2003; Fuller, 2000).

From a general perspective the introduction and extensification of agriculture into an area leads to fragmentation and increasing landscape heterogeneity to a point where agricultural lands start to become the dominant land cover. Often with traditional low input agriculture, supporting both mixed grazing and row crops, a high level of habitat heterogeneity is created. As the management of these lands are intensified it results in landscapes that are simplified in the number of habitat types, with fewer and larger patches of those habitats. With intensification, croplands tend towards monocultures that are often continually cropped and temporally synchronized, while the intensification of grazing is generally typified by the introduction of forage crops in lieu of natural pasture and livestock are maintained at higher densities. At the same time agrochemical use often reduces the diversity and availability of food resources (i.e. seeds, invertebrates) and the vegetative structure within habitats (Potts, 1997; Benton et al., 2003; Fuller, 2000; Robinson and Sutherland, 2002; Buckwell and Armstrong-Brown, 2004; Freemark and Boutin, 1995; Fuhlendorf and Engel, 2001; Orians and Lack, 1992). Here I review how the process of agricultural intensification affects habitat diversity and quality across spatial scales and highlight the temporal effects within spatial scales.

2.2.1 Region

Regional scales agricultural intensification leads to polarization in land use towards dominance by either grazing or row crops dependent upon climatic and edaphic conditions, and transportation infrastructure. For example, in North America the area known as the corn belt was formerly tall grass prairie and is now dominated by corn and soybean agriculture, whereas further west decreased precipitation produces conditions most suitable for wheat production in areas formerly supporting mixed-tall grass prairie. Additionally, regional specialization leads to livestock management that is dependent upon a small suite of species and breeds, as well as similar management practices among farms (Primdahl, 1990; Potts, 1997; Shrubb, 2003; Stoate, 1996; Matson et al., 1997).

2.2.2 Landscape

Farm machinery is capital intensive and thus leads to maximizing the return on investment through the consolidation of farms, which in turn leads to larger areas under common management systems (Hart, 1991; Fuller, 2000; Solbrig and Viglizzo, 2000; Robinson and Sutherland, 2002). Consolidation of farms, combined with regional specialization in land management, creates landscapes that are more homogeneous as to crops and livestock types. The introduction of farm machinery has also removed the need for pastures or feed crops (particularly oats) for draft animals, which in turn allows land for these purposes to be dedicated to the dominant regional land use. Moreover, landscape homogeneity is exacerbated by the commonality in regional management systems stemming from regional specialization.

2.2.3 Within-farm

With the advent of bigger and more powerful machinery, farm efficiency and profitability are increased through the consolidation of fields within farms by removing field boundaries and other non-cropped land (Potts, 1997; Fuller, 2000; Robinson and Sutherland, 2002). This creates larger fields, and due to the efficiency of modern farm machinery, allows for maximizing the area of land in production. Also, profitability is maximized when the diversity of within farm management systems are minimized, which leads to within-farm homogeneity by promoting monocultural production systems.

The availability of inorganic fertilizers eliminates the need for livestock manure for fertilizer in row crops and subsequently leads to the loss of pasture area as it is put into row crop production. Fertilizer use reduces or eliminates the need for fallow periods, allowing for continual cropping and also allows for the compensation of nutrient deficiencies in soils which enables marginal lands to be profitably put into row crops (Matson et al., 1997). Also, in regions dominated by grazing, fertilizer use promotes the management of pasture for silage over natural pasture for grazing and causes rapid growth, allowing for earlier and multiple cuttings per year (Bollinger et al., 1990; Green et al., 1997; Herkert, 1997; Fuller, 2000).

Agricultural intensification increases spatial homogeneity within farms since there is a trend for farms to 1) specialize in row crops or grazing, 2) consolidate fields and pastures by removing field boundaries and formerly unutilized lands, 3) move to specialized management systems, concentrated on a small variety of species (i.e. monocultures), 4) place land unsuitable for row crops into production through the use of inorganic fertilizers, and 5) the concentration of livestock in feed-lots, facilitating pasture conversion to row crops or pasture for silage (Naylor, 1996; Fuller, 2000; Robinson and Sutherland, 2002; Benton et al., 2003).

Temporally, intensification increases farm homogeneity through 1) maintaining more land in production over the course of a year (i.e., double cropping, autumn sowing) and 2) land use operations that are more synchronized which temporally places the land within farms at similar managerial states (i.e. planting, harvesting, haying) (Naylor, 1996; Fuller, 2000; Robinson and Sutherland, 2002; Benton et al., 2003).

2.2.4 Within-field

Within-field management determines the heterogeneity in vegetative structure, floral diversity, and the amount and diversity of food resources available over the course of the year. Intensification leads to greater homogeneity in vegetative structure and lower floral diversity within fields through more efficient sowing of seeds, denser sowing of seeds to shade out weeds, and herbicide use to depress weed populations. The use of inorganic fertilizers eliminates the need for undersowing and cover crops within fields and also facilitates the dominance of the weed community by fewer species (Naylor, 1996; Fuller, 2000; Robinson and Sutherland, 2002; Benton et al., 2003).

With intensification there is a trend towards the development of pastures planted with perennial forage crops over natural pasture. Floral diversity in planted pastures compared to natural hay fields is lower and vegetative structure reduced. These differences are further exacerbated with the use of inorganic fertilizers which, as in row crop fields, leads to homogeneity in vegetative structure and a less diverse weed community. Intensification also leads to increased density of livestock on pastures for longer periods of time, facilitated by planted forage and supplemental feeding, decreasing structural heterogeneity through over-grazing and the lack of unpalatable species (Naylor, 1996; Fuller, 2000; Robinson and Sutherland, 2002; Benton et al., 2003).

The management of fields also determines the amount and availability of food resources, which generally decreases with intensification. Weed populations and diversity are reduced by herbicide use and other weed control management practices which in turn determine the abundance of weed seeds (Draycott et al., 1997; Evans, 1997; British Trust for Ornithology, 2002). Furthermore, the reduction of weed density and diversity directly impact invertebrate populations and diversity by reducing the resources that they depend upon (Potts, 1986; Fuller, 2000; Ewald and Aebischer, 2000; Boatman et al., 2004; Cederbaum et al., 2004).

Aside from the indirect effects of herbicide use on invertebrate density and diversity there are also the obvious direct effects of insecticide use upon invertebrates. The impact of insecticide use on invertebrates is variable and dependent upon taxa, the type and efficacy of the insecticide, and the number and timing of applications (Aebischer, 1990; Ewald and Aebischer, 2000; Boatman et al., 2004). Although the use of pesticides can have significant negative impacts on wildlife populations, the heterogeneity of both insecticide and herbicide application can produce heterogeneity in invertebrate abundance and diversity that is sufficient to mitigate the negative effects (Rands, 1985; Sotherton, 1991; Potts, 1997; Haysom et al., 2004).

The types of crops grown, their management, and harvest efficiency determine the amount and availability of waste grain; an important food source, particularly in the winter (Draycott et al., 1997; Evans, 1997; British Trust for Ornithology, 2002; Shrubb, 2003; Siriwardena and Stevens, 2004). Harvest efficiency has increased with advances in harvest technology, which is dependent upon crop type (O'Connor and Shrubb, 1986; Fuller, 2000; Shrubb, 2003; Krapu et al., 2004; PRECOP, 2005). There have also been large changes in grain drying and storage practices. Grain is now usually dried and stored off the farm, where as formerly grain was cut and dried in the fields and stored on site, leading to higher amounts of waste grain (O'Connor and Shrubb, 1986; Fuller, 2000; Shrubb, 2003).

Even if the amounts of waste grain and weed seeds are high within a field, management practices will affect its availability. Burning or disking of stubble will change the availability of waste grain, as will livestock foraging (Baldassarre et al., 1983; Warner et al., 1985). An important aspect of agricultural intensification has been the development of quick maturing crops that, combined with farm mechanization, have facilitated double cropping systems using autumn sown cereals (Fuller, 2000; Shrubb, 2003), which is of significance because the preparation of fields for sowing in the autumn can reduce the availability of waste grain and weed seeds. Moreover the availability of waste grain and weed seeds is further reduce through the winter and spring as crops mature and decrease foraging efficiency (Fuller, 2000).

These hierarchical processes of agricultural intensification are manifested throughout all areas where the process has occurred, however, all the aforementioned effects have not been documented universally. However, the similarities in climate and the changes that have occurred due to intensification among temperate agroecosystems allow for strong comparisons and inferences to be drawn amongst and within regions. In the following chapters (Chapters 3 and 4) I review and quantify the spatial and temporal changes that have occurred in agricultural management in western Europe, North America, temperate Africa, and austral South America, concentrating on the aspects that are most relevant to the ecology of wildlife in these systems.

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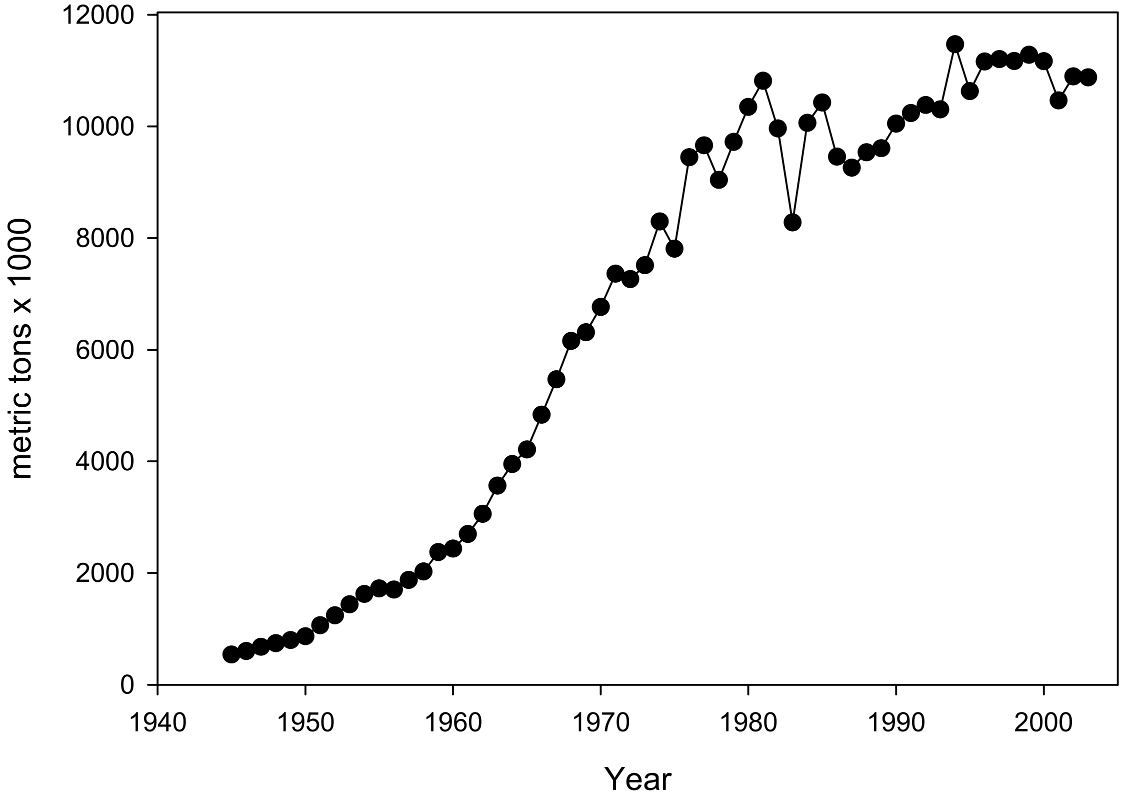

Figure 2.1: Fertilizer consumption in the United States during 1945-2003. Data combined from FAOSTAT (2006) and Alexander and Smith (1990).

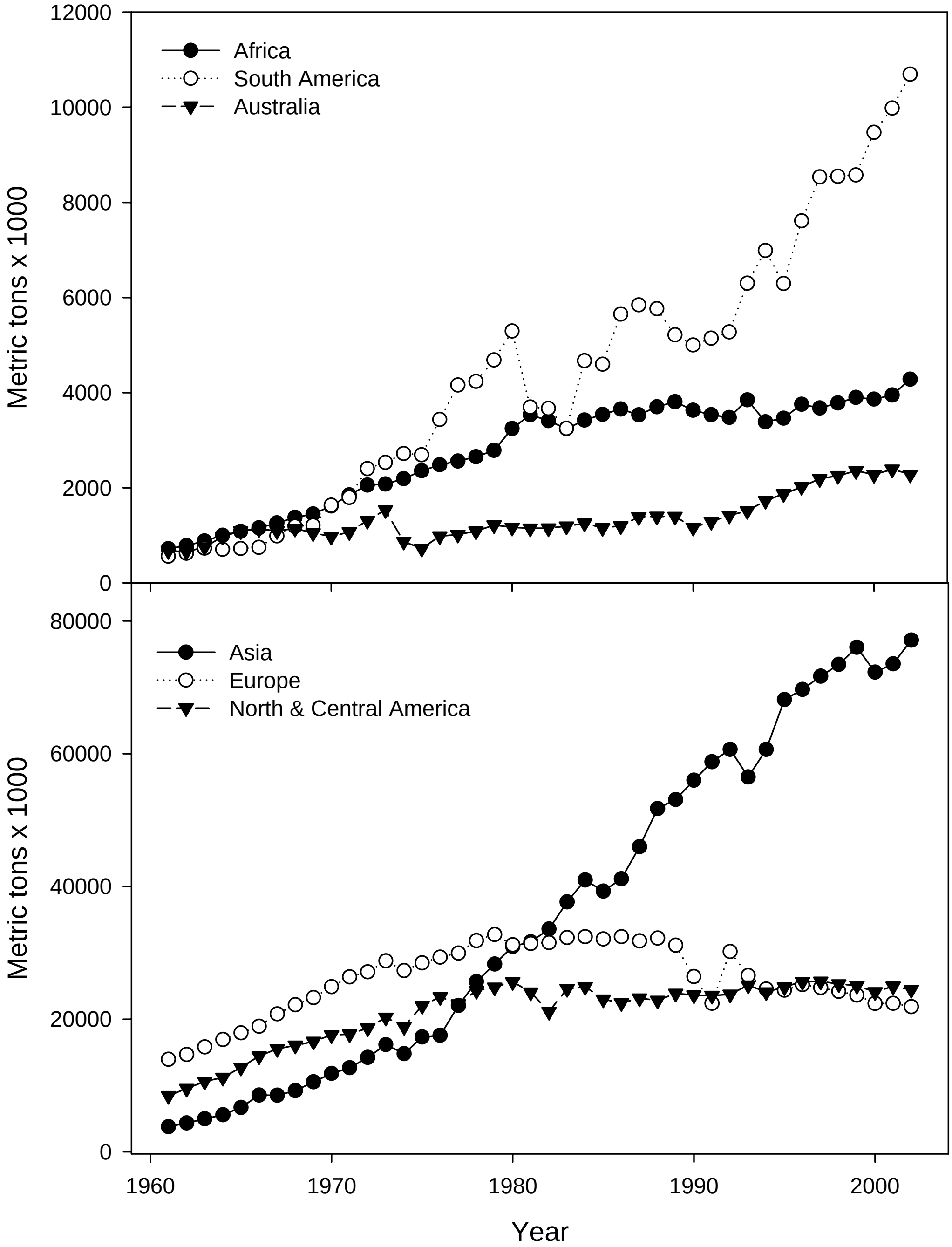

Figure 2.2: Continental level fertilizer use for Africa, South America, Australia, Asia, Europe, and North and Central America during 1961-2002 (FAOSTAT, 2006).

Chapter 3

Global examples of agricultural intensification in temperate ecosystems

The process of agricultural intensification has occurred globally but is most notable in the principal food producing regions of North America, Europe, temperate Africa, and austral South America. All the characteristics of agricultural intensification discussed in Chapter 2 are evident in these regions, although the temporal scales over which the process has occurred differs. Moreover, within regions there have been important spatial patterns that have developed stemming from biological, abiotic, economic, political, and cultural factors that are essential to understanding the ecological significance, and the implications for wildlife conservation, of agricultural intensification in these regions.

3.1 Europe

3.1.1 Western Continental Europe

Agriculture in Europe has a 10,000 year old history and is the dominant land use on the continent, developing a high diversity of management systems as a function of climatic and edaphic conditions in combination with socio-economic factors (Meeus et al., 1990; Potter, 1997; Pinto-Correia and Vos, 2004), and subsequently a large portion of European biodiversity is dependent upon agroecosystems. Until recently (prior to 1945) all of these management systems had been dependent upon large inputs of human and animal labor, while relying upon rotational land use systems that employed multiple cultivars and livestock to maintain soil fertility and minimize pestilence (Stoate, 1996). Furthermore, the management systems employed in many regions were highly integrated with, and dependent upon, forest management (Pinto-Correia and Vos, 2004).

However, since the end of the Second World War large changes in rural demographics and national economic agendas have driven, and been driven by, the intensification of agriculture. An additional factor has been the Common Agricultural Policy (CAP) that has promoted regional specialization in agriculture and the abandonment of row crops and grazing on marginal land (including forest) (Potter, 1997; Robson, 1997; Pinto-Correia and Vos, 2004). Combined, agricultural intensification and the influence of CAP have had profound effects upon European agroecosystems.

Regional specialization stemming from intensification is evident throughout Europe has been exacerbated by marginalization via CAP. The interaction between regionalization and marginalization is conspicuous in Mediterranean regions where there has been agricultural specialization with less productive land taken out of row crop agriculture and used for grazing or abandoned all together (Suárez et al., 1997; Mazzoleni et al., 2004; Roura-Pascual et al., 2005). These processes have lead to arable lands being converted to permanent pasture, and vice versa, while forest cover has increased due to natural forest regeneration or afforestation (Fig. 3.1). Although forests area is expanding, traditional agro-silvo-pastoral management systems are no longer employed frequently in these forests. Meanwhile due to decreasing rural populations and their uncompetitive nature, transhumance grazing systems have largely been abandoned, further accelerating forest expansion (Pinto-Correia and Vos, 2004).

The shift towards more intensive agricultural system in Western Europe is demonstrated by both increases in fertilizer use and increases in farm machinery. Fertilizer use on arable land increased dramatically, particularly in France and Italy, during 1961 through the 1970s (Fig. 3.2). In France and Italy this trend eventually leveled off and has been decreasing, where as the increases in fertilizer application in Spain and Portugal has been slower but continued through 2002 (Fig. 3.2).

The use of farm machinery, both harvesters and tractors, increased in western Europe following the Second World War. The total number of harvesters increased until the mid

1980s, when numbers slowly declined (Fig. 3.3). Compared to the regional trends, in France the increase in harvesters halted in the mid-1970s and slowly declined into the 2000s (Fig. 3.3). The number of tractors exhibited even greater increases in western Europe, although as with harvesters, in France numbers decreased in the latter part of the 20^{th} century (Fig. 3.3).

The leveling off of the numbers of farm machinery in western Europe is due to both market saturation and the increasing power of tractors, which reduced the number of smaller tractors. The number of hectares of arable land per both harvesters and tractors decreased throughout Western Europe for the period of 1961 - 2002, although in France and Portugal the number of hectares per tractor increased during the 1980's through 2002 (Fig. 3.4). Again, these increases are likely related to increased power of farm machinery leading to reduced numbers of smaller machines, but in the case of Portugal, a reduction in the area of arable land.

The intensification in the management of arable land in western Europe also included a large increase in the area of irrigated land (Fig. 3.5) and significant shifts in the types and proportion of cultivars occupying arable land. Most evident have been large increases in forage crops, particularly in France, Italy, and Portugal in the latter half of the 20^{th} century, and in Spain, barley, which is also often used as animal feed (Fig. 3.14). Concurrently, overall cropland diversity decreased, indicated by the amount of arable area in minor cultivars ("Other"; Fig. 3.14). Furthermore, there were decreases in the area of formerly important crops, such as oats and potatoes, whereas industrial oil crops (rape, sunflower) increased in importance (Fig. 3.14).

3.1.2 United Kingdom

As in continental western Europe, the process of agricultural intensification has occurred in the United Kingdom and central and eastern Europe (Meeus et al., 1990; Primdahl, 1990; Wrbka et al., 2004; Shrubb, 2003). In the United Kingdom (UK), unlike much of Europe, there are adequate data to more thoroughly quantify the process of intensification in both space and time.

The effects of regional specialization and marginalization are evident in agriculture in the UK, particularly so for England and Wales, concentrating grazing practices towards the west of England and in Wales, whereas row crops dominate eastern England (Fig. **??**-Fig. **??**). Of note is the increasing trend in the dominance of wheat cultivation (Fig. **??**). Moreover, farms have become specialized so that they are dedicated to one land use or the other, with mixed grazing and row crop management infrequently practiced (Shrubb, 2003). This is illustrated for England (Fig. **??**) and Wales (Fig. **??**) by the reduction in the proportion of arable land in rotational pasture, fodder crops, and bare fallow; suggesting the reduced importance of livestock with row crop farms.

Concurrent with regional specialization there have been significant changes in farm management, particularly the increased use of inorganic fertilizers and pesticides, planted pastures, increased frequency of cutting of pasture, increased use of silage, reduced diversity of cultivars, and the dominance of spring sowing. Meanwhile, livestock have generally become more numerous and concentrated, indicating intensified management (Ewald and Aebischer, 2000; Fuller, 2000; Shrubb, 2003; Stoate, 1996).

Fertilizer application in the United Kingdom increased by 46.1% during 1961 to 1986 based upon a five year running average, and then decreased by 13.3% during 1986 to 2002, for a mean increase over the entire period of 35.9% (Fig. **??**). Recent national level pesticide consumption has averaged 4.9 kg per hectare of arable land from 1990 - 2002 (FAOSTAT, 2006); however, data presented by Ewald and Aebischer (2000) illustrate dramatic increases in the amount, frequency, and efficacy of pesticide application between 1970 and 1995 for a 62 km^2 area in West Sussex, England, a principal cereal producing region of the United Kingdom and are indicative of national level trends.

By 1970, the area of summer cereals, winter wheat, and winter oats and barley treated with herbicides had already increased to nearly 100% in the West Sussex study area of

Ewald and Aebischer (2000) (Fig. 3.15). Between 1970 and 1995, however, there was a trend in winter wheat, winter oats, and barley for increasing numbers of herbicide applications, especially for winter wheat (Fig. 3.15).

In West Sussex, fungicide application was highly variable in summer wheat with no noticeable trends; however, winter wheat and winter oat and barley both exhibited increases in the area treated so that close to 100% of crop area was treated starting in the early 1980's. (Fig. 3.15). Moreover, the number of fungicide applications greatly increased through the 1980's, particularly for winter wheat (Fig. 3.15).

In West Sussex the use of insecticide in spring cereals was uncommon, the maximum percent area treated and percent area sprayed not exceeding 18% (Fig. 3.15). Comparatively, insecticide use in winter wheat and winter barley and oats was extremely high, both crops showing large increases beginning in the early 1980's. Winter barley and oats were consistently treated only once, whereas in winter wheat multiple applications became the norm starting in the late 1980's (Fig. 3.15; Ewald and Aebischer 2000).

As the management of row crop agriculture increased in intensity so did grazing practices. In England cattle numbers began increasing after 1930, peaking in 1980 at close to 8 million head and then decreasing to less than 6 million head in 2002 related to livestock culling due to an outbreak of hoof and mouth disease (Fig. 3.16). Meanwhile, sheep numbers remained stable from the 1920 until 1970, with a small decrease associated with the second World War, doubled between 1970 and 1990, and have exhibited a decline since 1990 (Fig. 3.16).

Similarly, in Wales there has been a substantial increase in the numbers of sheep and cattle. As in England, cattle numbers leveled of and then declined during the latter decades of the 20^{th} century, however, sheep numbers continued an upward trend until 2000. Both England and Wales showed similar patterns in the increase in stocking rates as measured by the number of animal units (AU) per hectare of farmland, with livestock densities leveling off at between 2.0-2.5 AU/ha around 1980 for both areas (Fig. 3.17).

An important factor driving the observed changes in agricultural management has been increasing mechanization into the 1970's, leading to reductions in the number of farm workers and eliminated the need for draft horses by the 1960's (Fig. 3.18-3.20). Also, associated with farm mechanization was an increase in farm size, with the number of farms in England less than 300 acres (121.4 ha) decreasing and the number of farms greater than 300 acres increasing between 1950 and 1970 as farms were amalgamated (Fig. 3.21). After 1970, governmental data are not comparable with previous periods since the minimum size class used for classification of small farms increased since the former classification was not relevant with the continuing loss of small farms. In the United Kingdom smaller farms have become increasingly unprofitable so that at the present the minimum mean farm size that is economically viable in Great Britain is 300-350 hectares (G.R. Potts pers. comm., based on University of Cambridge annual farm surveys).

3.2 United States of America

Compared to Europe the history of large-scale agriculture in what is now the United States is short, commencing with European settlement in the 16^{th} century. The pattern of land conversion and land use was related to the westward expansion of European settlement and the economic, socio-political, technological, and biological (i.e. soil fertility) factors driving land use (Hart, 1968; Rasmussen, 1982; Cochrane, 1993; Whitney, 1994; Gerard, 1995; Hall et al., 2002; Cunfer, 2004).

The expansion of agriculture in the United States was dependent upon deforestation initially in the eastern states and later in the states west of the Appalachian Mountains, peaking between 1880-1920 (Fig. 3.22). During the late 19^{th} row crop agriculture moved into the central and western prairie, followed by the western desert and inter-mountain regions during the 20^{th} century. In the southeastern and northeastern United States the early part of the 20^{th} century illustrated a decrease in the area in farms and cropland, exacerbated by the Great Depression, and a related increase in forest area as abandoned farmland was reforested (Fig. 3.22). In the southeastern states there was an anomaly in this pattern where from 1940 to 1960 the area in farms increased due to the rapid expansion of agriculture and the fencing of open rangeland in Florida.

In the upper midwestern states a reduction in the area in farms was not evident until 1950, however, cropland decreased considerably in association with the Great Depression. Furthermore, unlike the northeastern and southern states forest area did not increase considerably since farmland in this region also occupies former prairie grasslands rather than cleared forest land and large areas of forest in the region remain under timber management (Fig. 3.22).

The east to west expansion of agriculture was not homogenous, the majority of farmland being concentrated in the northeastern states and the Ohio River Valley between 1850 and 1870 (Fig. 3.23). By 1900 agriculture had expanded into the eastern Dakotas, Nebraska, Kansas, and Texas and also was prominent in the Central Valley of California and eastern Oregon and Washington (Fig. 3.23). During the first half of the 20^{th} century agriculture continued to expand, particularly in the southeastern states, and became the dominant land use in the Upper Midwest and throughout the eastern two-thirds of the Great Plains (Fig. 3.23). From 1950 to the present agricultural land has become increasingly polarized in its distribution, concentrated in the Upper Midwest, the Great Plains, the Central Valley of California, eastern Oregon and Washington, along the Mississippi River, and the southern half of Florida (Fig. 3.23).

Although in the United States the changes in the relative importance of principal crops grown during the period after World War II was not as pronounced as in Europe, there were some important changes that occurred. Most pronounced were increases in the proportion of farm area used for soybean production and for wheat and corn during the 1970's (Fig. 3.24). Concurrently, the proportion of farm area in oats declined substantially, as did cotton during the 1950's and barley starting during the late 1980's (Fig. 3.24).

As the area in farmland became increasingly polarized, so did the production of some of the principal crops. In 1950, corn was a common crop in much of the eastern and central United States, but occupied no more than 60% of farmland within any state (Fig. 3.25). By 2005, however, the proportion of farmland in corn had declined considerably in the southeastern states, while increasing in importance in the Upper Midwest and Central Plains states (Fig. 3.25).

Comparatively, wheat showed little change in the proportion of farmland occupied or region of production. Between 1950 and 2005 the central plains and the northwestern states remained the principal areas for production of wheat and the proportion of farmland in wheat per state changed little, although there was some contraction in the east (Fig. 3.26). The largest change in row crop production in the United States following World War II was the increased importance of soybeans in both the proportion of farmland that it occupies and in the distribution of its cultivation. In 1950, soybeans made up no more than 13% of farmland and were concentrated in the Upper Midwest and in the Mississippi River Valley (Fig. 3.27). By 2005 the Upper Midwest remained the center of production with over 30% of farmland in soybeans (Fig 2.20). Moreover, the cultivation of soybeans spread throughout the southeastern states and the central plains states (Fig. 3.27).

The post-World War II era not only saw the polarization of agricultural production in the United States, but as in Europe, intensification in agriculture. Yields of principal food crops at least doubled in the post-World War II period (Fig. 3.28) and fertilizer use greatly increased (Fig. 3.29). Also, during the later half of the 20^{th} century farm mechanization greatly increased with correlated decreases in draft animals on farms (Fig. 3.30), large decreases in farm population and labor (Fig.

The importance of farm machines in substitution for labor and in facilitating farm amalgamation can be seen by looking at the trends in farm sizes in the eastern United States from 1850-2002. During 1850-1860 mean farm size was significantly larger in slave states (Alabama, Arkansas, Delaware, Florida, Georgia, Kentucky, Louisiana, Maryland, Mississippi, Missouri, North Carolina, South Carolina, Tennessee, Texas, Virginia) compared to free states (Connecticut, Illinois, Indiana, Maine, Massachusetts, Michigan, New Hampshire, New Jersey, New York, Pennsylvania, Ohio, Rhode Island, Vermont) due to the large labor force created by slavery (Fig. 3.34). Following the Civil War and the abolition of slavery, mean farm size in the two regions were similar and did not show any increase until 1950 when the use of farm machinery became more prevalent (Fig. 3.34).

During the post-World War II era, livestock husbandry also intensified with increases in cattle numbers (Fig. 3.35) and significant increases in milk yield, which facilitated a large decrease in the overall number of dairy cows (Fig. 3.36). An important component of the intensification of cattle management was the increased use of managed forage crops. Although the total area of land harvested for hay decreased during the second half of the 20^{th} century, yields increased so that total production increased over the period (Fig. 3.37). Integral to increasing production of hay was an increased proportion of hayed land being planted in alfalfa (Fig. 3.38), as well as more intensive management of hay crops (fertilization, earlier and more frequent cuttings) that increased and maintained yields of alfalfa and all hay (Fig. 3.37; Fig. 3.38).

The change in area of hay and the proportion of hay land managed in alfalfa was not homogeneous across the United States. The area harvested for hay decreased considerably in the eastern United States (with the exception of Florida, Kentucky and Tennessee), the Upper Midwest, the Northern Great Plains, and along the Pacific Coast between 1950 and 2005 (Fig. 3.39). Conversely, the area harvested for hay increased in the Central and Southern Great Plains, the Desert Southwest, and the Great Basin states during the same period (Fig. 3.39). Meanwhile the proportion of hay land in alfalfa greatly increased in most of the northeastern and western states, the Northern Great Plains and the Upper Midwest and decreased in the southern states (Fig. 3.40).

3.2.1 Regional agricultural land use in the United States

As discussed above, agriculture in the United States has undergone significant changes in importance as a land use over the last 150 years. Moreover, there were important changes in agricultural management, particularly in the crops produced following the Second World War. Here, by selecting several states from the northeastern, Upper Midwest, southeastern and Central Plains regions that are representative of regional changes in land use, I further illustrate the evolution of agricultural land use in the eastern half of the United States.

The Northeast

New York and Pennsylvania are indicative of the trends in land use that have occurred in the northeastern United States. The number of farms reached a maximum by 1900, which generally declined consistently until 1970, and since has mostly leveled off (Fig. 3.41a). In both states mean farm size decreased during the second half of the 19^{th} century and remained constant until 1940 when the mean farm size began to increase dramatically (Fig. 3.41b). During the late 19^{th} century the area in cropland peaked, began a steep decline from 1920 to 1940, and then declined slightly to the present (Fig. 3.41c).

The principal use of crops land has, and continues to be for oats, wheat corn, and hay. Into the 1950's these crop types remained in relatively constant proportions despite the decreasing area of crop land, however, during the second half of the 20^{th} century until the present oats and wheat decreased in importance while the proportion of are in corn has approximately doubled (Fig. 3.42). Hay remained the dominant crop type in both New York and Pennsylvania although New York witnessed a decline in the proportion of crop land in hay while in Pennsylvania it remained relatively constant (Fig. 3.42). Moreover, starting in the 1950's the proportion of hay in alfalfa for both states increased dramatically (Fig. 3.43).

The number of cattle in New York and Pennsylvania has showed relatively small changes over the last century and a half. There was a decreasing trend until 1930, where it then started to increase until approximately 1960 (Fig. 3.44). In New York there has been a general decrease in the number of cattle since 1960 until the present, however, in Pennsylvania from 1960 into the early 1980's the number of cattle varied little but then started to decline (Fig. 3.44). Comparatively, the number of dairy cows changed little in both states until to the mid-1950's, then steadily declined until 2005 (Fig. 3.45).

The Southeast

Representative of agricultural trends in the southeastern United States are Georgia and Alabama where farm numbers increased throughout the second half of the 19^{th} century and peaked in Alabama in 1920 and in Georgia between 1910 and 1940 (Fig. 3.46a). For both states the number of farms decreased steadily from their peaks until 1990, most precipitously up to 1970, and showed small increases in 2000 (Fig. 3.46a).

As discussed above, mean farm size in the southern states, including Alabama and Georgia decreased following 1860 until 1920-1930, where size then began to increase and leveled off by 1970 (Fig. 3.46b). In both states the area of cropland peaked in 1920, declined rapidly until 1950, and has continued to decline up to the present (Fig. 3.46c).

The types and importance of principal crops changed significantly from the first half of the 20$\hat{t}$h century compared to the second half. From 1909 - 1951 corn and cotton were the dominant crop types in Alabama and Georgia, although hay and peanuts showed some small gains (Fig. 3.47). No data for cotton are available from 1951 through 1973, but by 1980 the importance of cotton and corn in both states had declined greatly, whereas soybeans increased significantly with associated increases in wheat (Fig. 3.47). By the late 1990's until the present soybean has decreased, as well as wheat, whereas cotton, hay, and peanuts have increased (Fig. 3.47).

Unlike the northeastern states alfalfa did not gain in importance as a forage crop in the southeast due to the climatic unsuitability of the region. Instead, other crops such as Bermuda grass (*Cynodon dactylon*), clover (*Trifolium* spp.), Bahia grass (*Paspalum* *notatum*), and Fescue (*Festuca* spp.) gained prominence as cultivated forage in monocultural pastures and contributed to significant yields in hay (Fig. 3.48).

Cattle numbers in Alabama and Georgia remained relatively constant from 1867 until about 1940 and increased steadily until approximately 1970 (Fig. 3.49). From the 1970's until the present cattle numbers in both states have declined rapidly (Fig. 2.44). Comparatively, the number of dairy cows in the states was fairly constant in the first half of the 20^{th} century and by 1960 had began a decline that continues until the present, although in Georgia the rate of decline slowed starting in 1970 (Fig. 3.50).

The Upper Midwest

In the Upper Midwest farm numbers were at their maximum in 1900 (Fig. 3.51a), as was the area of cropland, illustrated by Illinois and Indiana (Fig. 3.51a). The number of farms has decline steadily since with the exception of a small increase from 1930 - 1940 (Fig. 3.51a), whereas since 1940 mean farm size has at the least doubled (Fig. 3.51b). In both states the area of cropland decreased from 1920 - 1940 and then increased until 1980, although not to previous levels, where after cropland area has remained relatively constant (Fig. 3.51c).

The most notable change in crop production was the large increase in the proportional area of cropland in soybeans, which has become increasingly dominant, particularly in Indiana (Fig. 3.52). This increase has been mostly a function of a decrease in the area of oats and to a lesser extent area in hay (Fig. 3.52). Moreover, in Indiana there has been a significant decrease in the importance of corn, where as in Illinois the proportional area of corn has remained constant (Fig. 3.52).

The decreasing importance of the area in hay was, as in the Northeastern states, accompanied by an increased proportion of the area harvested for hay being composed of alfalfa (Fig. 3.53). Concurrently the number of dairy cows declined sharply starting in the late 1940's (Fig. 3.55). Throughout the late 19^{th} century until the 1930's cattle numbers fluctuated but stayed relatively constant, however, from about 1940 until 1960 numbers increased and then declined so that present numbers are below those of the late 1800's (Fig. 3.54).

The Central Plains States

In the Central Plains the number of farms increased dramatically during the second half of the 19^{th} century until 1910 as settlers moved westward into the prairies as was seen in Kansas and Nebraska (Fig. 3.56a). Farm numbers then slowly declined up to 1940 and then decreased more rapidly through to the present (Fig. 3.56a). Mean farm size, however, has steadily increased and more than tripled since the late 1800's (Fig. 3.56b).

As the number of farms increased as Kansas and Nebraska were settled the area in crops also increased, peaking around 1910 - 1920 (Fig. 3.56c). The area in cropland decreased sharply from 1920 - 1940, increasing and approaching a similar area in 1970 as was observed in the early 1900's, and since has mostly remained constant (Fig. 3.56c).

As throughout the rest of the United States, the importance of oats decreased greatly in both Kansas and Nebraska while the middle of the 20^{th} century saw the introduction and subsequent increase of soybeans, particularly in Nebraska, as well as the increased importance of sorghum (Fig. 3.57). In Kansas, the proportion of land in corn, wheat, and hay remained relatively constant, where as in Nebraska hay and wheat decreased while corn showed relatively large variations through time (Fig. 3.57).

In both Kansas and Nebraska, the proportion of harvested hay in alfalfa increased until about 1960, however, in Kansas this trend reversed so that at present the proportion of hay in alfalfa is approximately half than in 1960 (Fig. 3.58). Meanwhile, in Nebraska since 1960 the proportion of hay in alfalfa has remain nearly constant or slightly increased, composing about 45% of all area harvested for hay (Fig. 3.58).

As in other regions the number of dairy cattle in the central plains has shown a sharp decline during the second half of the 20^{th} century (Fig. 3.60). The number of cattle, however, consistently increased during the late 1800's, leveling off from about 1900 - 1940, close to doubling from 1940 - 1970, and had fluctuated but remained relatively constant since 1970 (Fig 3.59).

3.3 Republic of South Africa

The large-scale European expansion of agriculture and grazing in South Africa, which began with Dutch settlement of the southwestern coastal region of the country (the Western Cape)in 1652, has been highly dependent upon the geomorphic and climatic characteristics of the country. Due to a combination of these factors <15% of the national area is suitable for row crops, concentrated along the southern and eastern coast and the north central interior (Fig. 3.61). Subsequently, the remaining areas of the nation are devoted to animal husbandry (Fig. 3.61).

The major determinants of land use were, and continue to be, the strong gradients in both precipitation and temperature that are found in South Africa. Mean annual precipitation decreases from the southeast coast towards the northwest of South Africa (Fig. 3.62), as does the mean maximum temperature, although the northeastern region of the country can be similar to the northwestern interior (Fig. 3.63).

The original settlement of the South Africa was confined to the coastal region of the southwest of the country until the mid-1800's, where agricultural production included row crops, fruit, and livestock. In 1835 descendents of the original Dutch settlers, the Boers, began a eastward and northward migration as a response to land shortages and disagreement with British management of the Cape region (Feinstein, 2005). By 1843, ~12,000 Boer settlers had moved into the interior and the eastern Cape. In 1838 the Boer republic of Natal was formed, which was annexed by the British in 1843, leading to the emigration from the territory to the interior by its Boer inhabitants. In 1860 separate Boer communities in the interior combined to form a republic; the Transvaal, which was incorporated into the Republic of South Africa in 1910 (Feinstein, 2005).

The complex political and cultural relationships that existed in South Africa among British, Boer, and native African inhabitants, combined with climatic (drought in particular) and edaphic limitations, were a factors in limiting agricultural development. Conflict between European colonists and native Africans during the 19^{th} century, as were tensions between Boers and the British, which culminated in the Anglo-Boer War of 1899-1902, were additional obstacles to agricultural development in the early history of the country (Feinstein, 2005).

The lack of a transportation infrastructures effectively made live animals the only feasible alternative for producers in the interior of the country and until the 20^{th} century the majority of agricultural production in South Africa stemmed from pastoral activities. This production, however, remained limited compared to other temperate regions of the world due to the high variability in rainfall and the low fertility of the majority of the land. Additionally, livestock disease greatly impacted and set back the livestock industry in South Africa. Particularly significant was an outbreak of rindepest in 1896 that spread throughout the country killing in excess of 90% of the cattle in the country (Feinstein, 2005).

Following the formation of the Republic of South Africa in 1910 until the 1950's agricultural production in the country increased at a very slow pace and led to the federal government initiating a series of subsidy programs for the agricultural sector during this period. Following the Second World War the agricultural sector began to make gains in production. This trend was particularly apparent during the 1960's and 1970's, as the large-scale introduction of irrigation removed the limitations due to rainfall (Feinstein, 2005).

By the 1960's the area of arable land in South Africa had stabilized and remained so until the mid-1980's when there was a 21% increase in cultivated land between 1985 and 1996 (Fig. 3.64). The dominant crop is corn followed by wheat, however, the dominance of cereals has diminished whereas sunflower and sugar cane, and to a lesser extent vegetable crops, have increased in importance (Fig. 3.65).

As in other regions the use of farm machinery increased dramatically in South Africa during the period following World War II with an associated decrease in the number of draft animals (Fig. ??). Also, as would be expected, the increased mechanization of farming is correlated with farm consolidation. Through the mid-1950's the number of farms increased while mean farm size decreased (Fig. 3.67). In the late-1950's, however, this trend reversed and the number of farms decreased while mean farm size increased before leveling off in the mid 1980's (Fig. 3.67).

The consumption of fertilizer in South Africa increased dramatically up to 1980, decreased during the 1980's, and remained generally consistent through the 1990's and into the 2000's (Fig. ??). The decrease in fertilizer use can be partly attributed to the removal of farm subsidies, however, continued increases in the concentration of nutrients in fertilizers (Fig. ??) has also meant that total nutrient application has been increasing since 1990 (Fig. ??).

Because of limited and high variation in precipitation, irrigation is an important component of row crop agriculture in South Africa and effectively increased the area suitable for row crops from the 10% of the national area with suitable rainfall to 15%. During 1961 to 2005 the area irrigated increased steadily, exhibiting an 85% growth (Fig. 3.69). Irrigation, along with increased mechanization, increased use of fertilizers, and more efficacious fertilizers have combined to make significant gains in crop yields as illustrated by both cereal and oil crops (Fig. ??).

Through the second half of the 20^{th} century stocking rates of livestock have remained relatively constant, although exhibiting oscillation over this period (Fig. 3.71). The composition of livestock has changed greatly, however, with the number of sheep declining by over 30% and cattle and goats increasing by 10% and 25% respectively between 1961 and 2005 (Fig. ??). The cultivation of forage crops for livestock has been less important than in other regions. For example, the cultivation of alfalfa for forage and silage increased since 1961, but has constituted only between 1-2% of arable area (FAOSTAT, 2006). Furthermore, the area of permanent pasture in South Africa, although decreasing has remained at least 5.5 times greater than the area of arable land, which highlights the dependence on natural pasture for animal husbandry (FAOSTAT, 2006).

3.4 Austral South America

The development of agriculture by Europeans in eastern austral South America (Argentina, Bolivia, Brazil, Paraguay, Uruguay) was limited until the second half of the 19^{th} century. Starting in the mid 16^{th} century the development of coastal Brazil for sugar production was initiated and the early 1600's saw the development of Jesuit Missions in the region, producing livestock and food and industrial crops (Cushner, 1983; Bradford, 1993). During the early colonial period the principal region of agricultural activity was in northwestern Argentina, which was developed towards supplying the silver mines of Potosí, Bolivia (Barsky and Gelman, 2001).

The introduction of cattle in the 16^{th} century initiated the process of the expansion of the cattle industry into the grasslands of the region, although limited until the mid-1800's because of Indian hostility and logistical limitations (Bradford, 1993; Barsky and Gelman, 2001; Klink and Moreira, 2002). During late 19^{th} and early 20^{th} century row crop agriculture expanded throughout the Rio del la Plata grassland system of eastern Argentina, Uruguay, and Southern Brazil (Soriano et al., 1991). Over the last several decades row crop agriculture has expanded rapidly in northwestern Argentina, eastern Bolivia, eastern Paraguay, and central Brazil largely due to deforestation and/or the conversion of lands used for extensive cattle raising (Bickel and Dros, 2003; Brown et al., 2005; Cardille and Foley, 2003; Dros, 2004; Hecht, 2005; Kaimowitz and Smith, 2001; Grau et al., 2005; Fernside, 2001; Steininger et al., 2001; Catterson and Fragano, 2004; Zak et al., 2004). Concurrent with the expansion of agricultural production throughout the region has been its intensification, illustrated by similar trends as witnessed in other global regions such as increases in farm machinery, fertilizer use, and yields. The principal drivers of agricultural expansion and intensification have been increasing globalization and development of foreign markets, particularly for soybeans bound for export (Dros, 2004; Grau et al., 2005).

The area dedicated to agriculture increased most in Brazil, Bolivia, and Paraguay, increasing in area 69%, 23%, and 71%, respectively over the period from 1961-2003 (FAOSTAT, 2006). Comparatively, Argentina and Uruguay showed little change over the same period (0.5% and -1.6%, respectively), reflecting the historical emphasis of agriculture in these countries (FAOSTAT, 2006). Furthermore, for all the countries except Uruguay, the proportion of agricultural area in arable land increased over the same period (Fig. 3.73).

With the expansion and intensification of agriculture in austral South America, farm mechanization became more prevalent so that the hectares of agricultural land per tractor decreased by 289% from 1961 - 2003 (Fig. **??**). Also, the use of fertilizers in relation to the area of arable land increased dramatically in all countries except Bolivia, particularly starting in the 1990's (Fig. 3.75). In Argentina, Bolivia, and Paraguay the use of irrigation increased by about 50-100% during 1961-1980 and then changed little afterwards, whereas in Brazil and Uruguay the area of irrigated land had increased about 5-7 times by 2003 (Fig. **??**). With increased farm mechanization, irrigation, and agrochemical use crops yields have increased as illustrated by cereal and oil crops (Fig. 3.77).

The adoption of irrigation in austral South America has been variable based upon the combination of ecological and economic factors making its use feasible. In Uruguay and Brazil the adoption of irrigation since 1961 has been extensive and has included a proportionally larger area of arable land than in other countries in the region or in comparison to South Africa, the U.S.A, or western Europe (Fig. **??**). Comparatively, within austral South America, the use of irrigation in Argentina has been generally consistent, keeping pace with increases in the area of arable land (Fig. **??**). In Bolivia, and particularly

Paraguay, the expansion of agriculture has occurred with little increase in area of irrigated land (Fig. ??) suggesting that the use of more land, rather than employing irrigation is more economically viable.

Driving the expansion and intensification of agriculture in austral South America has been the increasing prominence of soybean production for export, particularly to China (Fig. 3.79). The most rapid period of growth in the area of soybean harvested started in the 1990's (since 2000 in Uruguay), with Argentina, Bolivia, Brazil, Paraguay, and Uruguay witnessing huge increases of 1,829%, 5,207%, 993%, 1,151%, and 7,666% respectively from 1990 - 2005 (Fig. ??); the majority of production concentrated in east-central and northwestern Argentina, eastern Paraguay and Bolivia, and southeastern and central Brazil (Fig. 3.81).

In both Argentina and Brazil, the countries with the greatest proportion of agricultural land in row crops and the greatest level of agricultural intensification, there is also evidence of concurrent intensification in livestock production. As in other global regions the intensification of row crop production has been accompanied by increased cattle densities, decreased area in natural pasture, and increased area in planted pasture in both Argentina (see Chapter 5) and Brazil. The Brazilian states of Goias, Minas Gerais, Mato Grosso, Mato Grosso do Sul, São Paulo, and the Federal District, where the agricultural expansion and the development of soybean cultivation have been greatest, have witnessed large increases in cattle densities and in the area of cultivated pasture while the area of natural pasture has declined considerably (Fig. 3.82)

3.5 Summary

All the temperate regions included in this analysis exhibit most, if not all the factors associated with the intensification of agricultural production discussed in Chapter 2. The large increases in crop yields which began after World War II are associated with, and dependent upon increased mechanization and agrochemical use, higher yielding cultivars, and increases in the use of irrigation. Concurrent with, and stemming from, the intensification in farm management has also been increases in farm size due to land consolidation and increased areas of managed pastures.

Overall, the intensification of agricultural land management within western Europe, the U.S.A., South Africa, and austral South America has been similar in process, however, the extent that various aspects of intensification are developed and/or utilized can be variable based upon different ecological, economic, and political factors associated with regions and countries. For example, the relatively late intensification of agriculture in South Africa and austral South America was largely due to economic and political policy, whereas the adoption of practices, such as irrigation, is dependent upon ecological need and potential economic gains.

The influence macro-economic factors deriving from economic and political forces is particularly important in determining how agricultural lands are managed. National agricultural policy in the United Kingdom during and following World War II produced large-scale shifts in land use throughout the country, and similarly the Common Agricultural Policy of the European Economic Union, federal subsidies and the Conservation Reserve Program in the U.S.A., and agricultural subsidies paid by the South African government have had, and continue to have, a significant influence over land use in those areas. Similarly, international markets for soybean have driven the expansion of soybean production in austral South America, which in Argentina has been principally facilitated by neoliberal trade policies imposed by the International Monetary Fund (Hall et al., 2001). The importance of national economic policy on influencing land use is apparent in the next chapter where I quantify the changes in agricultural sector in Argentina, showing how both former and present economic policy have influenced the spatial and temporal pattern of agricultural land use and its recent intensification.

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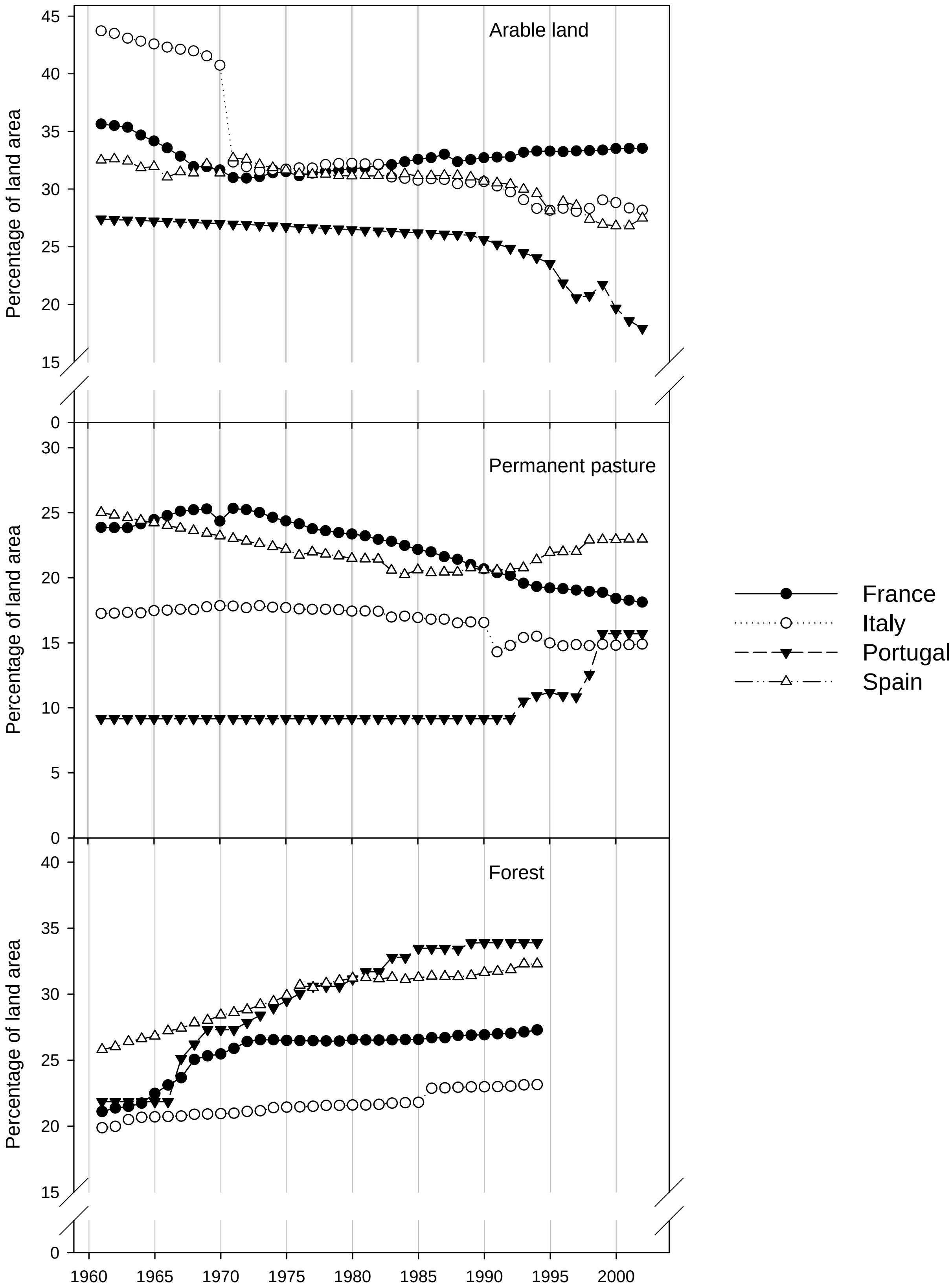


Figure 3.1: Percentage of national land area in a) arable lands (1961-2002), b) permanent pasture (1961-2002), and b) forest (1961-1995) in France, Italy, Portugal, and Spain(FAOSTAT, 2006).

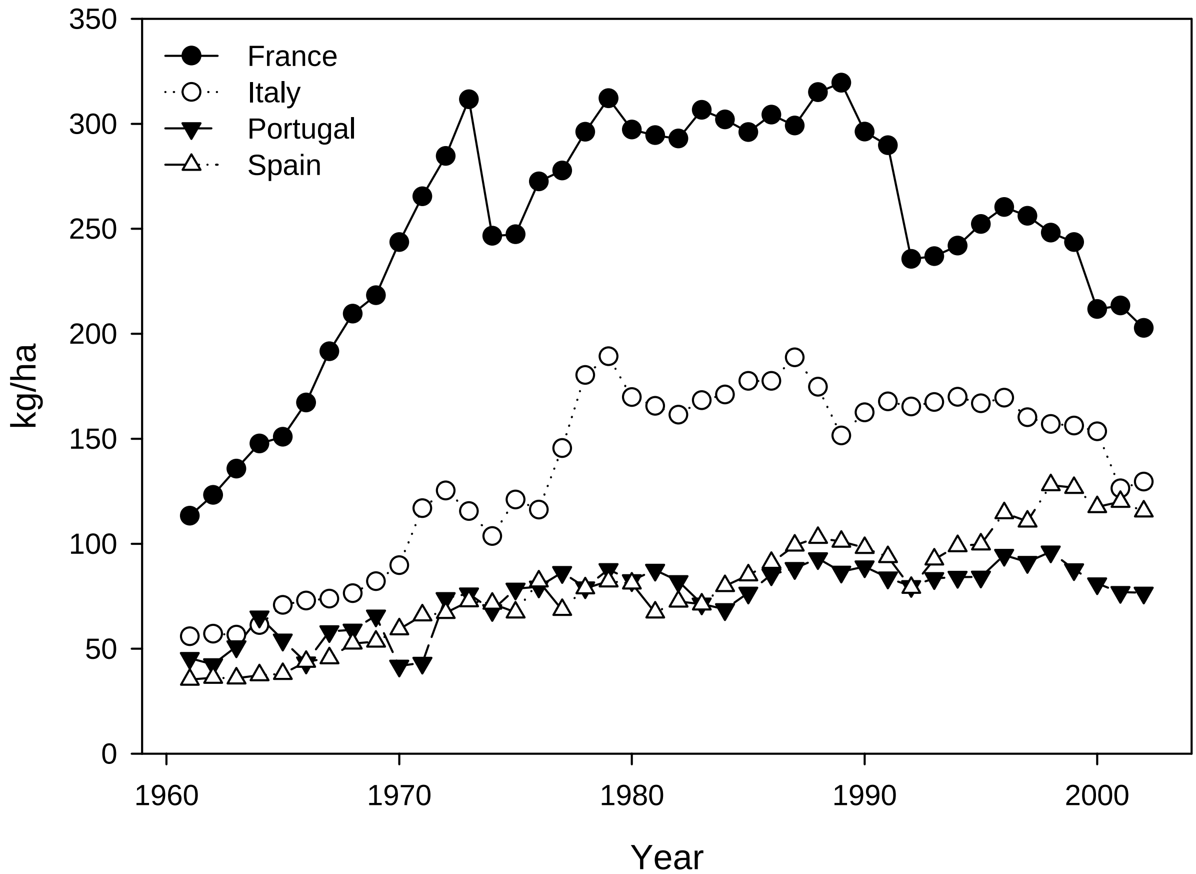


Figure 3.2: Fertilizer application per hectare of arable land during 1961 - 2002 for France, Italy, Portugal, and Spain (FAOSTAT, 2006).

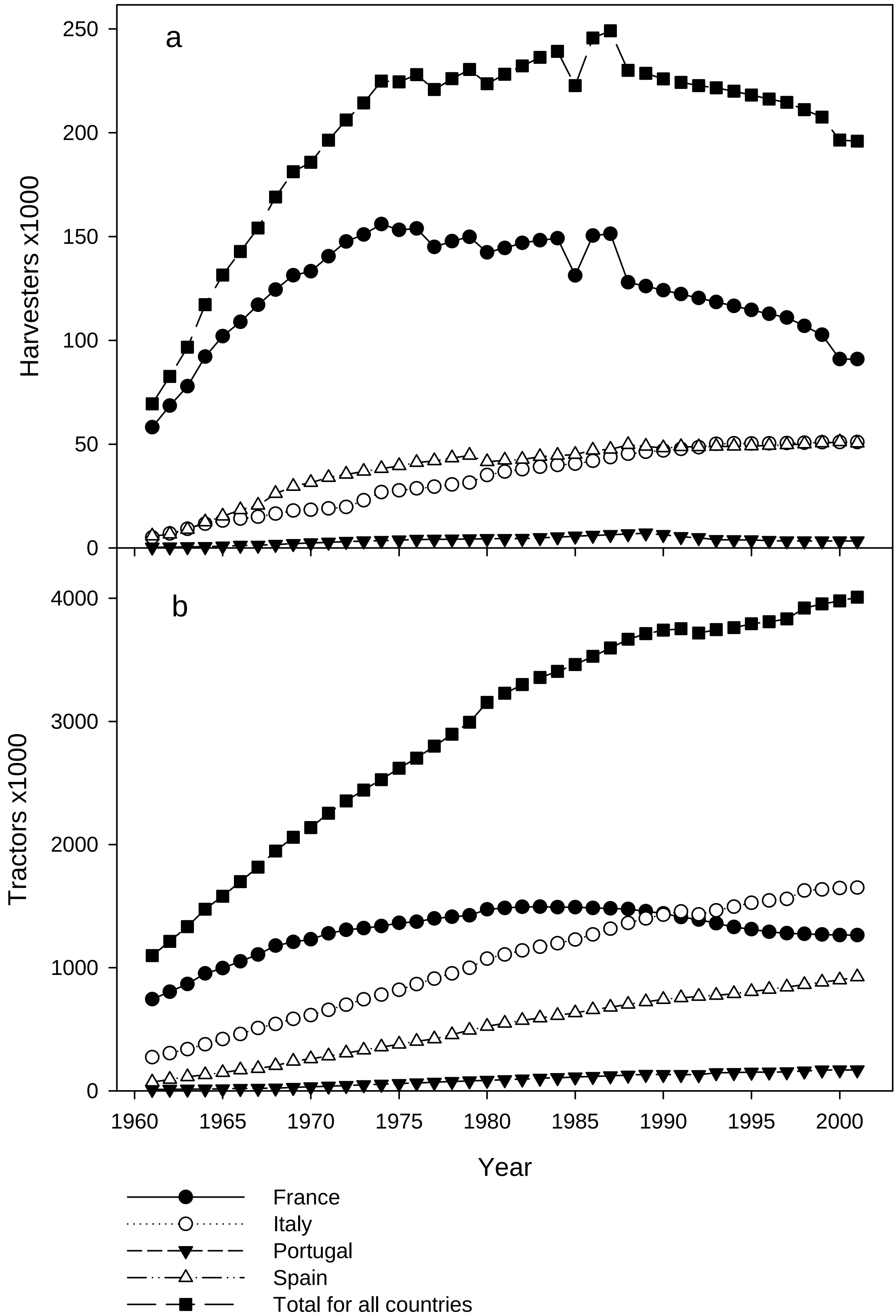


Figure 3.3: The number of harvesters (a) and tractors (b) during 1961 - 2002 in France, Italy, Portugal, Spain, and the totals for all countries (FAOSTAT, 2006).

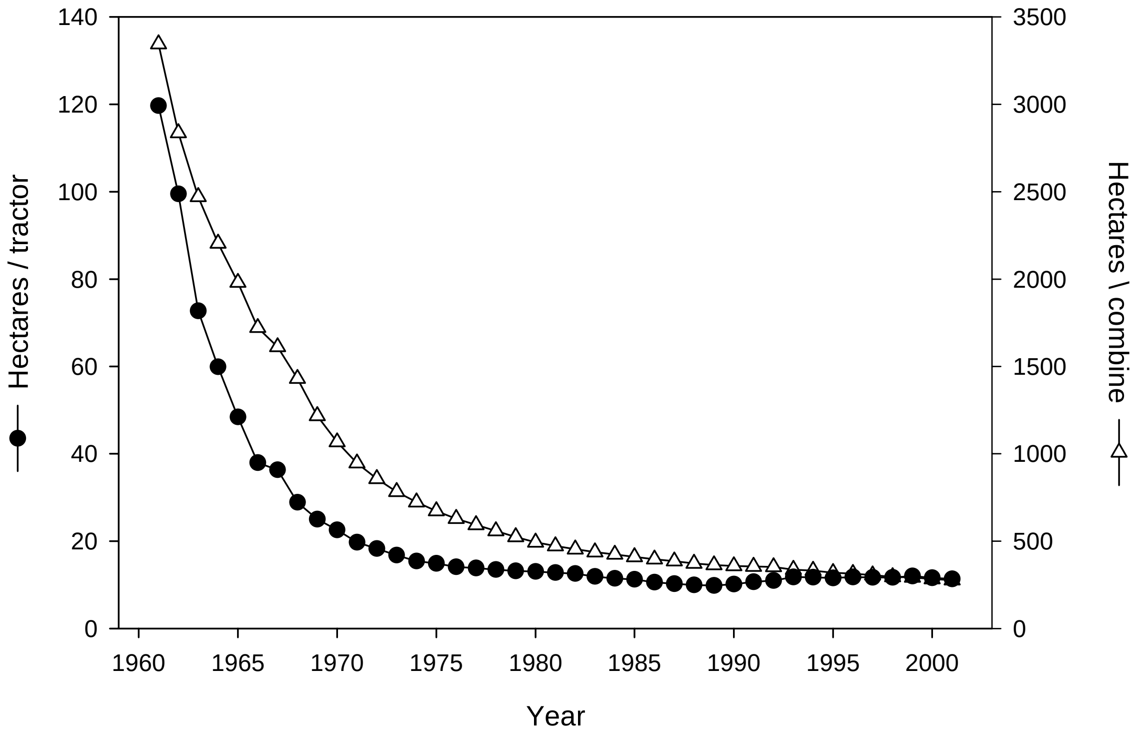


Figure 3.4: The number hectares of arable land per tractor and harvester during 1961 - 2002 for western continental Europe (France, Italy, Portugal, and Spain (FAOSTAT, 2006).

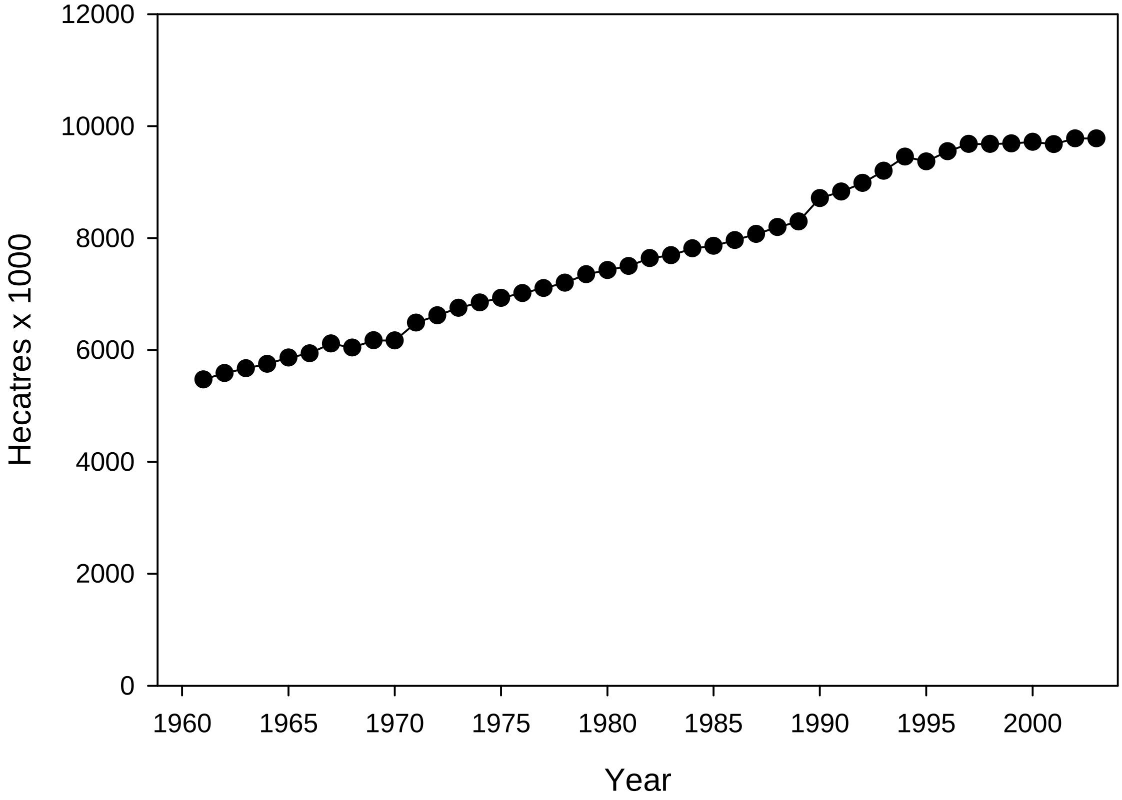


Figure 3.5: Total area of land under irrigation for France, Italy, Portugal, Spain during 1961 and 2001 (FAOSTAT, 2006).

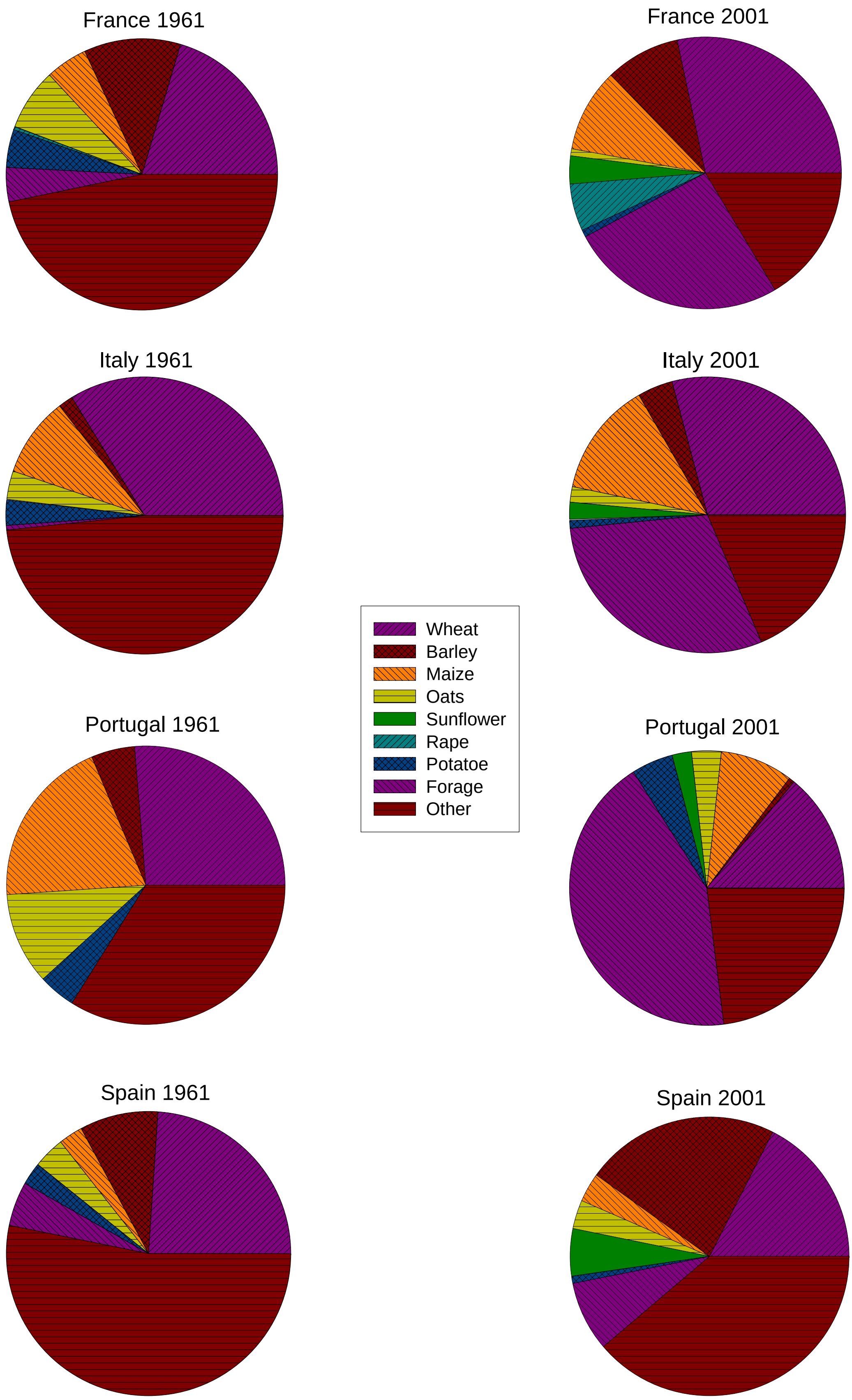


Figure 3.14: Proportional composition of crops in France, Italy, Portugal, Spain, and the totals for all countries during 1961 and 2001 (FAOSTAT, 2006).

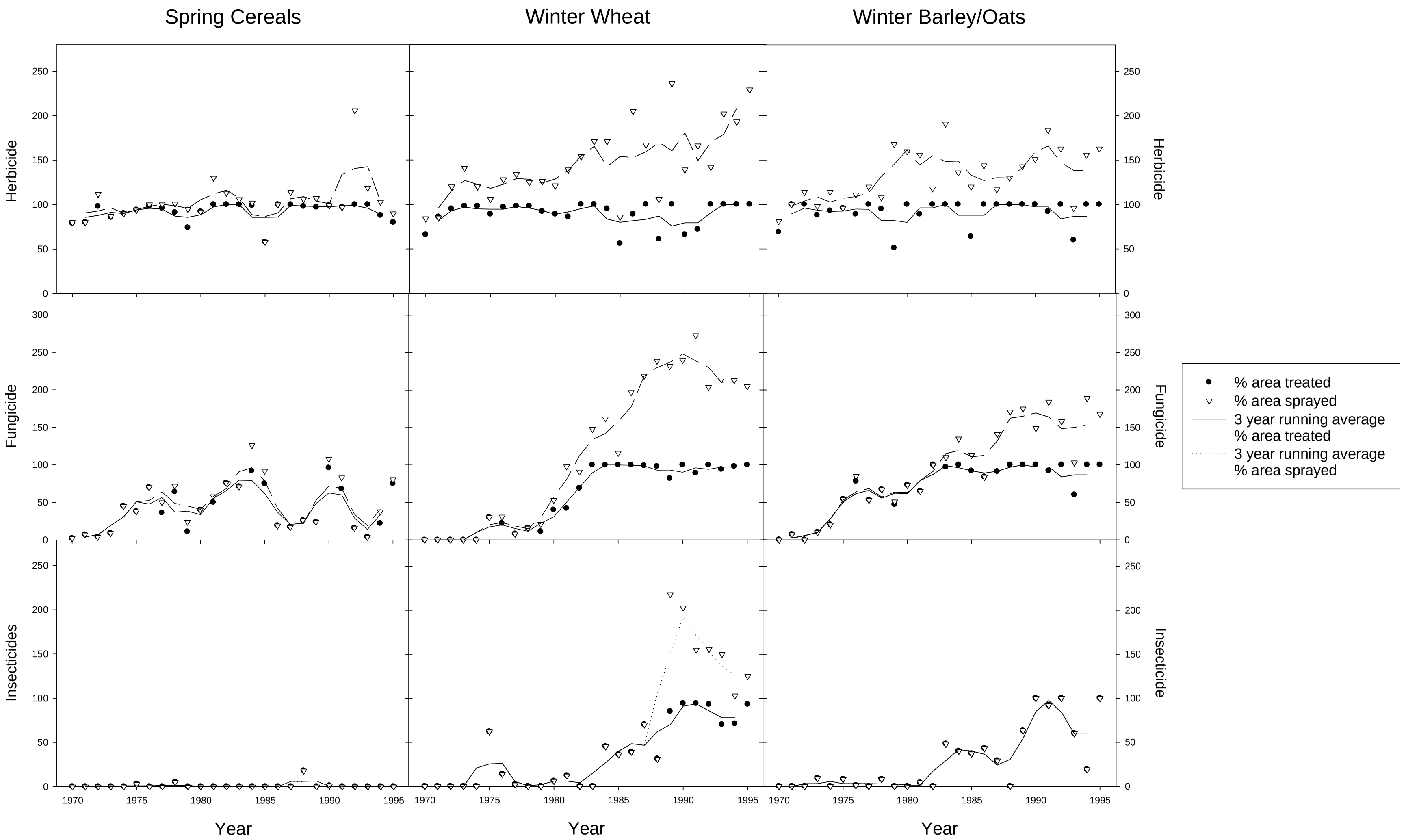


Figure 3.15: The area and frequency of herbicide, fungicide, and insecticide applications on spring wheat, winter wheat, and barley in Sussex, England (Ewald and Aebischer, 2000) during 1970-1995.

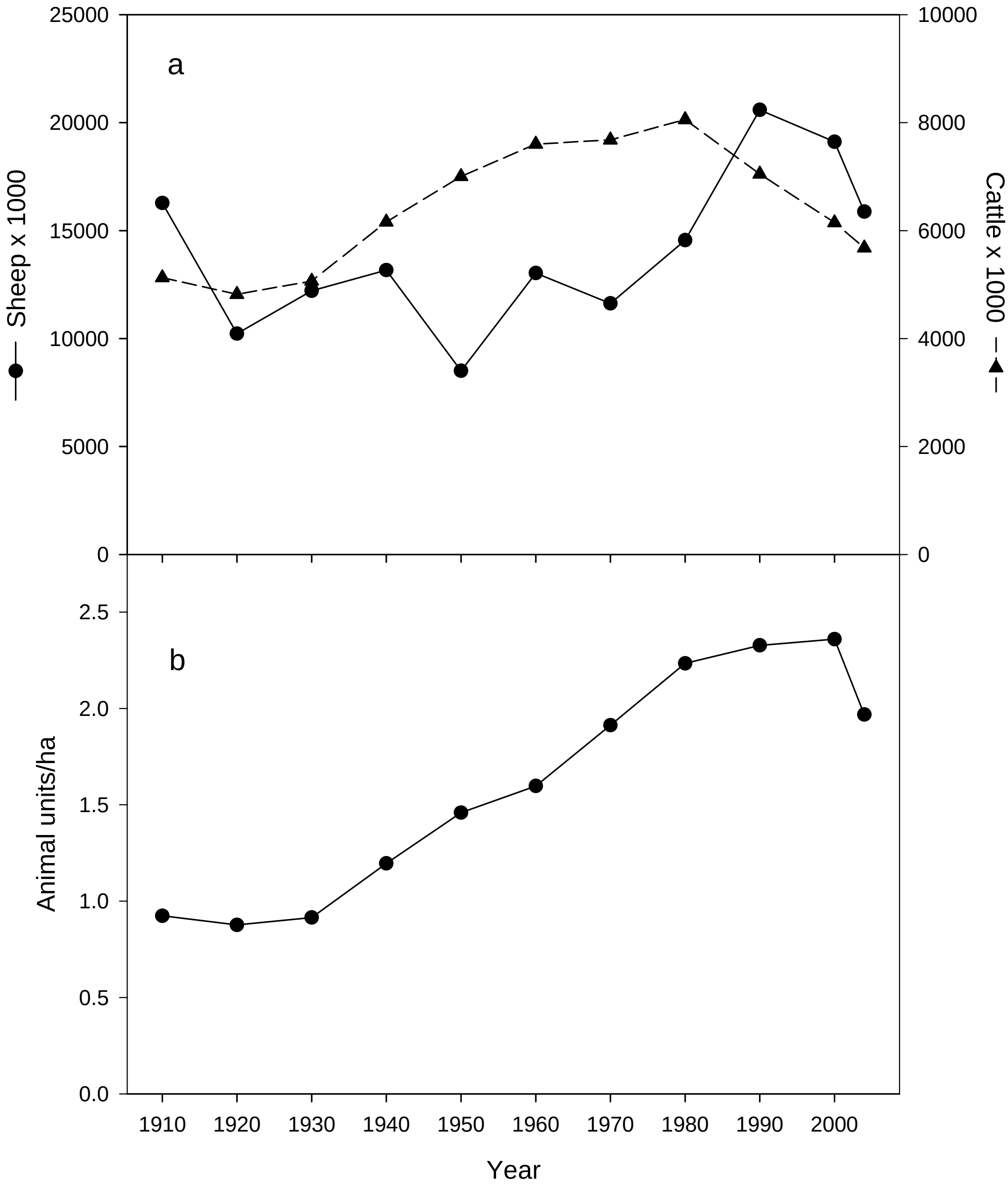


Figure 3.16: a) Sheep and cattle numbers and b) animal units per hectare of farmland for England 1910 - 2004 (DEFRA, 2006).

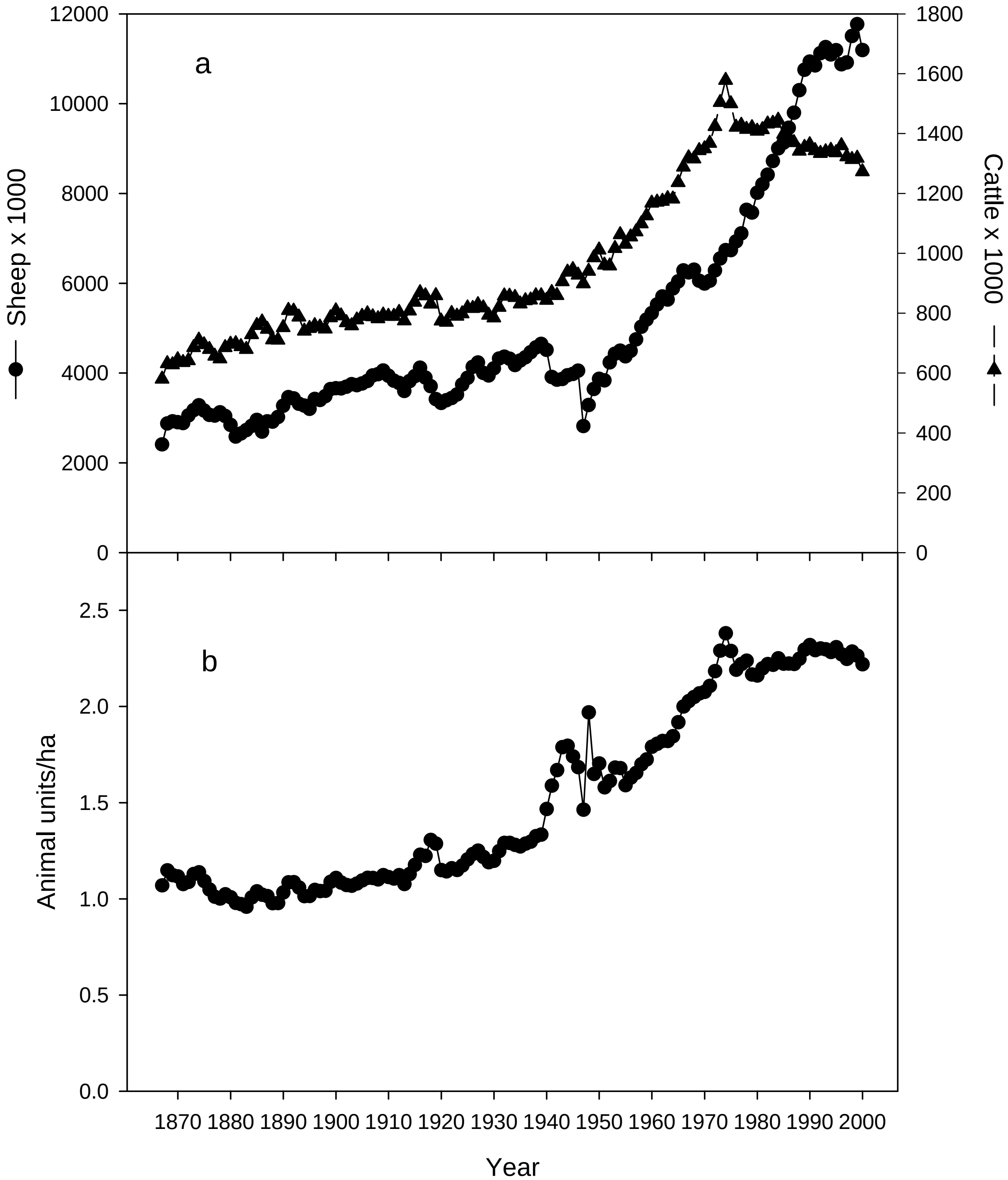


Figure 3.17: a) Sheep and cattle numbers and b) animal units per hectare of farmland for Wales 1867 - 2000 (DEFRA, 2006).

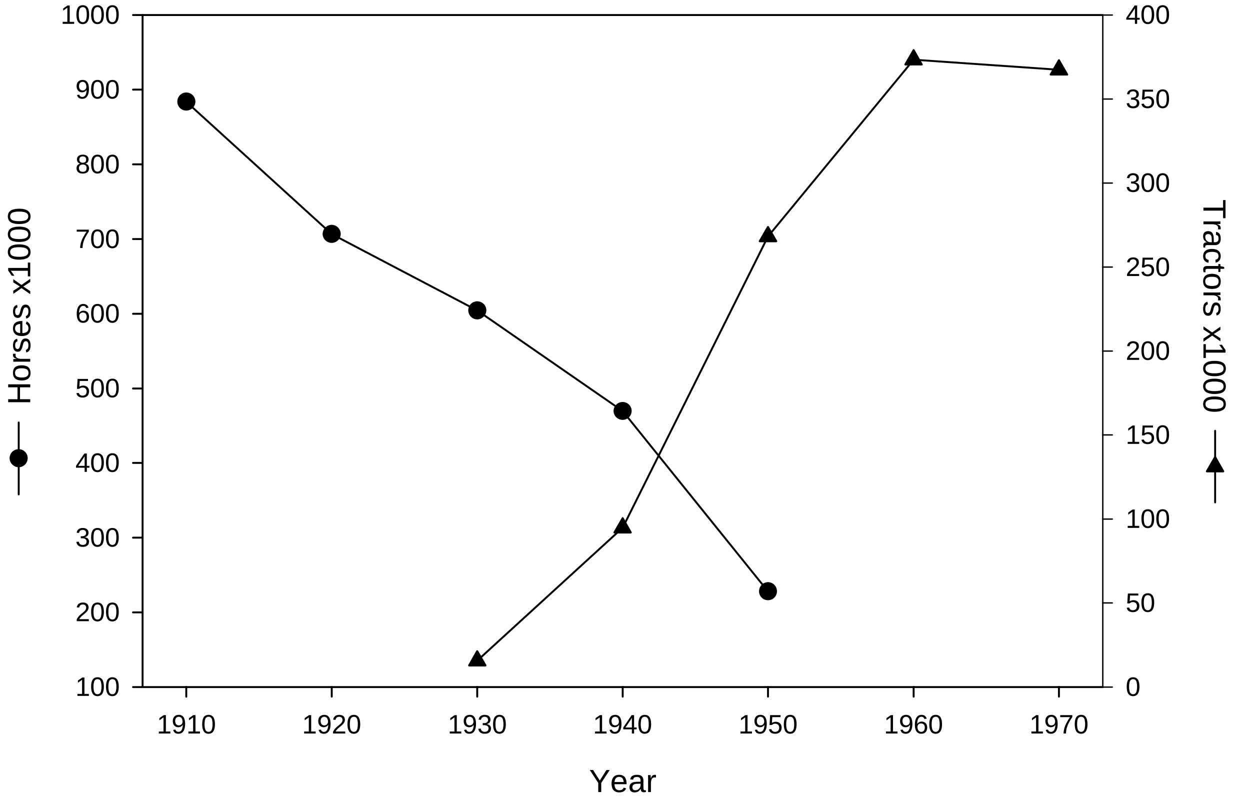


Figure 3.18: The number of draft horses and tractors on farms in England during 1910 - 1970 (DEFRA, 2006).

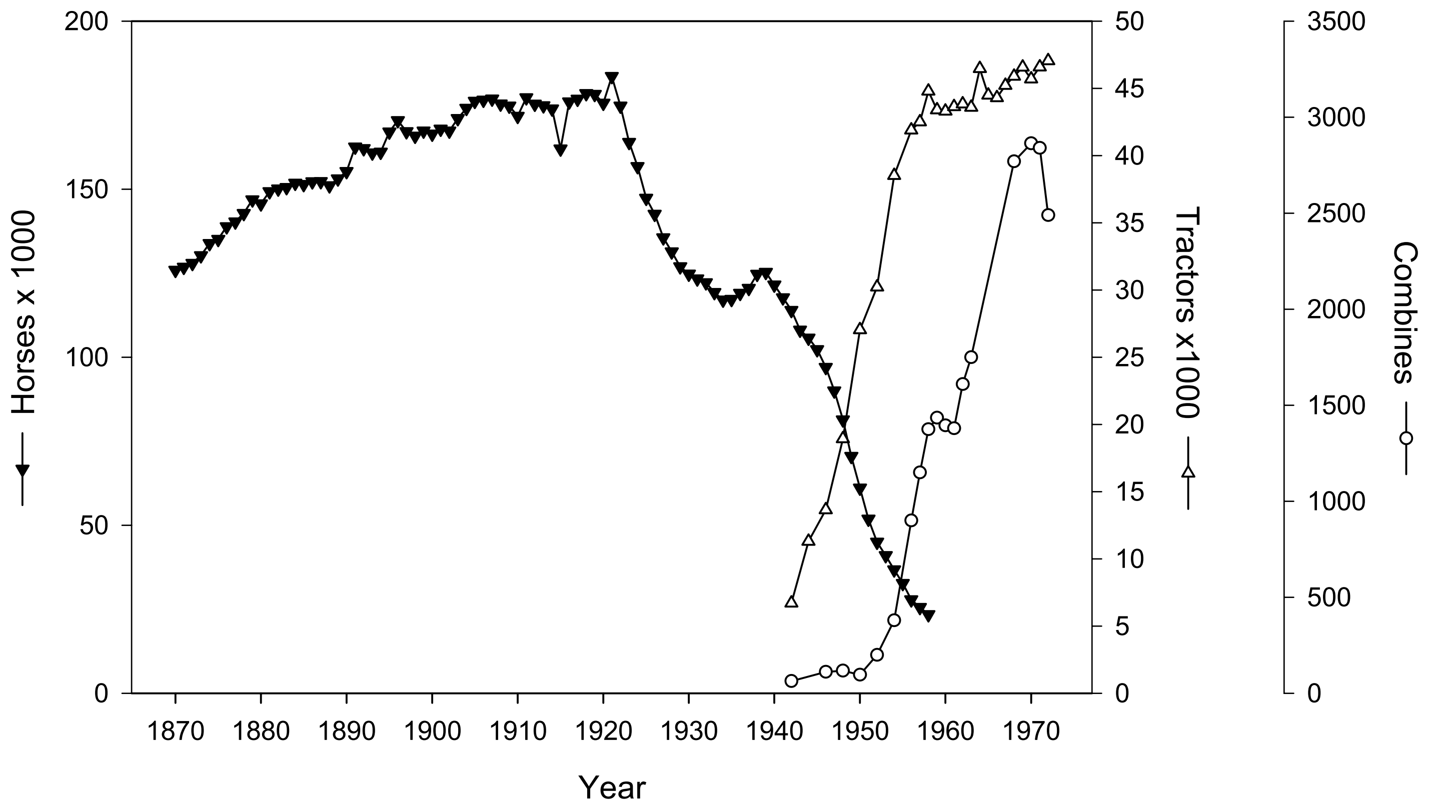


Figure 3.19: Number of draft horses, tractors, and combines on farms in Wales during 1867 - 1974 (DEFRA, 2006).

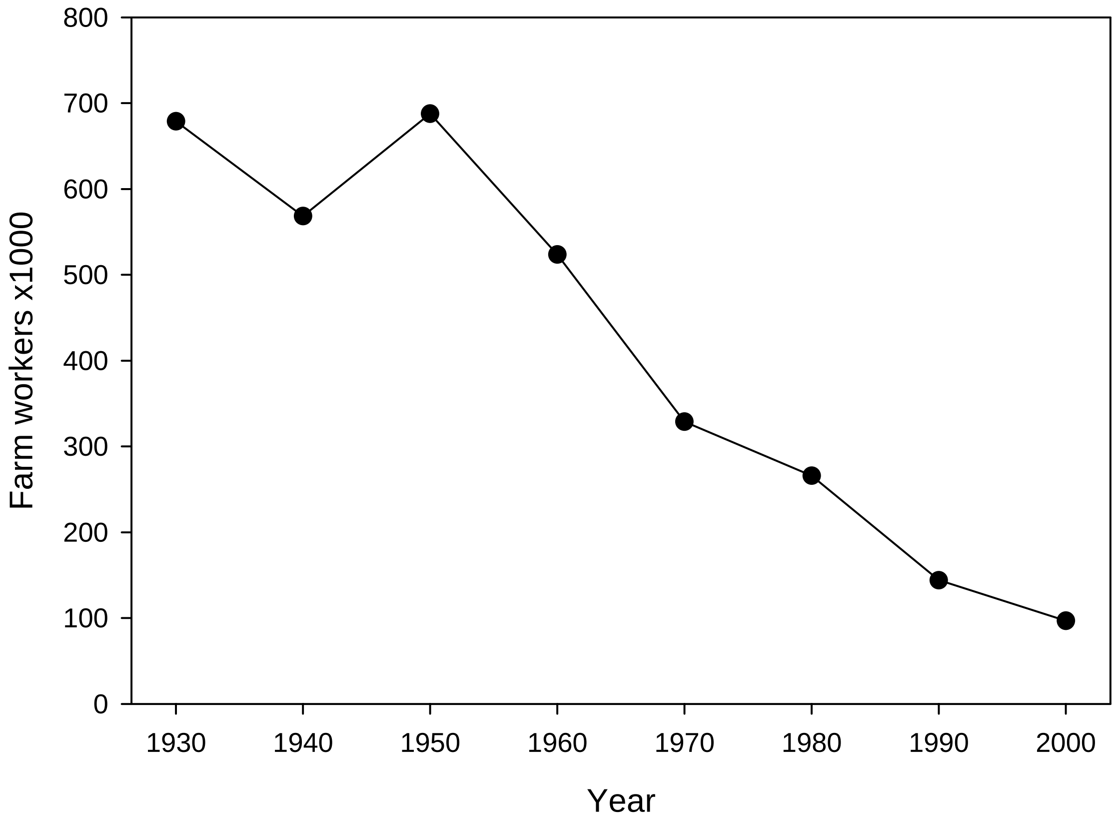


Figure 3.20: Number of farm workers on farms in England during 1930 - 2000 (DEFRA, 2006).

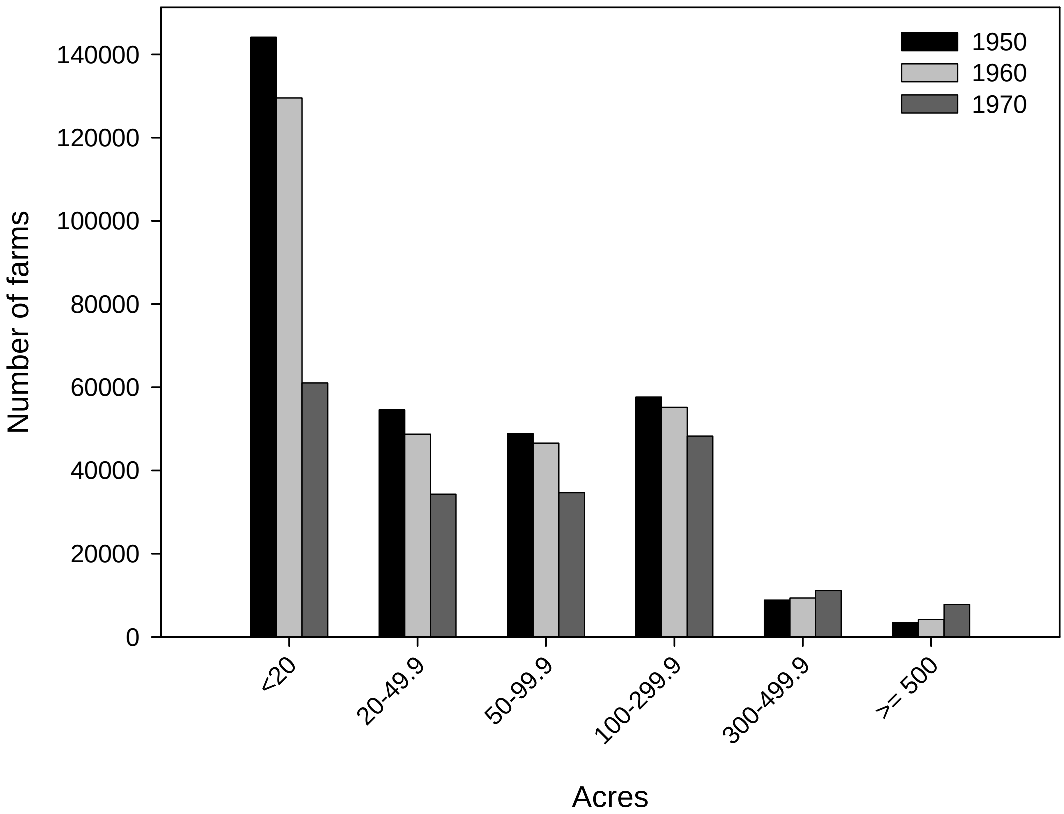


Figure 3.21: Number of farms in England by size class (acres, hectares) during 1950 - 1970 (DEFRA, 2006).

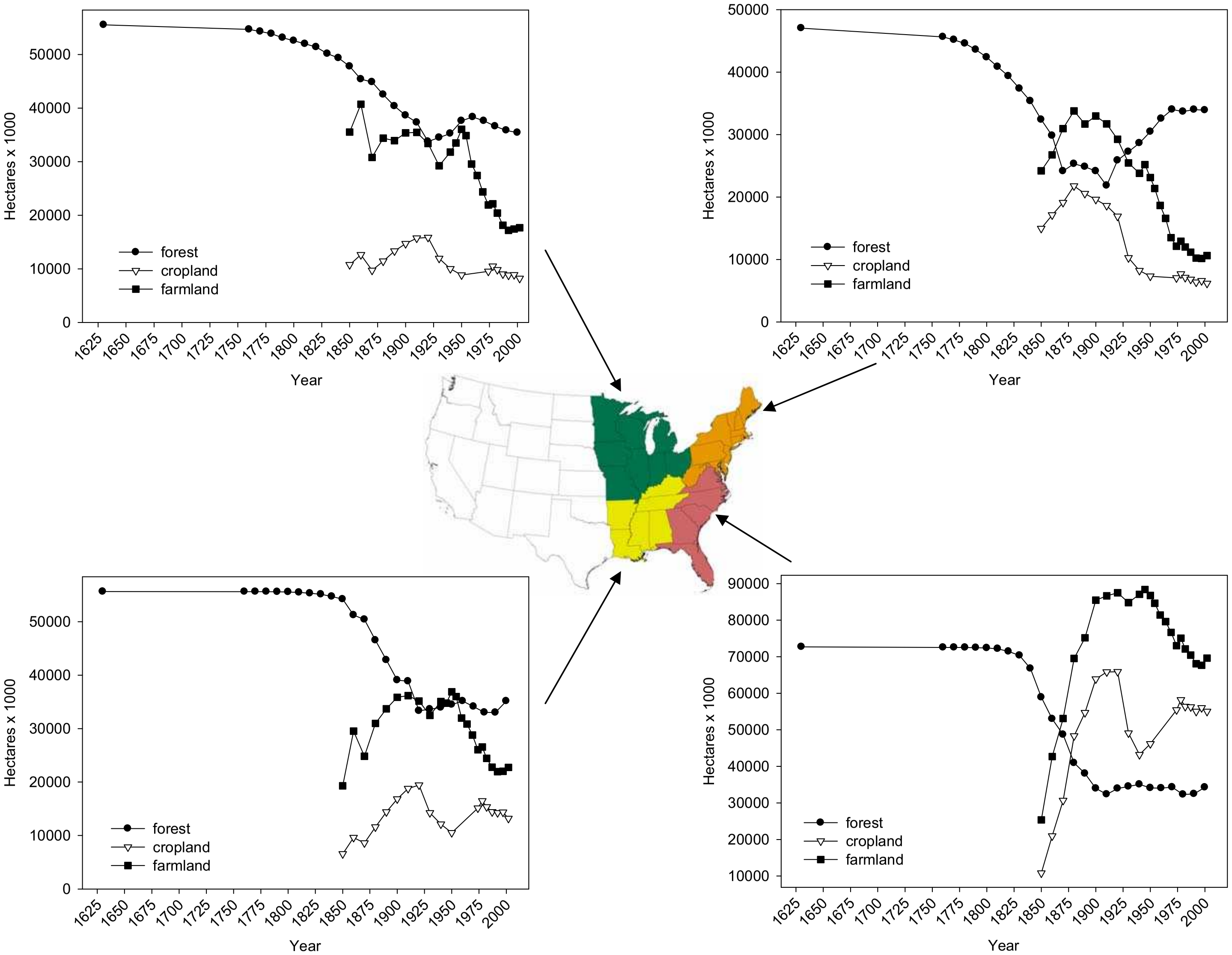


Figure 3.22: Regional (northeast, southeast, north central, and south central) land use changes in the United States east of the Mississippi River during 1620-2002 showing land are in forest, crops, and farms (U.S. Census Bureau, 1850-2000; USDA, 2006; Smith et al., 2004).

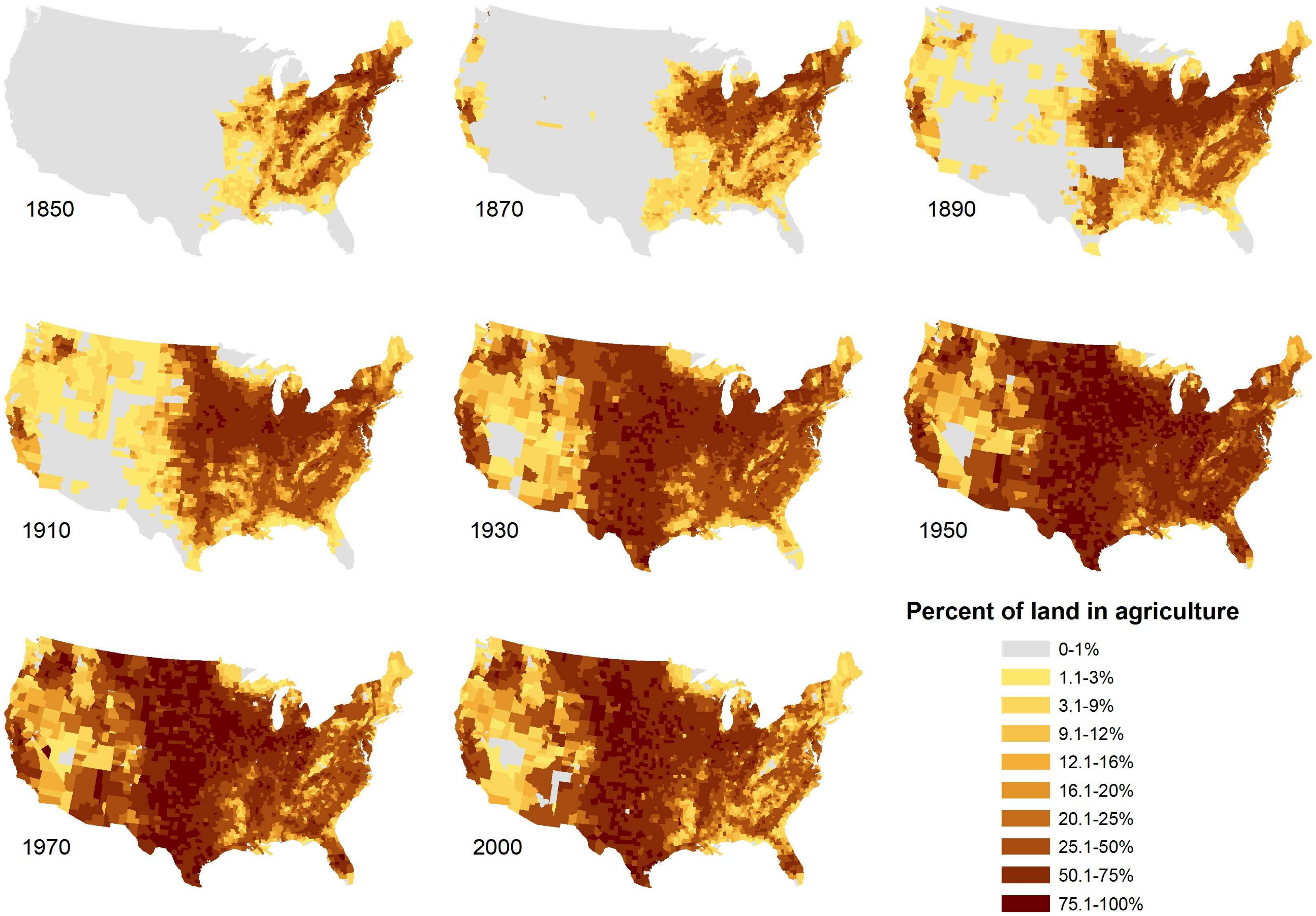


Figure 3.23: Percent of county area in farms for the United States during 1850 to 2000 (U.S. Census Bureau, 1850-2000; USDA, 2006).

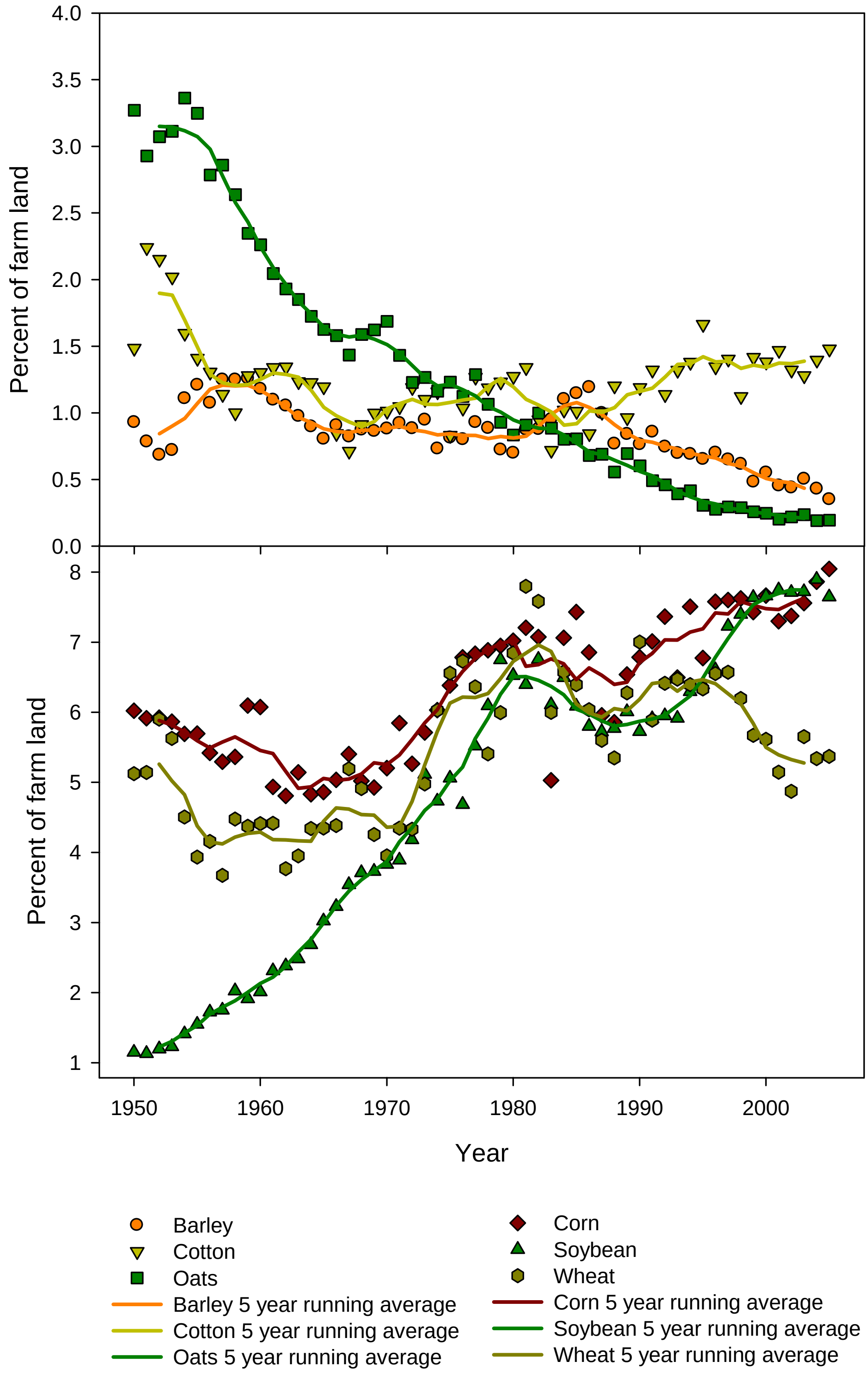


Figure 3.24: Proportion of United States farm land, and 5 year running average, in barley, cotton, oats, corn, soybean, and wheat during 1950 - 2000 (USDA, 2006).

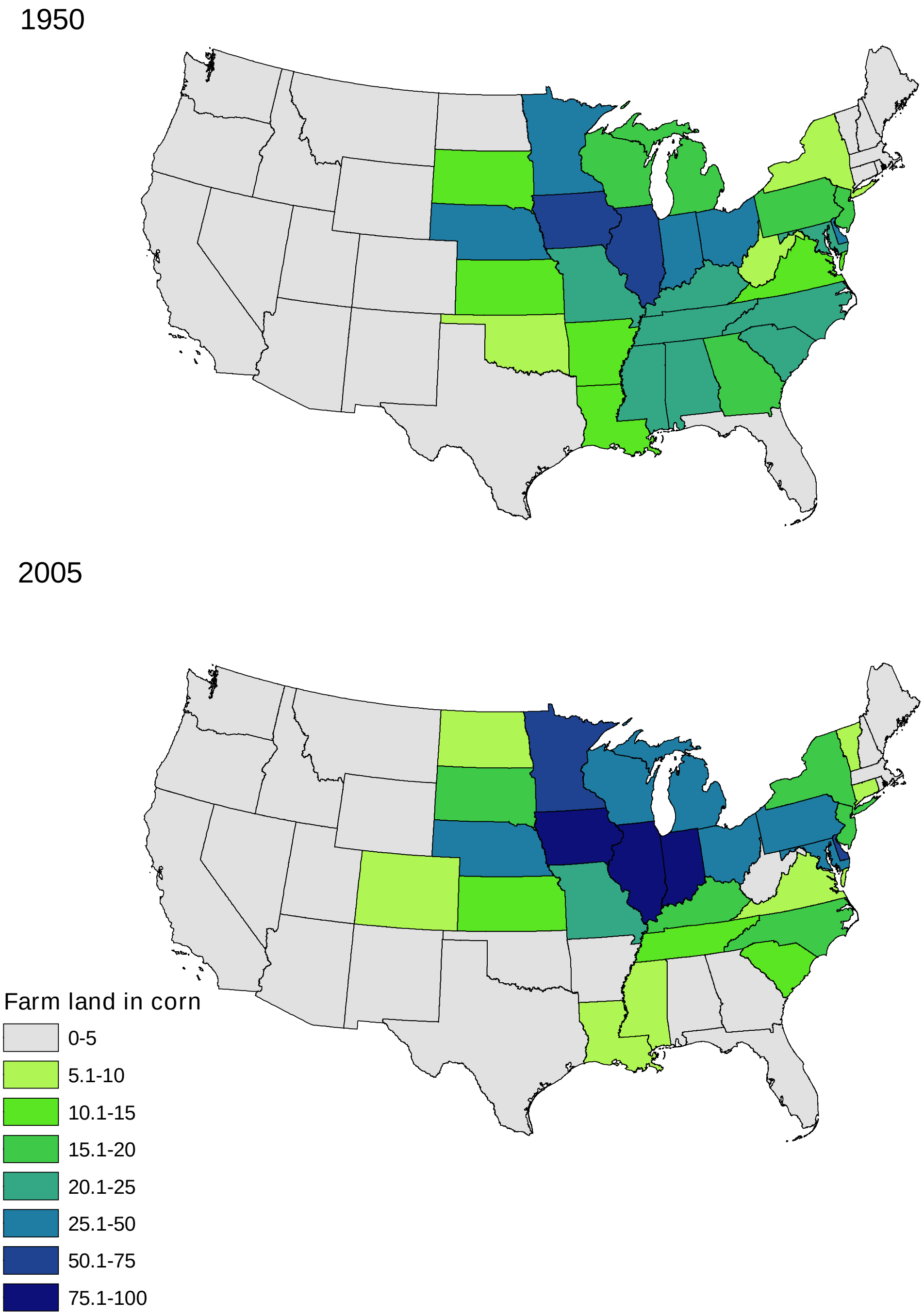


Figure 3.25: Percentage of farmland in corn by state for 1950 and 2005 (USDA, 2006)

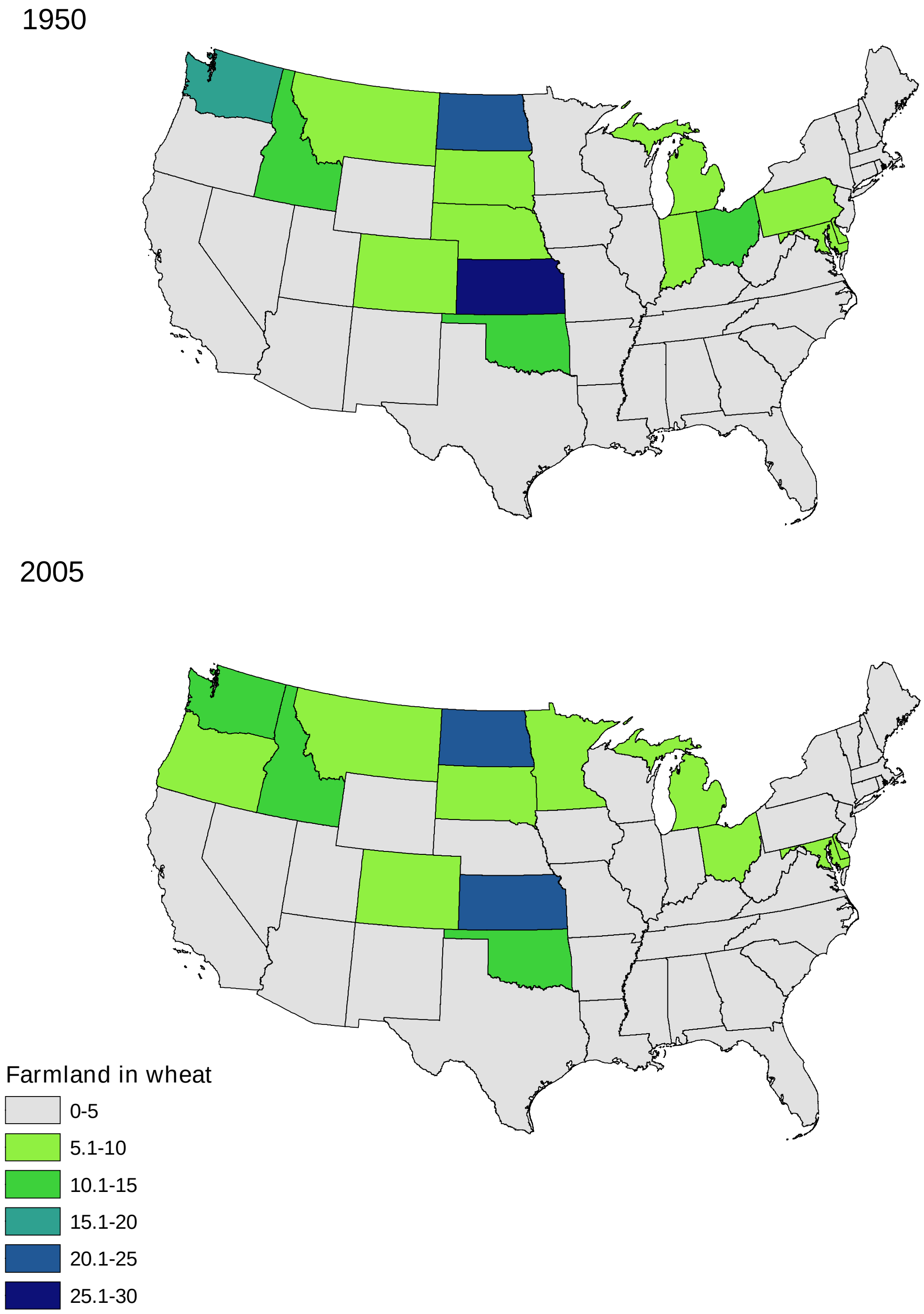
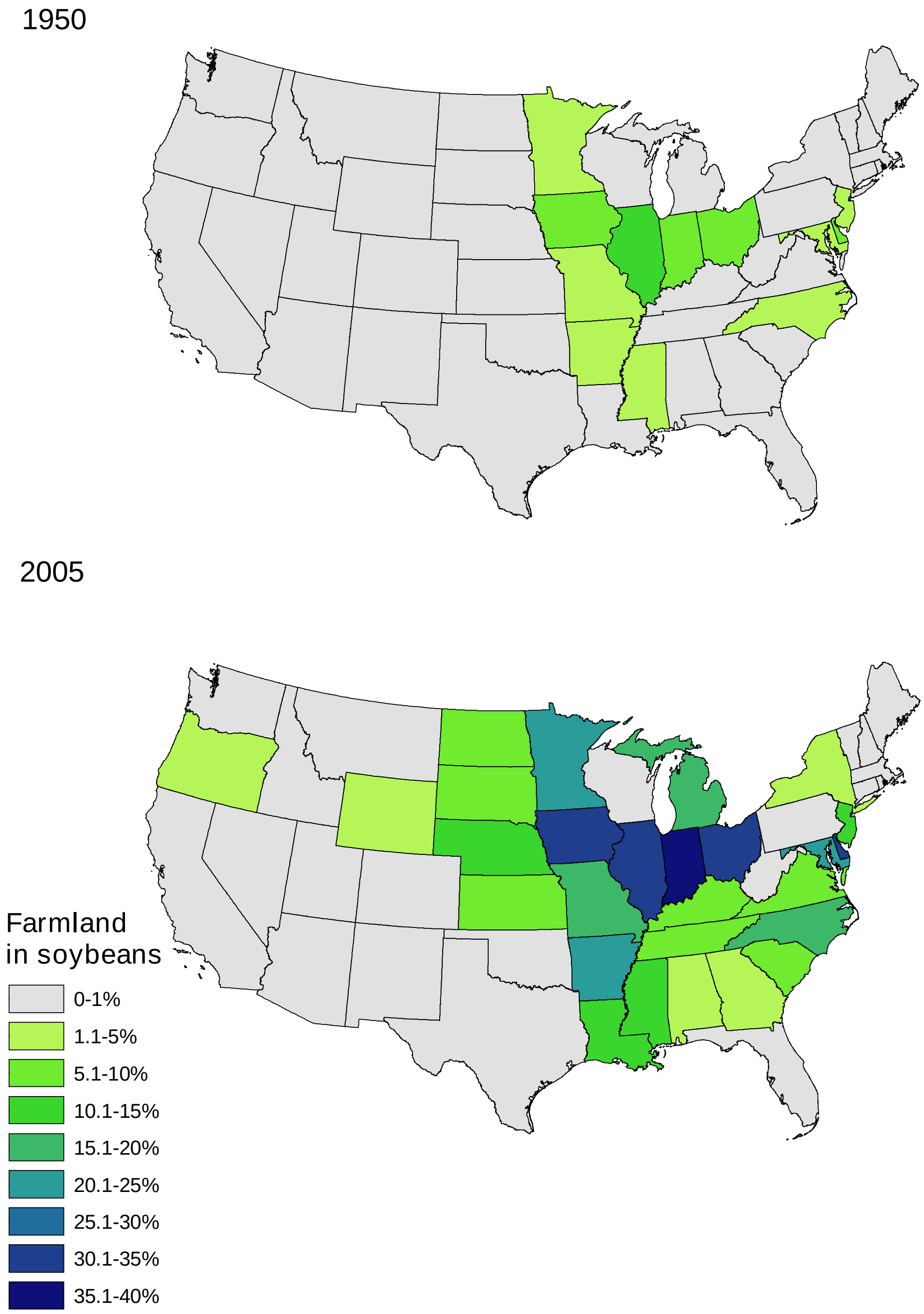


Figure 3.26: Percentage of farmland in wheat by state for 1950 and 2005 (USDA, 2006).



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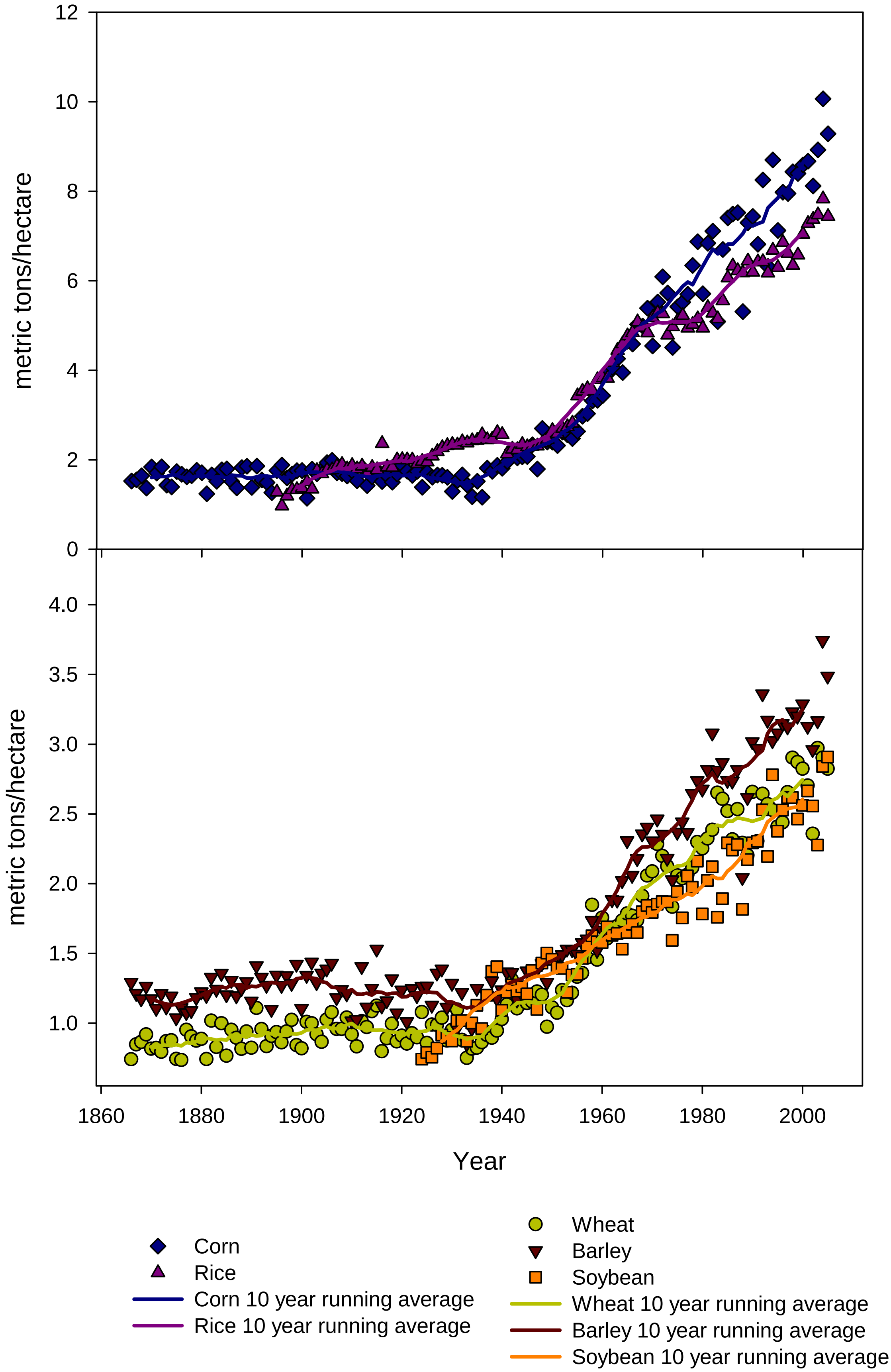


Figure 3.28: US yields of barley, corn, rice, soybean, and wheat during 1866-2005 (USDA, 2006).

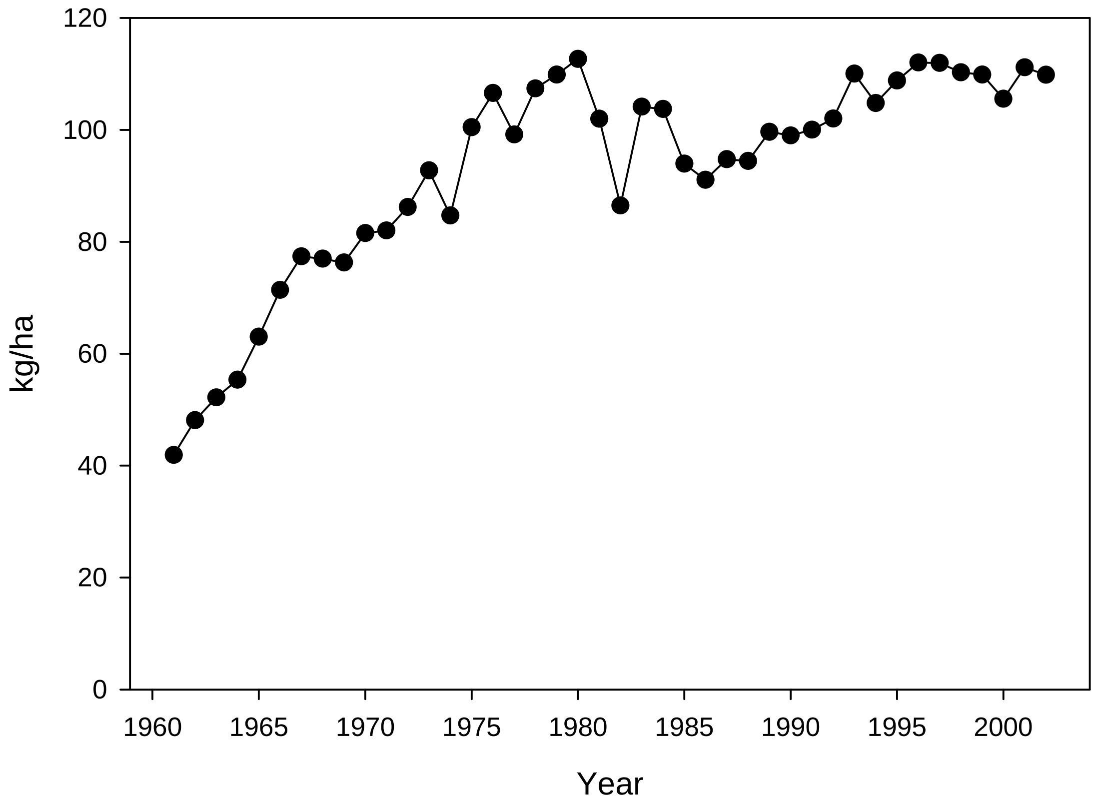


Figure 3.29: US fertilizer use per hectare of arable land during 1961-2002 (FAOSTAT, 2006).

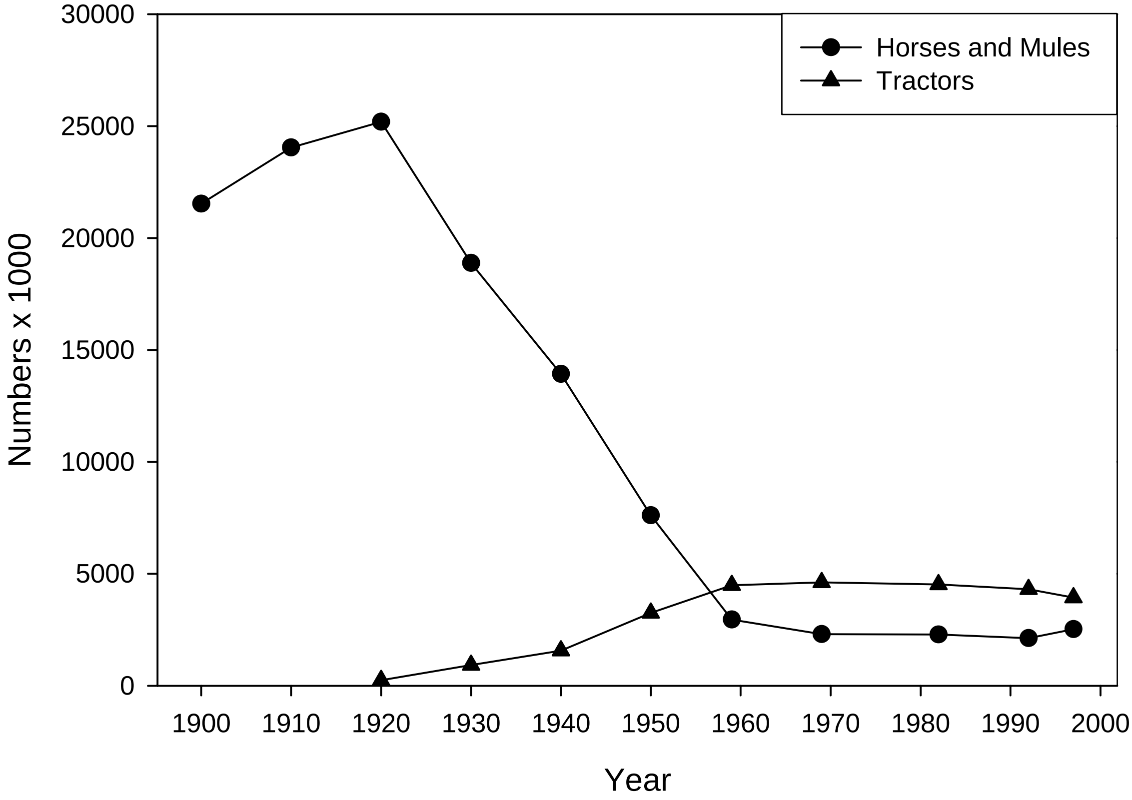


Figure 3.30: Numbers of horses and mules and numbers of tractors on US farms during 1910-1997 (USDA, 2006).

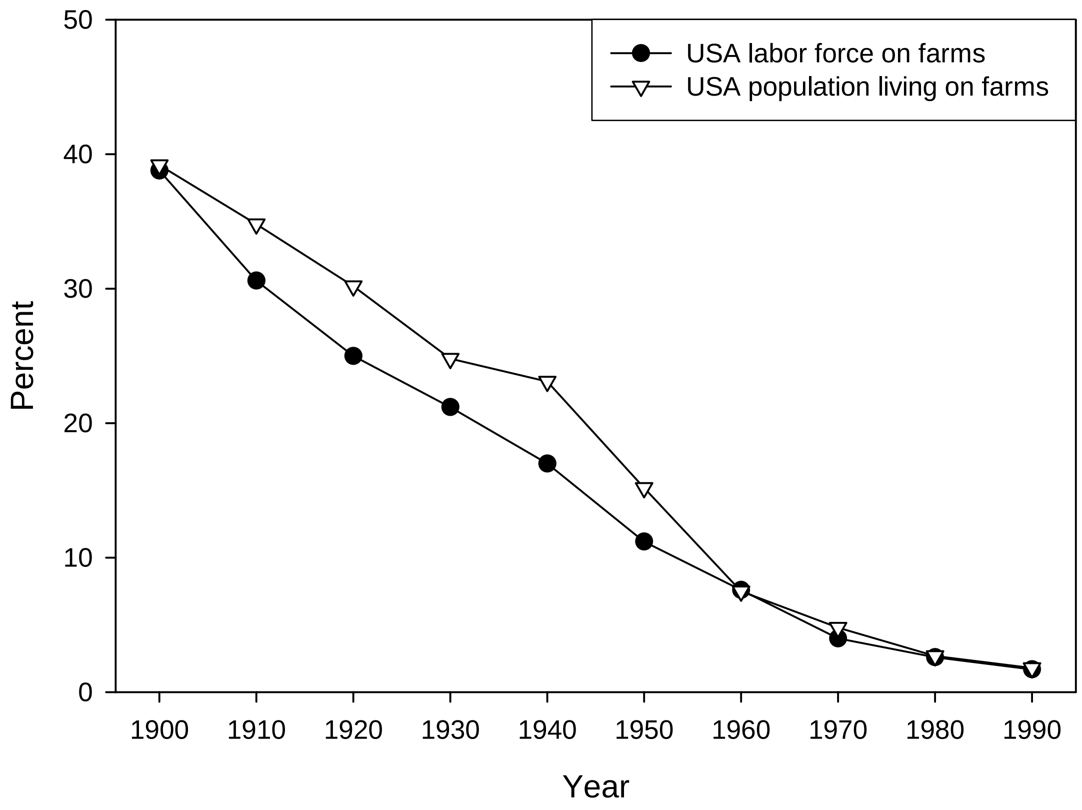


Figure 3.31: Labor force and population on US farms during 1900-1990 (USDA, 2006).

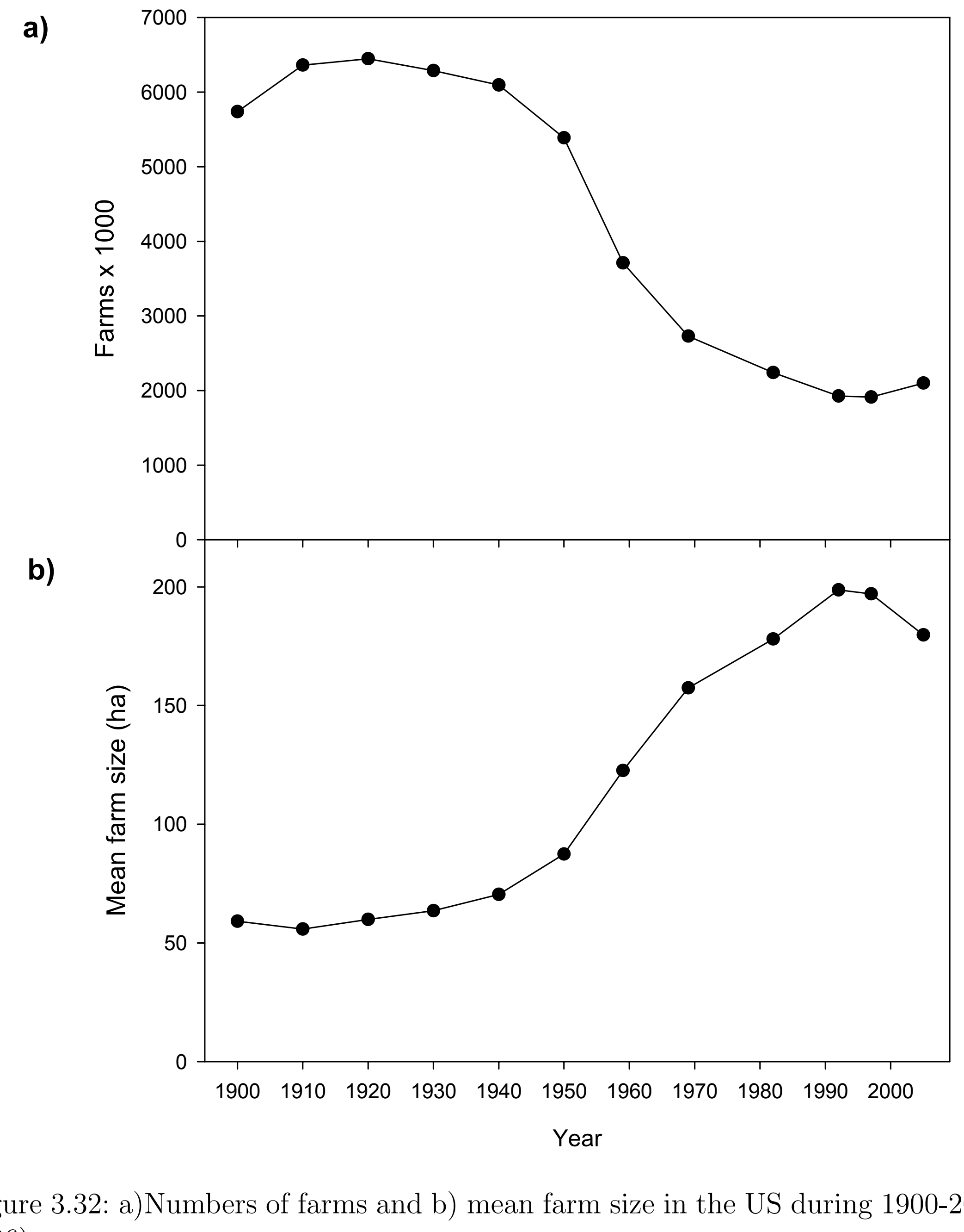


Figure 3.32: a)Numbers of farms and b) mean farm size in the US during 1900-2002 (USDA, 2006).

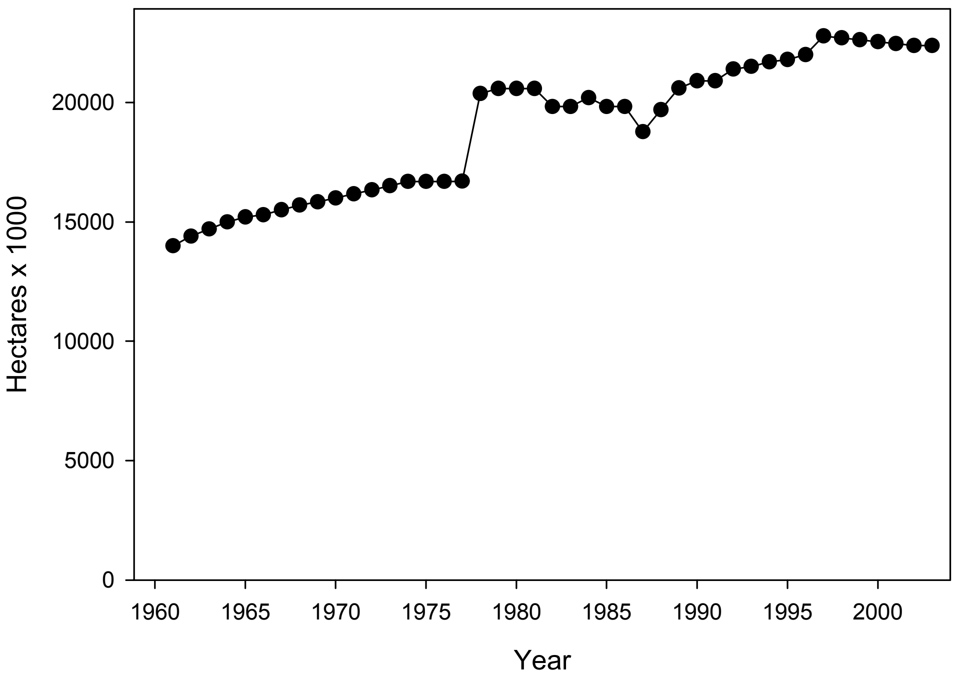


Figure 3.33: Area of irrigated land in the U.S.A. during 1961-2003 (FAOSTAT, 2006).

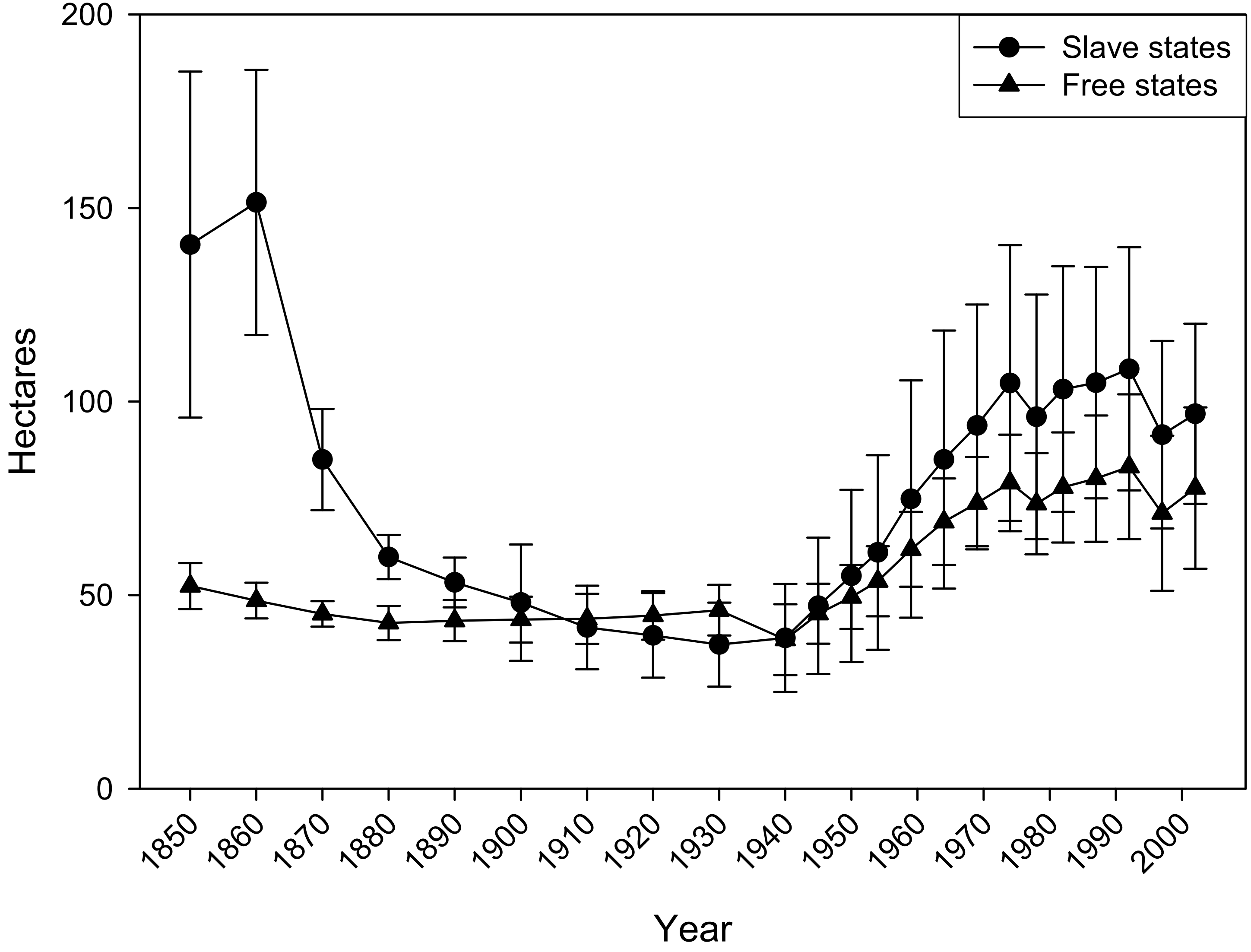


Figure 3.34: Mean farm size (±SE) for free states and slave states in the US during 1850-2002 (USDA, 2006).

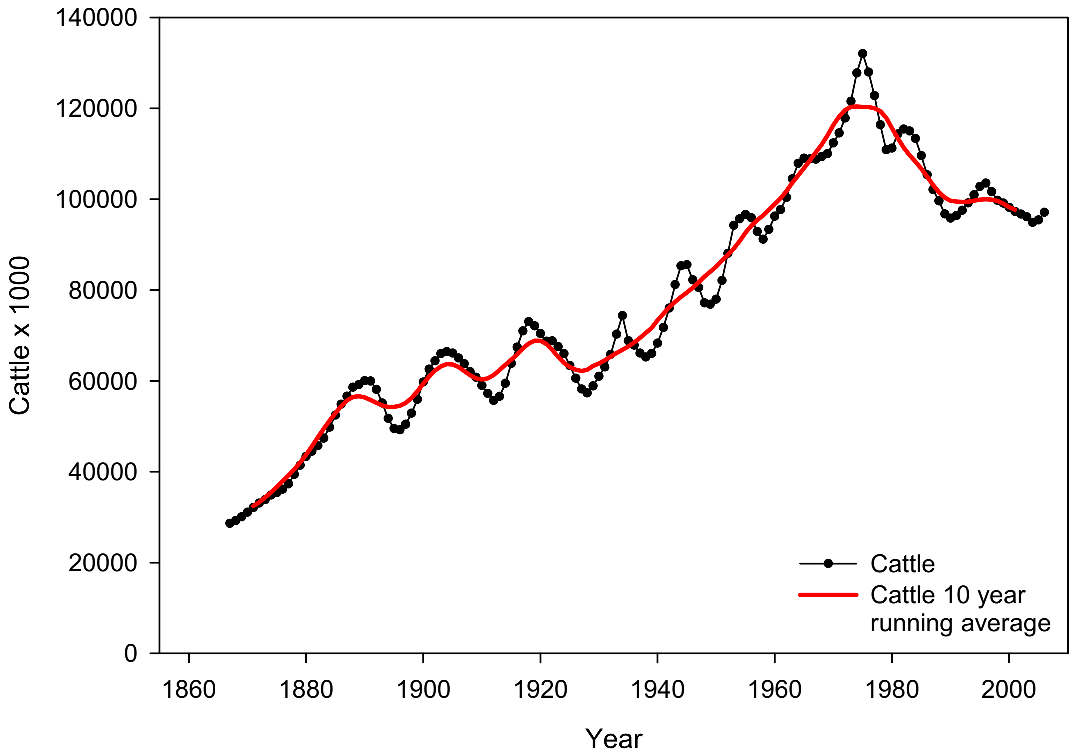


Figure 3.35: US cattle numbers and 10 year running average during 1866-2006 (USDA, 2006).

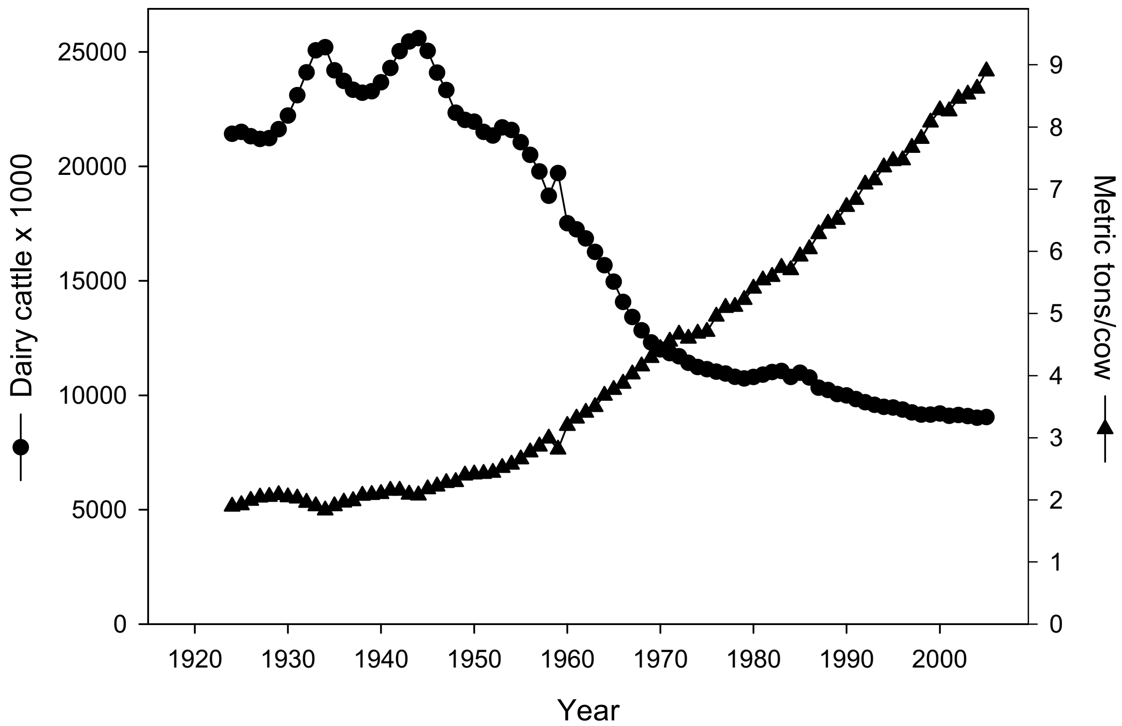


Figure 3.36: Number of dairy cows and milk yield in the US during 1924-2005 (USDA, 2006).

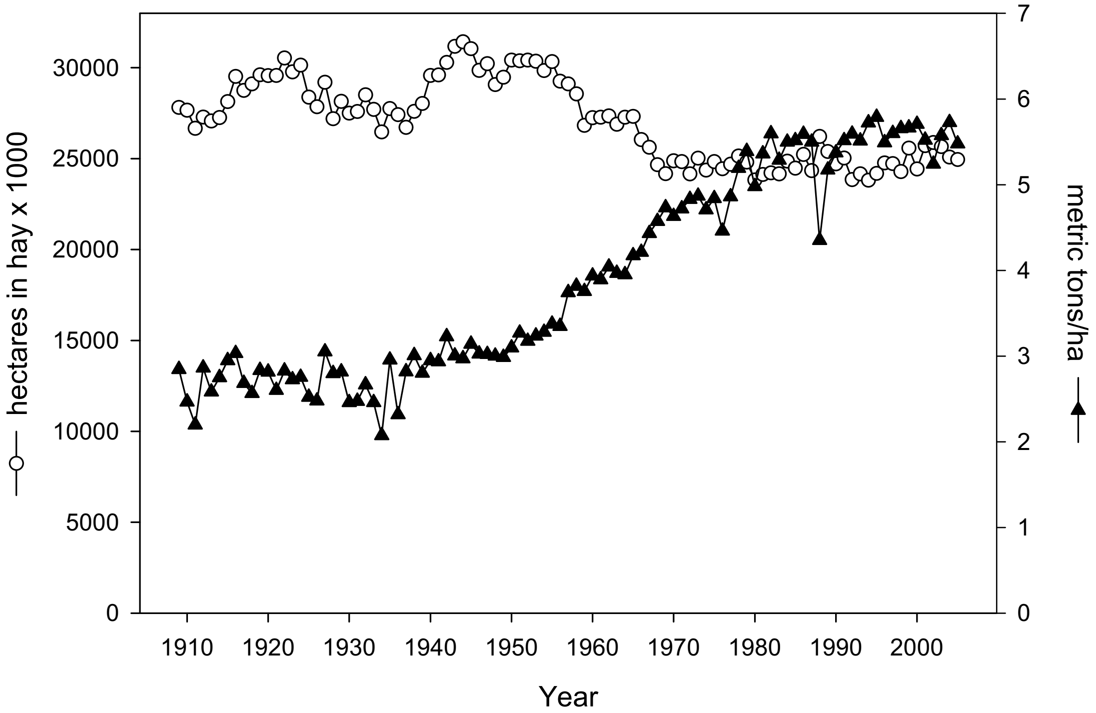


Figure 3.37: US hay area and yields during 1909-2005 (USDA, 2006).

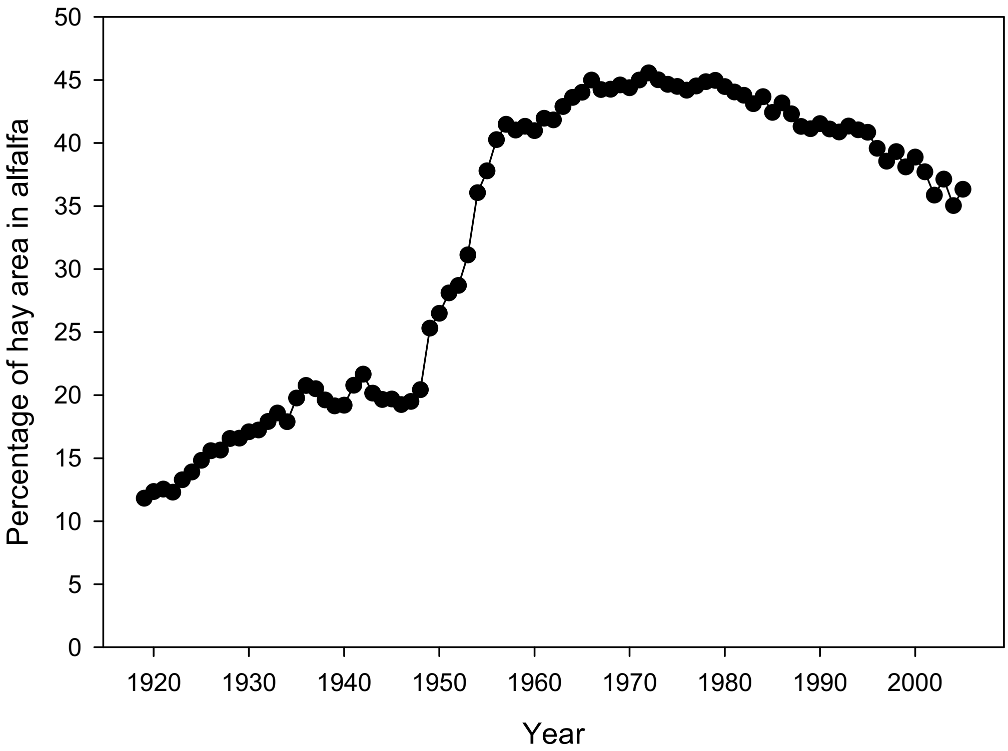


Figure 3.38: Percentage of US hay area in alfalfa during 1909-2005 (USDA, 2006).

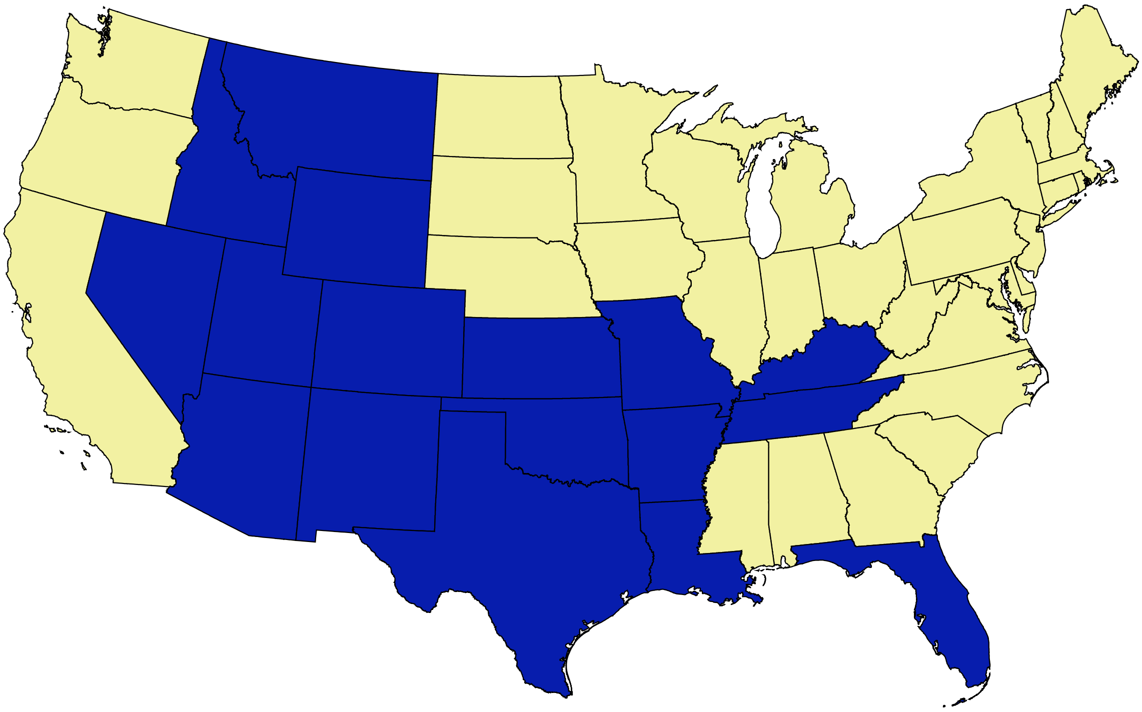


Figure 3.39: Change in proportion US agricultural land in hay by state between 1950 and 2005. Blue represents an increased proportion and yellow a decreased proportion (USDA, 2006).

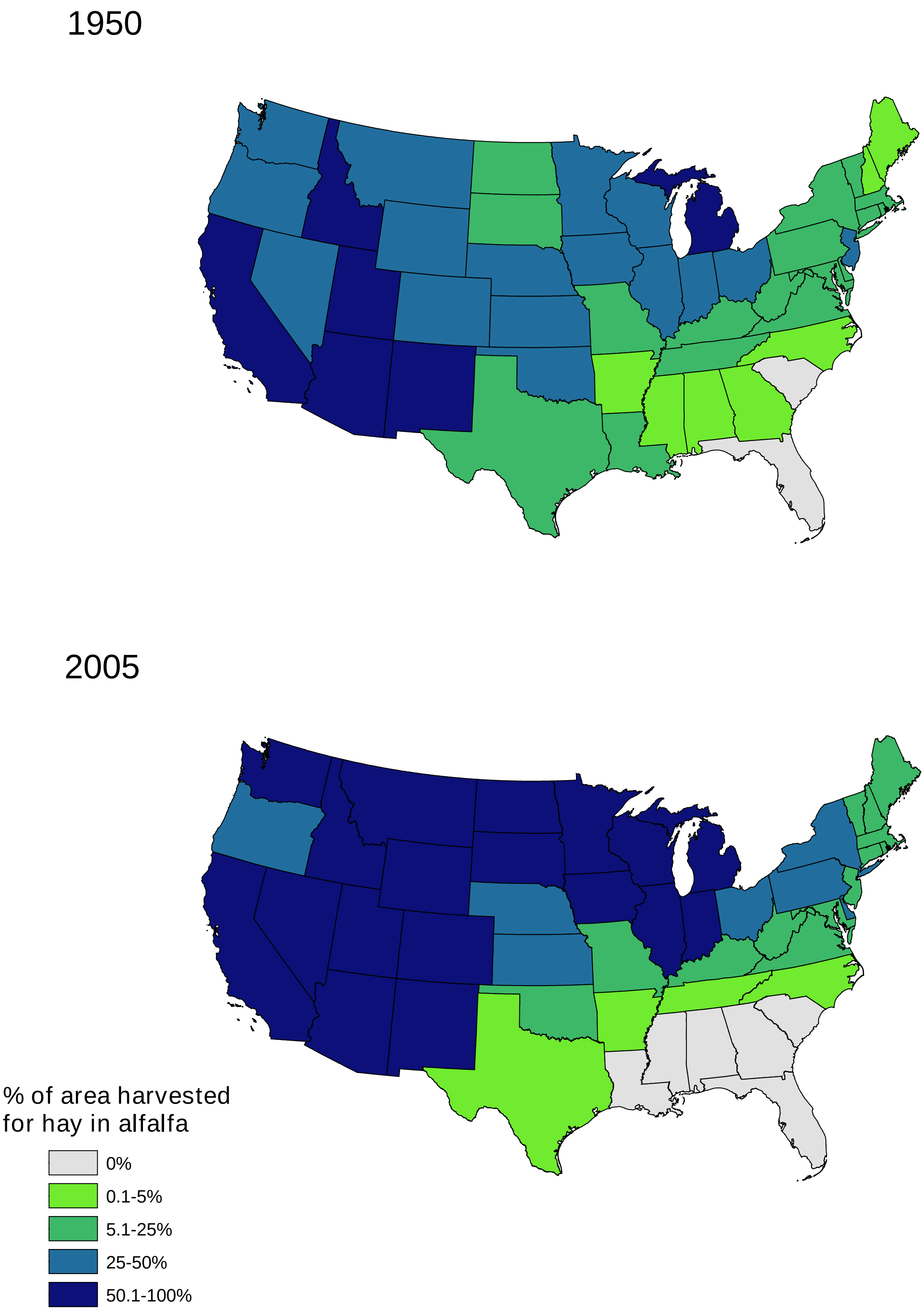


Figure 3.40: Percentage of US agricultural land in hay by state for 1950 and 2005 (USDA, 2006).

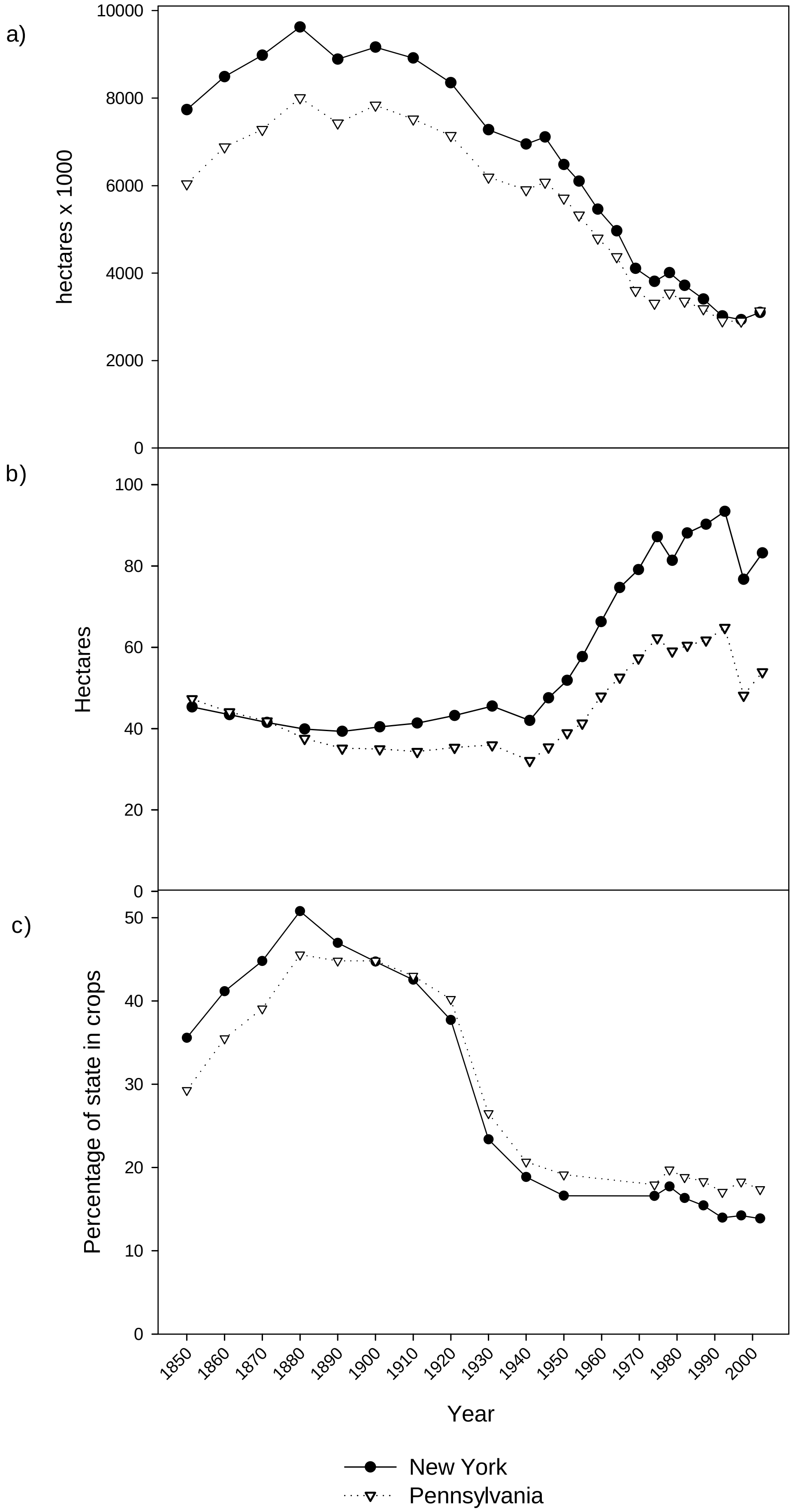
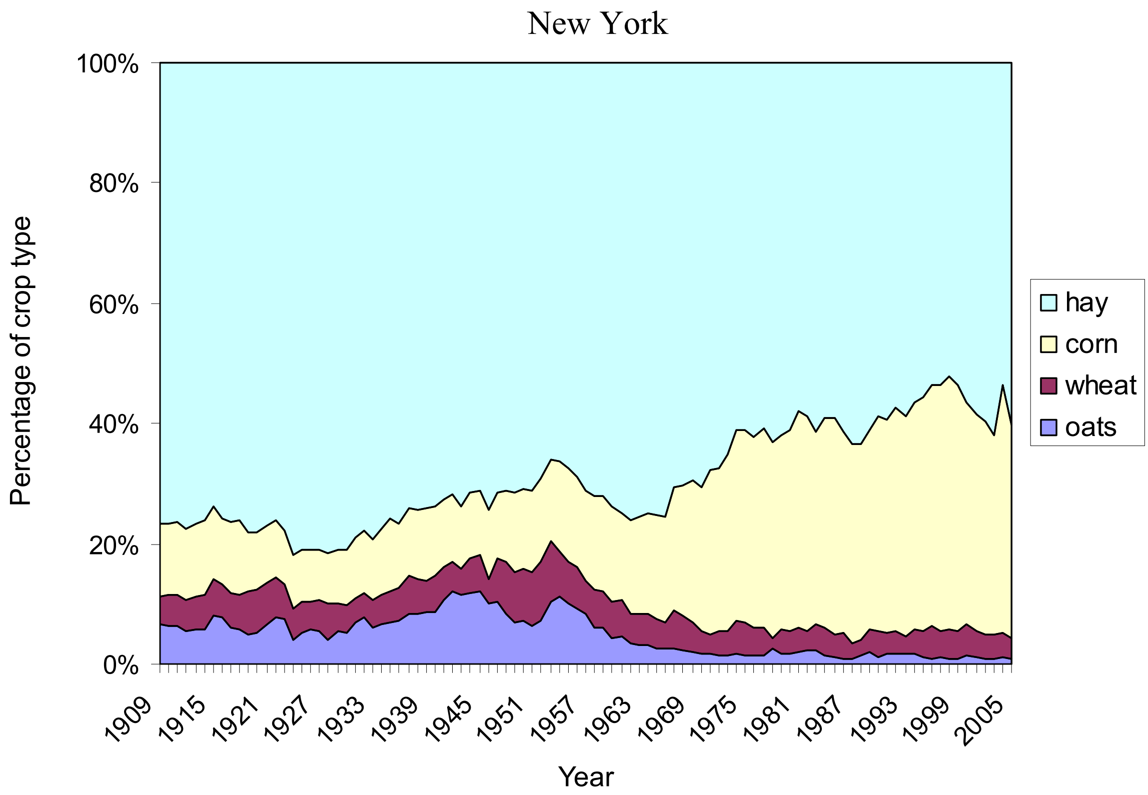


Figure 3.41: Number of farms a), mean farm size b), and percentage of land in crops c) for New York and Pennsylvania during 1850-2002 (USDA, 2006).



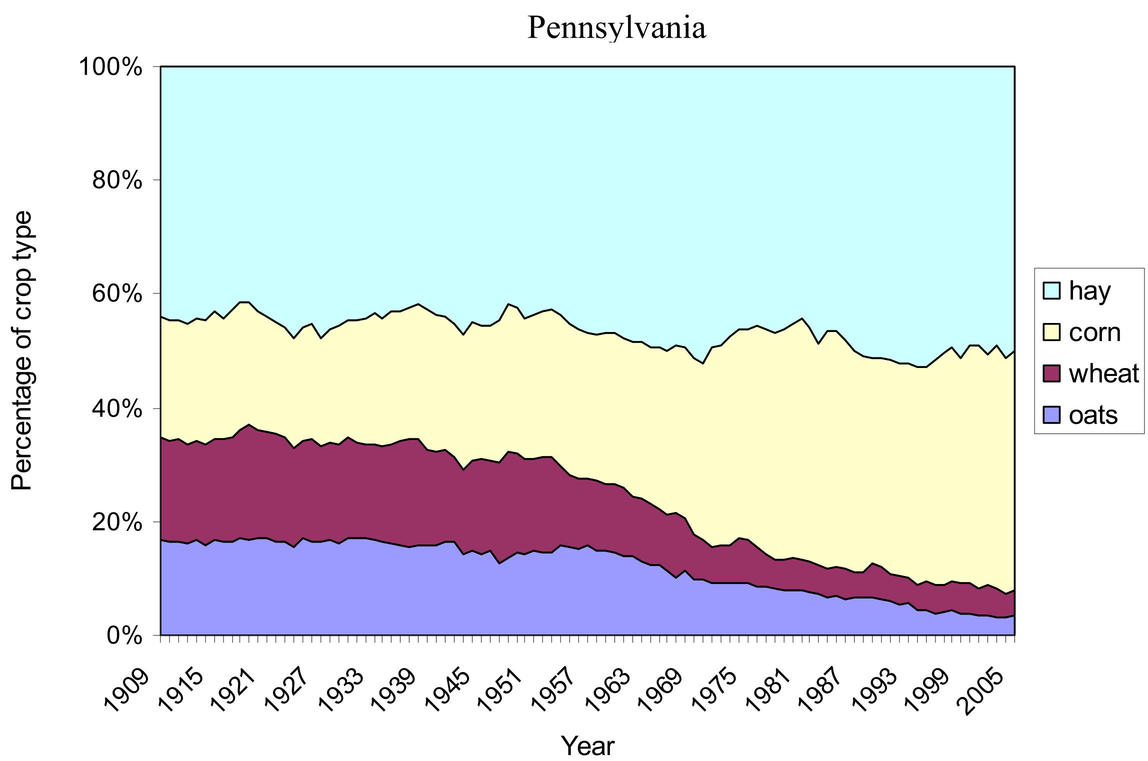


Figure 3.42: Crop composition in New York and Pennsylvania during 1909-2006 (USDA, 2006).

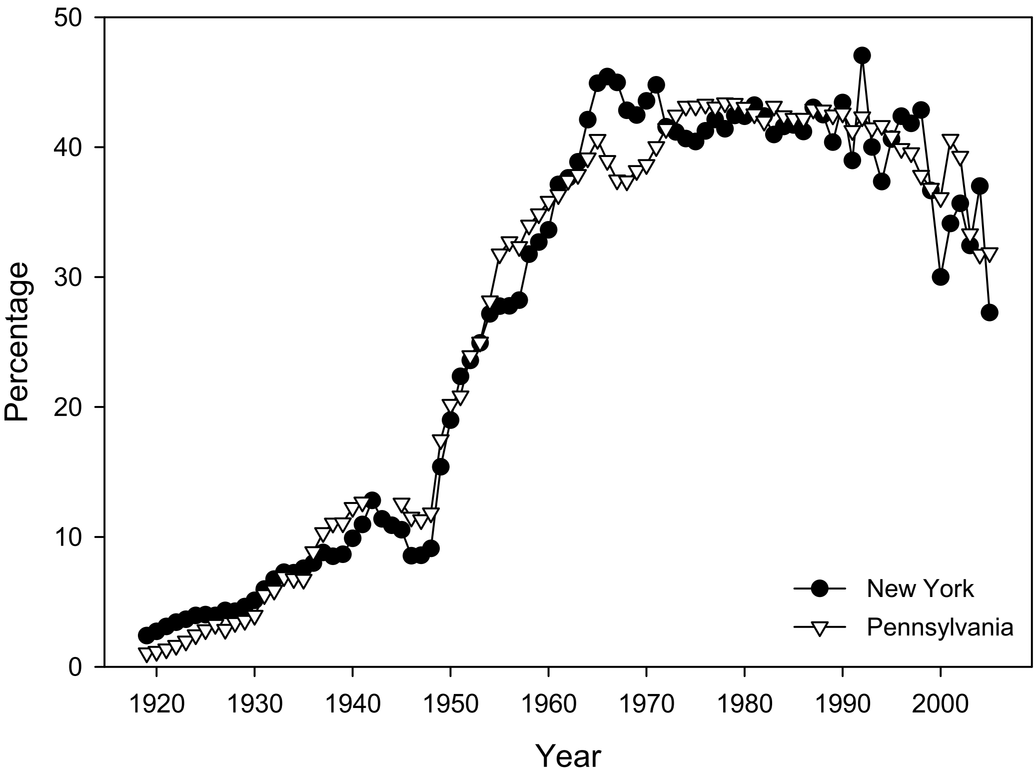


Figure 3.43: Percentage of hay in alfalfa for New York and Pennsylvania during 1920-2005 (USDA, 2006).

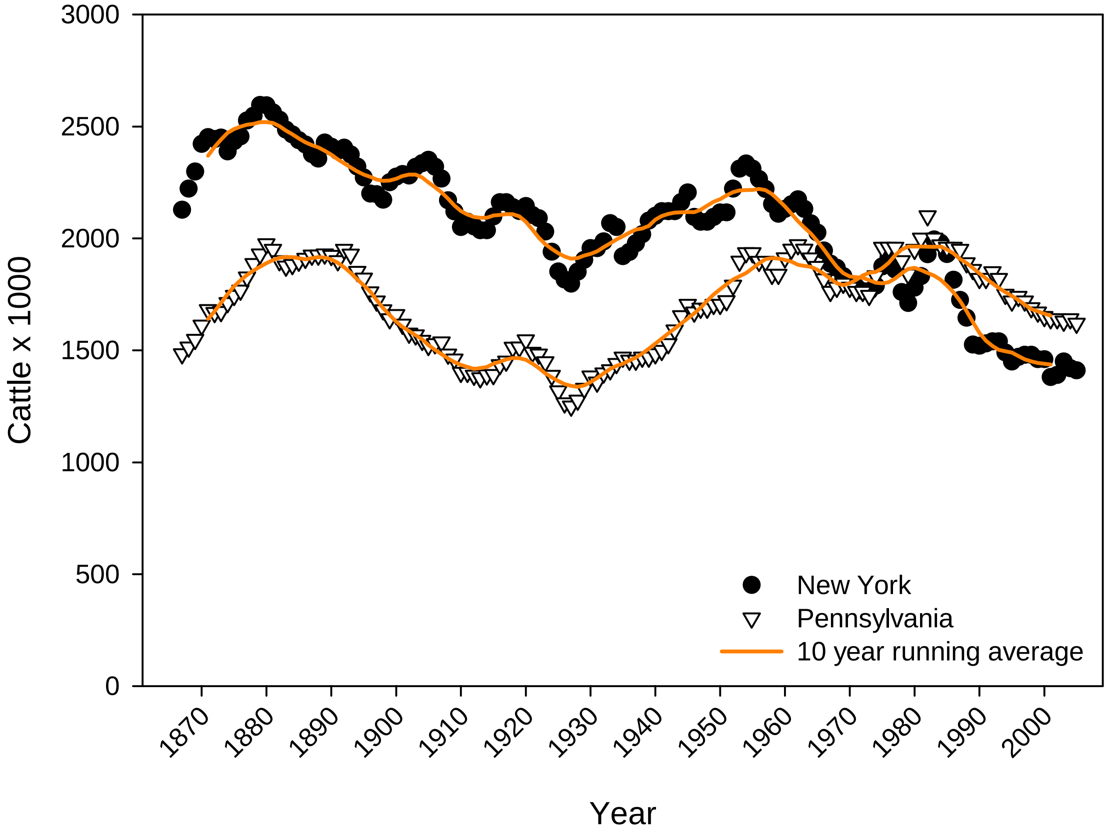


Figure 3.44: Cattle numbers and 10 year running average for New York and Pennsylvania during 1867-2005 (USDA, 2006).

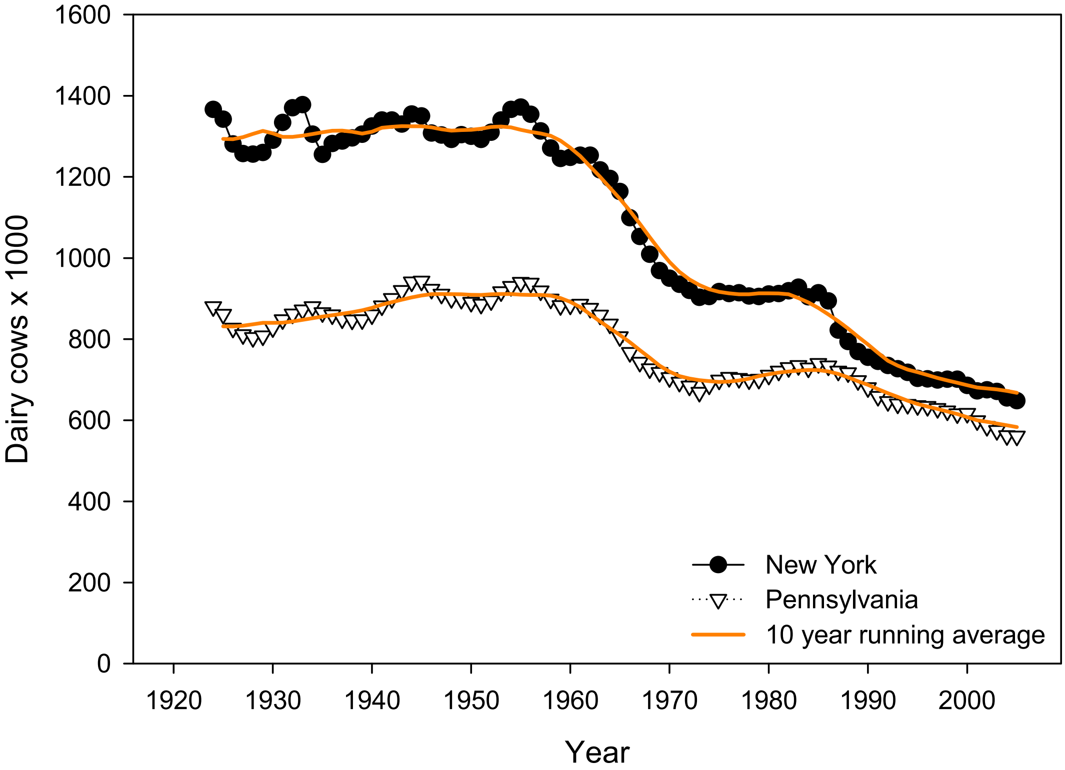


Figure 3.45: Number of dairy cows and 10 year running average for New York and Pennsylvania during 1924-2005 (USDA, 2006).

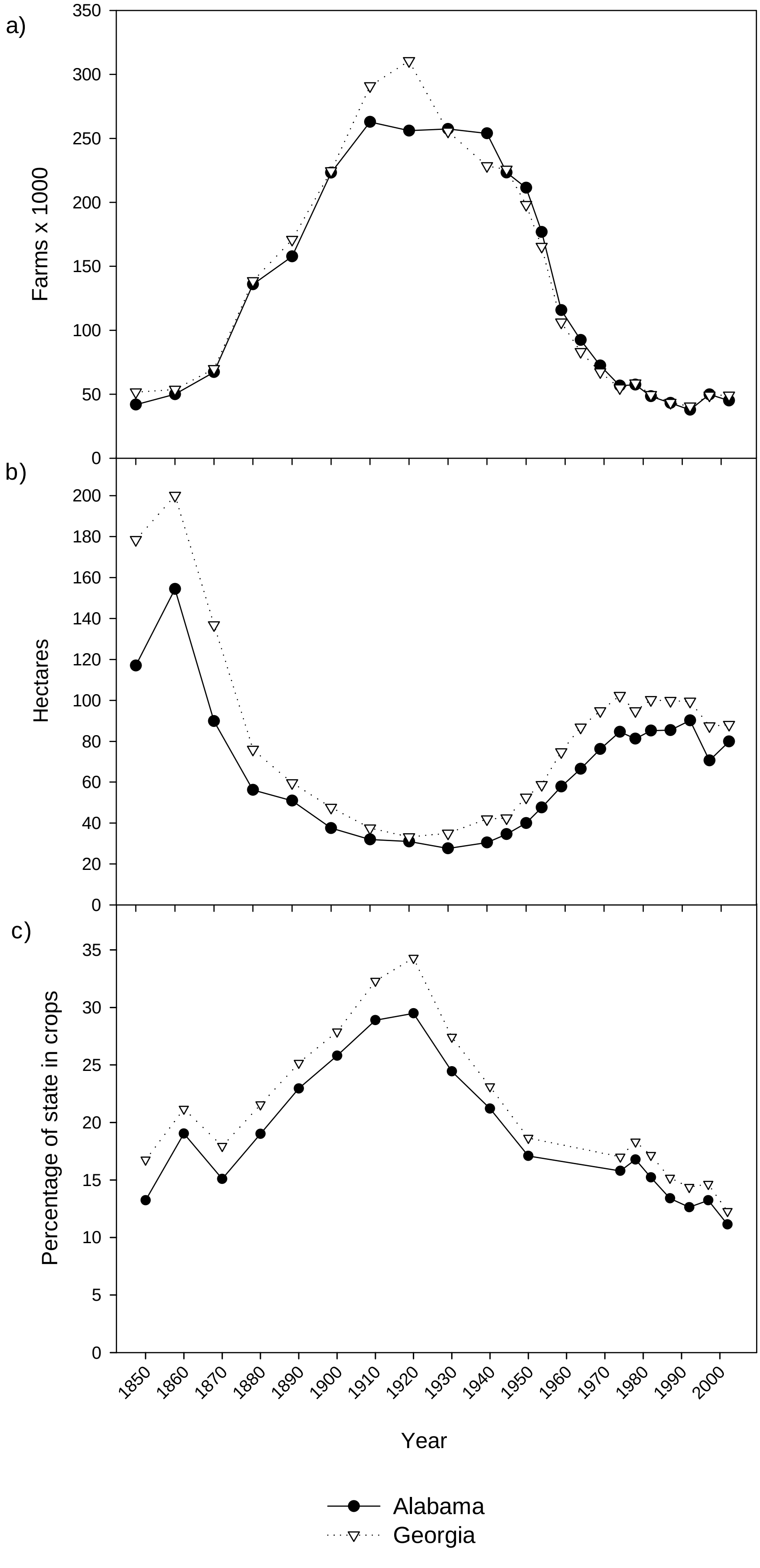


Figure 3.46: Number of farms a), mean farm size b), and percentage of land in cropland c) in Alabama and Georgia during 1850-2002 (USDA, 2006).

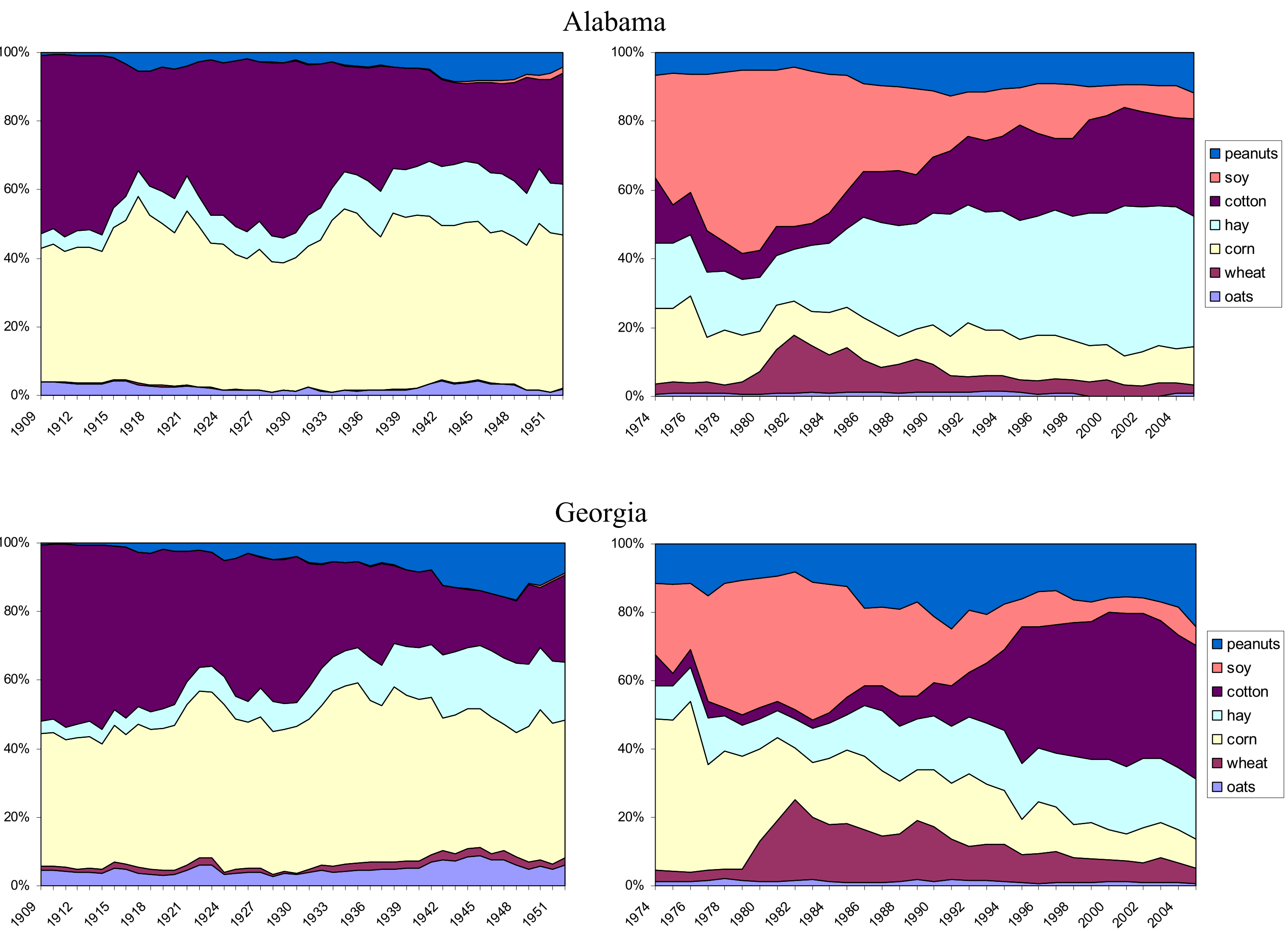
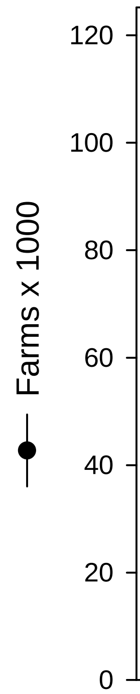


Figure 3.47: Crop composition in Alabama and Georgia during 1909-2006 (USDA, 2006). Data for cotton from 1954-1973 are not available.



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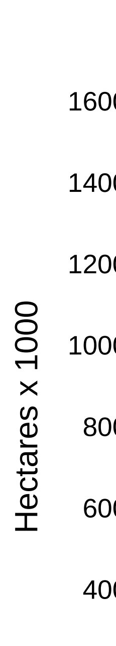

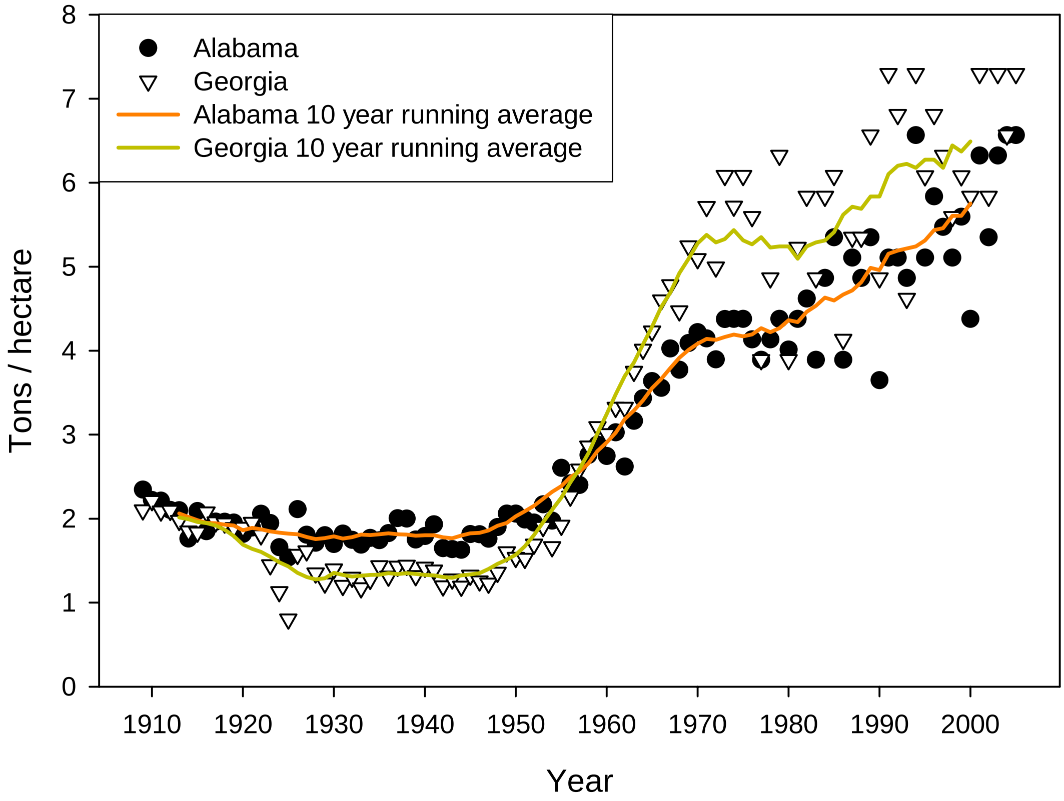

Figure 3.48: Hay yield and 10 year running average for Alabama and Georgia during 1909-2006 (USDA, 2006).

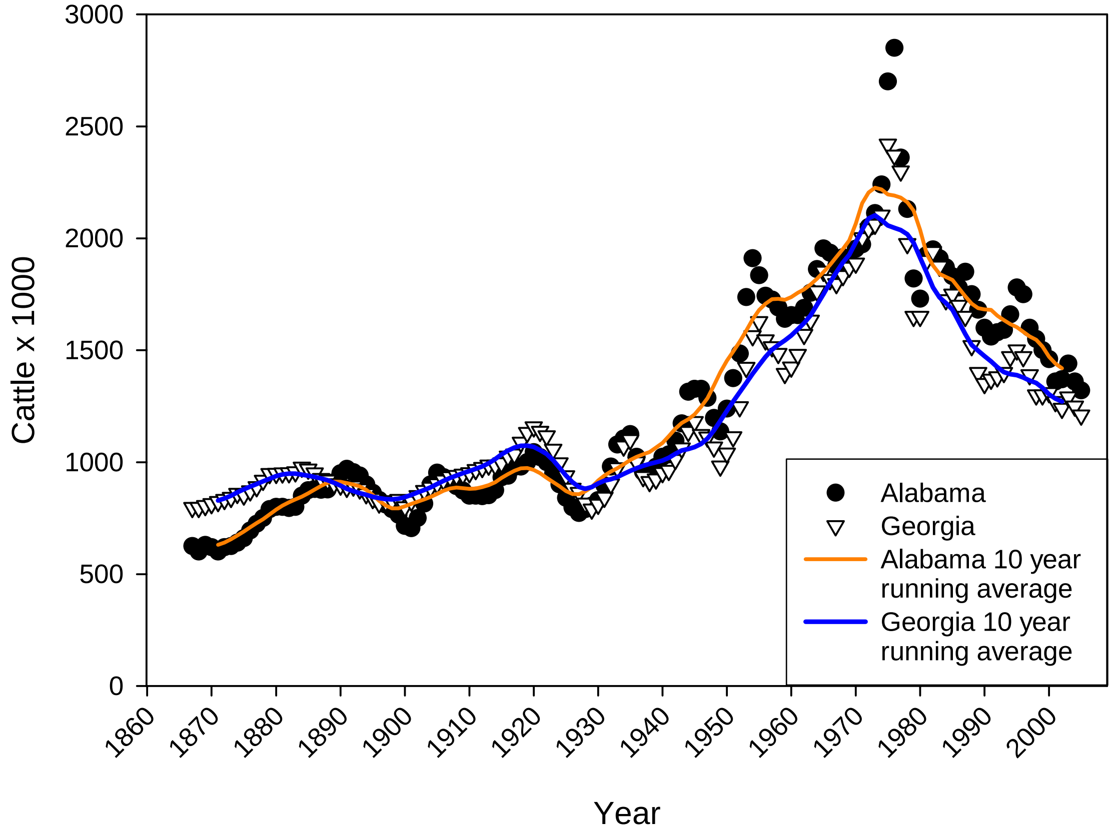

Figure 3.49: Cattle numbers and 10 year running average for Alabama and Georgia during 1867-2005 (USDA, 2006).

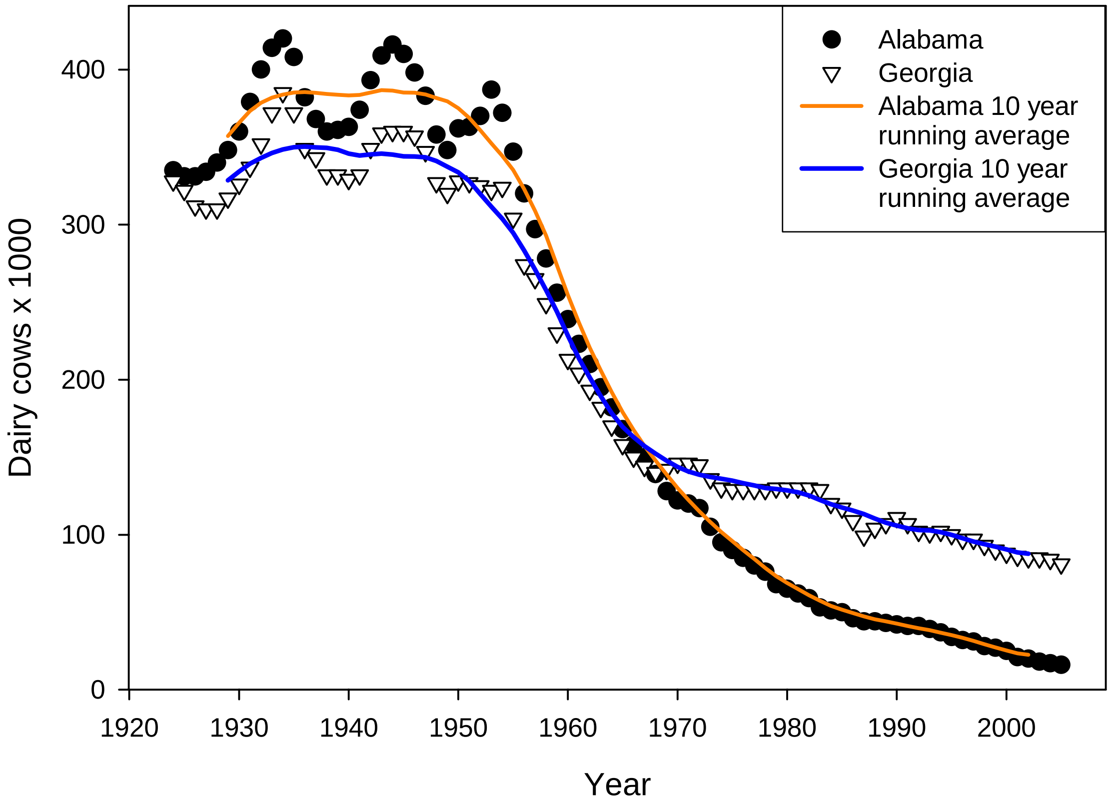

Figure 3.50: Number of dairy cows and 10 year running average for Alabama and Georgia during 1924-2005 (USDA, 2006).

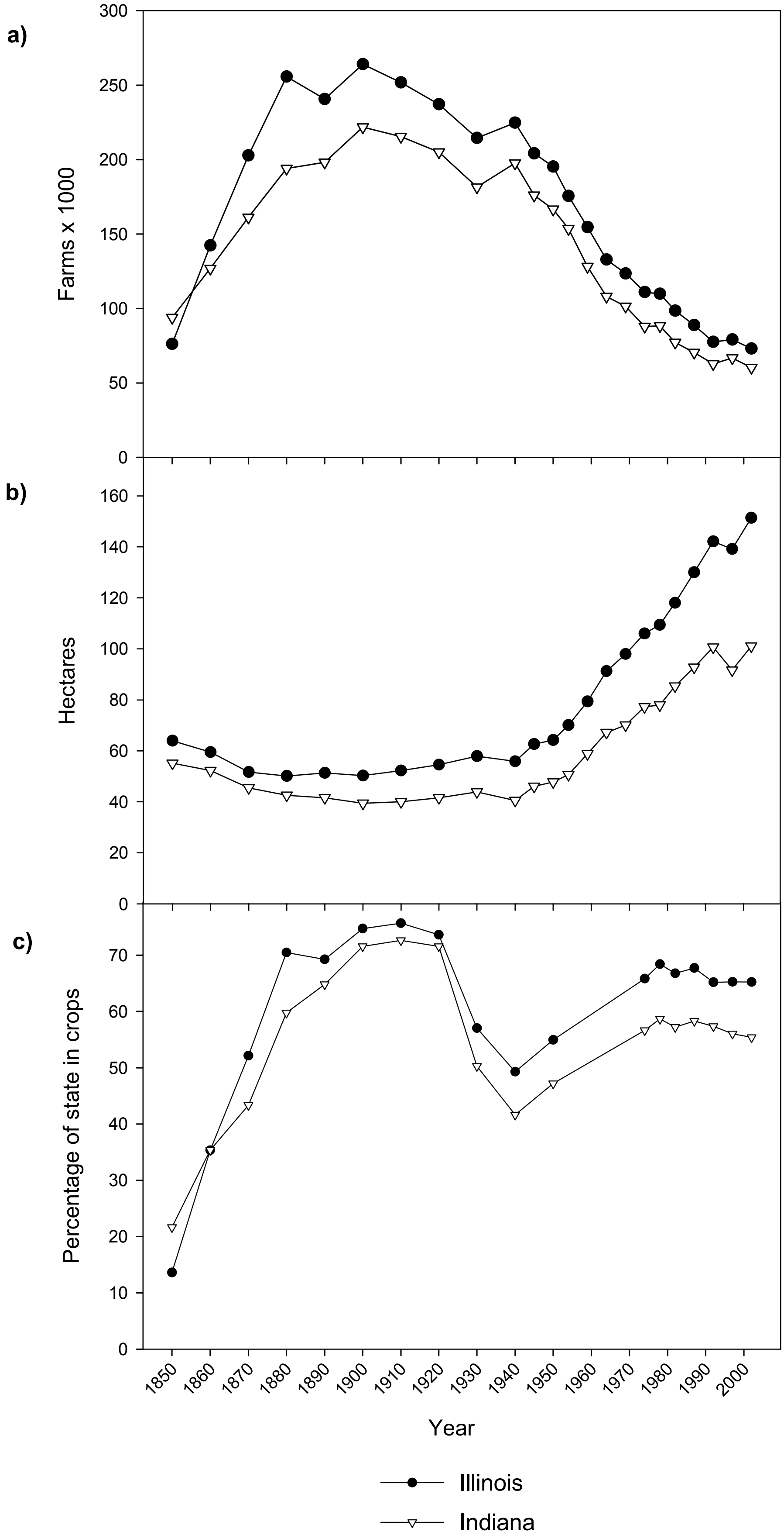

Figure 3.51: Number of farms a), mean farm size b), and percentage of land in crops c) in Illinois and Indiana during 1850-2002 (USDA, 2006).

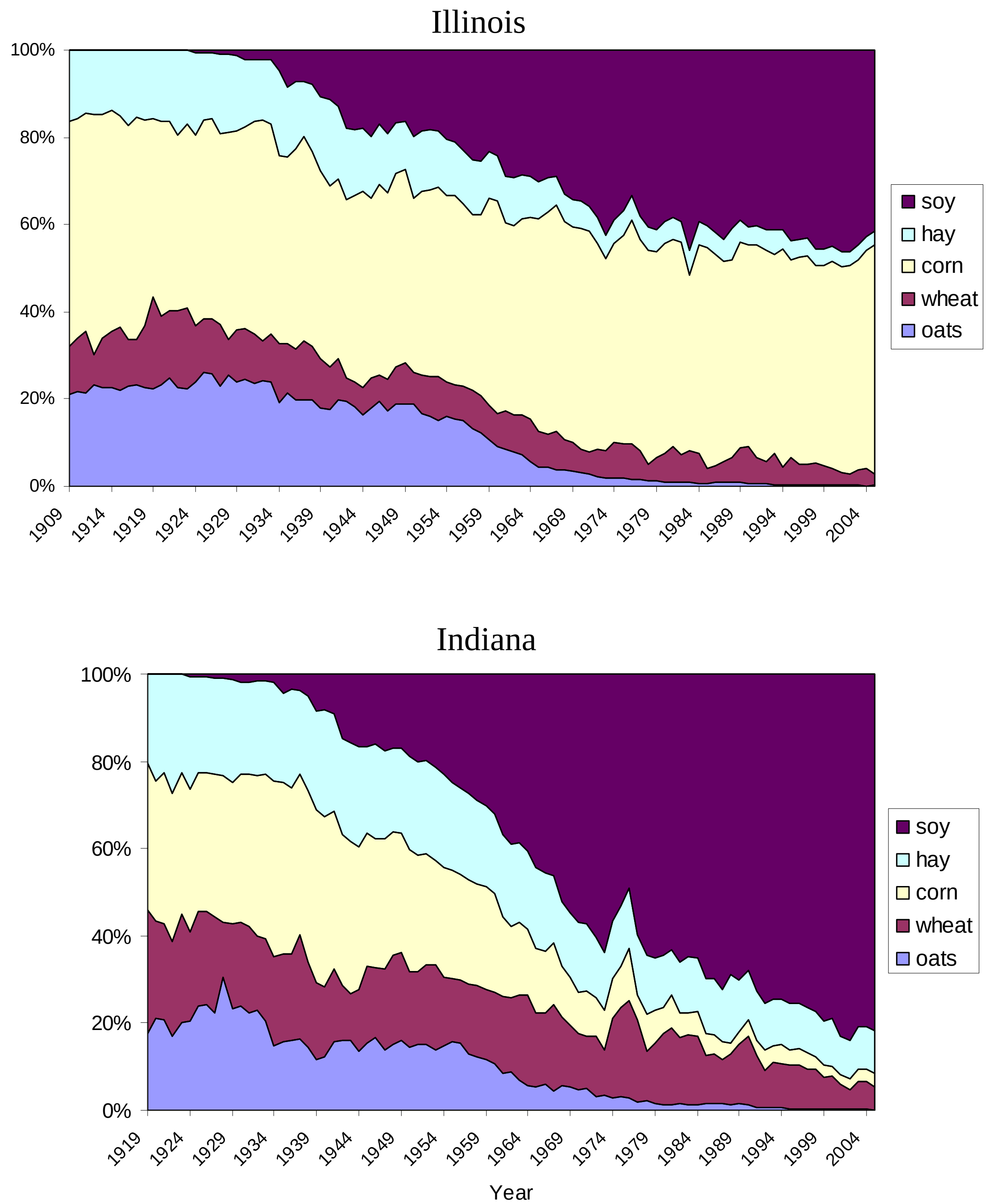

Figure 3.52: Crop composition in Illinois from 1909-2006 and Indiana during 1919-2006 (USDA, 2006).

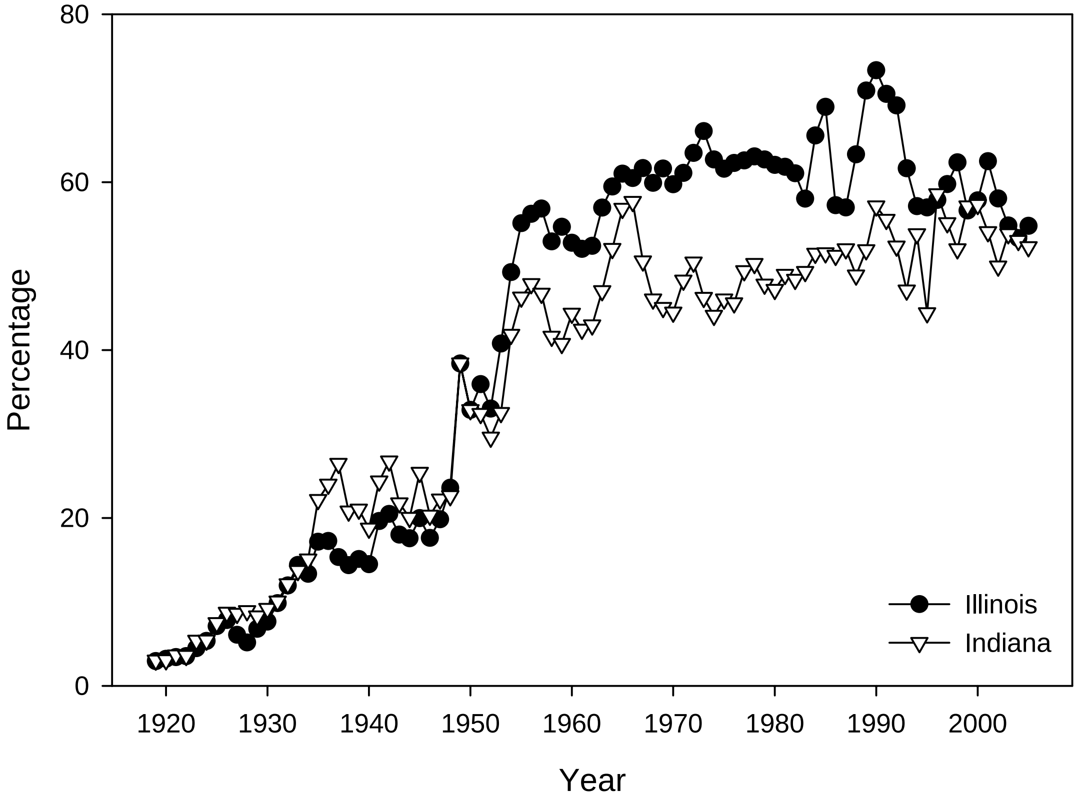

Figure 3.53: Percentage of hay in alfalfa for Illinois and Indiana during 1920-2005 (USDA, 2006).

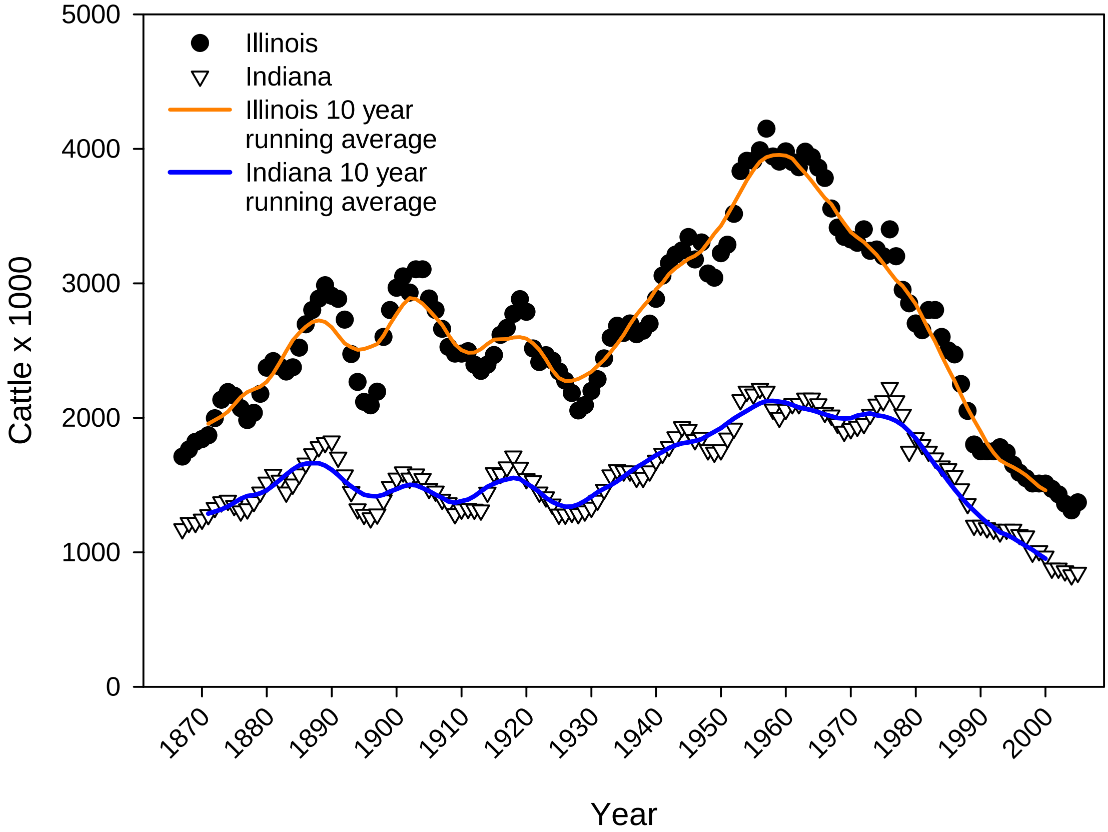

Figure 3.54: Cattle numbers and 10 year running average for Illinois and Indiana during 1867-2005 (USDA, 2006).

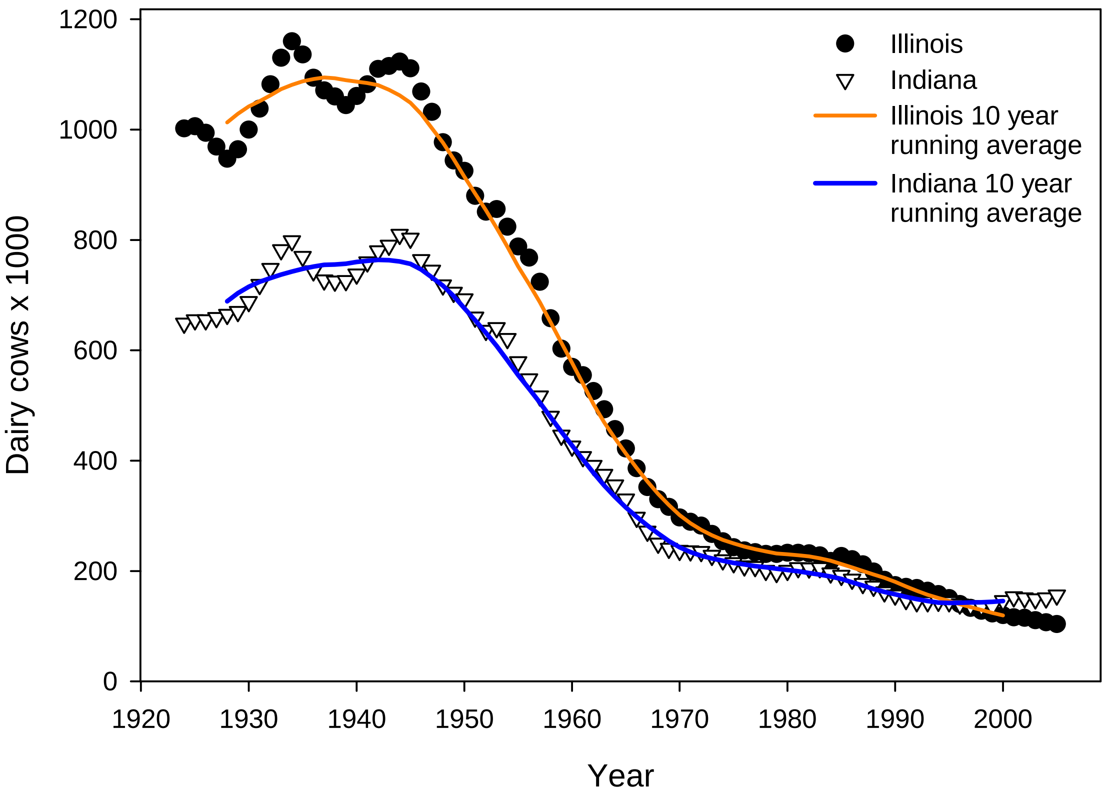

Figure 3.55: Number of dairy cows and 10 year running average for Illinois and Indiana during 1924-2005 (USDA, 2006).

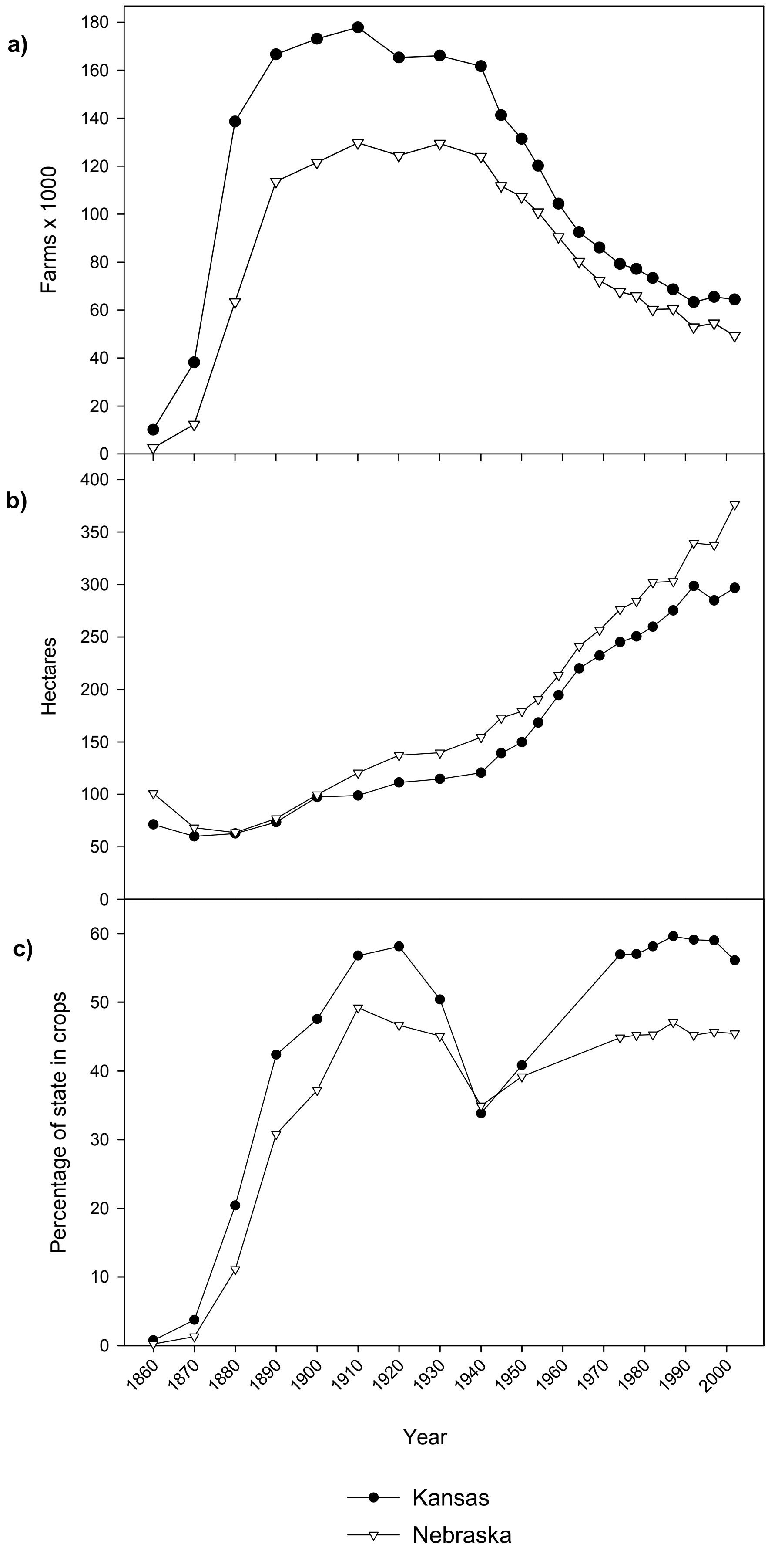

Figure 3.56: a) Number of farms, b) mean farm size, and c) percentage of land in crops in Kansas and Nebraska during 1860-2002 (USDA, 2006).

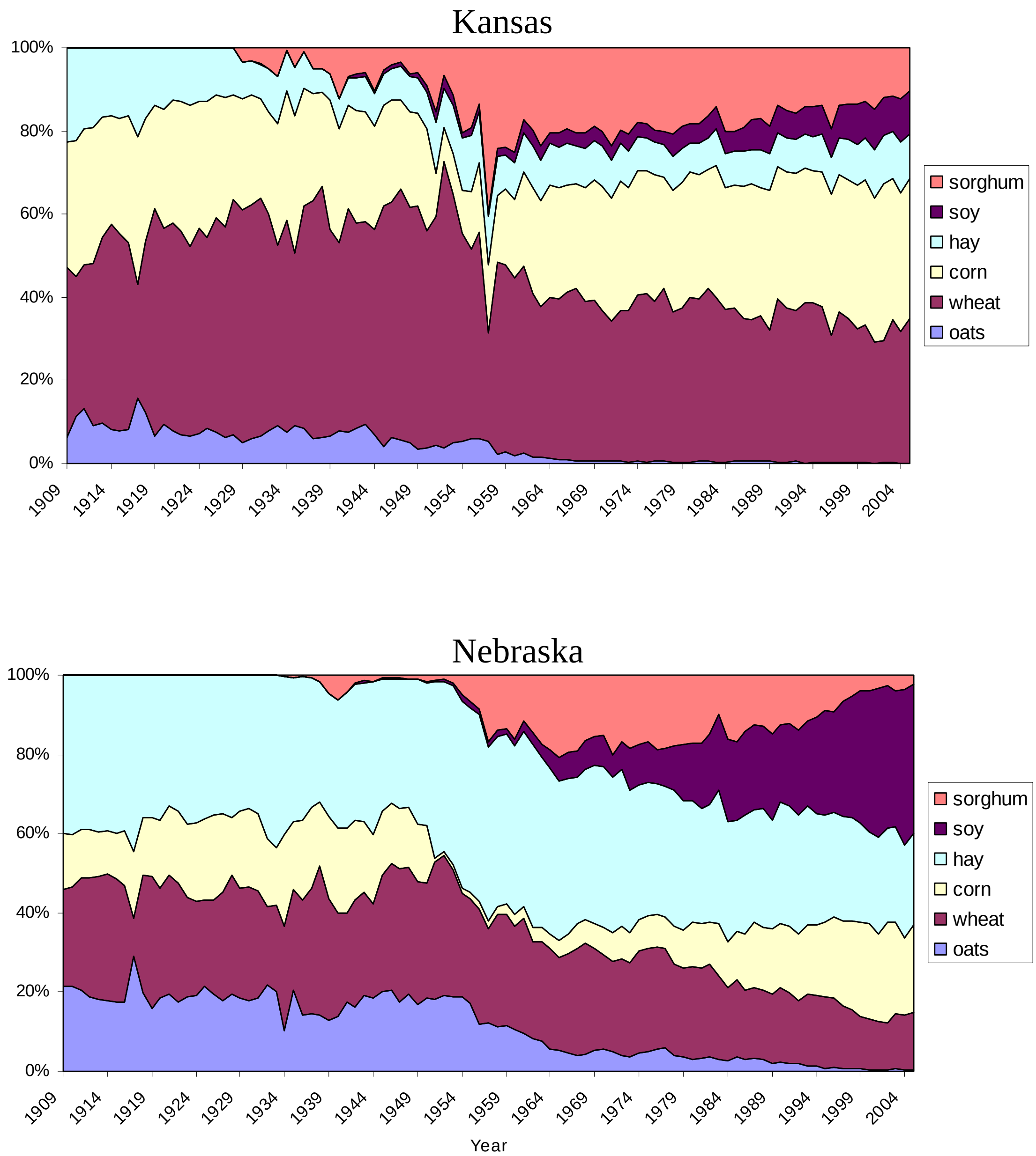

Figure 3.57: Crop composition in Kansas and Nebraska during 1909-2006 (USDA, 2006).

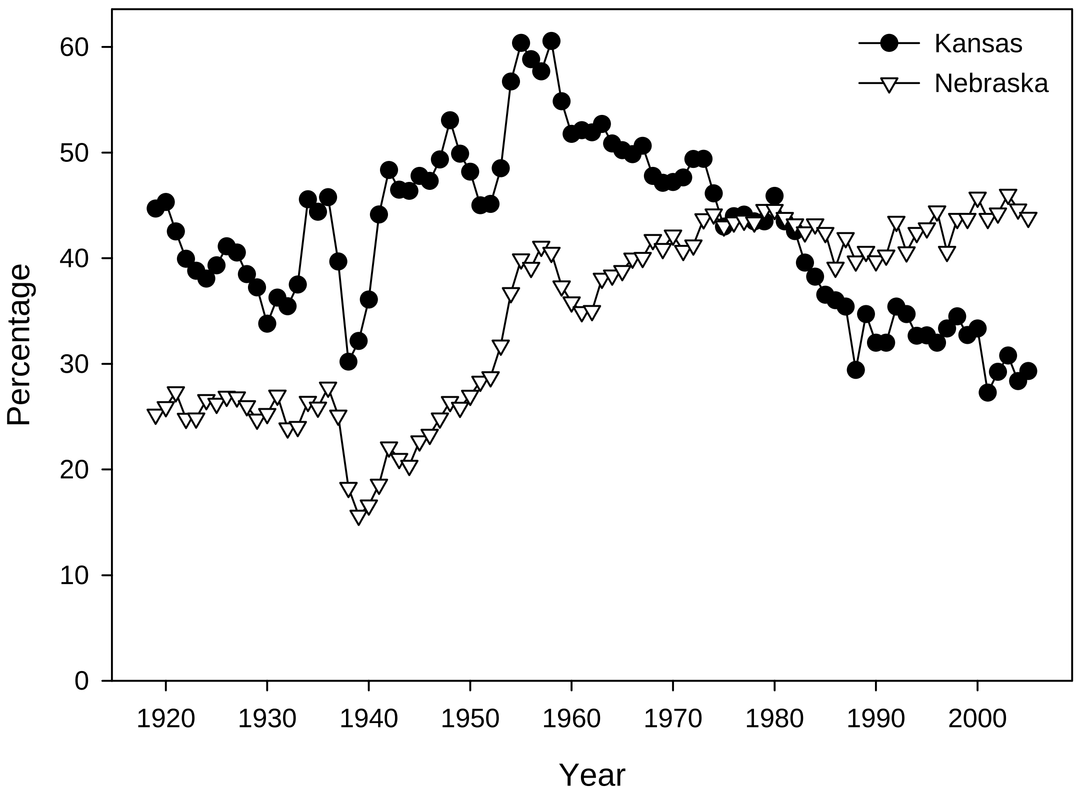

Figure 3.58: Percentage of hay in alfalfa for Kansas and Nebraska during 1920-2005 (USDA, 2006).

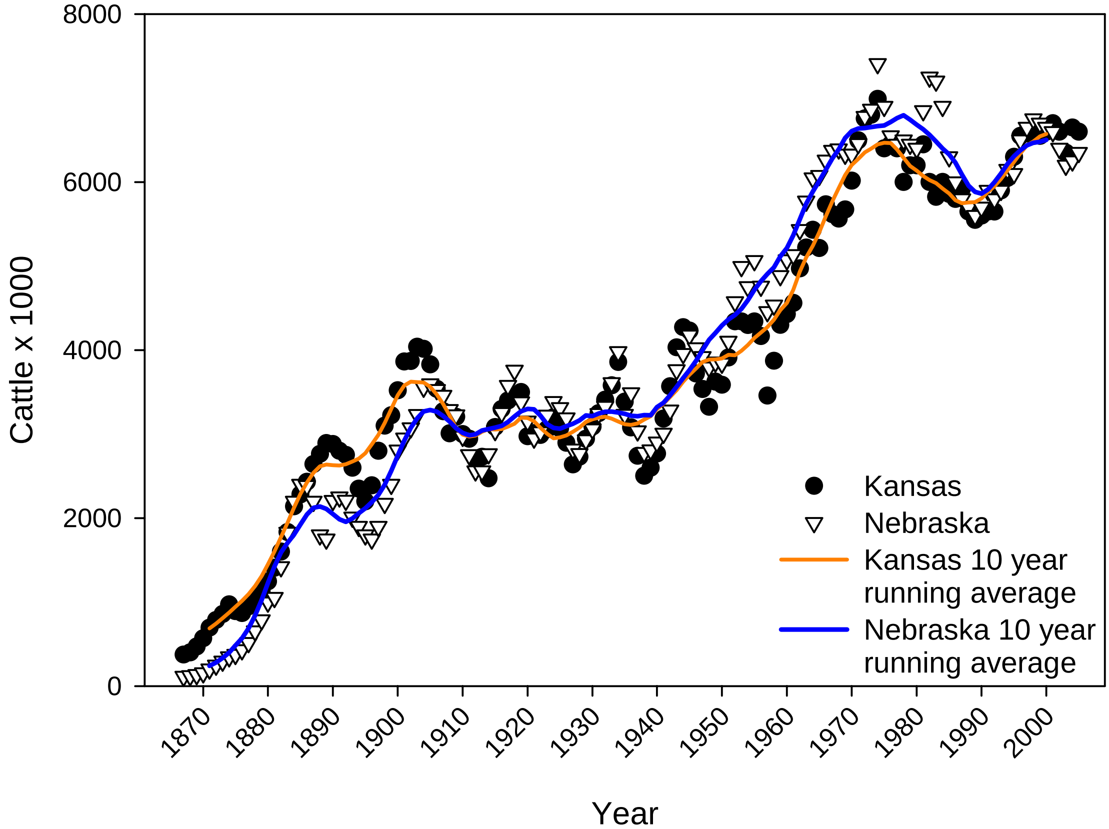

Figure 3.59: Cattle numbers and 10 year running average for Kansas and Nebraska during 1867-2005 (USDA, 2006).

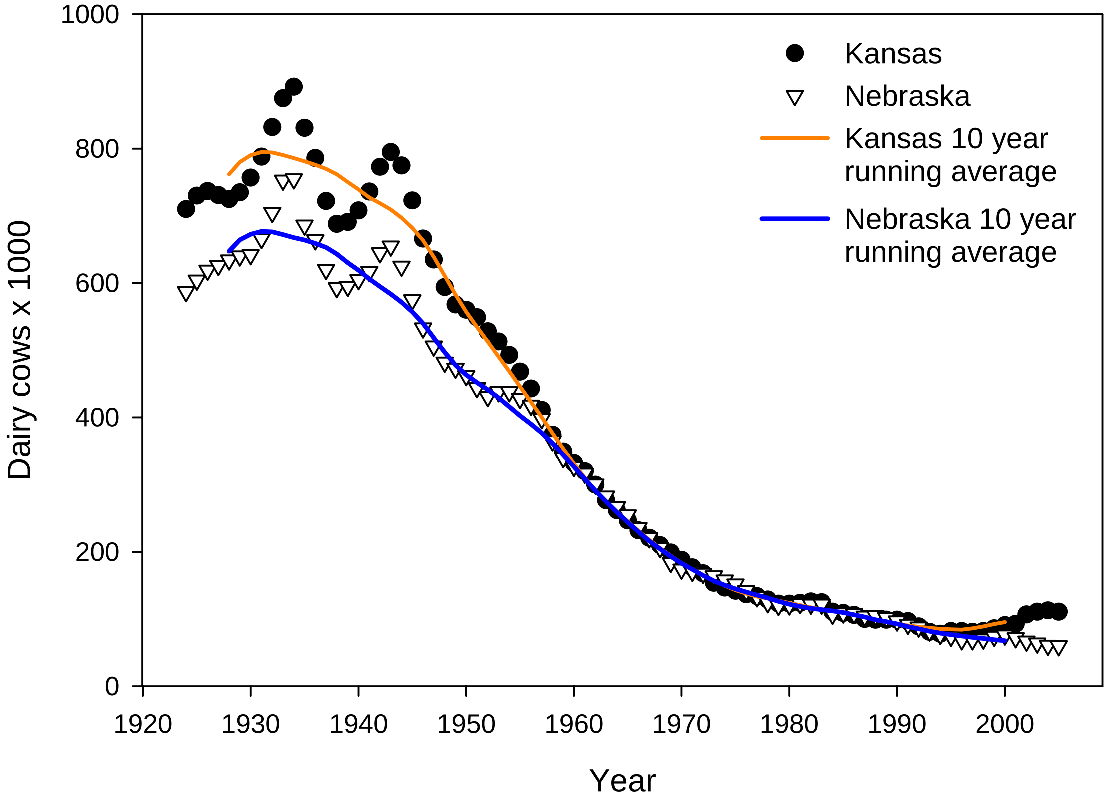

Figure 3.60: Number of dairy cows and 10 year running average for Kansas and Nebraska during 1924-2005 (USDA, 2006).

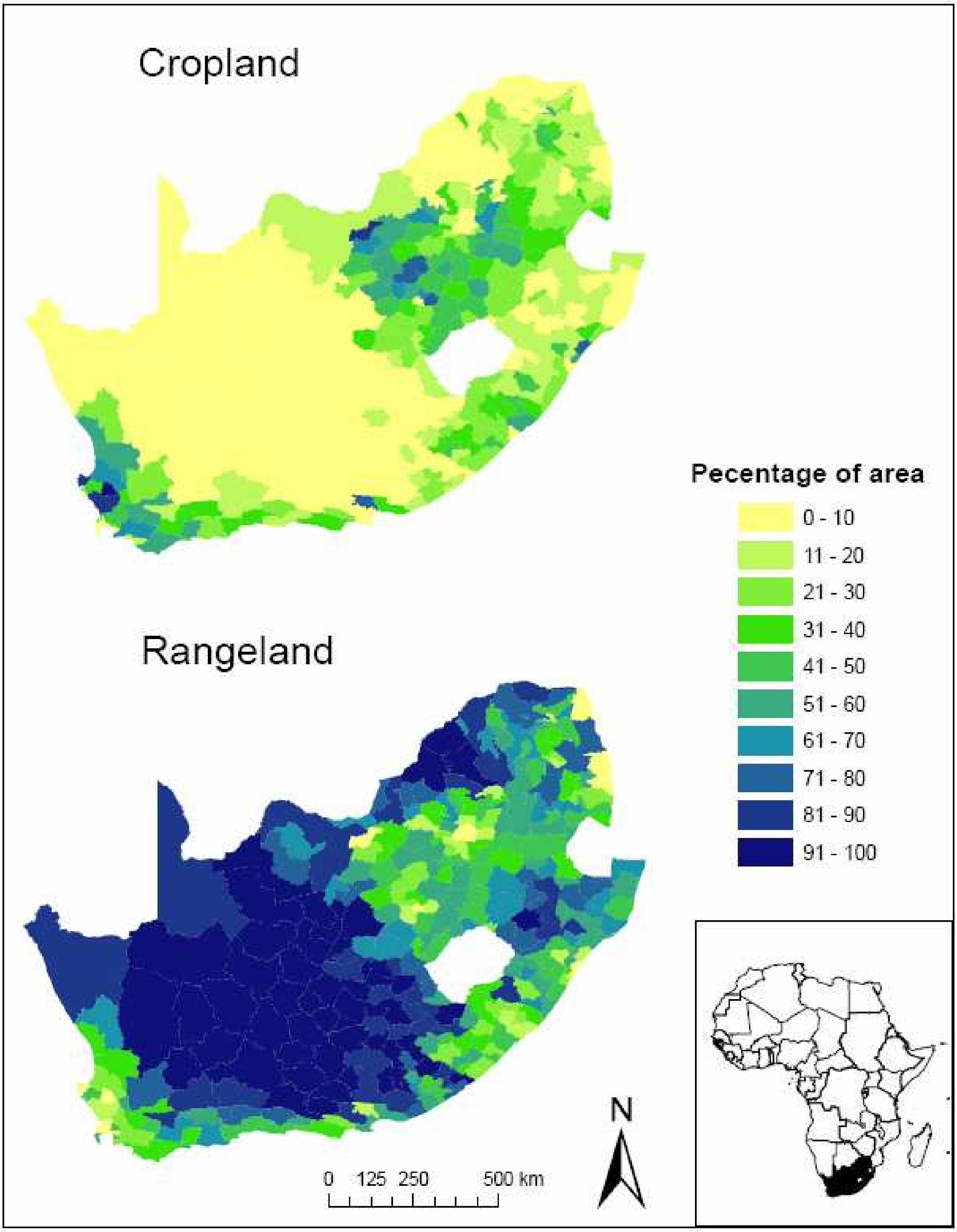

Figure 3.61: Distribution of rangeland and cropland in the Republic of South Africa (1999) (Hoffman et al., 1999).

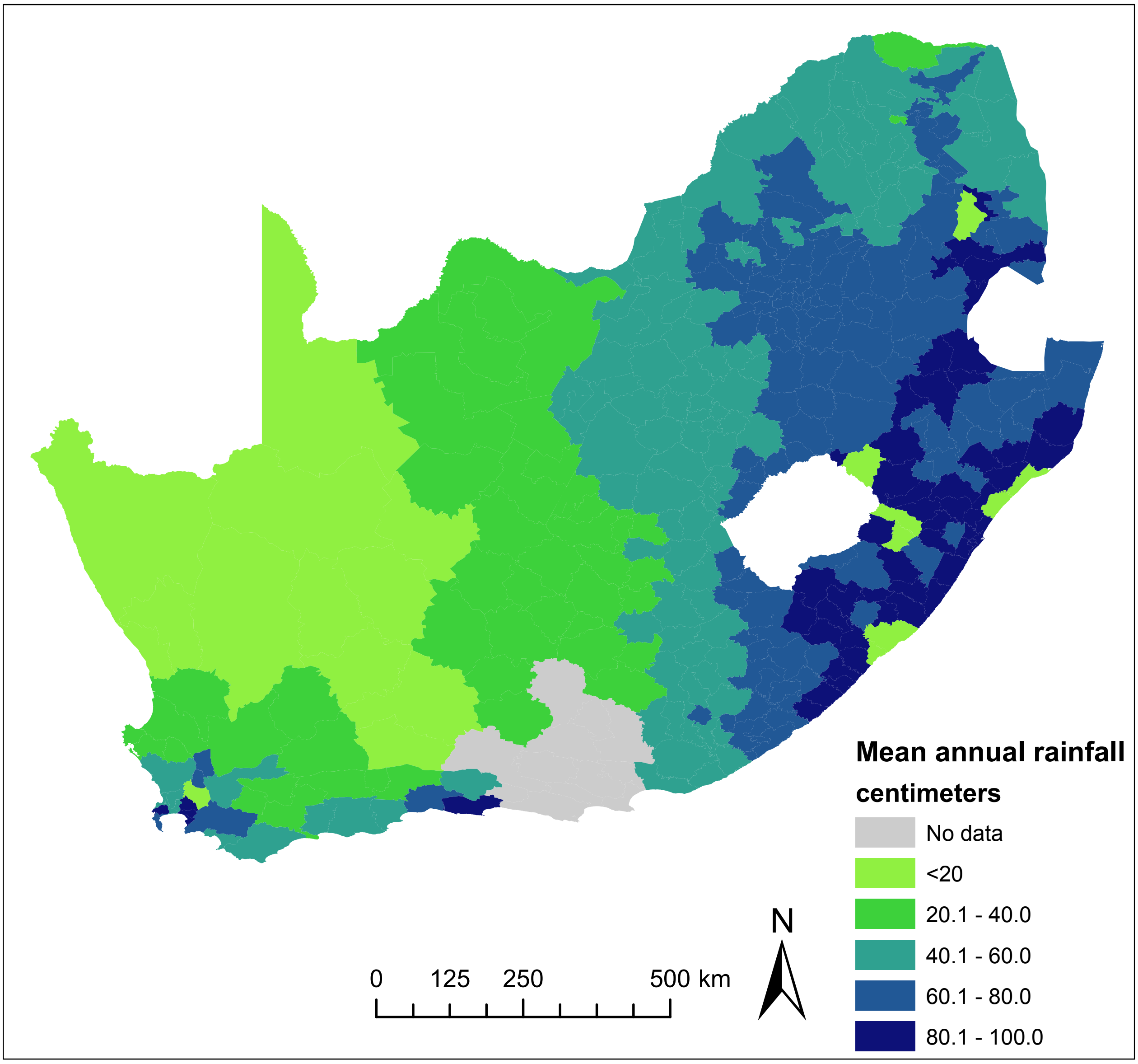

Figure 3.62: Distribution of rainfall in the Republic of South Africa (Hoffman et al., 1999).

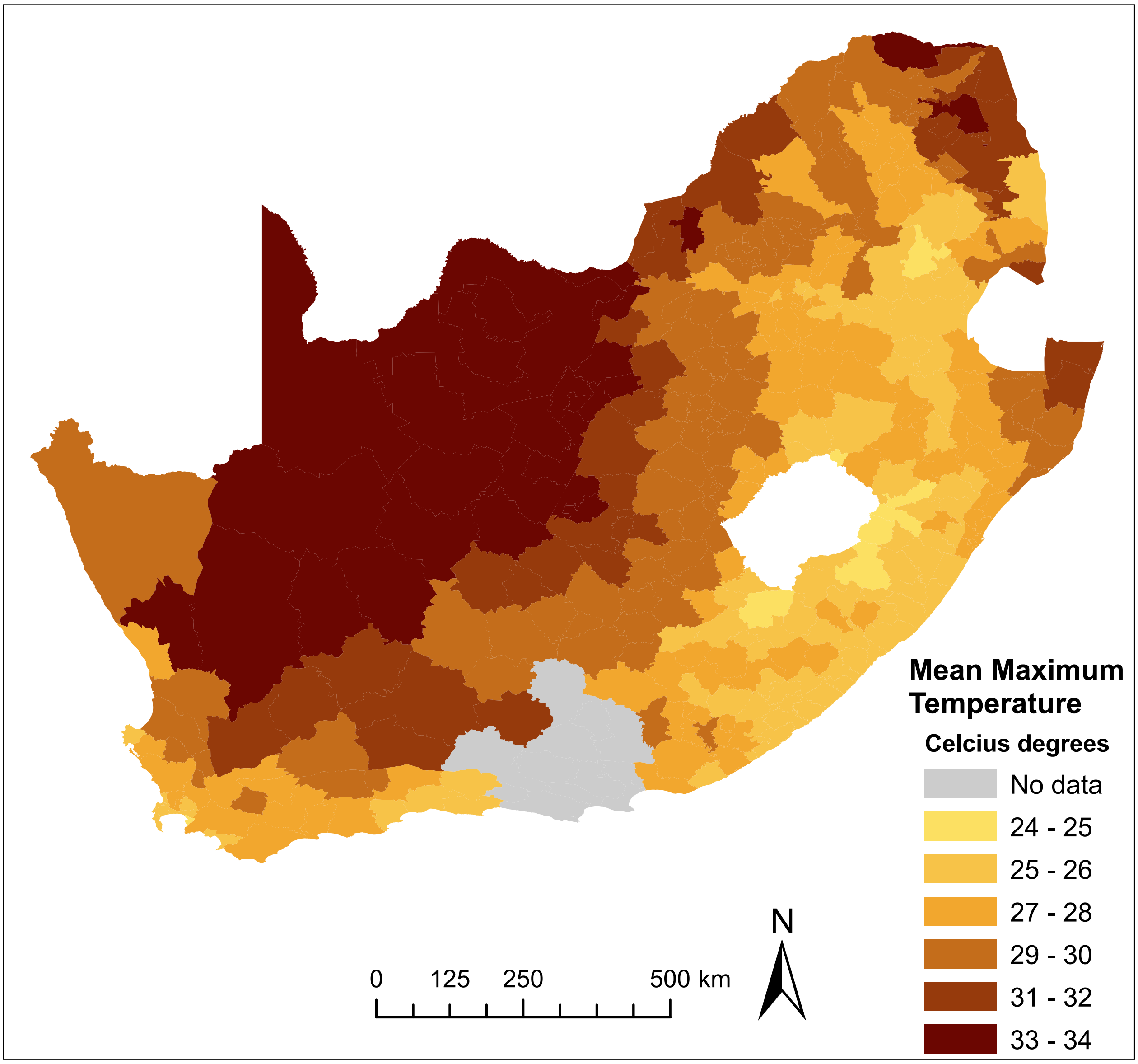

Figure 3.63: Mean maximum temperature for the Republic of South Africa (Hoffman et al., 1999).

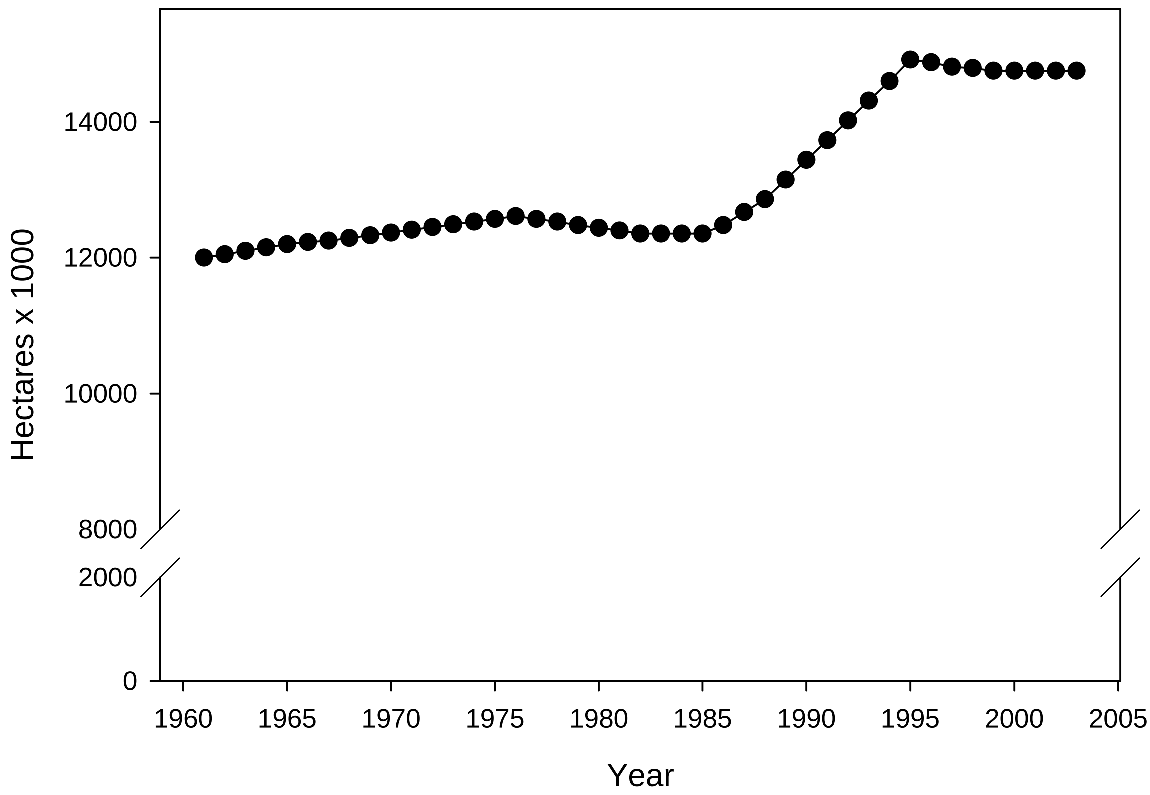

Figure 3.64: Area of arable land in the Republic of South Africa during 1961-2005 (FAOSTAT, 2006).

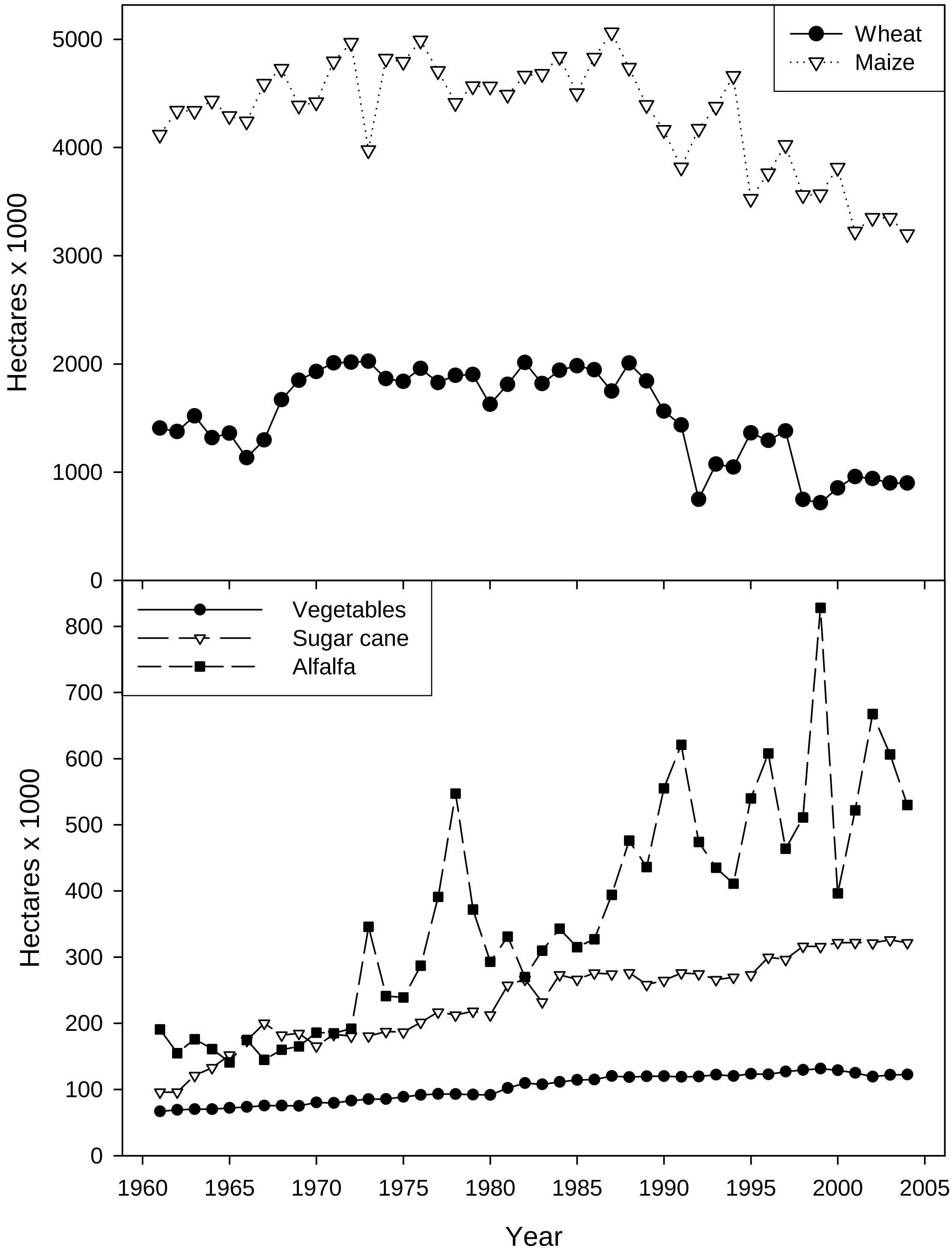

Figure 3.65: Area of wheat, maize, sunflower, sugar cane, and vegetable crops in the Republic of South Africa during 1961-2005 (FAOSTAT, 2006).

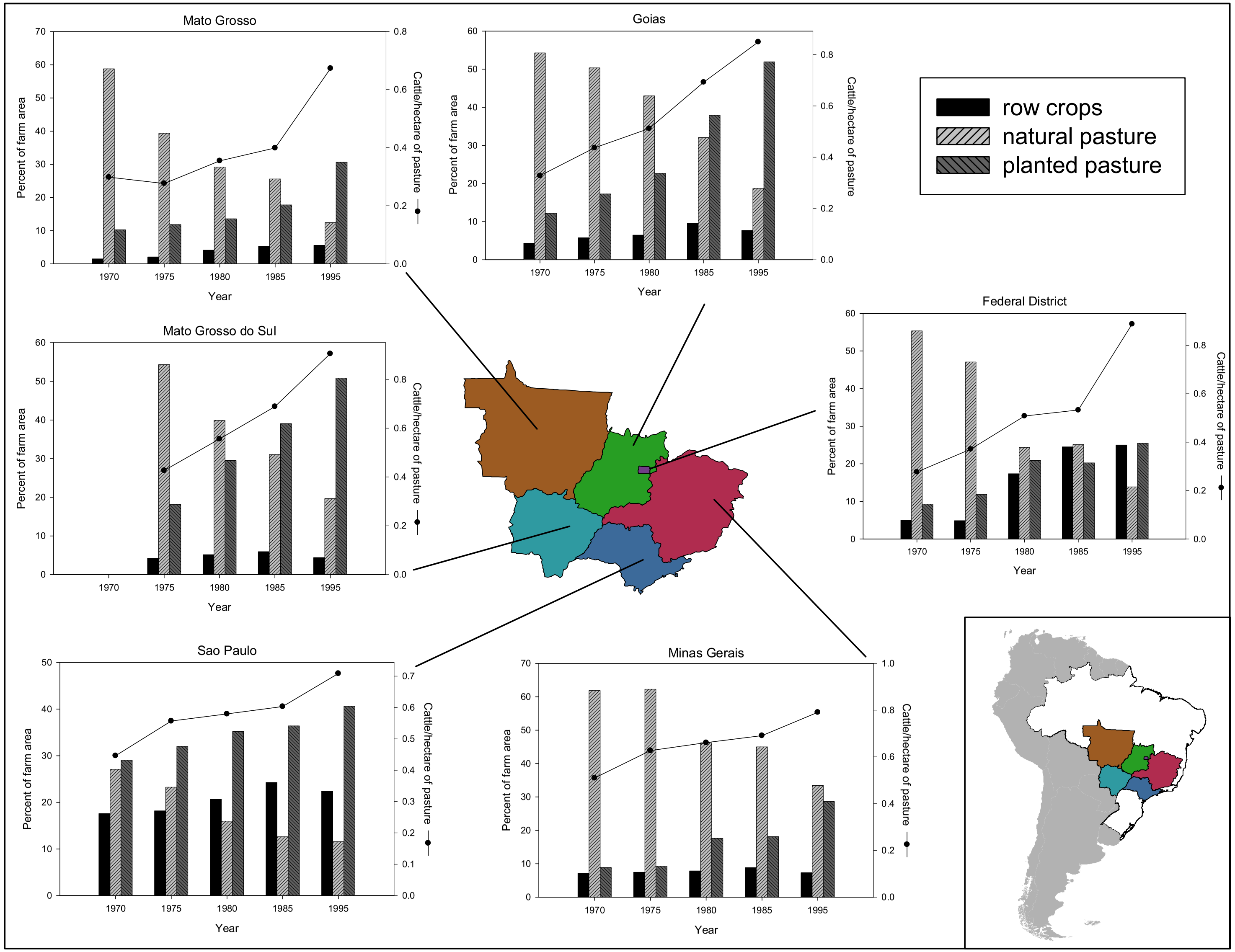

Figure 3.82: Percentage of agricultural area in natural pasture, planted pasture, and row crops and cattle densities in the Brazilian states of Goias, Minas Gerais, Mato Grosso, Mato Grosso do Sul, São Paulo, and the Federal District during 1970-1995 (Instituto Brasileiro de Geografia e Estatística, 2006).

Chapter 4

The expansion, development, and intensification of agriculture in Argentina

Argentina is the 8^{th} largest country in the world (2,776,890 km^2) and one of the world's principal food producing countries with approximately 10.5% of the land area devoted to agricultural crops (FAOSTAT, 2006). The two principal areas of agricultural production are the Pampas of east-central Argentina and the Chaco in the north-central and north-western regions of the country (Fig. 4.1). The Pampas cover over 240,000 km^2 in Argentina are part of the $\sim$760,000 km^2 Río de la Plata grassland system which covers eastern Argentina, Uruguay, and southeastern Brazil (Soriano et al., 1991; Dinerstein et al., 1995). The Chaco is a $\sim$1.2 million ha mosaic of seasonal dry forest, savanna, and grasslands in austral South America, mostly in northern Argentina, eastern Bolivia, and Paraguay (Bucher, 1982; Dinerstein et al., 1995).

The development of European agriculture in Argentina was initiated with the founding of the city of Potosí, Bolivia in 1545 after the discovery of silver. The city became the most important source of precious metals for the Spanish Empire and its population reached a maximum of close to 200,000 inhabitants. Potosí was not only one of the largest cities in the Spanish colonial empire but, at over 4,000 meters in elevation, the highest. Because of the high elevation, neither food nor wood for construction and fuel could be locally produced, and subsequently needed to be imported from lower elevations (Barsky and Gelman, 2001).

The production of food for, and particularly the raising of mules for transporting goods to, Potosí became an important industry for Tucumán and Salta in the Chaco of northwestern Argentina. The demand for mules became so high that they were transported from Santiago del Estero, Córdoba, and Buenos Aires more than 1,000 km to the southeast in the Pampas (Barsky and Gelman, 2001).

In 1573, cattle were introduced into the Pampas by Juan de Garay and, left wild, began to reproduce and populate the Pampas (Barsky and Gelman, 2001). In the 1700's these wild cattle formed the base of the first cattle industry in the Pampas with the wild animals harvested for their hides and later for the meat to be salted for export (Barsky and Gelman, 2001). Although row crop agriculture began with European settlement, the development of large-scale commercial agriculture did not begin until the mid-1800's (Barsky and Gelman, 2001).

Sheep ranching was an important activity during the first half of the 19^{th} century, but with falling wool prices and the introduction of refrigeration, cattle production increasingly became the dominant agricultural activity in Argentina during the second half of the century. In the later part of the 19^{th} century row crop agriculture grew substantially with the removal of hostile Indians, European immigration, improved farm technologies, and expanding railways; reaching it's maximum extent during the 1930's (Barsky and Gelman, 2001). This expansion occurred mostly in the Pampas, however, the eastern, more humid Chaco saw the development of farming during the 1930's, particularly for cotton (Barsky and Gelman, 2001).

The effects of the great depression, a decrease in global markets for row crop commodities, and domestic economic policies favoring industrialization shifted agricultural production towards meeting the demands of the domestic, and increasingly urban market which, subsequently favored beef production (Barsky and Gelman, 2001). This period lasted until the 1960's when the global market in meat was reduced and the first introductions of hybrid crop varieties, inorganic fertilizers and pesticides, and increased farm mechanization initiated an increasing emphasis upon row crop agriculture in Argentina (Barsky and Gelman, 2001).

Starting in the 1990's the implications of economic restructuring, imposed by the International Monetary Fund, began to become evident as most import and export tariffs were removed, which facilitated a rapid increase in the use of agrochemicals, new crop varieties, and technologies while increasing access to international markets for Argentine agricultural goods (Hall et al., 2001; Barsky and Gelman, 2001). During this period fertilizer consumption more than quadrupled (Fig. **??**), the area in soybeans more than doubled (Fig. **??**), and with the near total removal of export and import tariffs (previously 30-40% and 65%, respectively) (Food and Agriculture Organization of the United Nations, 2004; Salvador, 2001) the exports of cereals and soybeans increased substantially (Fig. **??**).

The mid-1990's saw the introduction of genetically modified (GM), herbicide tolerant soybeans commonly referred to by the trade name "Roundup Ready®", and the widespread use of associated glyphosate herbicides and conservation tillage (Fig. 4.5) (Qaim and Traxler, 2005). The same period saw also saw the introduction of other GM crops, however, at a much more limited scale. For example, *Bt* cotton comprises only 5% of the cotton grown compared to >80% for soybean (Qaim and de Janvry, 2003; Qaim and Traxler, 2005). The shift towards conservation tillage and glyphosate-resistant soybeans, although greatly increasing the amount and number of applications of pesticides, has also been associated with the large decreases in the use of high toxicity pesticides (Qaim and Traxler, 2005).

The introduction of GM soybeans, its associated technology, and foreign demand have been important components in the rapid intensification of agriculture in Argentina and has lead to increased specialization and polarization of agricultural production towards either row crops or livestock husbandry at the expense of more traditional mixed farm management. Furthermore, this specialization has also led to an increased monocultural landscape as the diversity of crops have decreased considerably as formerly important crops, such as alfalfa, have declined considerably in area (Fig. 4.6).

The increased area of cultivated soybean resulted from both a switch from other crops, but also from increasing the area of land under cultivation starting in the 1990's (Fig. 4.7). Although the mean yields of principal row crops increased with increasing area cultivated, the relationship is non-linear so that the relative increase in yields has decreased as increasingly more land has been cultivated (Fig. 4.8). Moreover, crop yields have increased relatively little in relation to external inputs of agrochemicals, improved farm machinery, and GM crops. For example, yields of principal crops increased little following 1991 when Argentine agriculture became more industrialized and fertilizer consumption greatly increased, however, the introduction of GM soybeans and increased use of no-till technology in the late 1990's does appear to have been beneficial (Fig. 4.9).

The intensification of agriculture in Argentina exhibits the same characteristics as have been witnessed in other regions, however, the temporal scale over which it has occurred has been very short. The loss of mixed farming systems, specialization, and decreases in landscape heterogeneity have important implications for biodiversity conservation in Argentine agroecosystems. Furthermore, the recent lucrative nature of row crop agriculture has facilitated its expansion into regions where formerly it was not practiced, converting grassland remnants in the Pampas to row crops and driving rapid deforestation of the Chaco (Gasparri and Parmuchi, 2003; Bilenca and Miñarro, 2004; LART-FAUBA, 2004; Grau et al., 2005b).

4.1 The intensification of agriculture in the Pampas

Although multiple delineations of the Pampas have been made, all of them locate the majority of the Pampas within the provinces of Buenos Aires, Córdoba, and Santa Fe; the remainder located in La Pampa and San Luis (Soriano et al., 1991; Dinerstein et al., 1995) (Fig. 4.10). Land use within the Pampas is largely dependent upon a strong gradient in precipitation declining from east to west that mostly excludes row crop agriculture from the westernmost portions of the Pampas in the provinces of San Luis and La Pampa (Hall et al., 1988) (Fig. 4.11). Additionally, decreasing mean annual temperature towards the south further determines the pattern of land use in the region (Hall et al., 1988).

Based upon these aforementioned climatic factors, as well as phsyiographic, edaphic, and hydrological characteristics, the Pampas can be divided into 5 regional subdivisions; the 1) Rolling Pampa, 2) Interior Pampa, 3) Southern Pampa, 4) Flooding Pampa, and 5) Mesopotamian Pampa (Soriano et al., 1991). These regions, and where they transition into the Chaco and Espinal forest at their northern and western extent, form the principal area of row crop agriculture and mixed row crop-grazing systems in Argentina (Fig. 4.12). The biotic and abiotic characteristics of these subdivisions have been critical in determining the development of agricultural land use in the Pampas (Hall et al., 1988).

As row crop agriculture expanded in the Pampas during the late 1800's it was principally in the Rolling Pampa, which is the best farmland in Argentina and has formed the nucleus of agricultural activity in the country (Hall et al., 1988; Morello and Solbrig, 1997). The Austral Pampa is the principal cereal zone since the growing season is more limited by temperature, while the Inland Pampa traditionally has been used for mixed agriculture and grazing due to the decreased and more variable rainfall, while historically most of the Flooding Pampa has been utilized for cattle grazing (Hall et al., 1988).

The expansion of row crop agriculture from the late 1880's to 1930 was a function of converting natural grasslands historically used for grazing and was primarily dependent upon soil types and precipitation patterns, having reached an equilibrium where all suitable land had been converted to row crops by the late 1980's and remaining lands maintained in natural grazing lands or in cultivated pastures (Solbrig, 1997; Viglizzo et al., 2001) (Fig. 4.13). The widespread adoption of farm machinery in the 1970's initiated a trend in farm consolidation and the subsequent increase in mean farm size and decrease in farm numbers in the Pampas (Fig. 4.14), mostly stemming form the loss of small farms (Fig. 4.15). Despite this, and some changes stemming from market demand, governmental policies, and technological improvements, the management of agricultural lands in the region generally remained a low-input/low-technological (little or no use of inorganic fertilizers, pesticides, irrigation, concentrated cattle feed, or high yield crop varieties) practice until the late 1980's (Solbrig, 1997; Solbrig and Viglizzo, 2000; Viglizzo et al., 2001).

The economic reforms of the 1990's initiated large scale changes in agricultural management in the Pampas. These changes are particularly evident in the area, yield, and/or distribution of the regions principal row crops; corn, wheat, sorghum, sunflower, and soybeans. Yields of all these crops increased dramatically from 1970 to 2004 (Fig. 4.16). Corn, sorghum, and soybean, however, showed the greatest rate of increase in yields after 1990, with the biggest increase in soybean yields occurring after 1995 correlated with the introduction of GM soybeans (Fig. 4.16).

The area planted in corn, wheat, and sunflower, although variable, has remained relatively stable since 1970, however, the area in sunflower has declined since the late 1990's (Fig. 4.17). Over the same period sorghum has undergone a downward trend while soybean increased in the area planted by greater than 3 orders of magnitude (Fig. 4.17). Although the expansion of soybeans in the Pampas has occurred over several decades, it greatly accelerated starting in 1990, illustrating the impacts of trade liberalization and intensification of management on pampaean agroecosystems.

The intensification of agriculture in the Pampas has not only shifted the emphasis on fewer crops, while increasing yields, but has also led to regional polarization in land use. The increased use of no-till technology and the introduction of GM cultivars has made continual cropping more common in many areas of the Pampas, resulting in less area laying fallow or managed for pasture. Subsequently, many farms have abandoned mixed row crop and grazing strategies and have specialized in one land use or the other depending upon the suitability of the land. Furthermore, as row crop agriculture has become more intensive, cattle raising has become more dependent upon supplemental feed and feed lots, which in turn reduces the need for pastures and forage crops.

Over the period from 1988 to 2002, representing pre- and post-market reforms, the area of annual crops increased by over 38% in the Pampas while the area in annual and perennial forage crops decreased by 32.1% and 22.8%, respectively over the same period (Table 4.1). Moreover, the variations among provinces illustrate that the magnitude of these trends vary spatially (Table 4.1). Of particular note is the large reduction in the proportional area of both annual and perennial forage crops from the western and northern extents of the Pampas (Fig. 4.18, Fig. 4.19).

The changes in area of forage crops were also related to changes in the principal row crops. Most obvious was the large expansion of soybeans in the northern half of the Pampas, and to a lesser extent in the south-central region (Fig. 4.20). Wheat also illustrated a significant expansion in the northern half of the Pampas, associated with its double cropping with soybean, and in the austral Pampa (Fig. 4.21). Conversely, the area in sunflower decreased in the northern and central Pampas, with some moderate increases in the southwest, while sorghum decreased throughout the Pampas, particularly in the north and west (Fig. 4.22, Fig. 4.23). The area planted in corn showed little change between the two years (Fig. 4.24).

Cattle densities changed little throughout the Pampas despite the large increases in the area of annual crops and the decreases in forage crops (Fig. 4.25), with decreases in the provinces of Buenos Aires, Córdoba, and Entre Ríos offset by increases in La Pampa, Santa Fe, and San Luis (Table 4.2). The number of land holdings with cattle, however, decreased substantially between the two years (Table 4.2). In comparison, the densities of sheep decreased dramatically, particularly in the southern half of the region (Fig. 4.26) stemming from the decrease in sheep numbers and properties with sheep throughout the Pampas (Table 4.3). These differences suggest the intensification of cattle production through the concentration of cattle, while sheep raising declined as the availability of pasture land decreased.

4.2 The expansion and intensification of the agricultural frontier in northern Argentina

The lucrative nature of row crop agriculture, particularly based upon soybeans, combined with new agrotechnologies has facilitated a large-scale changes in land use in northern Argentina (Gasparri and Parmuchi, 2003; LART-FAUBA, 2004; Grau et al., 2005b). Through deforestation there has been a large expansion in the area of row crops in the region and, moreover, the management of existing agricultural land has been become more intensive, as has livestock production.

The expansion of the northern agricultural frontier has been most evident in the 6 north Argentinean provinces of Chaco, Formosa, Jujuy, Salta, Santiago del Estero and Tucumán (Fig. 4.27) occurring mostly from deforestation of subtropical Yungas and Chaco forest (Gasparri and Parmuchi, 2003; Montenegro et al., 2003a; Brown and Malizia, 2004; LART-FAUBA, 2004; Grau et al., 2005b). Yungas forest, or Selva Tucumano-Boliviana, occupies a relatively small area (3 726 835 ha) in Argentina mostly in Jujuy, Salta, and Tucumán (3,665,436 ha) (Proyecto Bosques Nativos y Áreas Protegidas, 2002). Chaco forest covers a large portion of northern Argentina (23,367,984 ha) along a humid to semi-arid gradient from east to west, the majority of which (20,480,577 ha) is in the provinces of Chaco, Formosa, Salta and Santiago del Estero (Proyecto Bosques Nativos y Áreas Protegidas, 2002, 2003) (Fig. 4.28; 4.29).

The Chaco and Yungas have been under development pressures and suffered degradation for decades, if not centuries (Barsky and Gelman, 2001; Boletta et al., 2006; Brown and Malizia, 2004; Bucher and Huszar, 1999; Grau and Brown, 2000; Montenegro et al., 2003a; Zak et al., 2004). The eastern, more humid Chaco saw relatively large scale conversion to agriculture starting in the early part of the 20th century (Barsky and Gelman, 2001), where as the southern, more temperate Chaco witnessed large scale conversion and degradation for row crop agriculture and grazing starting in later half of the 20th century (Boletta et al., 2006; Zak et al., 2004). Over the last 15 years the rate of conversion of Yungas and Chaco forest to agriculture has greatly accelerated so that the annual rates of deforestation in these areas are some of the highest in the world (Gasparri and Parmuchi, 2003; Montenegro et al., 2003a; Gasparri, 2004; Manghi et al., 2004a,b; Parmuchi et al., 2004). Although deforestation has occurred throughout the distribution of both forest types it has in recent years been most prevalent in the northwestern (Province of Salta) and southeastern (Province of Santiago del Estero) Chaco and the premontane Yungas in the provinces of Jujuy, Salta, and Tucuman (Fig. 4.30) (Brown and Malizia, 2004; Manghi et al., 2004a,b; Parmuchi et al., 2004; Grau et al., 2005a).

In northwestern and north-central Argentina row crop agriculture remained mostly confined to the east of the province of Chaco and the pre-montane region of Tucumn, Jujuy, and Salta into the 1980's and livestock grazing (cattle, sheep, goats) remained an extensive practice, primarily on natural rangeland, with cattle densities concentrated in the humid Chaco in the eastern portions of the provinces of Chaco and Formosa and sheep and goats most common in more arid regions of Chaco forest (INDEC, 1988; LART-FAUBA, 2004). Starting in the late 1980's rapid deforestation in the region resulted in a 44% increase in cultivated land, mostly in the southwest of the province of Chaco, the east of Santiago del Estero, and the central portion of the province of Salta (INDEC, 1988, 2002) (Fig. 4.31). Associated with agricultural expansion have been significant shifts in the primary cultivars in the region, particularly to soybeans (Grau et al., 2005a; INDEC, 1988, 2002).

Although the increase in the area of soybean has been the most dramatic across the region, the changes in crop diversity have manifested itself differently among provinces (Fig. 4.32). The provinces of Chaco, Salta, Santiago del Estero, and Tucumán saw significant increases in the area of soybean starting in the later half of the 1990's. During the same period Salta, Santiago del Estero and Tucumán also exhibited a correlated increase in the area of wheat, which is expected since wheat is often rotated with soybeans in double cropping systems given the proper growing conditions. Cotton, formerly an important crop in Chaco, Formosa, and Santiago del Estero, declined markedly in those provinces, particularly during the late 1990's (Fig. 4.32).

In Jujuy and Salta beans grew in importance starting in the 1970's and showed marked increases during the 1990's, particularly in Jujuy, which more recently have been substituted by soybeans (Fig. 4.32) (LART-FAUBA, 2004; Grau et al., 2005b). Corn remained a relatively minor crop throughout the region and showed little variation in the area planted, while sunflower was uncommon, except in the Chaco where it declined in area until 1999 when, combined with soybean, it replaced cotton (Fig. 4.32).

One of the key drivers in facilitating the spread of row crop agriculture in northern Argentina has been abnormally high precipitation that has allowed soybean cultivation to become feasible over an increased area (Grau et al., 2005a). Also, as in the pampas an important component of modern row crop management in northern Argentina has been GM crops and no-till technology, which combined with judicious herbicide use for weed control, eliminates the need for tilling. By causing little soil disturbance through farm operations, no-till management maintains soil moisture, which is further enhanced by maintaining organic matter on the soil surface (Peiretti, 2001), and subsequently has been an important factor in facilitating the expansion of cultivation in northern Argentina.

Since areas of the eastern and westernmost Chaco have been exploited for row crop management for decades to centuries, it would be expected that the most accessible and productive lands have been those exploited up until the recent expansion, and that the expansion of cultivated land is taking place on more marginal lands that are less productive (*sensu* Ricardo 1817). Soybean yields from the northern provinces have been consistently lower compared to those from the pampas since 1990 and appear to be more disparate with time, suggesting that the land being converted for soybean cultivation in northern Argentina is less than optimal (Fig. 4.33). This is of significance because, if this is the case, proportionally more land in the Chaco needs to be cultivated compared to more fertile land to achieve equivalent total production.

The intensification of row crop agriculture in Argentina has changed livestock husbandry practices by concentrating livestock and use of supplemental feed. This process at the national level has increased the importance of northern Argentina as a livestock producer as pastures in the more fertile pampean region have been increasingly converted to row crops and cattle managed in feed lots. The effect of intensification on livestock husbandry is evident in northern Argentina in the large increases in the area of planted perennial pastures associated with expansion of row crop agriculture (4.34).

The large increase in planted pastureland in northern Argentina has allowed for an increase in cattle numbers in chaco forest (Chaco, Formosa, Salta, Santiago del Estero), despite the high rates of deforestation and expansion of row crops, where as in areas of yungas forest (Jujuy, Tucumán) cattle numbers remained the same or declined (Fig. 4.35). The same pattern can be seen in the number of sheep and goats, with the exception of Santiago del Estero where sheep numbers declined (Fig. 4.35). When viewed on a whole, based on Animal Units (1 cow = 10 sheep or 15 goats), overall livestock densities have increased within the provinces dominated by chaco forest (Chaco, Formosa, Salta, Santiago del Estero) and have remained constant or slightly declined in those provinces with a high proportion of Yungas forest (Jujuy, Tucumán) (Fig. 4.35).

4.3 Summary

Argentine agriculture has traditionally been dominated by livestock production and mixed farming systems, largely as a function of national economic policies imposing high export tariffs on cereal and oil crops. This emphasis, particularly in the grassland ecosystem of the Pampas favored either extensive livestock production or mixed row crop grazing systems, both which are relatively favorable for wildlife conservation in grassland systems. A switch to a free market economy in the early 1990's and overseas demand for soybeans facilitated the development of an export-based agricultural sector which, incorporated and embraced imported genetically-modified crops and no-till agricultural technology as a fundamental component. This change in production goals and technology has led to a rapid and large-scale change in the conservation utility of agricultural land in Argentina that is similar to other temperate regions of the world (Chapter 3).

Aside from increasing the area of land in row crop production and reducing landscape diversity, the recent changes that have occurred in agricultural management in Argentina have also greatly increased the use of inorganic fertilizers and pesticides, as well as intensified livestock production, particularly in former mixed-cropping areas. Moreover, areas formerly utilized for extensive cattle production and considered marginal or unsuitable for row crops (semi-arid Chaco, Interior Pampa) are increasingly being converted for that purpose.

The implications for wildlife in the agricultural systems of Argentina are profound since former mixed-cropping areas are increasingly dominated by row crop agriculture that is more intensively managed and rangeland areas are increasingly being converted to intensive row crop agriculture, which equates to both a decrease in habitat availability and quality over a large portion of the country. The changes are particularly acute in the Pampas since these ongoing process continue to remove and degrade areas serving as refugia within this system. Moreover, the deforestation of the Chaco and Yungas is occurring at a rate that surpasses most areas in the world and possess a sever threat for the conservation of the relatively high biodiversity supported by these forest.

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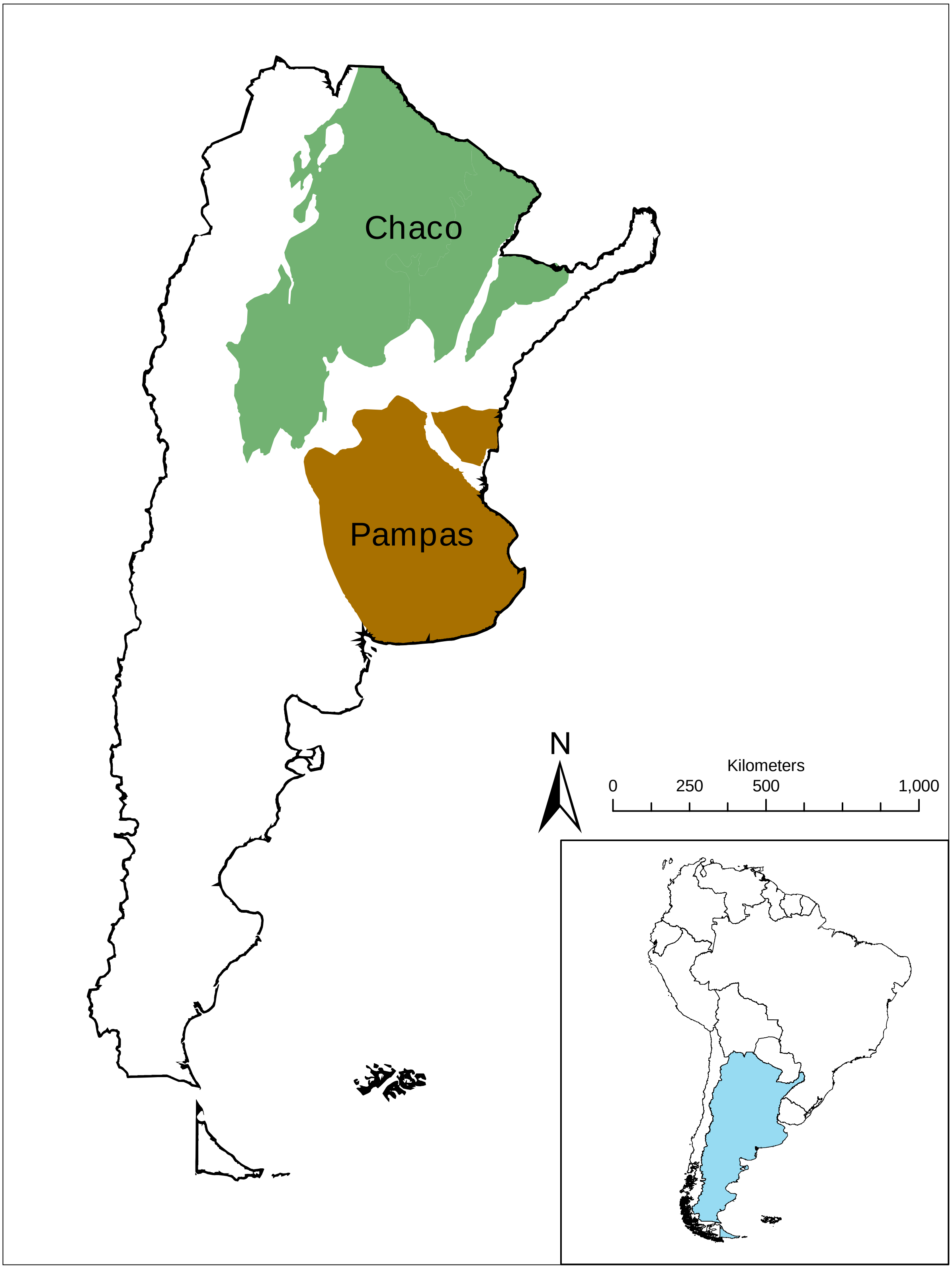

Figure 4.1: Map of Argentina showing the distribution of the Pampas and the Chaco. Based on data from Olson et al. (2001).

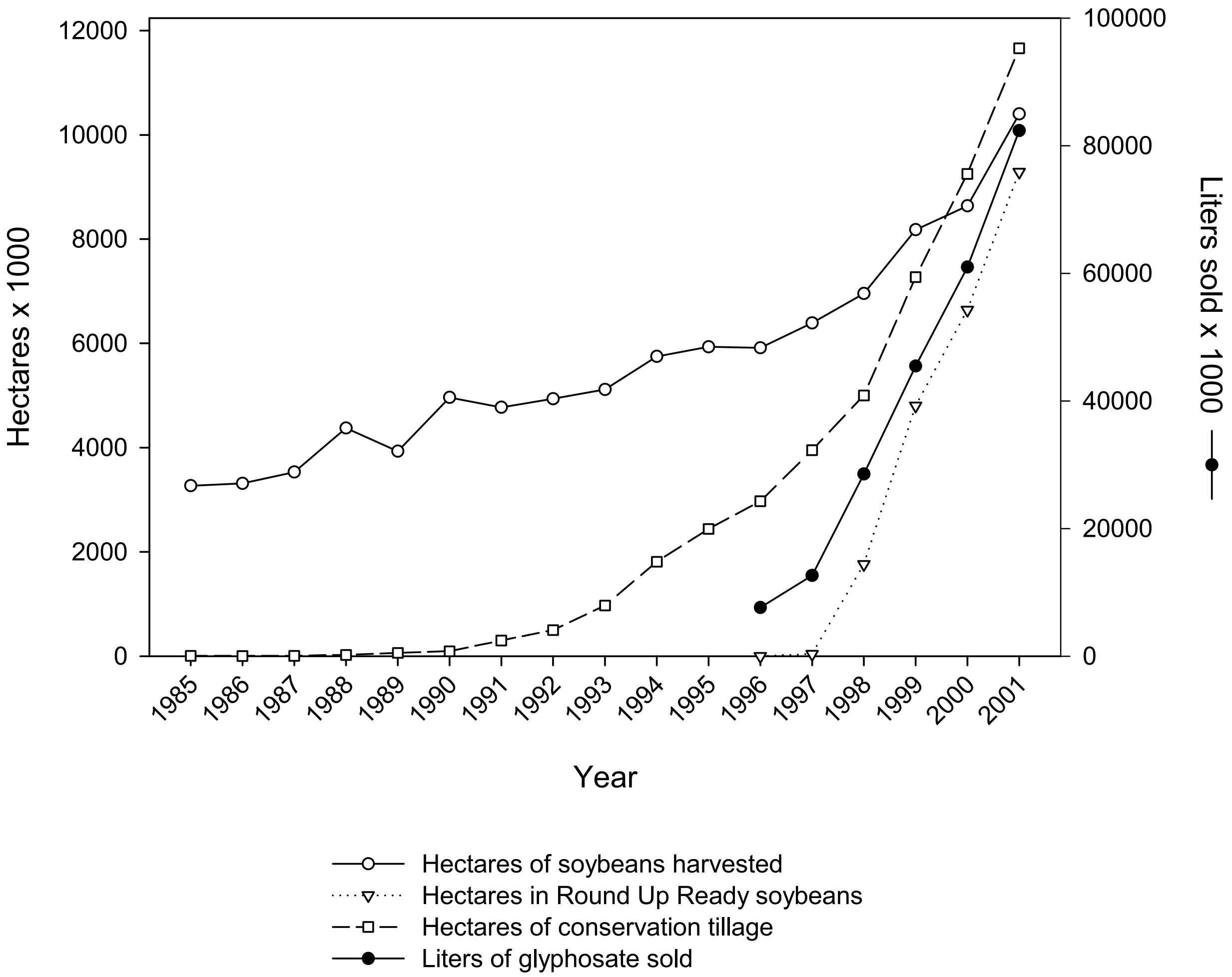

Figure 4.5: Use of zero tillage, Round Up® ready soybeans, and glysophate herbicide in Argentina during 1984-2002 (AAPRESID, 2006; Qaim and Traxler, 2005).

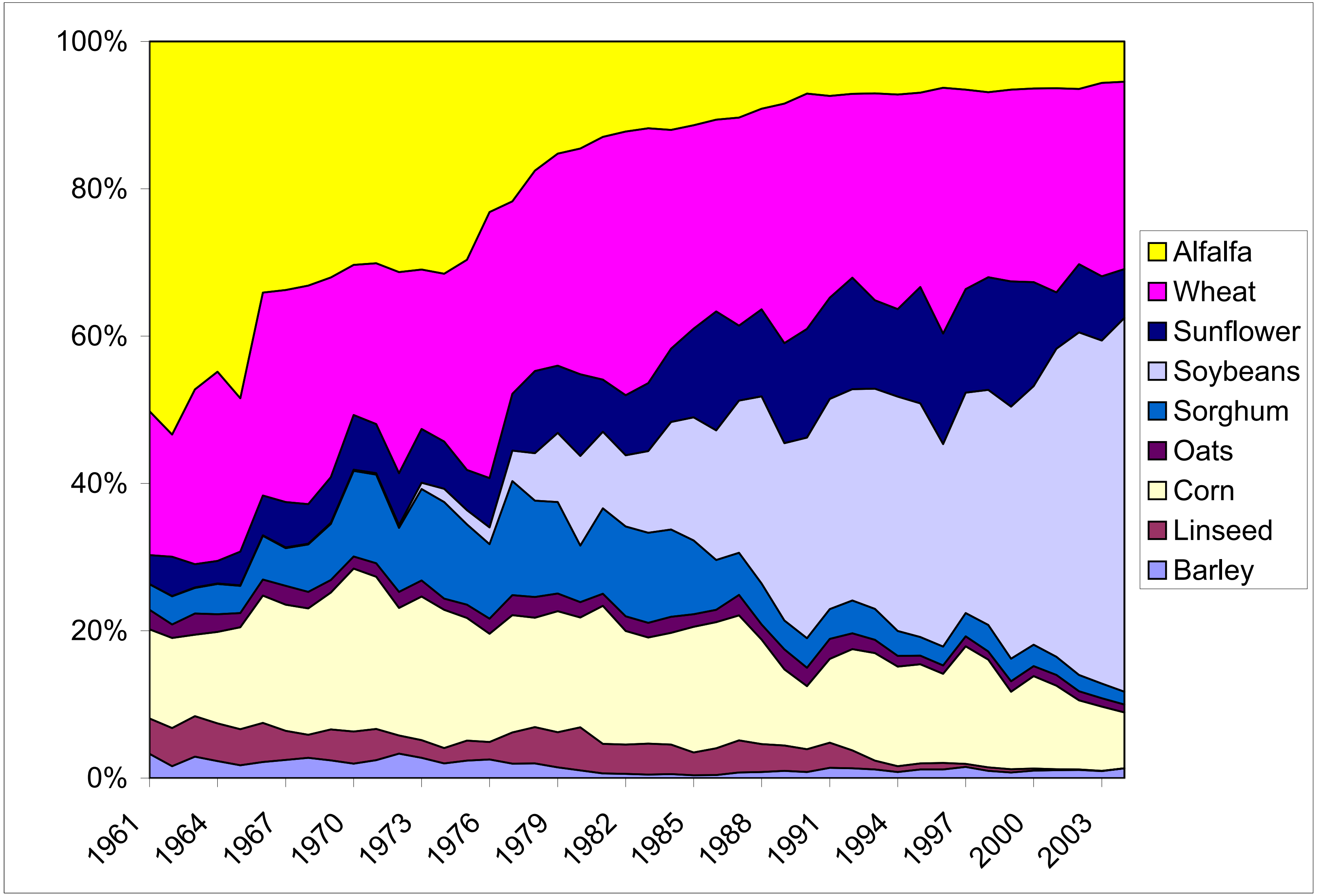

Figure 4.6: Proportional area of principal rows crops harvested in Argentina during 1961-2004 (FAOSTAT, 2006).

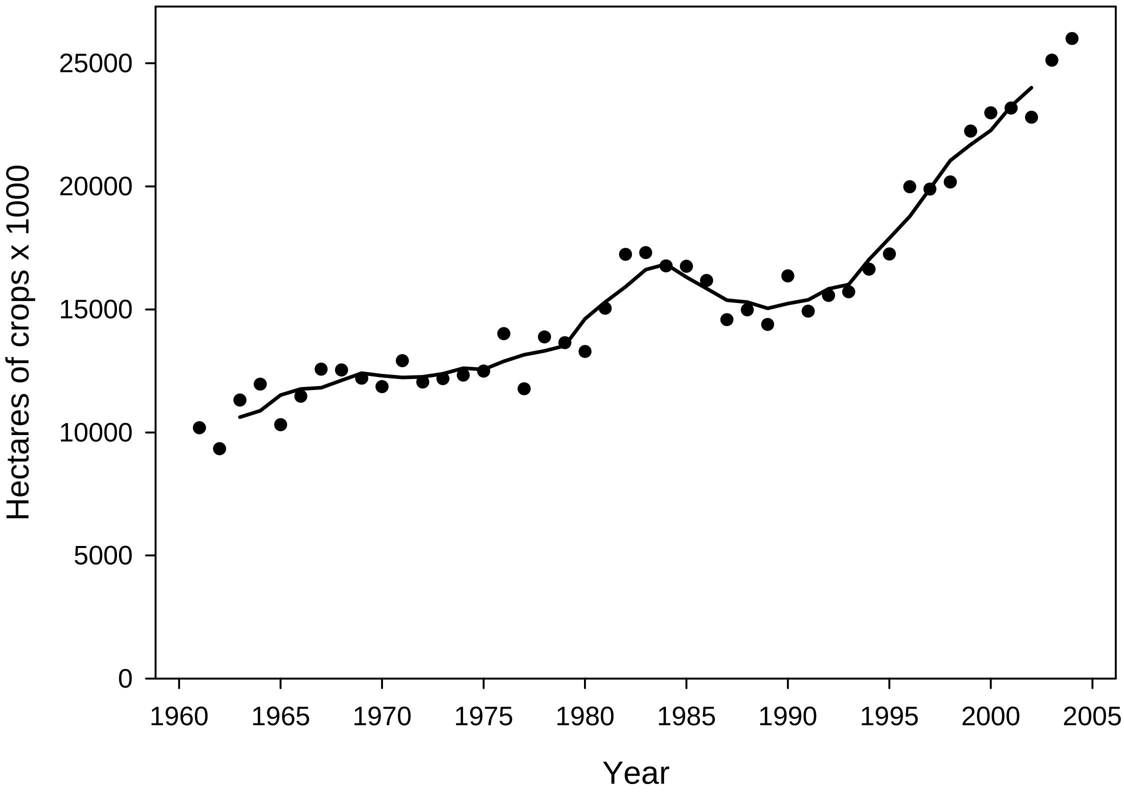

Figure 4.7: Area of in cereals (barley, corn, oats, sorghum, wheat) and oil crops (soybean, sunflower) in Argentina during 1961-2004 (FAOSTAT, 2006). Line represents the 5-year running average.

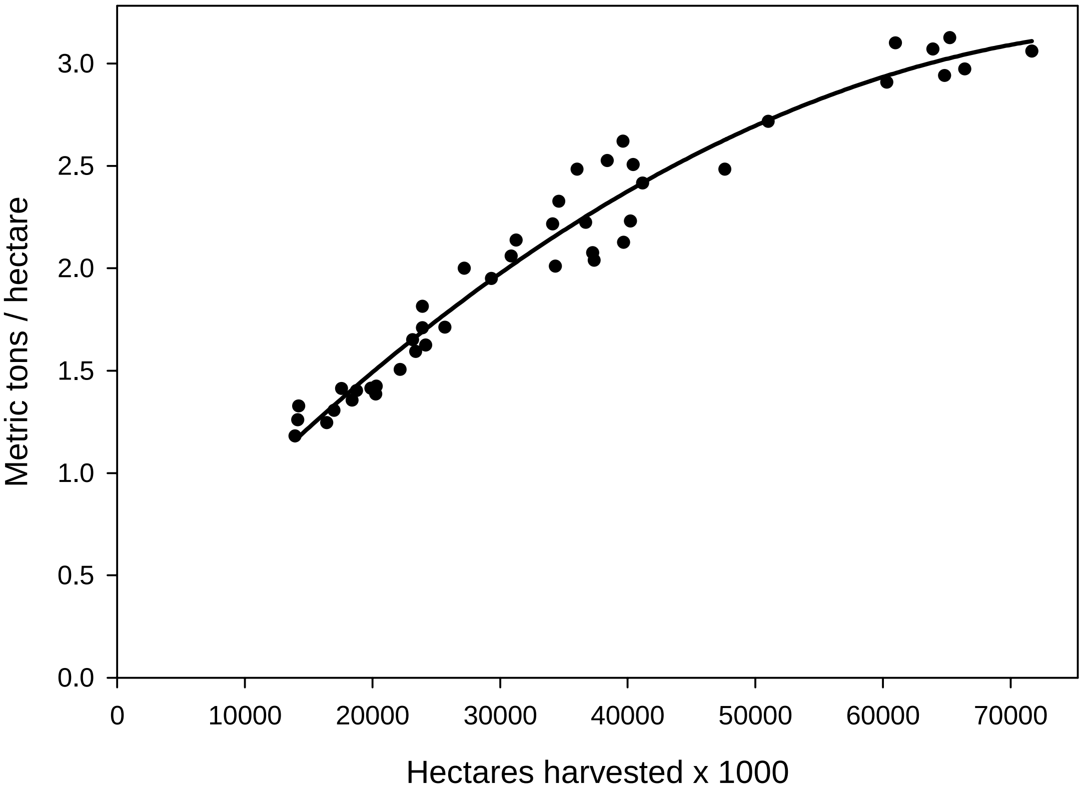

Figure 4.8: Yields of cereals (barley, corn, oats, sorghum, wheat) and oil crops (soybean, sunflower) in Argentina during 1961-2004 (FAOSTAT, 2006). Trend line represents best fit (r^2=0.96).

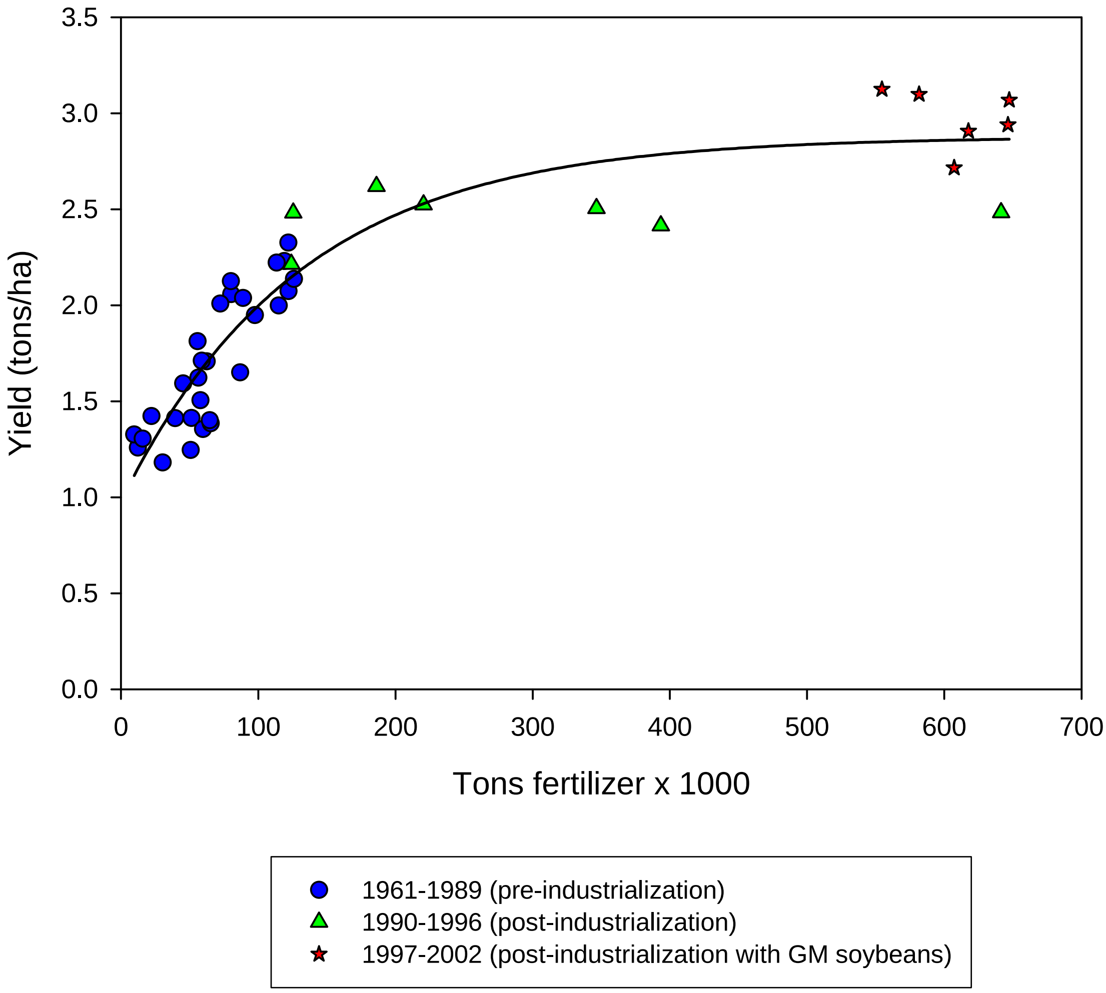

Figure 4.9: Yields of cereals (barley, corn, oats, sorghum, wheat) and oil crops (soybean, sunflower) in relation to fertilizer consumption in Argentina during 1961-2004 (FAOSTAT, 2006). Time periods represent the periods before industrialization of agriculture (pre-1992), post-idustrialization (1992-1996), and post-industrialization and the introduction of GM soybean. Fertilizer consumption for these crops was considered to be 75% of the national total based on Food and Agriculture Organization of the United Nations (2004). Line is best fit, r^2=0.87.

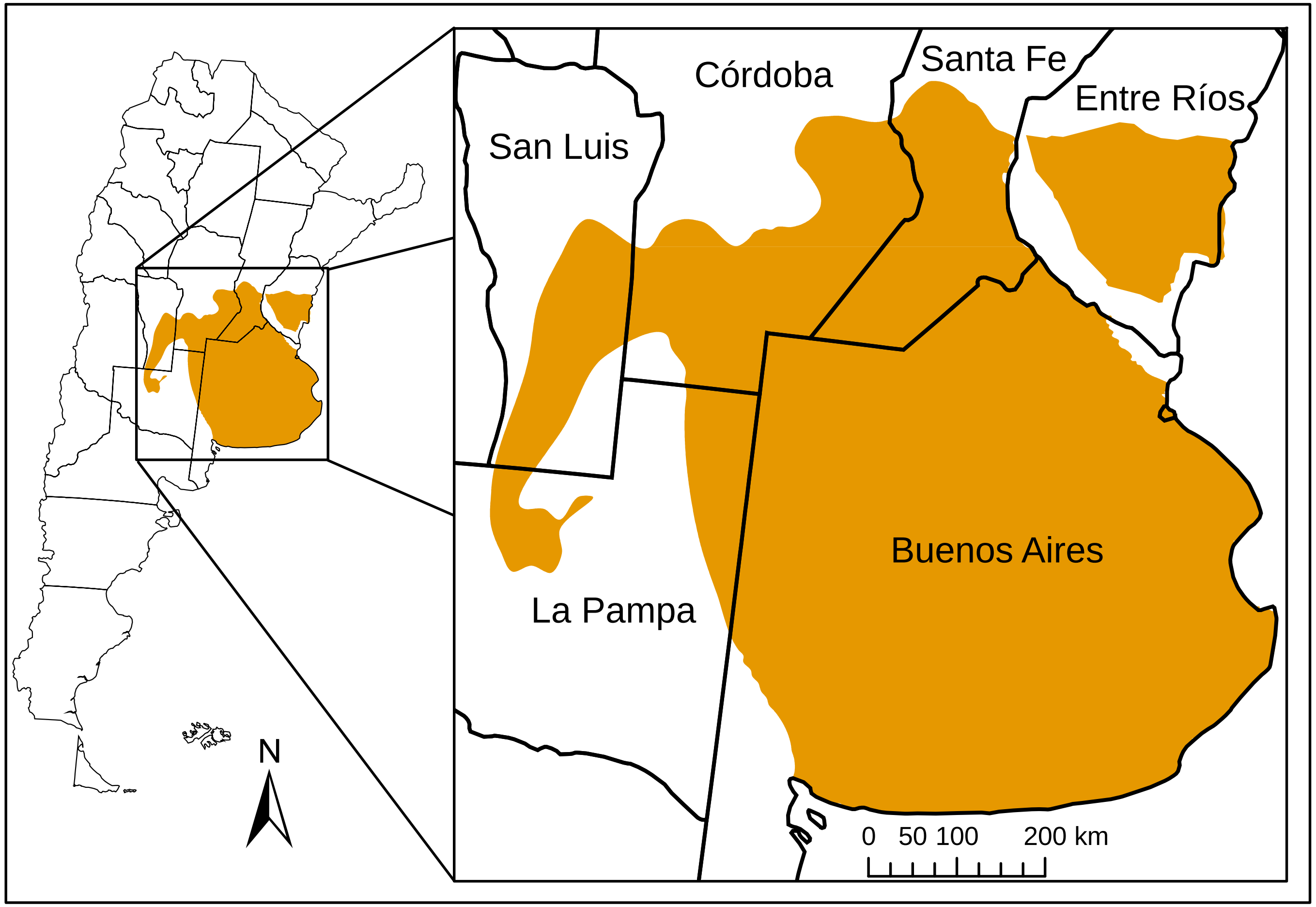

Figure 4.10: Distribution and location of the Pampas showing province boundaries. After Soriano et al. (1991).

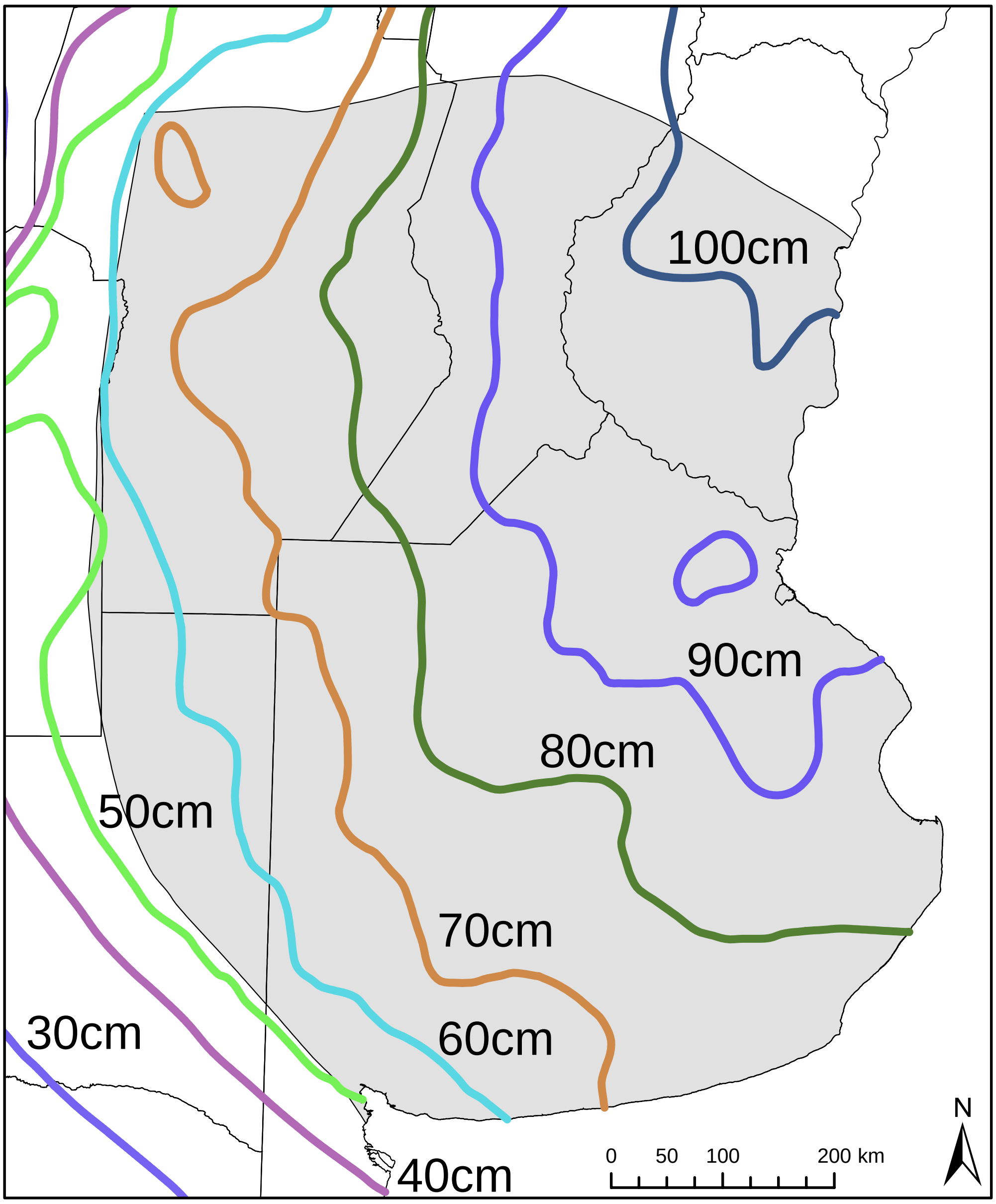

Figure 4.11: Annual mean precipitation isopleths for the principal row crop production region of the Pampas (INTA, 1995).

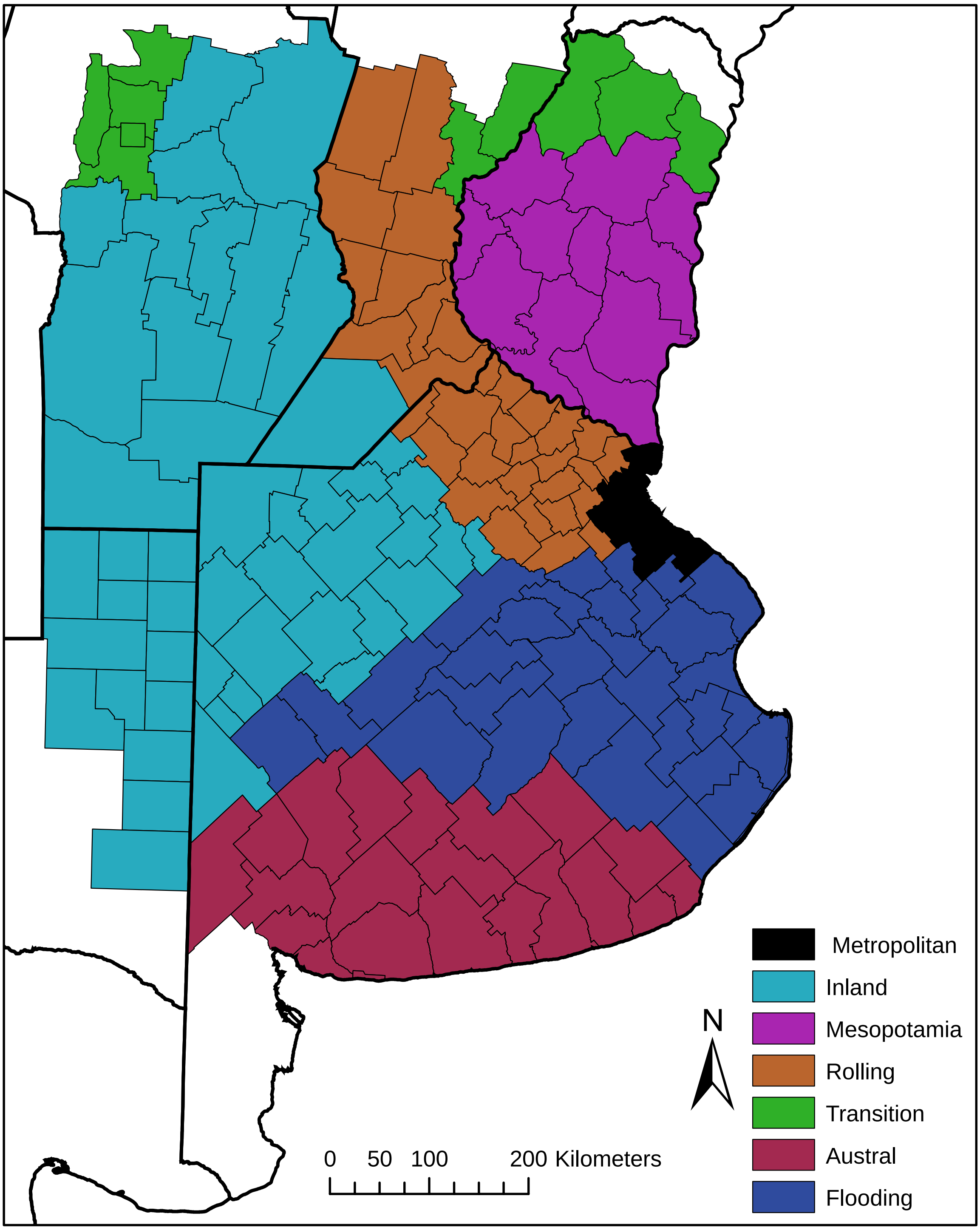

Figure 4.12: Regional subdivisions of the Pampas by district. District boundries from INTA (1995).

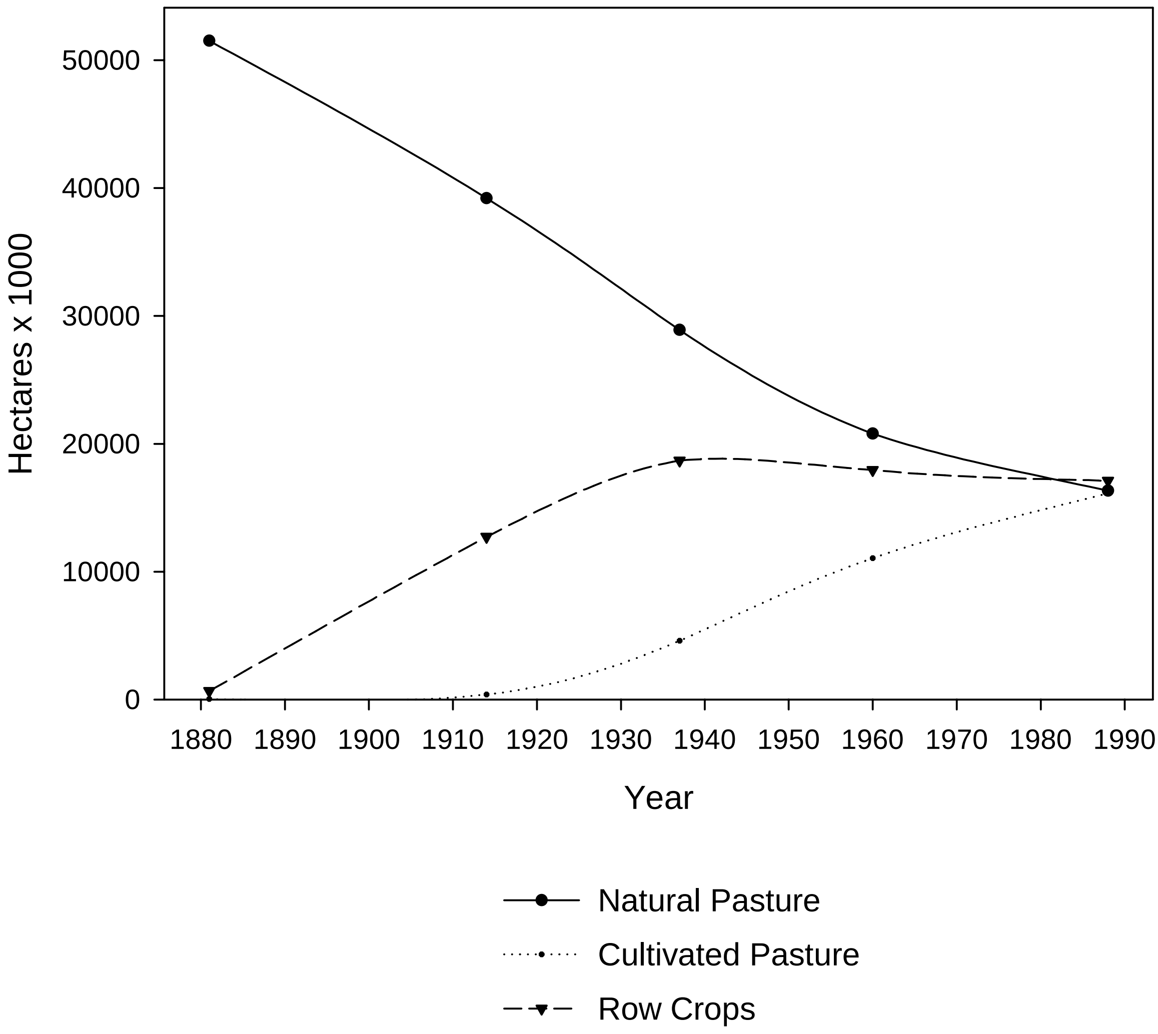

Figure 4.13: Area of the Pampas dedicated to row crops, grazing, and planted pasture during 1888-1988. Based on data from Viglizzo et al. (2001).

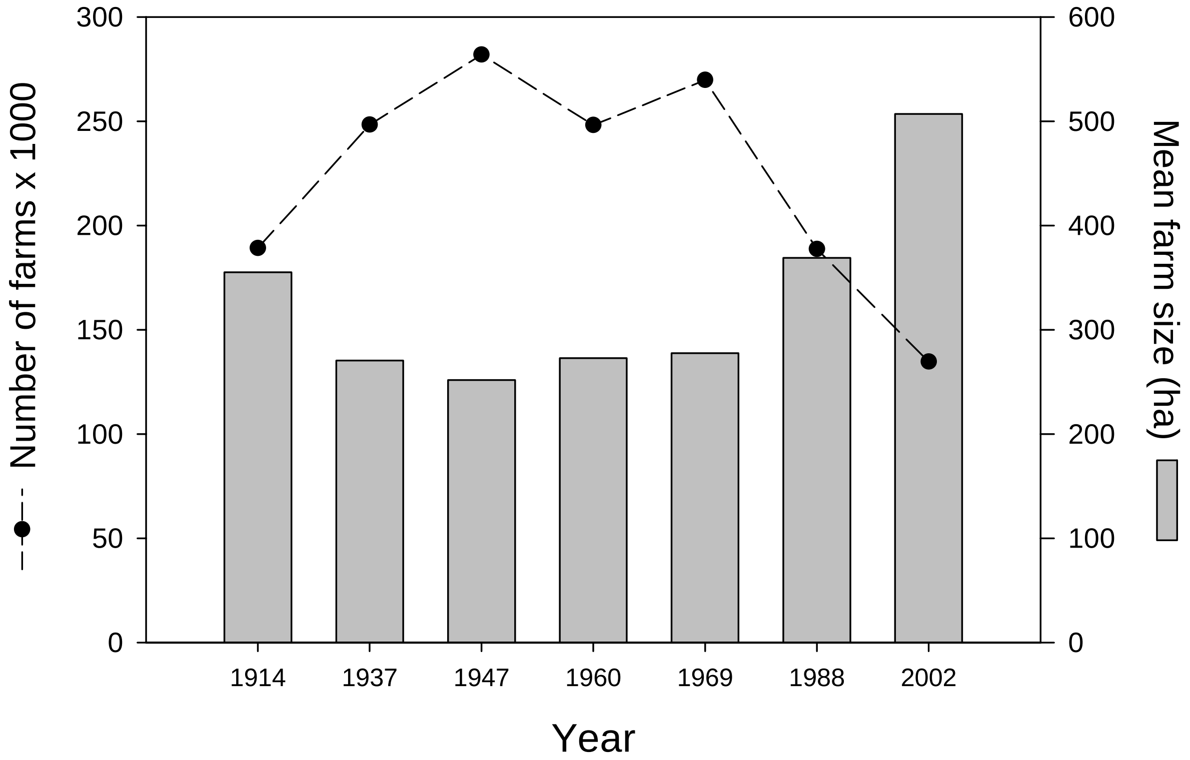

Figure 4.14: Mean farm size and number of farms in the provinces of Buenos Aires, Córdoba, Entre Ríos, La Pampa, and Santa Fe during 1914-2002 (INDEC, 1914-2002).

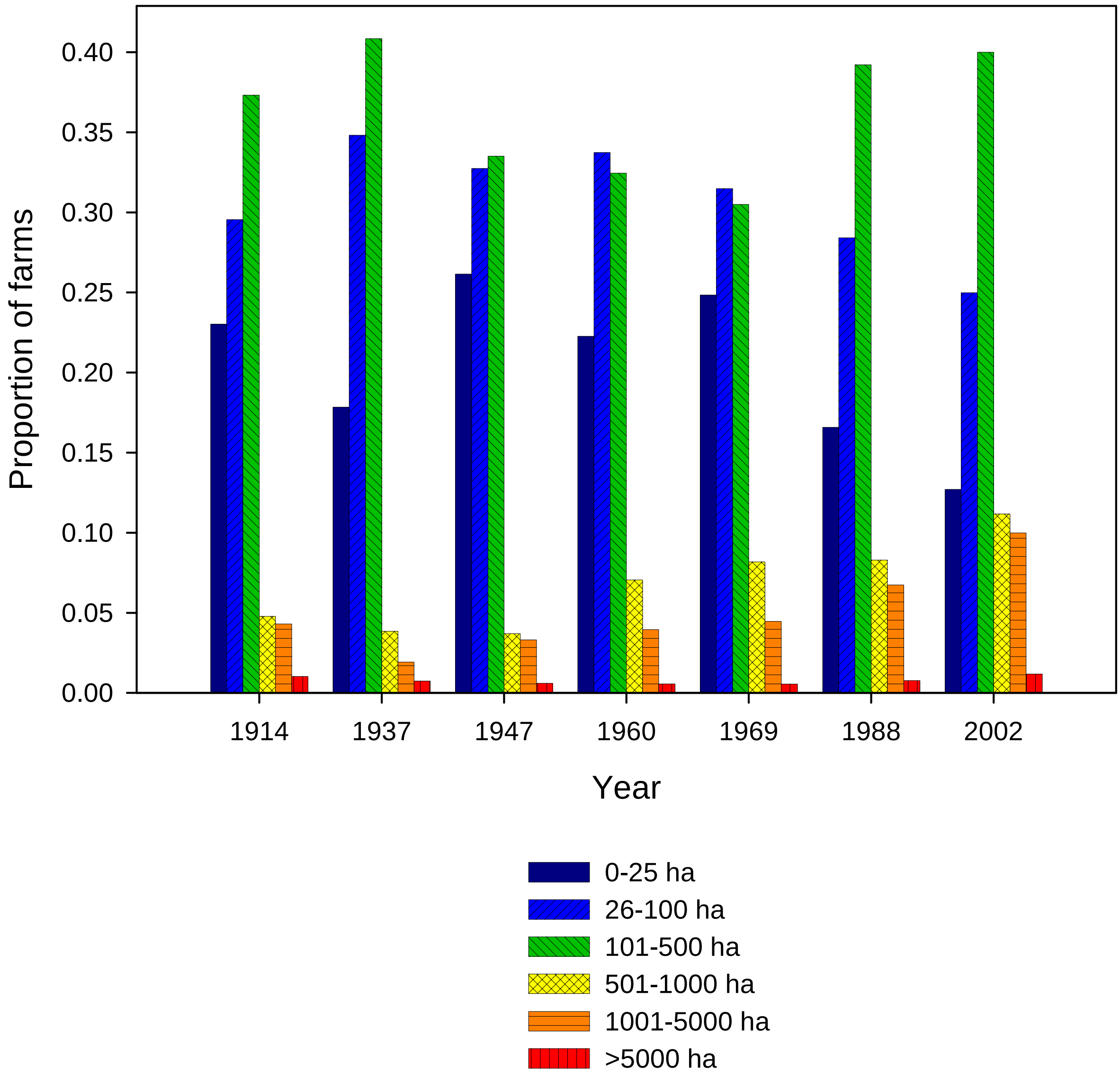

Figure 4.15: Proportion of farms by size class in the provinces of Buenos Aires, Córdoba, Entre Ríos, La Pampa, and Santa Fe during 1914-2002 (INDEC, 1914-2002).

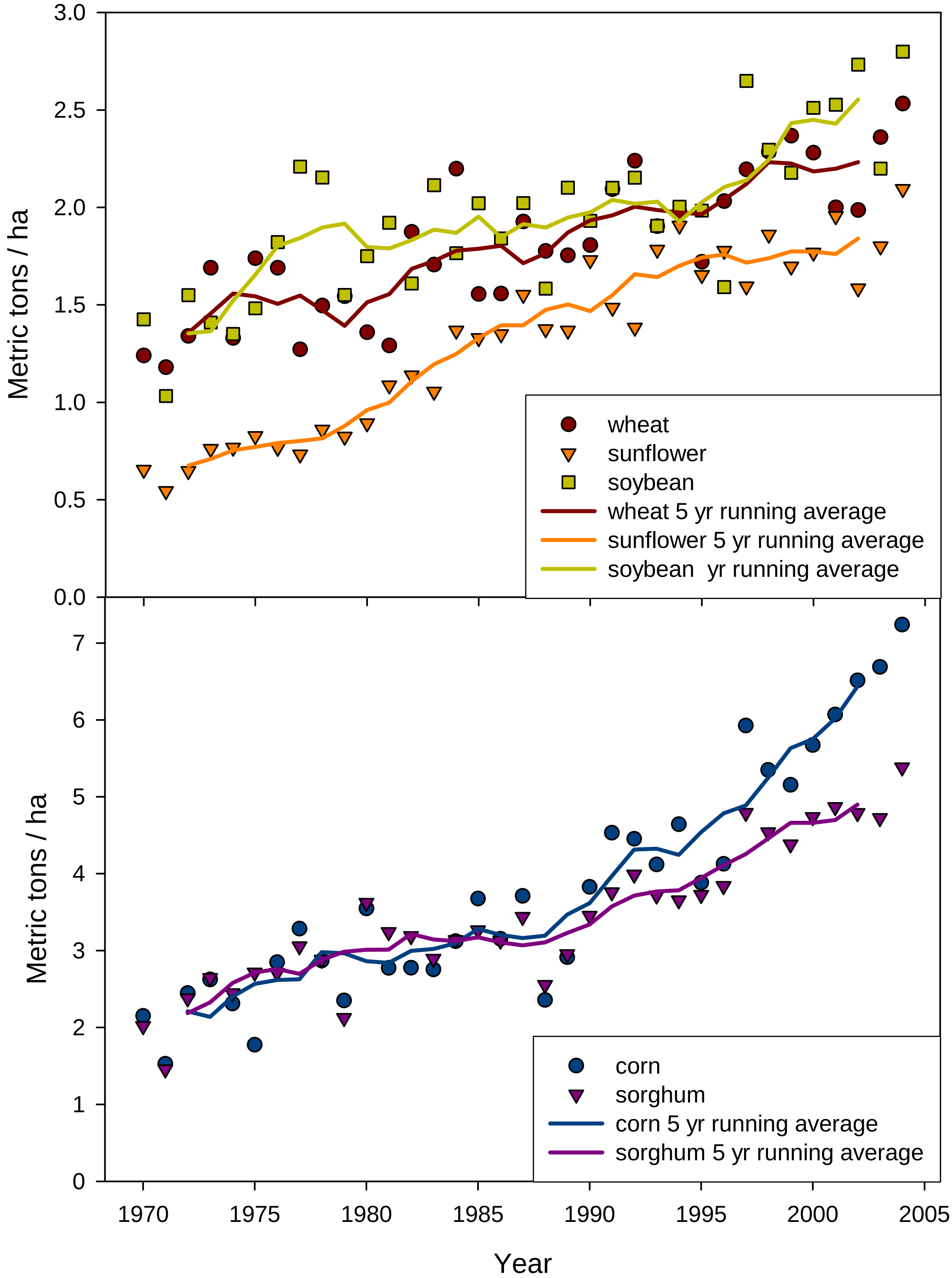

Figure 4.16: Yields of principal row crops from the provinces of Buenos Aires, Córdoba, Entre Ríos, La Pampa, and Santa Fe during 1970-2004 (SAGPyA, 2006).

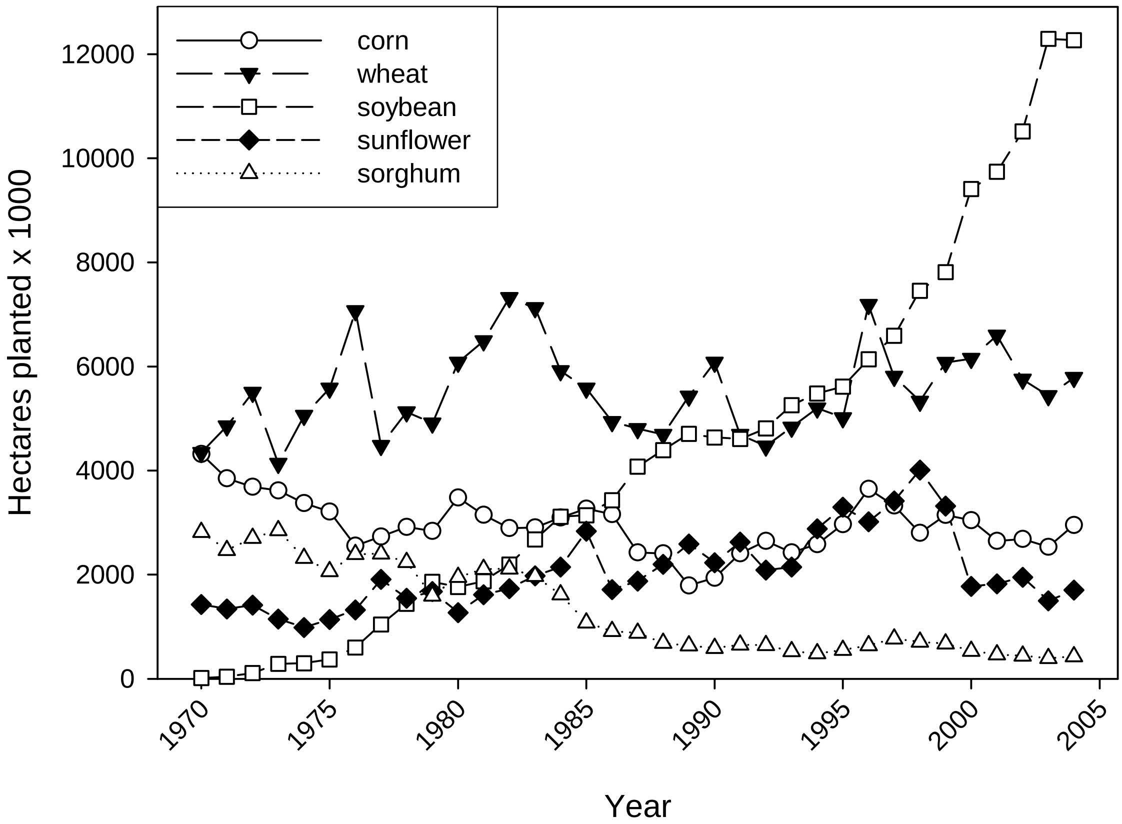

Figure 4.17: Area of principal row crops from the provinces of Buenos Aires, Córdoba, Entre Ríos, La Pampa, and Santa Fe during 1970-2004 (SAGPyA, 2006).

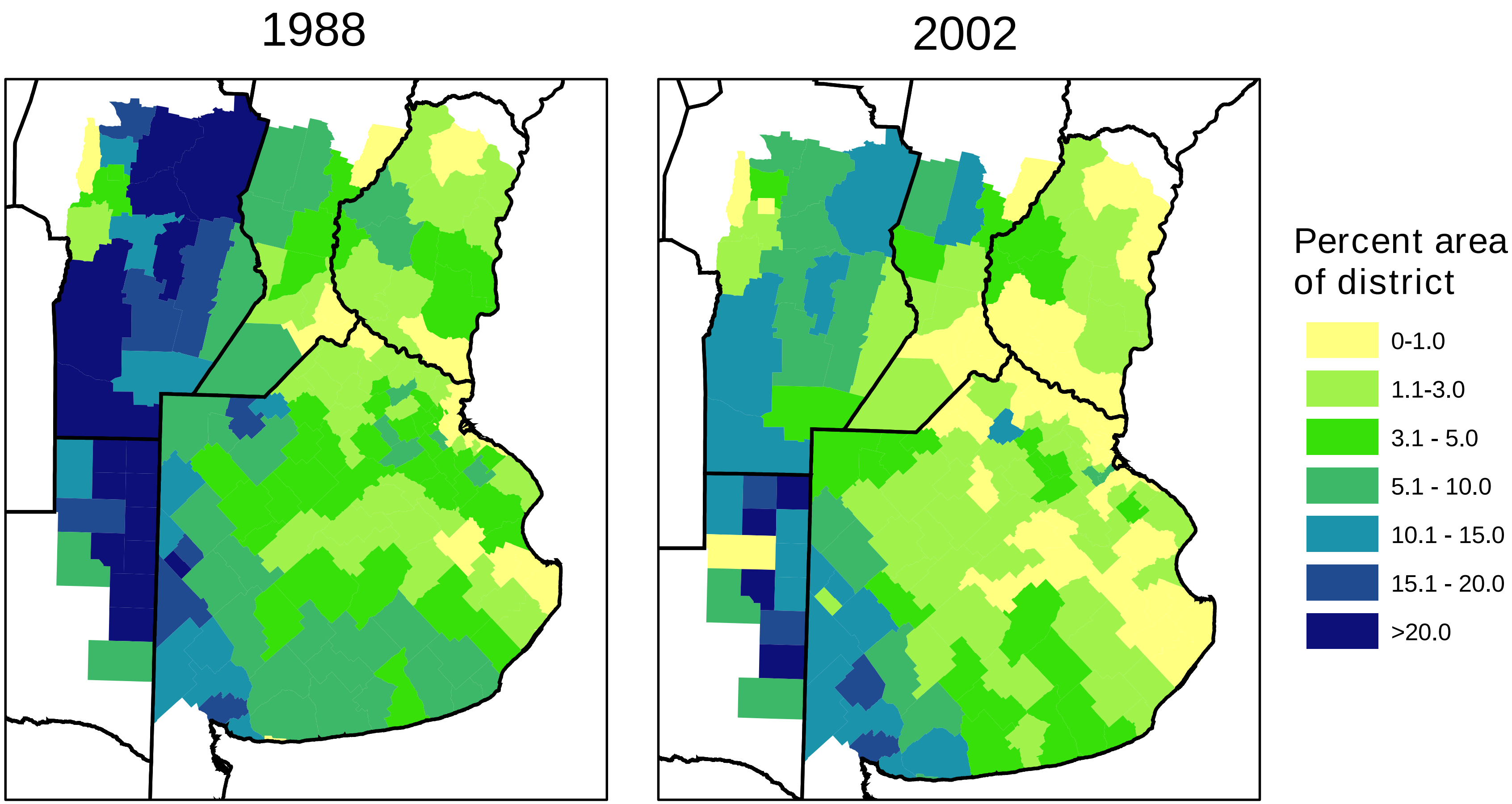

Figure 4.18: Proportional area by district in annual forage crops for 1988 and 2002 (INDEC, 1988, 2002).

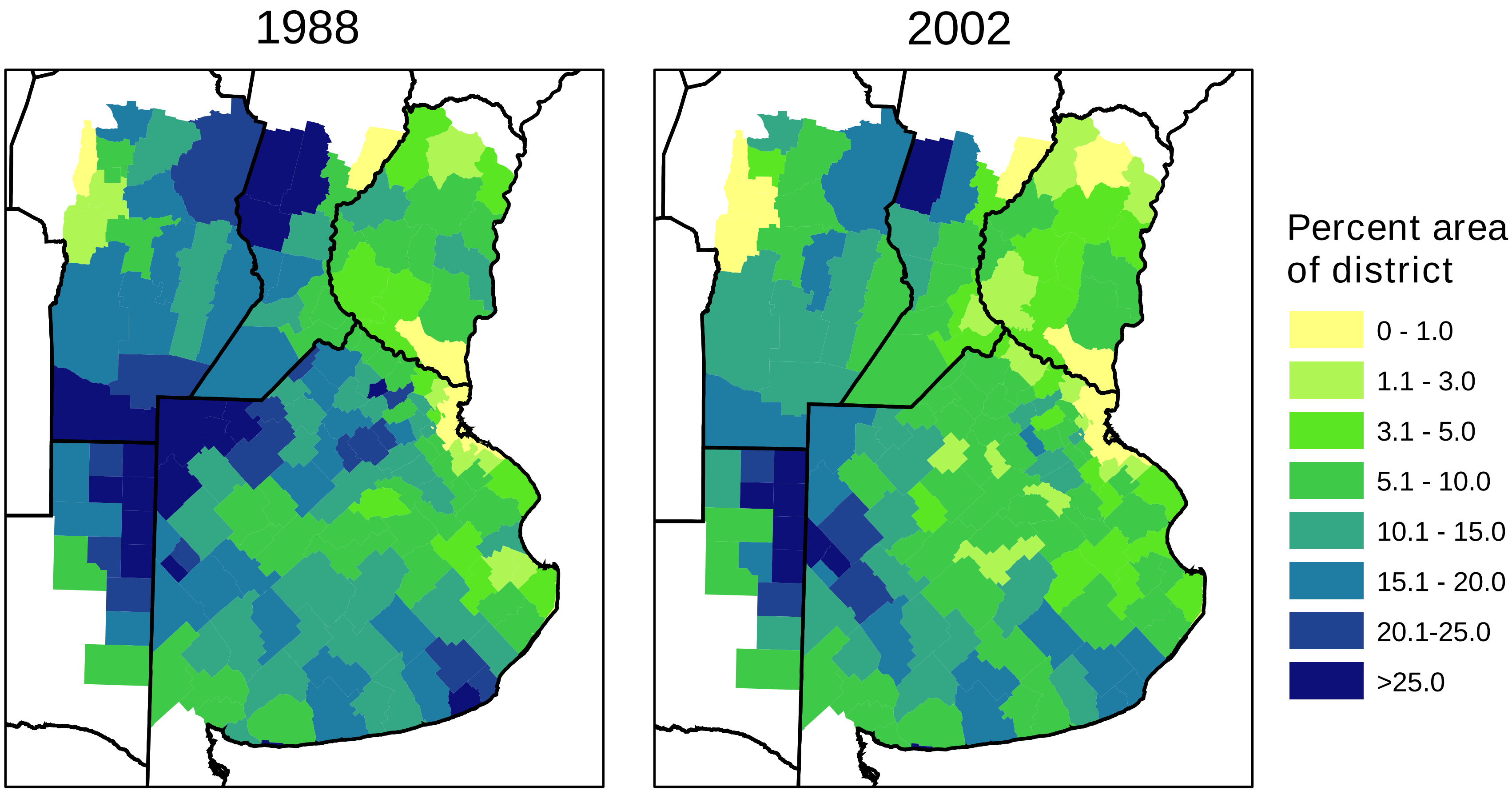

Figure 4.19: Proportional area by district in perennial forage crops for 1988 and 2002 (INDEC, 1988, 2002).

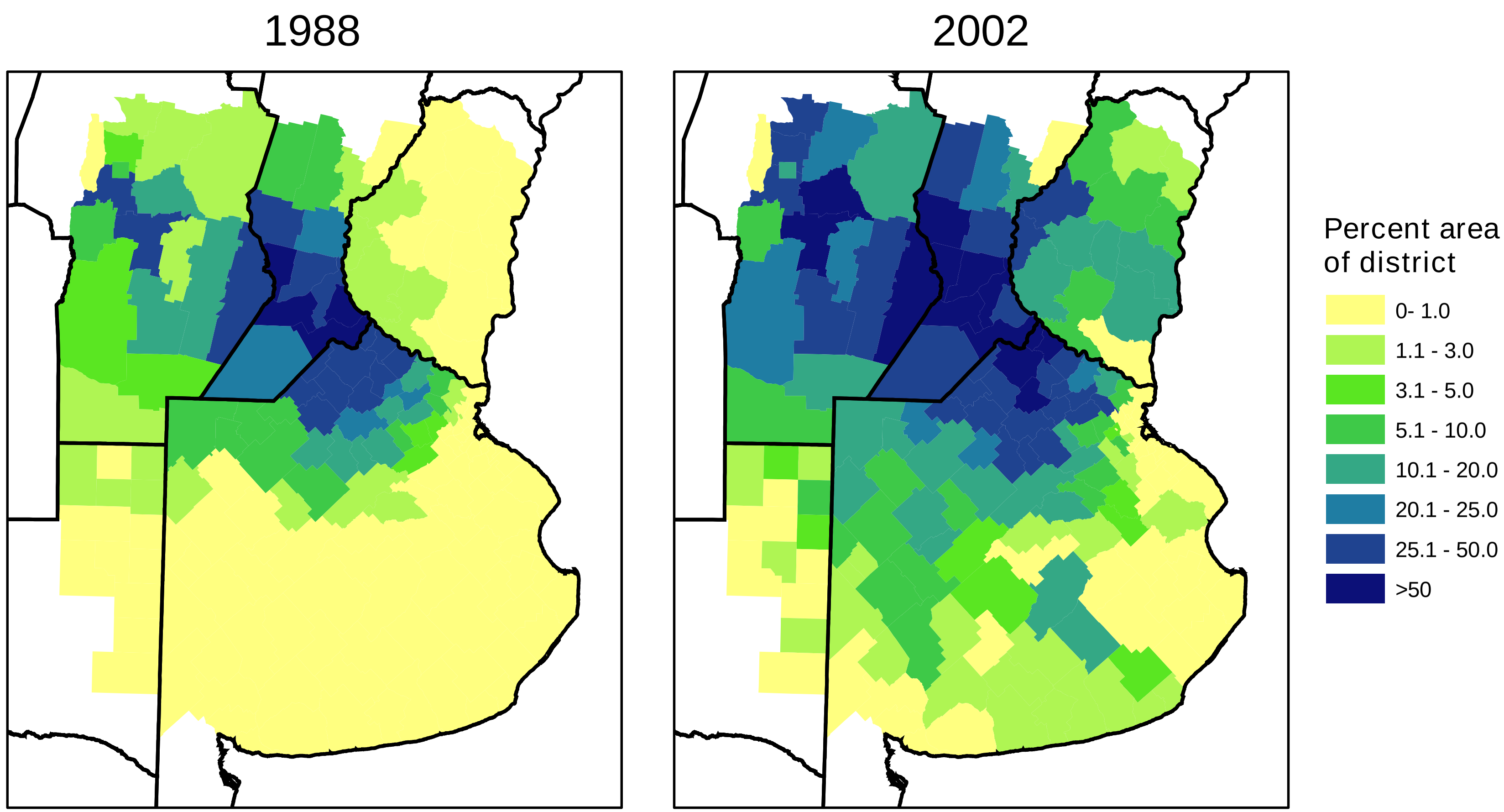

Figure 4.20: Proportional area by district in soybeans for 1988 and 2002 (INDEC, 1988, 2002).

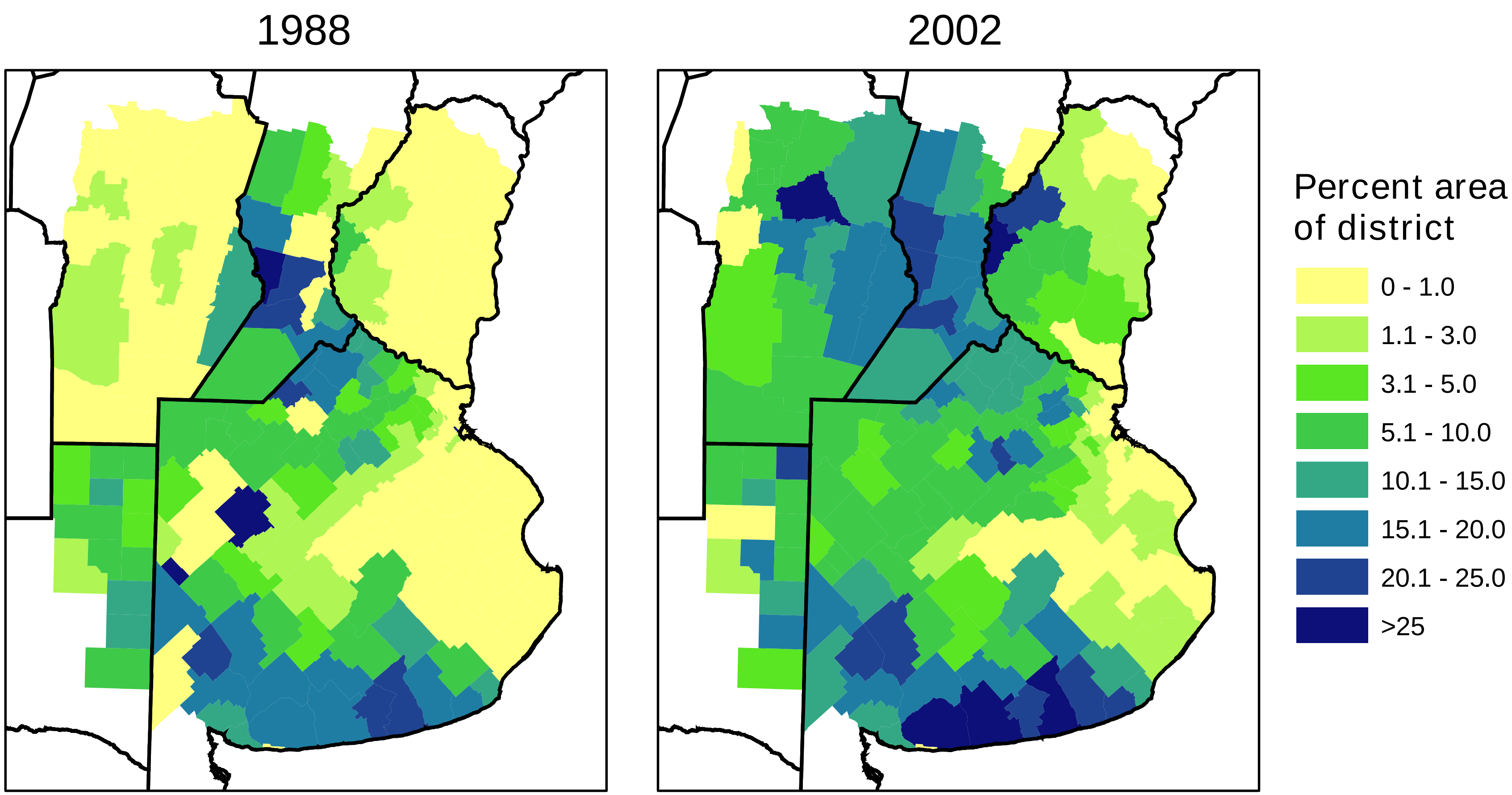

Figure 4.21: Proportional area by district in wheat for 1988 and 2002 (INDEC, 1988, 2002).

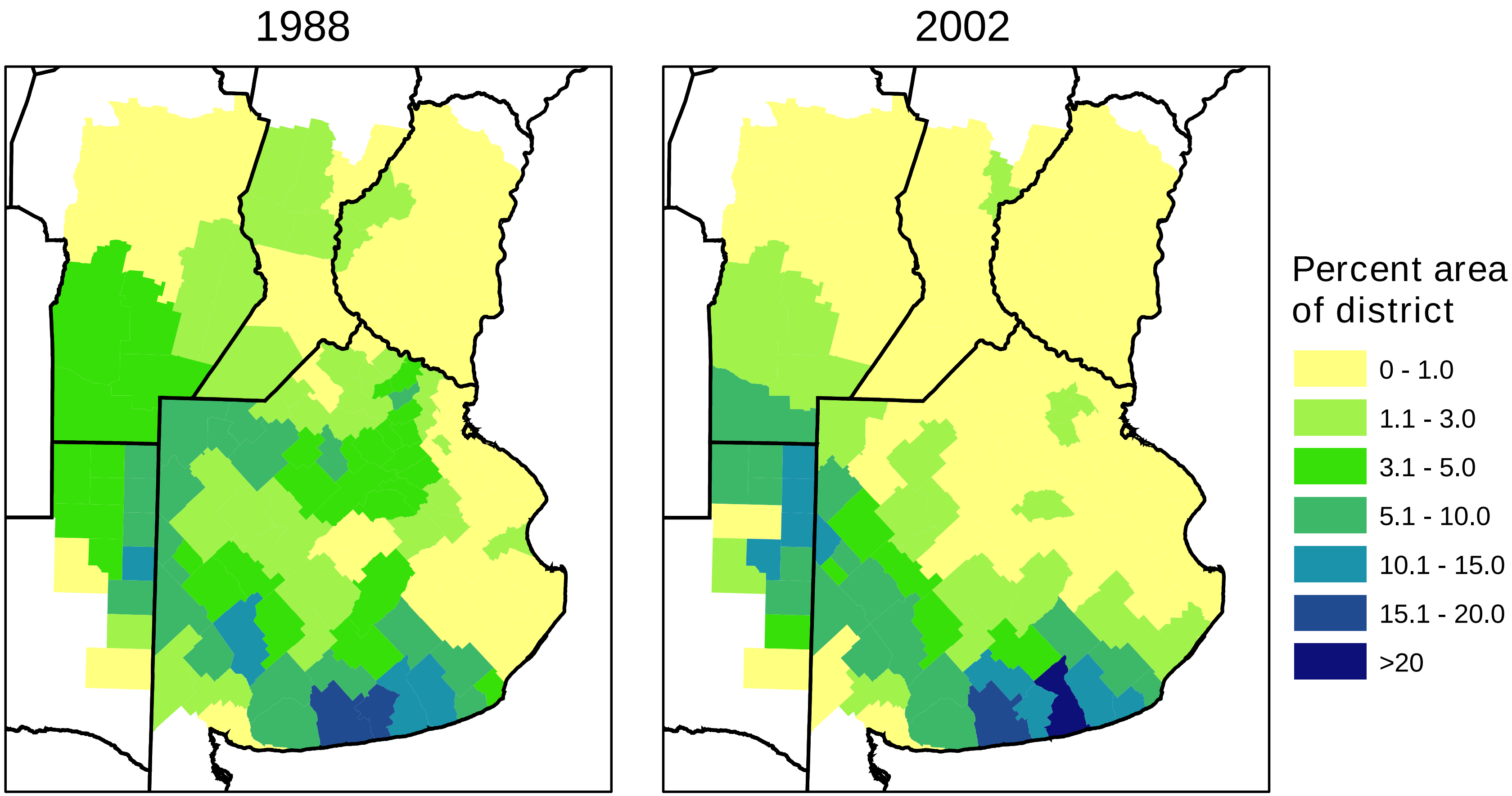

Figure 4.22: Proportional area by district in sunflower for 1988 and 2002 (INDEC, 1988, 2002).

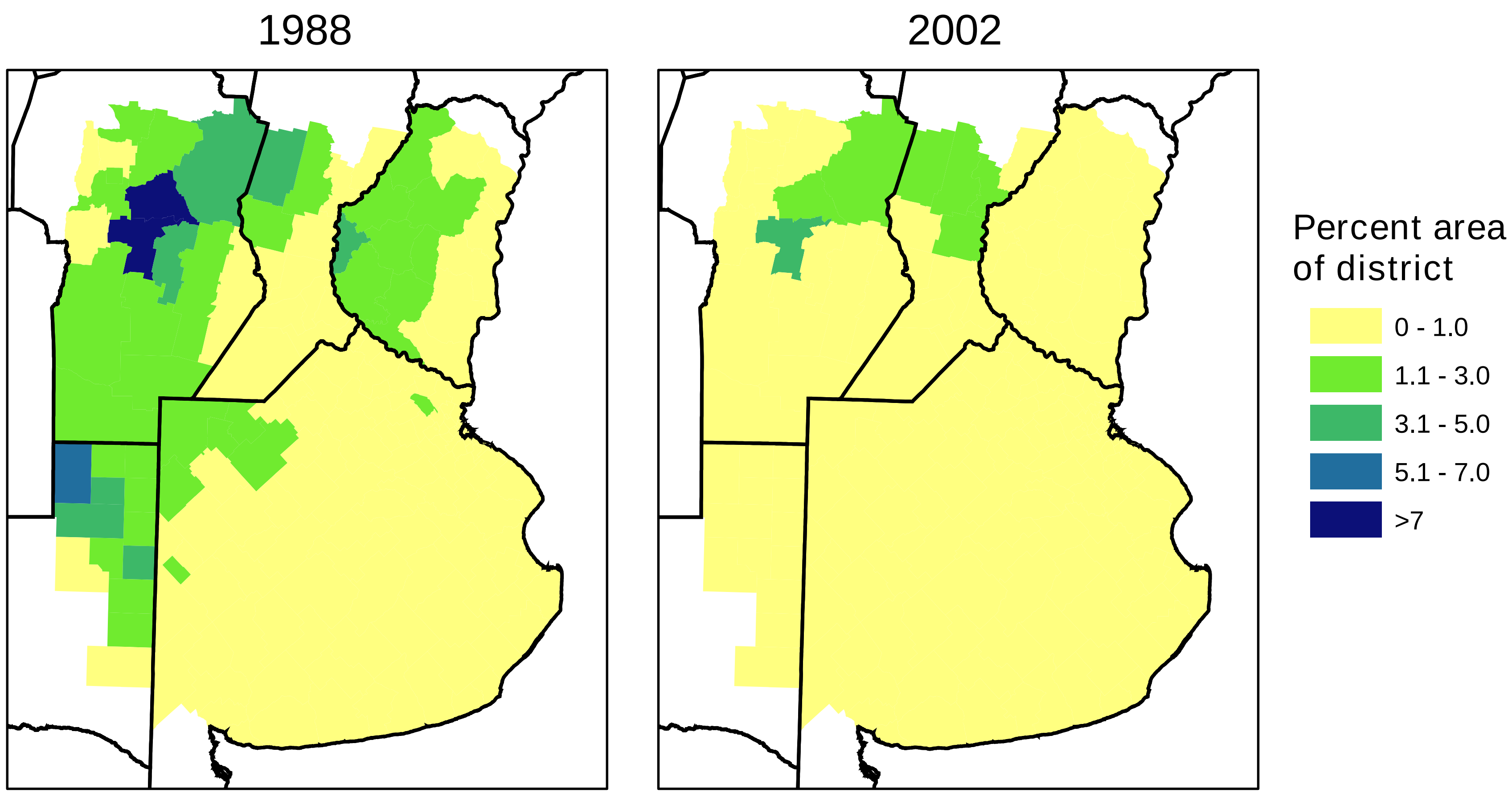

Figure 4.23: Proportional area by district in sorghum for 1988 and 2002 (INDEC, 1988, 2002).

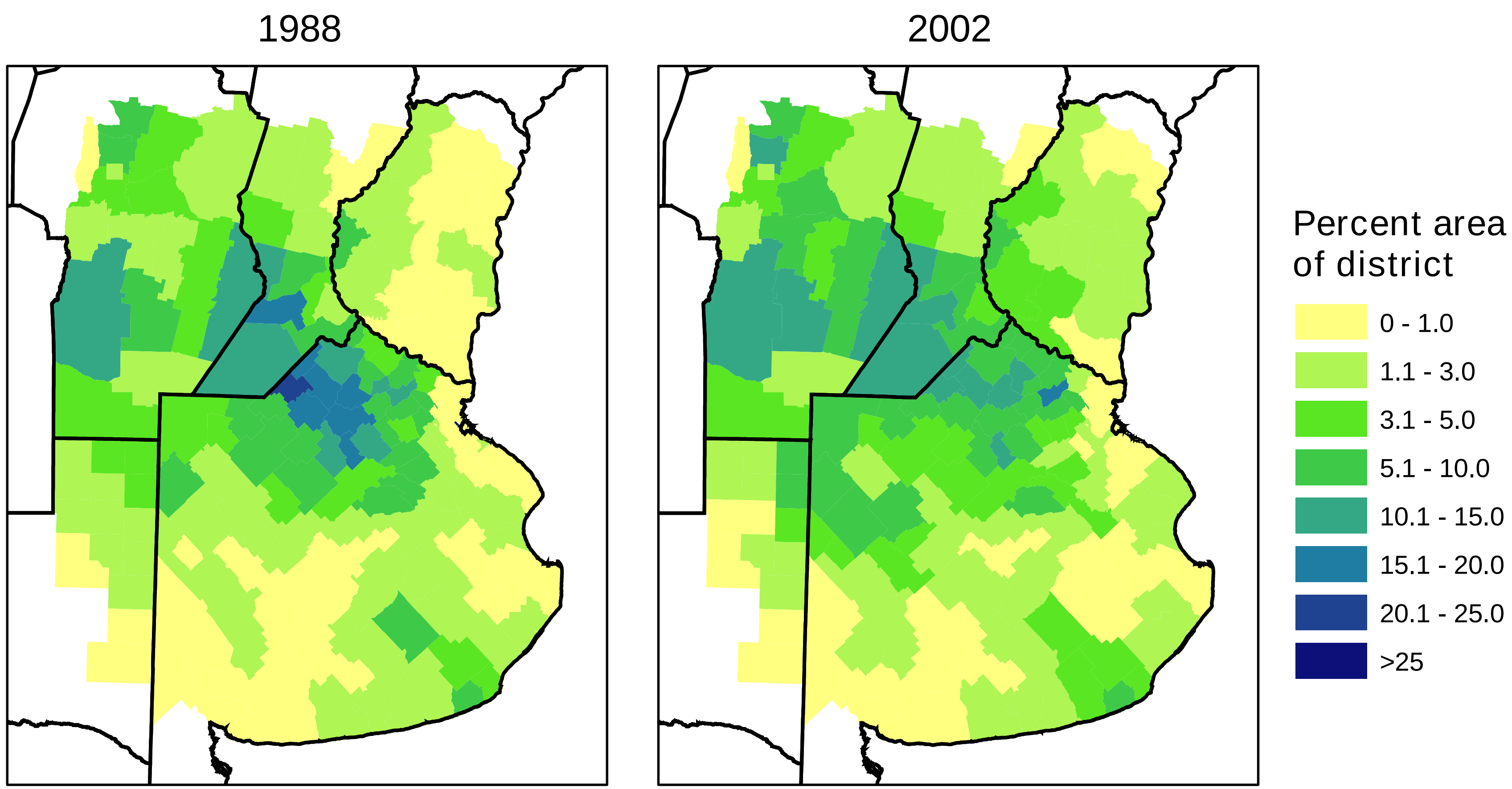

Figure 4.24: Proportional area by district in corn for 1988 and 2002 (INDEC, 1988, 2002).

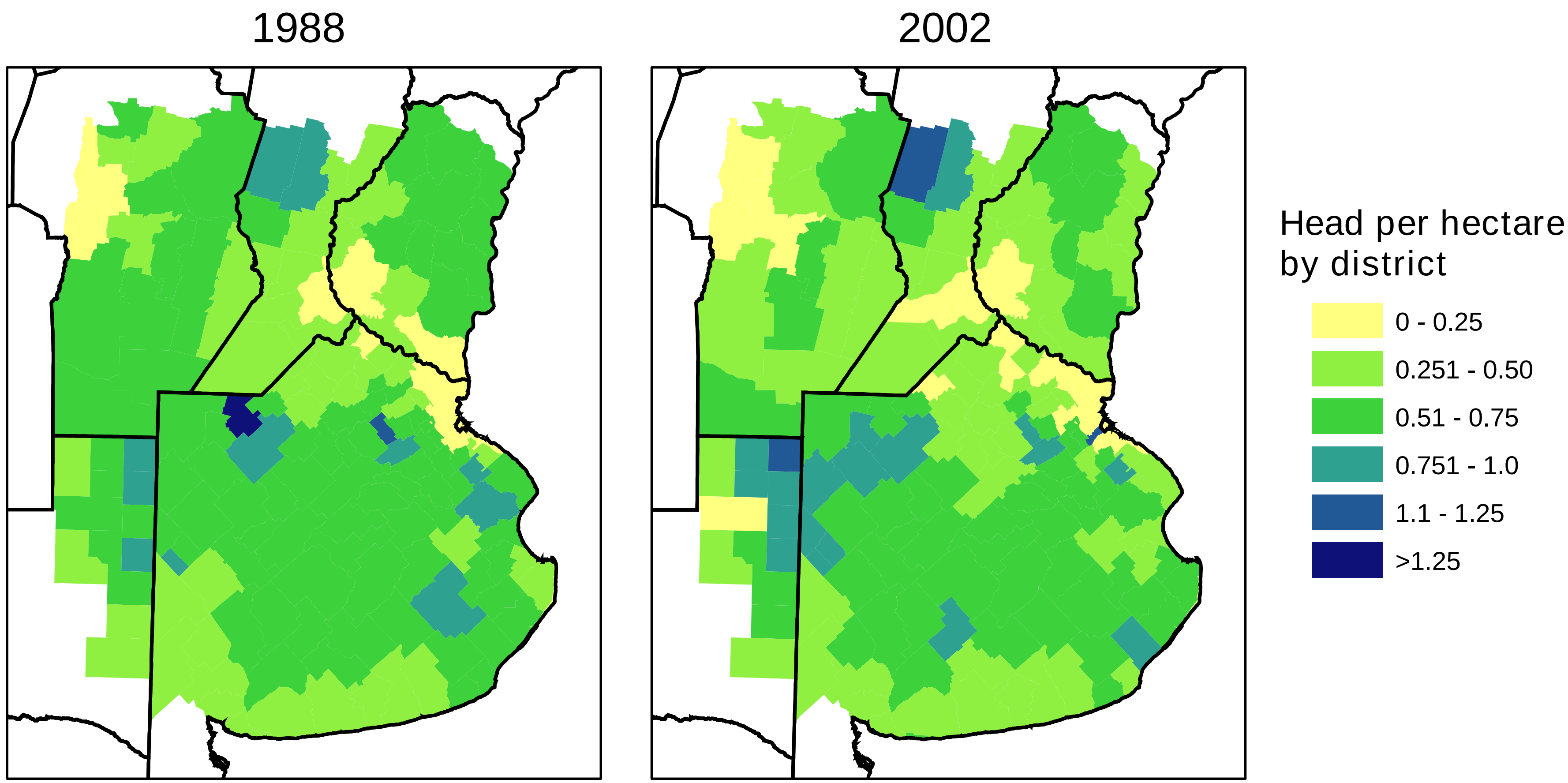

Figure 4.25: Number of cattle by district for 1988 and 2002 (INDEC, 1988, 2002).

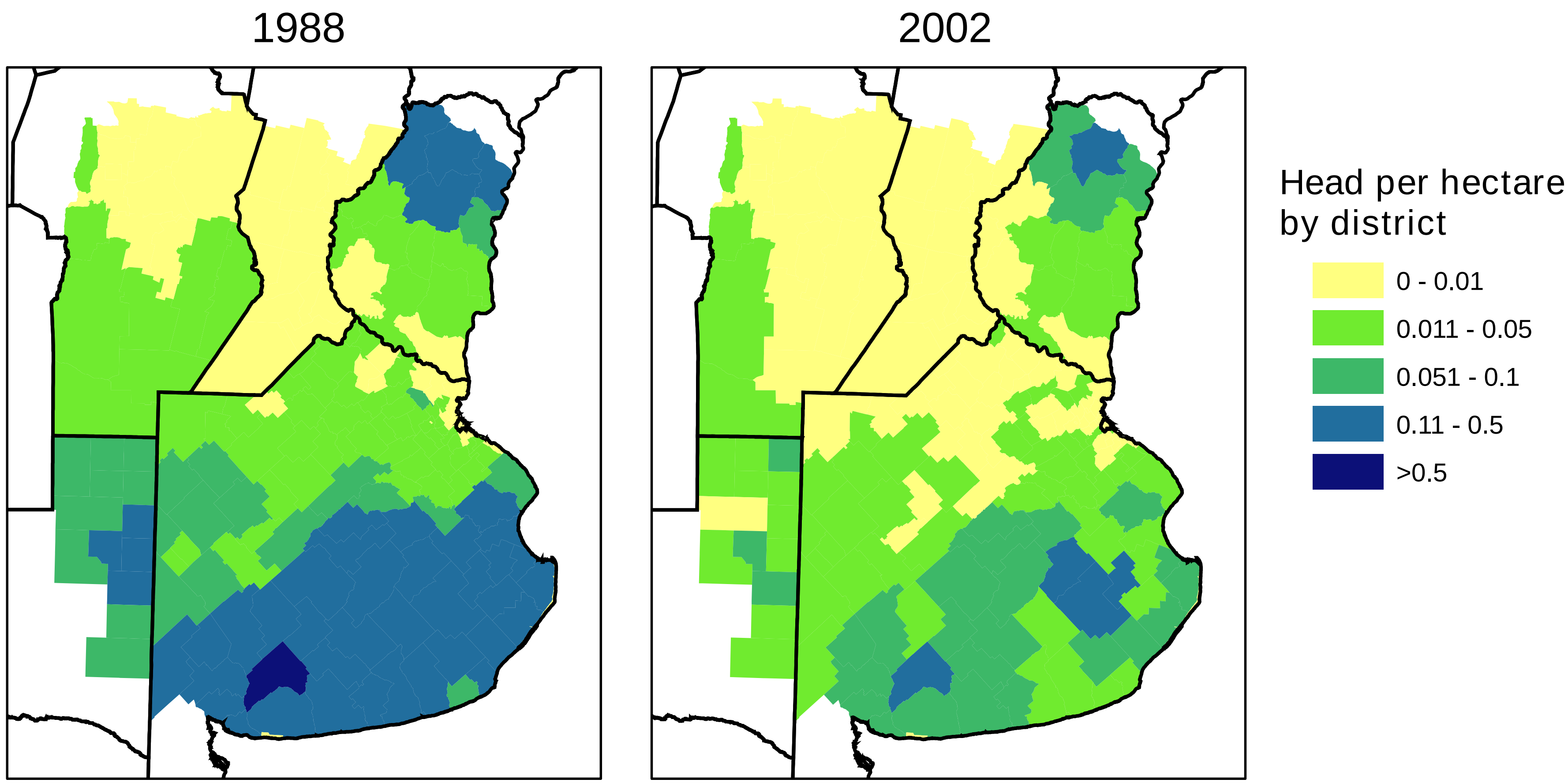

Figure 4.26: Number of sheep by district for 1988 and 2002 (INDEC, 1988, 2002).

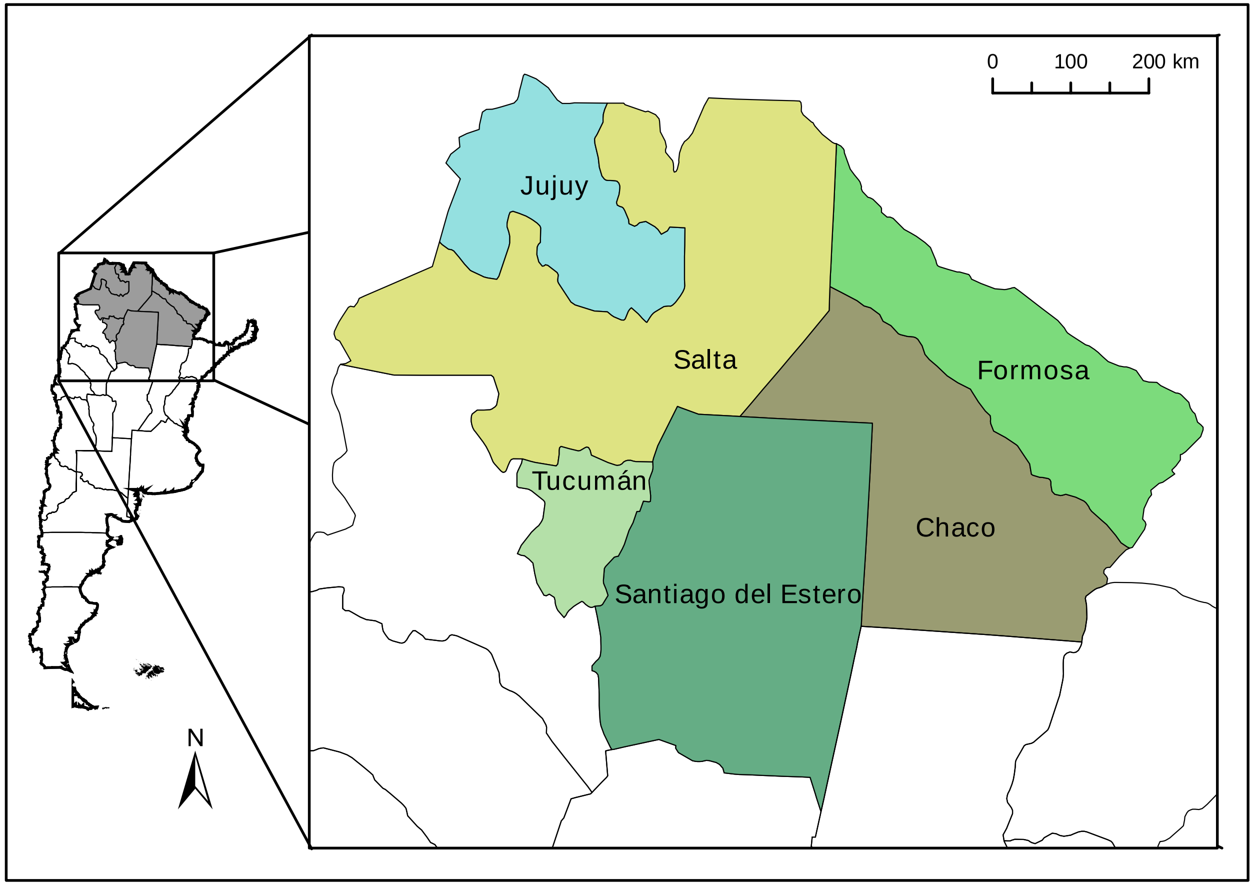

Figure 4.27: Location of the Argentine provinces of Chaco, Formosa, Jujuy, Salta, Santiago del Estero, and Tucumán

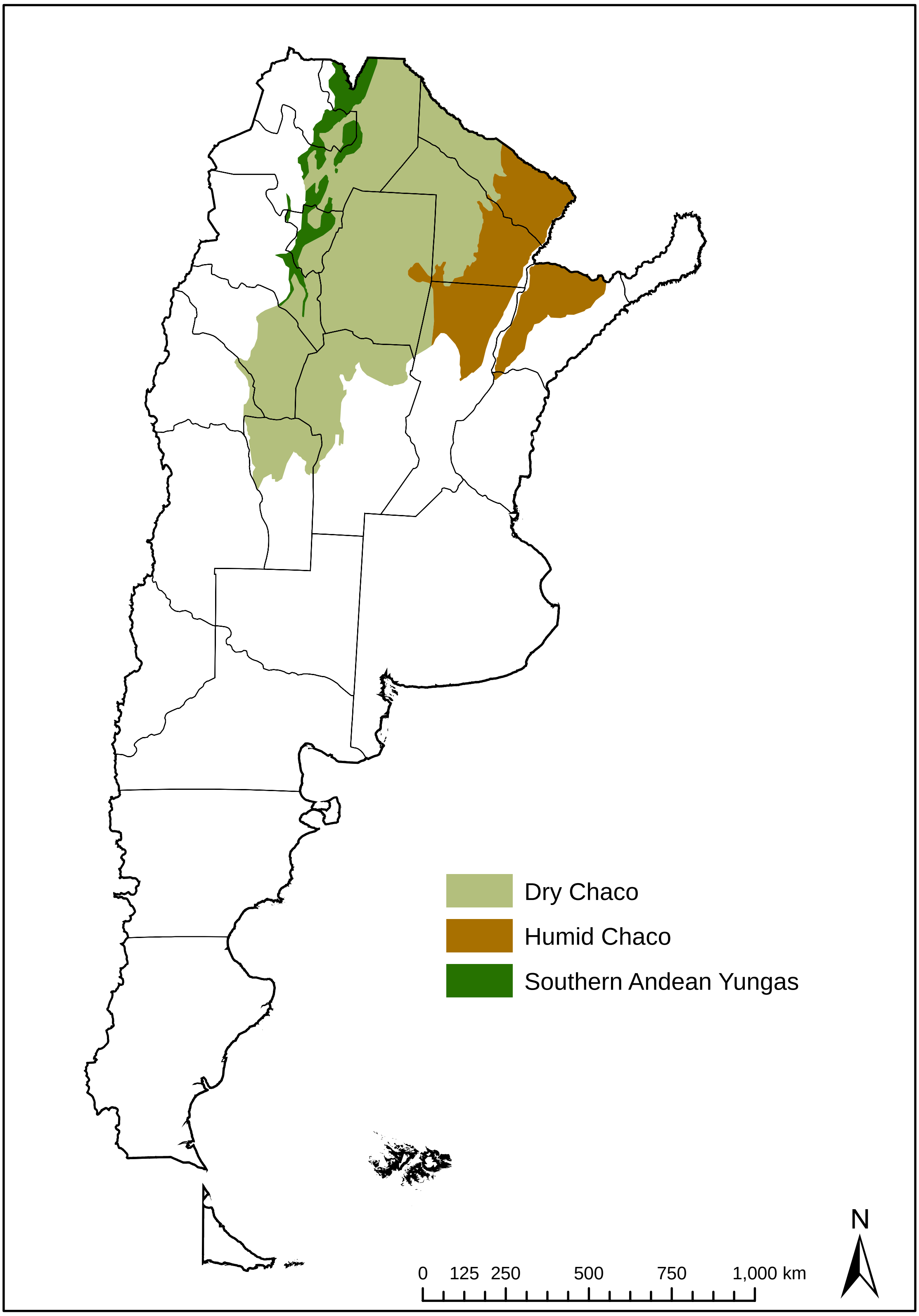

Figure 4.28: Distribution of Chaco and Yungas forest in Argentina.

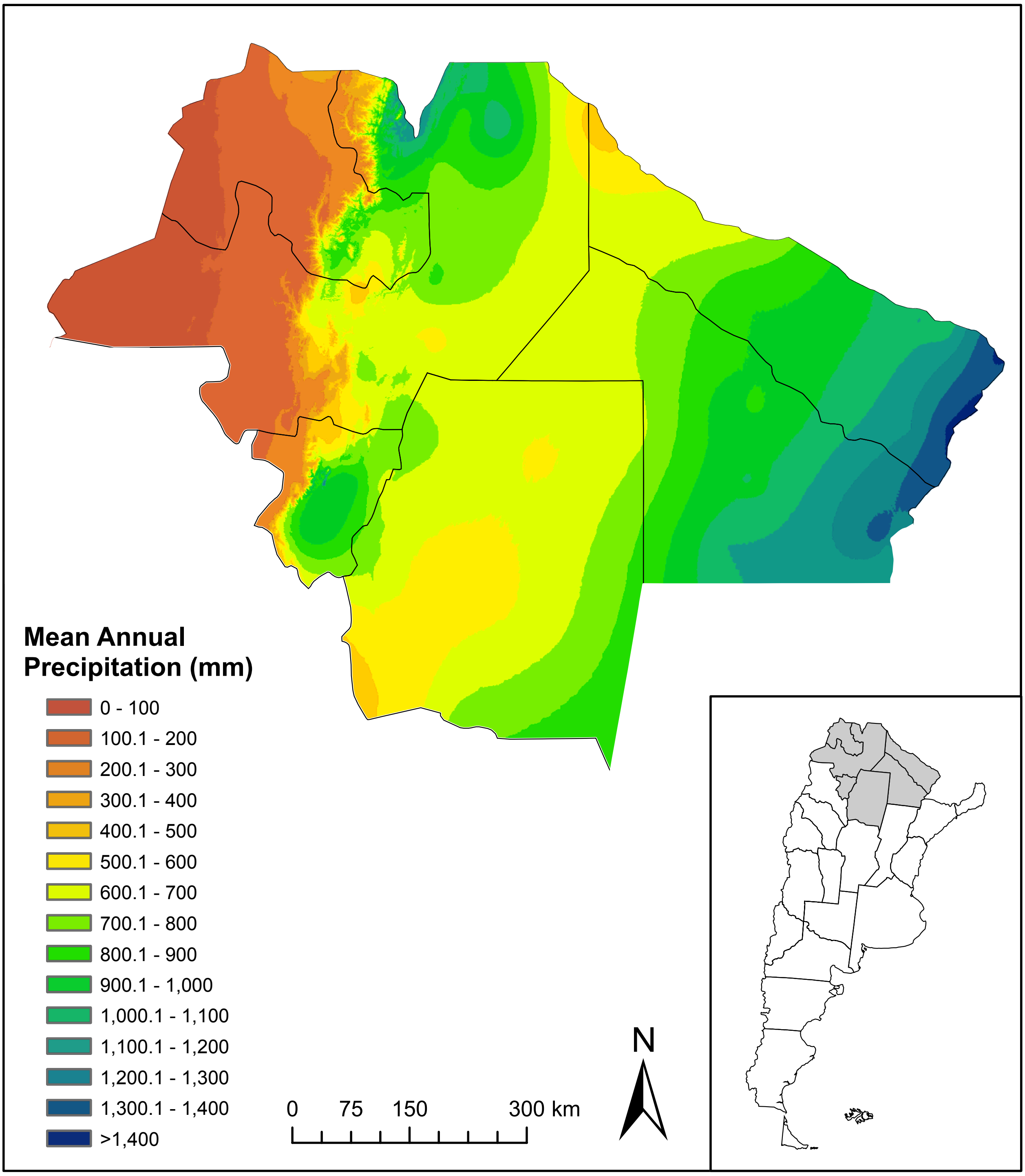

Figure 4.29: Mean annual precipitation in the provinces of Chaco, Formosa, Jujuy, Salta, Santiago del Estero, and Tucumán. Derived from data from (Hijmans et al., 2005).

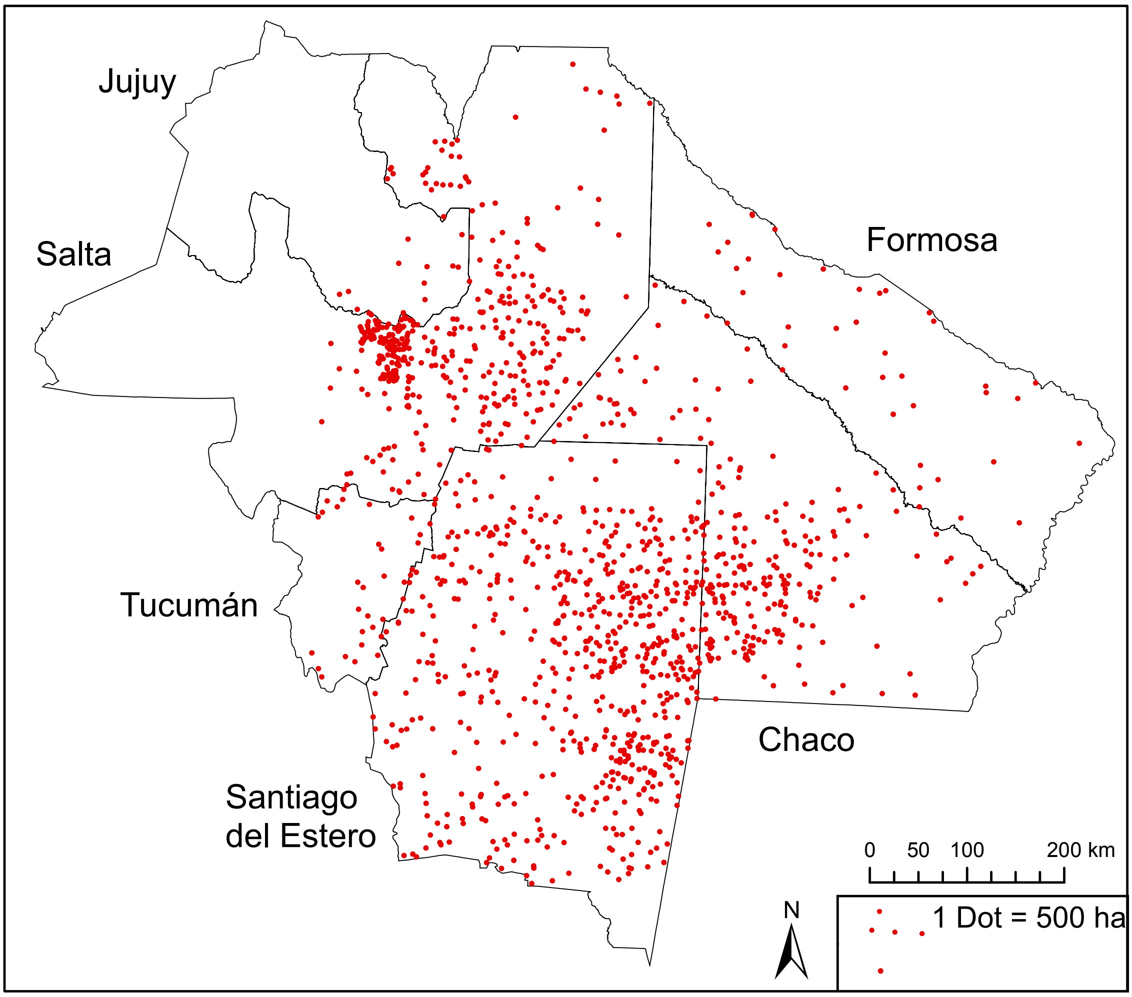

Figure 4.30: Deforestation of Chaco and Yungas forest in northwestern Argentina between 1998 and 2002 by district. Each dot represents 500 hectares (Bono et al., 2005; Gasparri et al., 2004a,b,c; Montenegro et al., 2003b; Parmuchi et al., 2004).

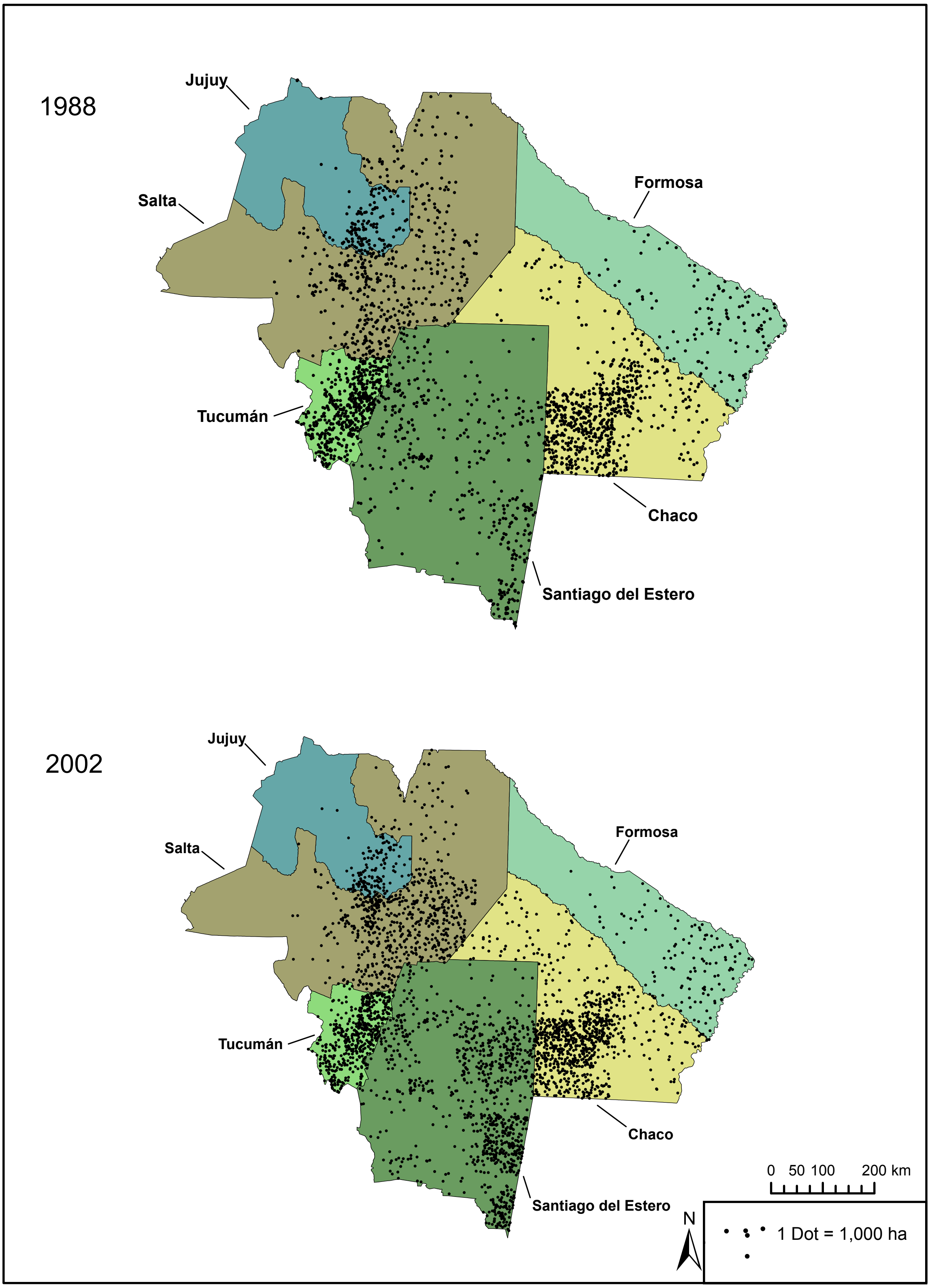

Figure 4.31: Distribution and area of cultivated land by district for 1988 and 2002 for the provinces of Chaco, Formosa, Jujuy, Salta, Santiago del Estero, and Tucumán (INDEC, 1988, 2002). Each dot represents 1,000 hectares.

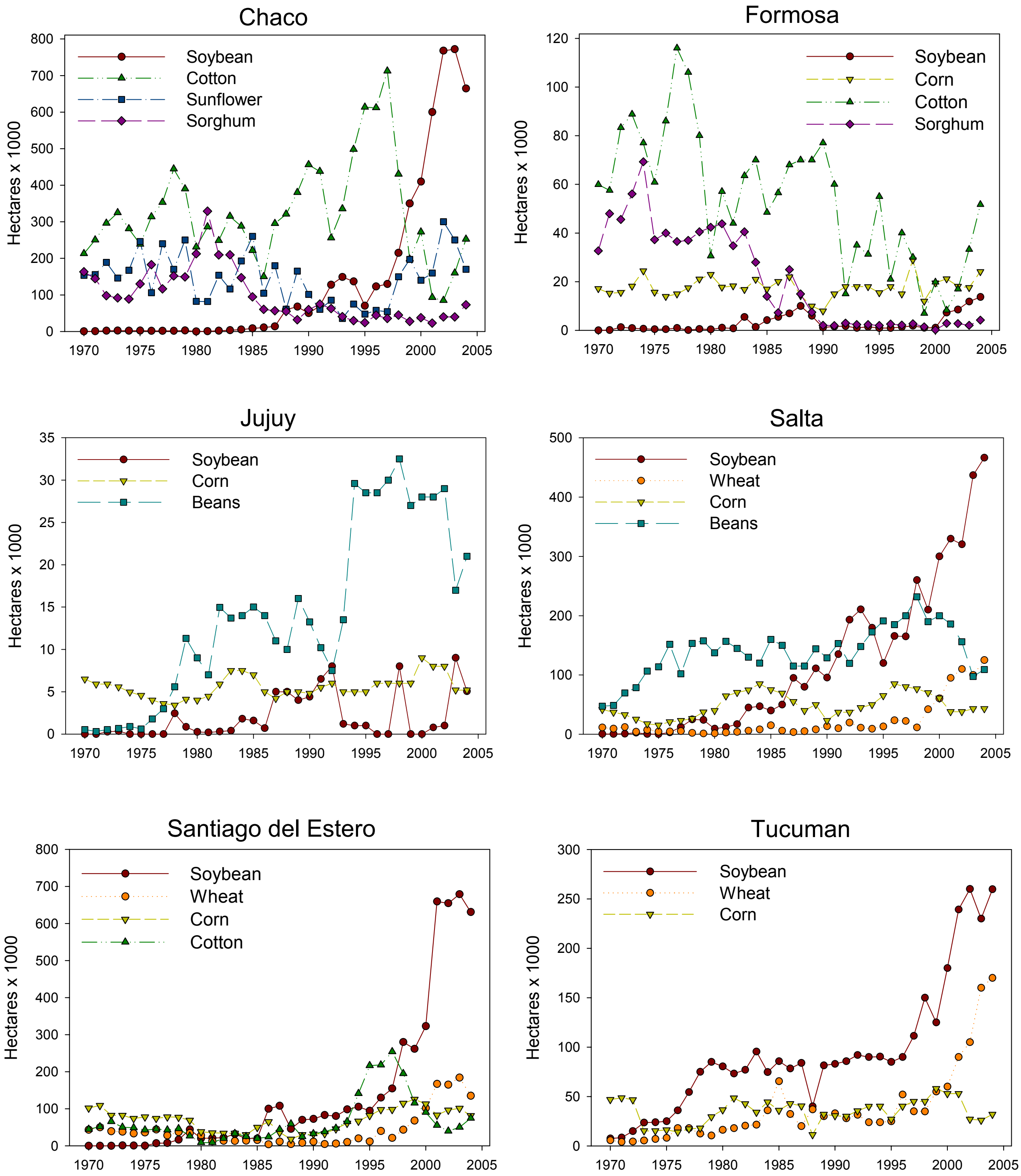

Figure 4.32: Total planted area of beans, corn, cotton, soybeans, sunflower, and wheat during 1970-2004 for the provinces of Chaco, Formosa, Jujuy, Salta, Santiago del Estero, and Tucumán (SAGPyA, 2006).

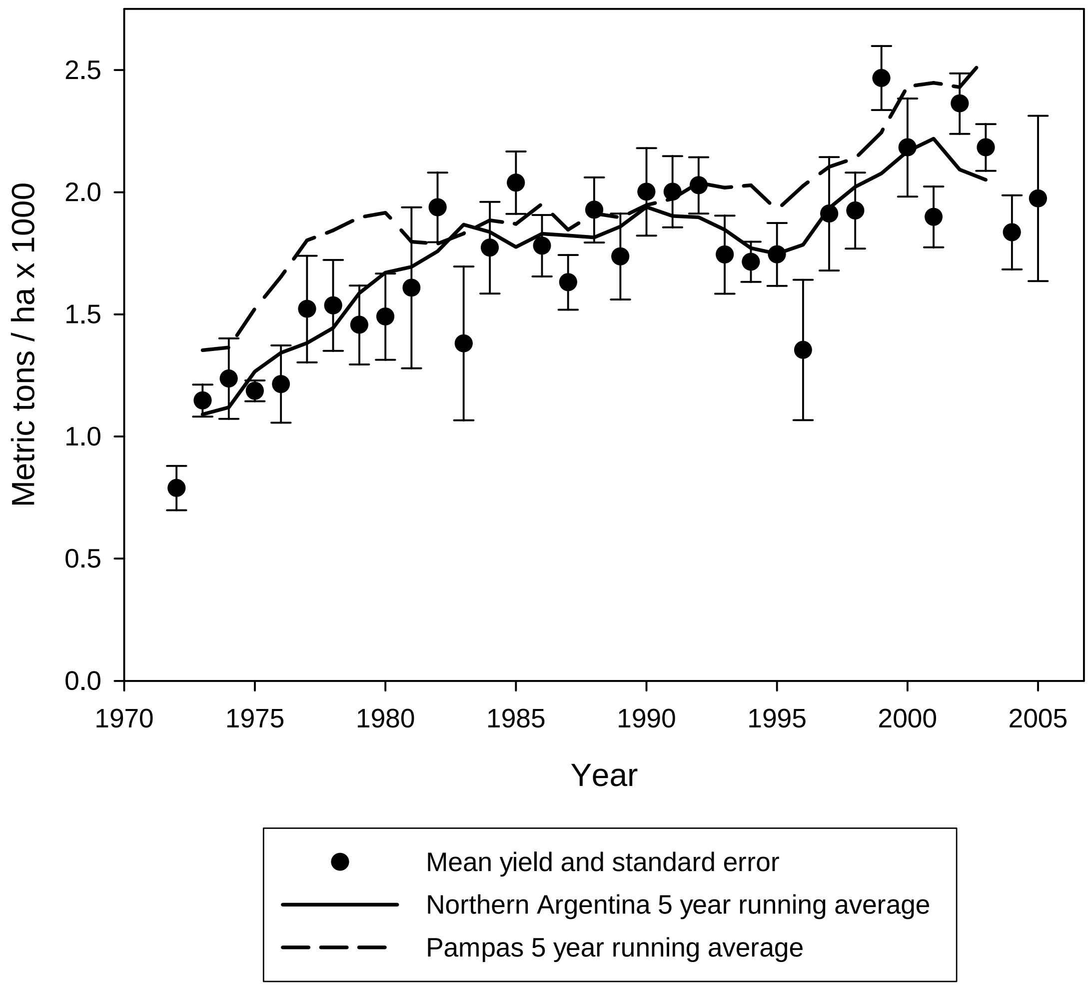

Figure 4.33: Mean yield of soybeans for northern Argentina (Chaco, Formosa, Jujuy, Salta, Santiago del Estero) during 1973-2006, error bars represent standard error. Solid line is the 5 year running average of soybean yields for northern Argentina and the dashed line is the 5 year running average of soybean yields in the Pampas (Buenos Aires, Córdoba, Entre Ríos, La Pampa, Santa Fe).

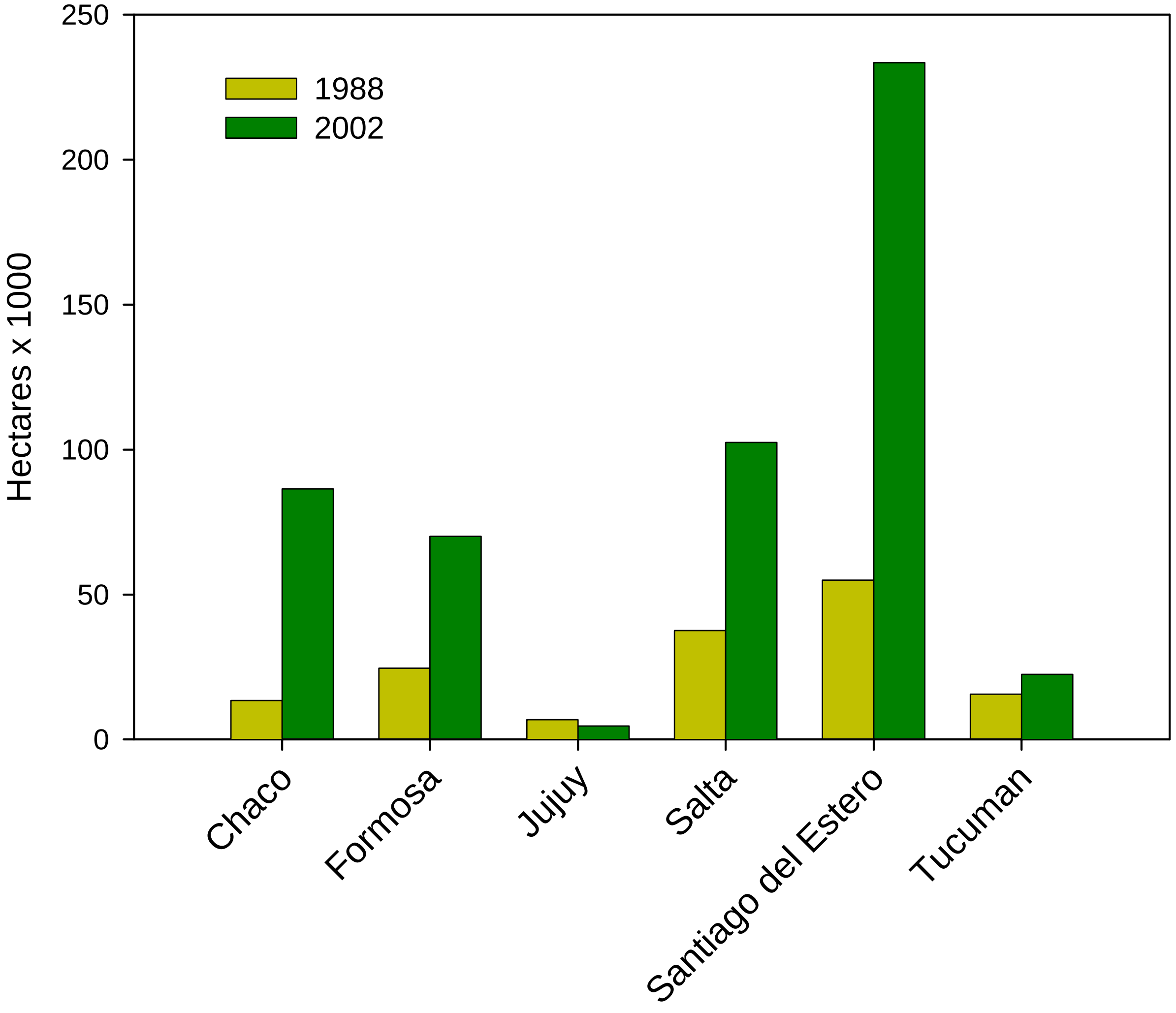

Figure 4.34: Distribution of perennial pastures by district for 1988 and 2002 for the provinces of Chaco, Formosa, Jujuy, Salta, Santiago del Estero, and Tucumán (INDEC, 1988, 2002).

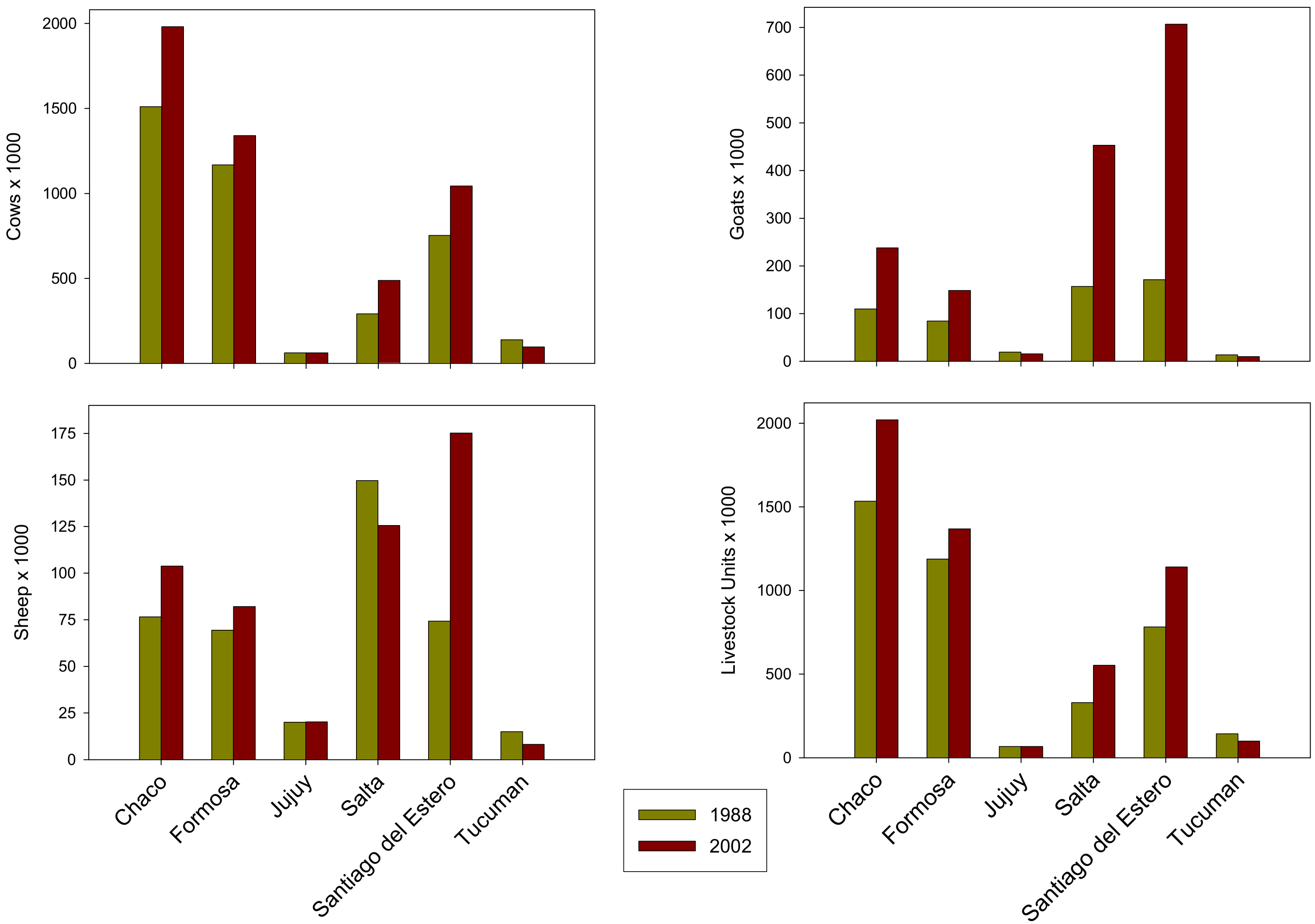

Figure 4.35: Cattle, sheep, goats, and Livestock Unit numbers for 1988 and 2002 for the provinces of Chaco, Formosa, Jujuy, Salta, Santiago del Estero, and Tucumán (INDEC, 1988, 2002).

Chapter 5

Implications of agricultural intensification for gamebird conservation in temperate agroecosystems

In Chapter 2, I reviewed how the process of agricultural intensification impacts wildlife in agroecosystems through changes in the quantity and quality of habitat and resources. In Chapters 3 and 4, I illustrated the common trends in agriculture intensification in the principal temperate agricultural regions of the world, stressing the factors that are responsible for and indicative of the changes in the quantity and quality of habitat and resources for wildlife in agroecosystems. Here, I examine the ecological implications of agricultural intensification and extensification from the perspective of the avifauna in temperate agroecosytems, with particular reference to the Galliformes.

5.1 Temperate agriculture and birds: a review and synthesis

I reviewed 158 articles from the primary literature, published from 1963-2007, that directly investigated the effects of agricutural land use upon terrestrial avian ecology (excluding raptors) to assess the extent and level of knowledge concerning the relationship between agriculture and the avifauna of agroecosystems. Articles were selected by searching the *Wildlife and Ecology Studies Worldwide* and *BIOSIS* databases, using combinations of the search terms *agriculture*, *birds*, *farmland*, and *grazing*, as well as from the references in the articles selected with the databases.

I grouped articles by the spatial scale(s) addressed; those being regional, landscape, among–farms, within–farm, and within–field. Studies at the region scale were those that investigated trends in land use and avian populations and/or distributions at the state, province, national, or physiognomic level, or interpreted research results to those extents. Landscape scale studies included investigations of multiple sites in relatively close proximity to one another which were ecologically similar, but potentially under different management regimes. Among–farms studies were those that compared differences among farms, within–farm studies dealt with spatial and temporal changes within farms, and within–field studies addressed factors related to differences within individual fields. Because of the hierarchical nature of these scales, many of these studies were not exclusive to a single hierarchical level.

The studies were relatively evenly distributed among spatial scales (Fig. 5.1a), with those addressing habitat types being dominant, followed by those dealing with the amount of habitat (Fig. 5.1b). Studies of habitat use were nearly 3 times more common than other types (Fig. 5.1c) and studies related to the reproductive season more common than those outside the breeding season (Fig. 5.1d).

5.1.1 Regional level

Multiple investigations from Europe and North America revealed strong correlations at the regional level between avian abundance and the intensification of agricultural land use. These studies and others illustrated general patterns in the relationship between avian abundance and agricultural land use as a function of the amount and type of habitat available, as well as factors related to the management of those habitats.

Of note in all these studies was the importance of multiple factors interacting to influence avian populations in agrarian systems, with no single cause for negative trends in populations. Moreover, many influential factors tended to be species or guild specific. Additionally, differences in management may overshadow the effects of changes in land cover, further complicating interpretations. Despite this, all of the studies at the regional scale that investigated the relationship between agricultural intensification and farmland birds (52% of all publications at this scale) showed a correlation between agricultural intensification and declining avian abundance.[1]

Of the 31 studies at the regional scale, 58% addressed the relationship between land cover and avian abundance[2] and 54% dealt with issues related to the intensification of agricultural management and avian abundance.[3] Several habitat types emerge as being important in relation to grassland and farmland " species; these being arable land, woodland, grassland, and non-cropland (particularly fallow, hay, and crop stubble), while the diversity of these land covers are also important (Fig. 5.2).

For grassland or farm-dependent birds positive relationships were illustrated for presence or increase in grassland[4], as well as for non-cropland habitats.[5] For farmland dependent species, increases in arable area[6] and decrease in woodlands [7] were positively associated with presence or population trends while grassland dependent species showed negative trends with both decreasing grassland area and increasing area of woodlands.[8]

[1](Aebischer and Ewald, 2004; Báldi and Faragó, 2007; Benton et al., 2002; Chamberlain and Fuller, 2001a; Chamberlain et al., 2000; Donald et al., 2001b, 2006; Fox, 2004; Green and Rayment, 1996; Gregory et al., 2004; Haberl et al., 2005; Herzon and O'Hara, 2007; Kujawa, 2002; Murphy, 2003; Warner and Etter, 1989; Wretenberg et al., 2006)

[2](Aebischer and Ewald, 2004; Ambrosini et al., 2002; Atkinson et al., 2004; Báldi et al., 2005; Chamberlain and Gregory, 1999; Chamberlain et al., 2000; Chamberlain and Fuller, 2001b; Fox, 2004; Hancock and Wilson, 2003; Herzon and O'Hara, 2007; Johnson and Igl, 1995; Johnson and Schwartz, 1993; Laiolo, 2005; McDonald and Reese, 1998; Murphy, 2003; Siriwardena et al., 2000; Suárez et al., 1997; Wretenberg et al., 2006)

[3](Aebischer and Ewald, 2004; Ambrosini et al., 2002; Atkinson et al., 2004; Báldi et al., 2005; Báldi and Faragó, 2007; Benton et al., 2002; Chamberlain et al., 2000; Chamberlain and Fuller, 2001b; Donald et al., 2001b, 2006; Green and Rayment, 1996; Haberl et al., 2005; Herzon and O'Hara, 2007; Kujawa, 2002; Siriwardena et al., 2000, 2001; Warner and Etter, 1989)

[4](Báldi et al., 2005; Johnson and Igl, 1995; Johnson and Schwartz, 1993; McDonald and Reese, 1998; Murphy, 2003; Wretenberg et al., 2006)

[5](Aebischer and Ewald, 2004; Ambrosini et al., 2002; Fuller et al., 2004; Hancock and Wilson, 2003; Herzon and O'Hara, 2007; Laiolo, 2005; Silva et al., 2004; Siriwardena et al., 2000; Warner and Etter, 1989; Wretenberg et al., 2006)

[6](Chamberlain and Gregory, 1999; Fuller et al., 2004; Hancock and Wilson, 2003; Murphy, 2003)

[7](Chamberlain and Gregory, 1999; Hancock and Wilson, 2003; Murphy, 2003)

[8](Fuhlendorf et al., 2002; Hancock and Wilson, 2003; Murphy, 2003; Siriwardena et al., 2000)

Despite the ambiguity in these studies from regional and species-specifc differences there are some general patterns that are evident at the regional level that impact the avian community in agroecosystems. These are:

1. The process of agricultural intensification in general, via increased yields and inputs, is correlated with declines of farmland bird populations in agroecosystems at national scales, although with species-specific variation.[9]

2. Habitat diversity in space and time stemming from management of agricultural systems is a key aspect of stable avian populations in those systems.[10]

3. Grassland or natural pasture which is ungrazed to moderately grazed, with little or no management has a positive effect on avian populations.[11]

4. There appears to be thresholds in the amount and/or quality of habitat at large scales that lead to population declines.[12] Grassland conversion of more than 10% of land area was shown by Fuhlendorf et al. (2002) to negatively affect greater prairie-chicken (*Tympanuchus cupidus*) populations while a mean land cover of 2% woody cover also had negative effects. Also, Hancock and Wilson (2003) showed an optimum of 0-10% woodland cover benefits passerines preferring open habitats.

[9](Aebischer and Ewald, 2004; Ambrosini et al., 2002; Báldi and Faragó, 2007; Benton et al., 2002; Chamberlain and Fuller, 2001a; Chamberlain et al., 2000; Donald et al., 2001b, 2006; Fox, 2004; Green and Rayment, 1996; Gregory et al., 2004; Haberl et al., 2005; Herzon and O'Hara, 2007; Kujawa, 2002; Murphy, 2003; Warner and Etter, 1989; Wretenberg et al., 2006)

[10](Báldi et al., 2005; Chamberlain and Gregory, 1999; Chamberlain et al., 2000; Chamberlain and Fuller, 2001b; Fox, 2004; Herzon and O'Hara, 2007; Murphy, 2003; Wretenberg et al., 2006)

[11](Atkinson et al., 2004; Báldi et al., 2005; Báldi and Faragó, 2007; Chamberlain and Gregory, 1999; Chamberlain and Fuller, 2001a; Johnson and Igl, 1995; Johnson and Schwartz, 1993; Laiolo, 2005; Murphy, 2003; Siriwardena et al., 2000; Wretenberg et al., 2006)

[12](Chamberlain et al., 2000; Fuhlendorf et al., 2002; Hancock and Wilson, 2003; McDonald and Reese, 1998; Siriwardena et al., 2000)

5. The presence of non-cropland habitats (stubble, fallow, field borders, hay) for both breeding and foraging habitat has a disproportionately positive effect of avian abundance within agricultural systems.[13]

5.1.2 Landscape level

A total of 30% of the publications reviewed directly investigated aspects of avian ecology in agricultural systems at the landscape scale or interpreted results in the context of the landscape. These studies specifically addressed or illustrated relationships to habitat diversity, habitat types (including intra-annual preferences), landscape configuration, food resources, land management, habitat characteristics, and recruitment (Fig. 5.3).

In terms of cover types the presence and amounts of natural grassland, steppe, or agricultural grasslands within the landscape, and the matrix in which they were distributed, was of shown to be of primary importance in multiple studies. In grassland and steppe areas there were positive associations with increasing area of natural habitats[14] while in areas dominated by agriculture, non-cropland (fallow, idle crop fields [particularly cereal stubble], hay, and hedgerows) generally had a positive influence on farmland dependent species[15], however, Aebischer and Potts (1998) and Marshall et al. (2006) showed no or negative effects of unmanaged uncropped land. In both grassland and

[13](Aebischer and Ewald, 2004; Ambrosini et al., 2002; Fuller et al., 2004; Hancock and Wilson, 2003; Herzon and O'Hara, 2007; Laiolo, 2005; Silva et al., 2004; Siriwardena et al., 2000; Warner and Etter, 1989; Wretenberg et al., 2006)

[14](Ammann, 1963; Coppedge et al., 2001; Fernandez et al., 2003; Fletcher, Jr. and Koford, 2003; Fuhlendorf et al., 2002; Herkert, 1994; Mangnall and Crowe, 2003; Manzer and Hannon, 2005; McDonald and Reese, 1998; Niemuth, 2000; Novoa et al., 2002; Ribic and Sample, 2001; Ryan et al., 1998; Woodward et al., 2001)

[15](Atkinson et al., 2002; Bergin et al., 2000; Bollinger et al., 1990; Bradbury et al., 2004; Brotons et al., 2005; Browne and Aebischer, 2003; Eraud and Boutin, 2002; Fuller et al., 2001; Malan and Benn, 1999; Moreira et al., 2005; Murtha, 1967; Pinto et al., 2005; Rands, 1987; Silva et al., 2004; Suárez et al., 1997; Taylor et al., 1999; Virkkala et al., 2004; Yeatter, 1963)

agricultural landscapes bird species most dependent upon those landscapes were negatively affected by the presence of woody vegetation or through its encroachment or afforestation.[16]

The effect of row crops[17] and pasture[18] is dependent upon the species, the landscape matrix, and management regimes; with the value of row crops and pasture decreasing with increasing management intensity. Within agricultural landscapes a higher proportion of pasture land had positive effects on avian abundance but, this utility decreased with increasing grazing intensity.[19] For some species row crops were illustrated to have positive effects, especially when not intensively managed[20], while others showed intolerance to row crop expansion beyond a proportional threshold of the landscape.[21]

Land management, mostly associated with the intensification of food production (*i.e.* increased livestock densities, irrigation, haying, pesticide use), affects habitat characteristics and availability of required resources and were shown to have significant influences on avian abundances within landscapes in several instances.[22] These studies all illustrated the importance of the type and management of grazing and lands mowed for hay, particularly relating negative effects to excessive grazing[23] and a shift from grass dominated hay fields to alfalfa and the earlier and more frequent mowing of those fields.[24] Management impacts

[16](Ammann, 1963; Bergin et al., 2000; Coppedge et al., 2001; Cunningham and Johnson, 2006; Fuhlendorf et al., 2002; Moreira et al., 2005; Niemuth, 2000; Novoa et al., 2002; Ribic and Sample, 2001; Woodward et al., 2001)

[17](Fuhlendorf et al., 2002; Mangnall and Crowe, 2003; Manzer and Hannon, 2005; Murtha, 1967; Niemuth, 2000; Pinto et al., 2005; Ratcliffe and Crowe, 2001a; Taylor et al., 1999; Vargas et al., 2006; Wolff et al., 2001; Woodward et al., 2001)

[18](Ambrosini et al., 2002; Bergin et al., 2000; Jobin et al., 1996; Pinto et al., 2005; Ribic and Sample, 2001; Söderström et al., 2001; Suárez et al., 1997)

[19](Ambrosini et al., 2002; Bergin et al., 2000; Jobin et al., 1996; Pinto et al., 2005; Ribic and Sample, 2001; Söderström et al., 2001; Suárez et al., 1997)

[20](Mangnall and Crowe, 2003; Murtha, 1967; Pinto et al., 2005; Ratcliffe and Crowe, 2001a; Taylor et al., 1999; Vargas et al., 2006; Wolff et al., 2001)

[21](Fuhlendorf et al., 2002; Manzer and Hannon, 2005; Ratcliffe and Crowe, 2001a; Woodward et al., 2001)

[22](Aebischer and Potts, 1998; Ammann, 1963; Bollinger et al., 1990; Fernandez et al., 2003; Green et al., 1997; Müller et al., 2005; Novoa et al., 2002; Pinto et al., 2005; Powell, 2006; Rands, 1987; Silva et al., 2004; Warner et al., 1984; Yeatter, 1963)

[23](Ambrosini et al., 2002; Fernandez et al., 2003; Silva et al., 2004; Söderström et al., 2001)

[24](Ammann, 1963; Bollinger et al., 1990; Green et al., 1997; Müller et al., 2005; Yeatter, 1963)

bird species by affecting habitat quality across the landscape via determining vegetative structure[25], while increased use of pesticides negatively impact recruitment.[26]

Increasing land use diversity was of importance in many cases for maintaining avian abundance and/or diversity[27], however, in the case of grassland and steppe obligates a decrease in landscape diversity favoring natural vegetative cover or extensive agriculture had positive implications.[28] These differences illustrate the sensitivity of grassland and steppe obligates to threshold levels of habitat conversion, as well as the complexity in habitat relationships among avian guilds in agricultural systems.[29]

Related to landscape diversity is the configuration of landscapes, with both landscape complexity and the juxtaposition of habitats being of importance.[30] These relationships are complex; varying among species[31], different spatial resolutions across the landscape[32], related to species–specific differences in seasonal habitat requirements.[33]

Large scale changes in landscape composition can impact reproductive success through affecting clutch size, nest failure and/or brood mortality.[34] In all these studies the increased

[25](Ammann, 1963; Bollinger et al., 1990; Eraud and Boutin, 2002; Novoa et al., 2002; Powell, 2006; Rands, 1987; Silva et al., 2004)

[26]Aebischer and Potts (1998); Warner et al. (1984)

[27](Atkinson et al., 2002; Coppedge et al., 2001; Clark and Bogenschutz, 1999; Eraud and Boutin, 2002; Jobin et al., 1996; Mangnall and Crowe, 2003; Malan and Benn, 1999; Moreira et al., 2005; Ratcliffe and Crowe, 2001a; Ribic and Sample, 2001; Söderström et al., 2001; Taylor et al., 1999; Virkkala et al., 2004; Wolff et al., 2001)

[28](Fuhlendorf et al., 2002; Herkert, 1994; Manzer and Hannon, 2005; McDonald and Reese, 1998; Moreira et al., 2005; Niemuth, 2000; Novoa et al., 2002; Ribic and Sample, 2001; Ryan et al., 1998; Suárez et al., 1997; Wolff et al., 2001; Woodward et al., 2001)

[29](Atkinson et al., 2002; Coppedge et al., 2001; Cunningham and Johnson, 2006; Herkert, 1994; Moreira et al., 2005; Ribic and Sample, 2001)

[30](Bergin et al., 2000; Brotons et al., 2005; Coppedge et al., 2001; Cunningham and Johnson, 2006; Fuhlendorf et al., 2002; Fuller et al., 2001; Herkert, 1994; Inglis et al., 1994; Malan and Benn, 1999; McDonald and Reese, 1998; Moreira et al., 2005; Ratcliffe and Crowe, 2001a; Ribic and Sample, 2001; Ryan et al., 1998; Woodward et al., 2001)

[31](Bergin et al., 2000; Cunningham and Johnson, 2006; Herkert, 1994; Moreira et al., 2005; Ribic and Sample, 2001)

[32](Bergin et al., 2000; Fuhlendorf et al., 2002; Woodward et al., 2001)

[33](Atkinson et al., 2002)

[34](Aebischer and Potts, 1998; Bergin et al., 2000; Clark and Bogenschutz, 1999; Manzer and Hannon, 2005; Taylor et al., 1999)

presence of grassland habitats, whether perennial[35], rotational[36], or natural[37] was related to greater recruitment. In agriculturally dominated areas, greater landscape diversity was also associated with higher recruitment[38], while in grassland dominated areas a maximum threshold of habitat conversion was exhibited past which recruitment decreased.[39]

Although the impacts of agricultural land use varies geographically and is dependent species, or guild–dependent, there exist some commonalities at the landscape scale. These are:

1. In agriculturally dominated areas, landscape diversity tends to have positive effects while in areas dominated by natural grasslands decreasing diversity in favor of natural land cover is favorable.

2. Grassland and steppe (including extensive cereal cultivation) dependent species exhibit optimal levels of habitat conversion that are species–dependent.

3. Forest encroachment or afforestation have negative impacts upon populations of bird species dependent upon grasslands or open landscapes

4. The juxtaposition of habitats and the landscape configuration influences both diversity and nest survival.

5. Intensification in land management in the form of switches from grass hay to alfalfa, earlier and more frequent haying, loss of rotational grasslands, fallow, and idle fields, and the increased use of pesticides have all been implicated in the declines of avian populations in agroecosystems.

6. Increasing livestock densities generally have negative effects for most bird species in agroecosystems.

[35](Clark and Bogenschutz, 1999; Taylor et al., 1999)
[36](Aebischer and Potts, 1998)
[37](Manzer and Hannon, 2005; Taylor et al., 1999)
[38](Aebischer and Potts, 1998; Bergin et al., 2000; Clark and Bogenschutz, 1999; Taylor et al., 1999)
[39](Manzer and Hannon, 2005; Taylor et al., 1999)

5.1.3 Among farms

Studies addressing differences among farms were 33% of all the publications reviewed and due to the hierarchical nature of these 37% were also interpreted within the context of the landscape.[40] Among farm studies investigated topics of seasonal ecology, habitat use, habitat diversity and configuration, and land management, food resources, and habitat characteristics (Fig. 5.4).

Investigations of breeding ecology[41] were nearly twice as common as those concerning winter ecology.[42] Of publications on breeding ecology, 62% investigated questions related to the nesting period[43]and 54% brood ecology.[44] Comparisons among farms showed negative relationships in recruitment with intensification of cropping[45], excessive grazing[46], haying[47], and the presence of woody vegetation[48], while the presence of non–crop habitats had positive effects.[49] Winter habitat use was illustrate to favor non–crop habitats[50], cereal stubble with higher weed densities.[51] and unimproved grasslands[52]

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[40](Aebischer and Potts, 1998; Ambrosini et al., 2002; Bergin et al., 2000; Bollinger et al., 1990; Bradbury et al., 2004; Browne and Aebischer, 2003; Eraud and Boutin, 2002; Fuhlendorf et al., 2002; Fuller et al., 2001; Malan and Benn, 1999; Marshall et al., 2006; Moreira et al., 2005; Müller et al., 2005; Rands, 1987; Silva et al., 2004; Söderström et al., 2001; Warner et al., 1984; Yeatter, 1963)

[41](Aebischer and Potts, 1998; Ambrosini et al., 2002; Avilés and Parejo, 2004; Bergin et al., 2000; Green, 1984; Lusk et al., 2003; Malan, 1998; Morris et al., 2005; Warner, 1994; Warner and Etter, 1989; Warner et al., 1984; Wilson et al., 1997)

[42](Barnett et al., 2004; Bruner et al., 2005; Moorcroft et al., 2002; Rodgers, 2002; Silva et al., 2004; Smith et al., 2005; Stoate et al., 2000)

[43](Avilés and Parejo, 2004; Bergin et al., 2000; Lusk et al., 2003; Malan, 1998; Müller et al., 2005; Warner, 1994; Warner and Etter, 1989; Wilson et al., 1997)

[44](Ambrosini et al., 2002; Avilés and Parejo, 2004; Green, 1984; Morris et al., 2005; Müller et al., 2005; Warner et al., 1984; Wilson et al., 1997)

[45](Avilés and Parejo, 2004; Green, 1984; Morris et al., 2005; Warner et al., 1984; Wilson et al., 1997)

[46](Ambrosini et al., 2002; Lusk et al., 2003; Malan, 1998; Wilson et al., 1997)

[47](Müller et al., 2005; Warner and Etter, 1989; Wilson et al., 1997)

[48](Bergin et al., 2000; Wilson et al., 1997)

[49](Bruner et al., 2005; Warner, 1994; Wilson et al., 1997)

[50](Bruner et al., 2005; Smith et al., 2005; Silva et al., 2004; Stoate et al., 2000)

[51](Moorcroft et al., 2002; Rodgers, 2002)

[52](Barnett et al., 2004)

Three-quarters of the studies among farms were directed at or included issues related to land management (burning[53], crop management[54], grazing[55], habitat creation[56], pesticide use[57], and organic farming[58]) 27% to habitat metrics (habitat diversity[59], habitat juxtaposition[60]), 46% to non–cropland[61], 23% to habitat characteristics[62], and 14% to food resources.[63]

Many of the same factors that are important at the landscape level are relevant among farms, however, several other factors emerge as being important. These are:

1. Farms with greater habitat diversity, particularly fallow, crop stubble and non-crop habitats, have higher avian abundance.[64]

[53](Jansen et al., 1999, 2001; Lueders et al., 2006; Malan, 1998; Mentis and Bigalke, 1981)

[54](Aebischer and Potts, 1998; Cederbaum et al., 2004; Lokemoen and Beiser, 1997; Mason and Macdonald, 2000; Malan and Benn, 1999; Rodgers, 2002; Stoate et al., 2000; Warner et al., 1984)

[55](Ambrosini et al., 2002; Atkinson et al., 2002; Lusk et al., 2003; Jansen et al., 1999, 2001; Lueders et al., 2006; Malan, 1998; Söderström et al., 2001; Verhulst et al., 2004)

[56](Aebischer and Potts, 1998; Bradbury et al., 2004; Bruner et al., 2005; Cederbaum et al., 2004; Eraud and Boutin, 2002)

[57](Aebischer and Potts, 1998; Avilés and Parejo, 2004; Morris et al., 2005; Warner et al., 1984)

[58](Belfrage et al., 2005; Beecher et al., 2002; Freemark and Kirk, 2001)

[59](Belfrage et al., 2005; Eraud and Boutin, 2002; Freemark and Kirk, 2001; Fuhlendorf et al., 2002; Goławski and Dombrowski, 2002; Malan, 1998; Moreira et al., 2005; Warner, 1994; Woodhouse et al., 2005)

[60](Bruner et al., 2005; Bergin et al., 2000; Malan and Benn, 1999; Ratcliffe and Crowe, 2001b; Warner, 1994; Wilson et al., 1997)

[61](Aebischer and Potts, 1998; Barnett et al., 2004; Bradbury et al., 2004; Bruner et al., 2005; Eggebo et al., 2003; Eraud and Boutin, 2002; Freemark and Kirk, 2001; Goławski and Dombrowski, 2002; Henderson et al., 2000a,b; Klansek, 2002; Lokemoen and Beiser, 1997; Marshall et al., 2006; Mason and Macdonald, 2000; Moreira et al., 2005; Palmer et al., 2005; Parish et al., 1994; Rands, 1987; Rodgers, 2002; Silva et al., 2004; Smith et al., 2005; Stoate et al., 2000; Warner, 1994; Wilson et al., 1997)

[62](Atkinson et al., 2002; Barnett et al., 2004; Eggebo et al., 2003; Henderson et al., 2000a; Jansen et al., 2001; Lusk et al., 2003; Malan, 1998; Parish et al., 1994; Rands, 1987; Rodgers, 2002; Silva et al., 2004; Wilson et al., 1997)

[63](Browne and Aebischer, 2003; Cederbaum et al., 2004; Green, 1984; Jansen et al., 2001; Marshall et al., 2006; Moorcroft et al., 2002; Morris et al., 2005)

[64](Aebischer and Potts, 1998; Belfrage et al., 2005; Bollinger et al., 1990; Bradbury et al., 2004; Bruner et al., 2005; Eraud and Boutin, 2002; Freemark and Kirk, 2001; Goławski and Dombrowski, 2002; Klansek, 2002; Lokemoen and Beiser, 1997; Mentis and Bigalke, 1981; Moreira et al., 2005; Parish et al., 1994; Ratcliffe and Crowe, 2001b; Silva et al., 2004; Söderström et al., 2001; Stoate et al., 2000; Warner et al., 1984; Warner, 1994; Woodhouse et al., 2005; Yeatter, 1963)

2. The presence of pasture land, residual crops, and fallow are important, but heterogeneity in the height and density of vegetation is key to the utility of those habitats.[65]

3. In row crop dominated regions fallow, undersown crops and rotational grasslands are generally preferred to permanent grassland due to structural properties.[66]

4. Although fire is critical in rangelands for maintaining habitat heterogeneity, there are negative effects on avian abundance from frequent burning.[67]

5. Loss of hay-lands, switches from grass to alfalfa for hay, and earlier mowing dates depress nest and brood success.[68]

6. Collectively, organic farming practices increase avian diversity and abundance, particularly through increasing heterogeneity.[69]

7. Edge density and proximity of habitat types to one another can be important but are species dependen.t [70]

8. Habitat heterogeneity increases avian diversity but more homogeneous farms may be most beneficial to grassland and open area dependent species.[71]

[65](Ambrosini et al., 2002; Atkinson et al., 2004; Barnett et al., 2004; Bollinger et al., 1990; Cederbaum et al., 2004; Eggebo et al., 2003; Henderson et al., 2000a; Jansen et al., 2000, 2001; Klansek, 2002; Lueders et al., 2006; Lusk et al., 2003; Moorcroft et al., 2002; Moreira et al., 2005; Rodgers, 2002; Silva et al., 2004; Verhulst et al., 2004; Wilson et al., 1997)

[66](Aebischer and Potts, 1998; Barnett et al., 2004; Bradbury et al., 2004; Cederbaum et al., 2004; Eraud and Boutin, 2002; Freemark and Kirk, 2001; Fuller et al., 2001; Goławski and Dombrowski, 2002; Henderson et al., 2000a,b; Klansek, 2002; Lokemoen and Beiser, 1997; Mason and Macdonald, 2000; Silva et al., 2004; Smith et al., 2005)

[67](Jansen et al., 1999, 2001; Lueders et al., 2006; Malan, 1998; Mentis and Bigalke, 1981)

[68](Bollinger et al., 1990; Müller et al., 2005; Warner and Etter, 1989; Wilson et al., 1997; Yeatter, 1963)

[69](Beecher et al., 2002; Belfrage et al., 2005; Freemark and Kirk, 2001)

[70](Bruner et al., 2005; Fuhlendorf et al., 2002; Malan and Benn, 1999; Parish et al., 1994; Ratcliffe and Crowe, 2001b; Smith et al., 2005; Warner, 1994; Wilson et al., 1997)

[71](Belfrage et al., 2005; Eraud and Boutin, 2002; Freemark and Kirk, 2001; Malan and Benn, 1999; Mentis and Bigalke, 1981; Moreira et al., 2005; Parish et al., 1994; Wilson et al., 1997; Woodhouse et al., 2005)

9. The increasing presence of weeds and their seeds and arthropods have positive effects on survival and recruitment.[72]

10. Increasing use of chemical pesticides have negative impacts on avian abundance and diversity and is negatively correlated with nest success.[73]

5.1.4 Within farm

Approximately a quarter of the publications reviewed addressed questions measured at the scale of the farm. Of these 30% were related to reproduction[74], 23% habitat structure[75], 60% related to use of non-cropland[76], and 56% to use of cropland and the implications of intensive farm management.[77] Non-crop habitat in the form of rotational set-aside and game crops [78] and fallows[79] were shown to have positive effects on avian abundance and represented 14% and 7% of the publications respectively at the within farm scale. Meanwhile, 24% of publications illustrated positive effects of permanent pasture and/or

[72](Browne and Aebischer, 2003; Cederbaum et al., 2004; Green, 1984; Jansen et al., 2001; Marshall et al., 2006; Moorcroft et al., 2002; Morris et al., 2005)

[73](Aebischer and Potts, 1998; Avilés and Parejo, 2004; Beecher et al., 2002; Belfrage et al., 2005; Malan and Benn, 1999; Morris et al., 2005; Rodgers, 2002; Warner et al., 1984)

[74](Avilés and Parejo, 2004; Cederbaum et al., 2004; Donald et al., 2002; Hart et al., 2006; Hill, 1985; McMaster et al., 2005; Parek, 1997; Patterson and Best, 1996; Peach et al., 2004; Robertson et al., 1993; Wakeham-Dawson et al., 1998; Warner, 1984; Wilson et al., 1997)

[75](Barnett et al., 2004; Diaz and Telleria, 1994; Eggebo et al., 2003; Jansen and Crowe, 2002; Jansen et al., 2000, 2001; McMaster and Davis, 2001; Olsson et al., 2002; Rands, 1987; Robinson and Sutherland, 1999; Wakeham-Dawson et al., 1998)

[76](Barnett et al., 2004; Donald et al., 2001a, 2002; Eggebo et al., 2003; Evans and Smith, 1994; Henderson et al., 2000a,b; Leif, 2005; McMaster and Davis, 2001; McMaster et al., 2005; Orlowski, 2006; Parek, 1997; Parish et al., 1994; Peach et al., 2004; Patterson and Best, 1996; Rands, 1987; Robertson et al., 1993; Sage et al., 2005; Smith et al., 2005; Stoate et al., 2000, 2004; Wakeham-Dawson et al., 1998; Williams et al., 2000; Wilson et al., 1997; Woodhouse et al., 2005)

[77](Avilés and Parejo, 2004; Cederbaum et al., 2004; Diaz and Telleria, 1994; Donald et al., 2001a, 2002; Hart et al., 2006; Henderson et al., 2000a; Hill, 1985; Jansen and Crowe, 2002; Jansen et al., 2001; Mentis and Bigalke, 1981; Messick et al., 1974; Moorcroft et al., 2002; Orlowski, 2006; Ratcliffe and Crowe, 2001c; Robinson and Sutherland, 1999; Robertson et al., 1993; Stoate et al., 2000; Warner, 1984; Warner et al., 1984; Wilson et al., 1997; Woodhouse et al., 2005)

[78](Donald et al., 2001a; Henderson et al., 2000a,b; Sage et al., 2005; Stoate, 2002; Stoate et al., 2000)

[79](Leif, 2005; Orlowski, 2006; Stoate et al., 2000)

farmland taken out of production[80] and negative impacts from the intensification in pasture management were shown in 5% of the studies.[81]

The negative effect of the intensification of crop management on avian abundance and/or diversity was shown in 17% of studies[82] and 17% of studies showed positive associations with less intensive farming (*i.e.*, mixed farming).[83] Also, 21% of the publications illustrated that within farms the presence of crop stubble, waste grain, and standing cover crops were positively associated with avian abundance[84] and the negative impacts of excessive grazing and/or frequent pasture burning had negative effects in 10% of investigations[85]. Increased pesticide use was associated with negative population effect in 14% of studies.[86]

From these studies several additional generalities can be elucidated, these being:

1. In row crop systems cereal crops and stubble are of importance, particularly spring sown cereals.[87]

2. The presences of grassland, pasture, or fallow are associated with positive trends in abundance, diversity, and survival with grassland habitats most beneficial in

[80](Eggebo et al., 2003; McMaster and Davis, 2001; McMaster et al., 2005; Parek, 1997; Parish et al., 1994; Patterson and Best, 1996; Smith et al., 2005; Wakeham-Dawson et al., 1998; Williams et al., 2000; Wilson et al., 1997)

[81](Barnett et al., 2004; Woodhouse et al., 2005)

[82](Avilés and Parejo, 2004; Donald et al., 2001a, 2002; Orlowski, 2006; Ratcliffe and Crowe, 2001c; Stoate et al., 2000; Warner et al., 1984)

[83](Jansen and Crowe, 2002; Peach et al., 2004; Robertson et al., 1993; Wakeham-Dawson et al., 1998; Warner et al., 1984; Williams et al., 2000; Wilson et al., 1997)

[84](Bradbury et al., 2000; Cederbaum et al., 2004; Diaz and Telleria, 1994; Donald et al., 2001a; Evans and Smith, 1994; Moorcroft et al., 2002; Ratcliffe and Crowe, 2001c; Robinson and Sutherland, 1999; Wakeham-Dawson et al., 1998)

[85](Jansen and Crowe, 2002; Jansen et al., 2000, 2001; Mentis and Bigalke, 1981; Woodhouse et al., 2005)

[86](Hart et al., 2006; Cederbaum et al., 2004; Hill, 1985; Messick et al., 1974; Ratcliffe and Crowe, 2001c; Wilson et al., 1997)

[87](Diaz and Telleria, 1994; Donald et al., 2001b, 2002; Orlowski, 2006; Ratcliffe and Crowe, 2001b; Robinson and Sutherland, 1999)

grassland regions and rotational grasslands, fallows, non-cropland, and game crops best in row crops dominated systems.[88]

3. Within farm habitat diversity in both space and time, although dependent upon species, is influential and has an optimum.[89]

4. Structural heterogeneity of habitats influences its suitability.[90]

5. Selection of foraging habitat is dependent upon both food availability, foraging efficiency, and predator avoidance.[91]

6. Fields untreated with pesticides are preferred over treated fields [92] or have higher arthropod abundance and nest or brood success than treated fields.[93]

5.1.5 Within field

Within field studies included 33% that investigated the indirect effects of pesticide use[94], 56% related to foraging success and/or food availability[95], 37% were associated with the physical characteristics of habitats, and 30% studied reproductive success.[96]

[88](Avilés and Parejo, 2004; Donald et al., 2001a; Freemark and Kirk, 2001; Henderson et al., 2000a; Hill, 1985; Jansen et al., 2001; McMaster and Davis, 2001; McMaster et al., 2005; Olsson et al., 2002; Parish et al., 1994; Peach et al., 2004; Rands, 1987; Sage et al., 2005; Smith et al., 2005; Stoate, 2002; Wakeham-Dawson et al., 1998; Warner, 1994; Wilson et al., 1997; Yeatter, 1963)

[89](Barnett et al., 2004; Bradbury et al., 2004; Eggebo et al., 2003; Jansen et al., 2000; Leif, 2005; Mentis and Bigalke, 1981; Moorcroft et al., 2002; Orlowski, 2006; Parek, 1997; Parish et al., 1994; Ratcliffe and Crowe, 2001c,b; Robertson et al., 1993; Warner, 1984; Williams et al., 2000)

[90](Donald et al., 2002; Jansen et al., 2000; Olsson et al., 2002; Patterson and Best, 1996; Wakeham-Dawson et al., 1998; Wilson et al., 1997)

[91](Diaz and Telleria, 1994; Donald et al., 2001b; Robinson and Sutherland, 1999)

[92](Robinson and Sutherland, 1999)

[93](Hart et al., 2006; Hill, 1985; Messick et al., 1974)

[94](Basore et al., 1987; Boatman et al., 2004; Borg and Toft, 2000; Brickle et al., 2000; Chiverton, 1999; Moreby and Southway, 1999; Pulliainen, 1984; Rands, 1985; Taylor et al., 2006)

[95](Basore et al., 1987; Boatman et al., 2004; Borg and Toft, 2000; Brickle et al., 2000; Britschgi et al., 2006; Butler and Gillings, 2004; Chiverton, 1999; Moorcroft et al., 2002; Moreby and Southway, 1999; Pulliainen, 1984; Rands, 1985; Robinson and Sutherland, 1999; Stoate et al., 2004; Taylor et al., 2006; Thomas et al., 2001)

[96](Boatman et al., 2004; Broyer, 2003; Bradbury et al., 2000; Brickle et al., 2000; Britschgi et al., 2006; Chiverton, 1999; Green et al., 1997; Riley et al., 1992)

Pesticide use was associated with smaller brood size, lower availability of preferred foods, and lower chick survival in all studies investigating the within field effects of pesticides.[97] Additionally, management of haylands that includes leaving refugia of unmowed areas was associated with greater recruitment in ground nesting species.[98]

Of articles concerning food availability, 60% included the effects of pesticide use[99] and 40% the effects of habitat characteristics and management on food availability.[100] Physical characteristics of fields determined their suitability, particularly through spatial and temporal differences in vegetational structure[101], but also through field size and edge proximity.[102]

Despite some species–specific differences in studies at the within field level, the overall results of these studies are generally unequivocal. The effects of field management be can be summarized as follows:

1. Uncultivated or unharvested areas adjacent or within fields increases feeding efficiency and survival.[103]

2. The spatial and temporal structure of the vegetation in crop fields, pastures, and uncropped land determines its utility to a species.[104]

[97] (Basore et al., 1987; Boatman et al., 2004; Borg and Toft, 2000; Brickle et al., 2000; Chiverton, 1999; Moreby and Southway, 1999; Pulliainen, 1984; Rands, 1985; Taylor et al., 2006)

[98] (Broyer, 2003; Brickle et al., 2000; Green et al., 1997)

[99] (Basore et al., 1987; Boatman et al., 2004; Borg and Toft, 2000; Brickle et al., 2000; Chiverton, 1999; Moreby and Southway, 1999; Pulliainen, 1984; Rands, 1985; Taylor et al., 2006)

[100] (Britschgi et al., 2006; Butler and Gillings, 2004; Moorcroft et al., 2002; Robinson and Sutherland, 1999; Stoate et al., 2004; Thomas et al., 2001)

[101] (Best, 2001; Britschgi et al., 2006; Butler and Gillings, 2004; Devereaux et al., 2004; Hughes et al., 1999; Morris et al., 2004; Odderskær et al., 1997; Riley et al., 1992; Whittingham et al., 2006)

[102] (Best et al., 1990; Hughes et al., 1999; Sparks et al., 1996)

[103] (Best et al., 1990; Broyer, 2003; Bradbury et al., 2004; Brickle et al., 2000; Devereaux et al., 2004; Green et al., 1997; Sparks et al., 1996; Thomas et al., 2001)

[104] (Best, 2001; Britschgi et al., 2006; Butler and Gillings, 2004; Devereaux et al., 2004; Hughes et al., 1999; Morris et al., 2004; Odderskær et al., 1997; Riley et al., 1992; Whittingham et al., 2006)

3. The intensive management of farmland, particularly the application of pesticides, reduces the abundance of food resources, particularly arthropods that are preferred for food, while increasing the dominance of the remaining arthropod community by less preferred arthropods that are also nutritionally inferior. These reductions in food availability and quality are responsible for reduced survival and recruitment in birds within these agroecosystems.[105]

5.1.6 Summary

Although much of the research reviewed suggests species-specific relationships with agricultural land management and avian populations, there are some generalities that appear to occur across scales. Particularly obvious is the important role that heterogeneity plays from landscape composition to within-field vegetative structure [*sensu* Benton et al. (2003)]. Furthermore, optimal heterogeneity in farmlands tends to be higher for species that are associated most with row crop agriculture (early successional species) in comparison to grassland species that do best where a higher percentage of natural habitat exists within landscape or farms.

The importance of heterogeneity occurs not only spatially, but temporally, since the dynamic nature of agricultural management results in large-scale and rapid changes in land cover. Polarization and mechanization of agriculture has resulted in increasingly synchronized management among farms, as well as monocultural landscapes, resulting in increased landscape homogeneity in both space and time. Concurrently, a dependence upon inorganic fertilizers has allowed for the extensive use of double cropping systems, reducing the area in fallow and cover crops, while intensified livestock husbandry via supplemental feeding has reduced the need of rotational grasslands, all of which are preferred habitats for birds associated with agroecosystems.

[105](Basore et al., 1987; Boatman et al., 2004; Borg and Toft, 2000; Brickle et al., 2000; Britschgi et al., 2006; Chiverton, 1999; Moreby and Southway, 1999; Pulliainen, 1984; Rands, 1985; Taylor et al., 2006)

The important role of ephemeral habitats for the avian community in agroecosystems is often not appreciated in the development of dual-purpose governmental policies designed for the optimization of agricultural production and the conservation agroecosystem biota. In set-aside programs in the European Economic Union, the Conservation Reserve Program (CRP) in the United States, and the Permanent Cover Program in Canada, agricultural land is taken out of production to reduce agricultural subsidies, with the intended side benefit being an increased suitability of agricultural land for species dependent upon those systems. However, the effect of these programs on farmland and grassland dependent birds has been mixed due to a lack of planning in placing and managing these habitats, particularly related to heterogeneity in landscape context and vegetative structure, although where species-specific needs are met these programs are successful (McMaster and Davis, 2001; Ryan, 2000; Sotherton, 1998).

The presence of woodlands and woody vegetation within agroecosystems often is correlated with increased avian diversity since it supplies habitats for woodland and woodland edge dependent species. However, the presence of woody vegetation within agroecosystems is often related to the absence or lower abundance of species dependent upon grasslands or open field systems, which are generally the species of highest conservation priority. Subsequently, measures of conservation value of agroecosystems should not depend upon overall avian biodiversity but rather abundance of species dependent upon those systems and prioritized for conservation. Moreover, the encroachment of woody vegetation and afforestation plans need to be included in regional conservation assessments and management plans for agroecosystems.

Based upon this analysis, much of the intensification in within farm row crop and grazing management discussed in prior chapters can be directly attributed to have caused declines in populations of birds in agricultural systems. Of particular note are the increased use of pesticides, reduced availability of waste grain and weed seeds in winter, and intensified stock management with the associated increase use of silage and its production.

The effects of pesticides on food availability to birds is particularly important since the use of insecticides and fungicides directly kill arthropods, herbicides kill weeds that arthropods are dependent upon, and the cumulative effect of pesticide use increases the dominance of the arthropod community in row crop fields towards species that are the least desirable food species for birds.

The extensive use of herbicides has, by reducing the abundance of weeds within row crops, significantly decreased the availability of weed seeds to birds. Furthermore, increased efficiency in grain harvesting has greatly reduced the abundance of waste grain in harvested fields. Also, the increased use of double cropping systems has greatly decreased the availability of weed seeds in waste grain during winter through soil preparation or crop growth.

Livestock husbandry has become increasingly intensified, more frequently depending upon supplemental feeding. This often leads to higher stocking rates, but also requires the production of supplemental feed. Of particular importance is the increased intensification of management of hay fields. Hay fields have increasingly been converted to monocultures of forage crops and fertilizers are used more often to promote more and faster growth. Together, these practices allow for earlier and more frequent cutting of hay, which has led to significant negative impacts on recruitment by species using those habitats for breeding.

Across continents, the changes in the management of agroecosystems has produced similar patterns in both the evolution of land management and their effects on the avian communities in those systems. Based upon the literature the major impacts of agricultural land management can be grouped into those that affect the diversity of habitat across scales and those that influence habitat quality (Fig. 5.27). Although the magnitude and sensitivity of these relations may be species and/or system dependent, they can be sufficiently generalized to illustrate the common patterns illustrated across systems (Fig. 5.27).

The conversion of natural habitat, or the conversion of one land use type to another, leads to an asymptotic rise in habitat heterogeneity that decreases as conversion passes a threshold, as do the effects of disturbances by fire and grazing pressure (Fig. 5.27). As the diversity of land cover and/or crop types increase, the heterogeneity of habitat increases linearly, while habitat heterogeneity decreases nonlinearly as farm and field sizes increase (Fig. 5.27).

The quality of habitat types is dependent upon the management of those habitat types and is nonlinear in its relationships. Both the frequency of mowing and the increasing dependence on planted pastures over natural pastures have a negative impact on habitat quality, which increases past a threshold (Fig. 5.27). Even low levels of pesticide application can greatly reduce habitat quality, as does the small scale introduction of woody vegetation into agroecosytems for species dependent on open habitats (Fig. 5.27). Earlier mowing dates reduces habitat quality for breeding and is detrimental when mowing occurs prior to the termination of nest and/or fledging (Fig. 5.27).

Although in all the reviewed systems these generalized impacts of agroecosystem management are evident across avian species, ground nesting species with precocial young have been some of the most negatively impacted. In particular the Galliformes (pheasants, partridges, grouse) associated with farm and/or grassland systems, but also Pteroclidiformes (sandgrouse) and Gruiformes (bustards). Because the Galliformes have economic and cultural significance as gamebirds they have been some of the most studied birds and, due to this and their sensitivity to changes in management of agroecosystems, they serve as an excellent study group to assess the effects of these changes on the underlying ecological process that drive population dynamics.

5.2 Agroecosystem ecology and gamebirds

The management of agricultural and grazing land reviewed has, and continues to have, important influences on the distribution and abundance of gamebirds (Galliformes) within temperate agroecosystems throughout the world by determining the amount and quality of habitat across spatial and temporal scales. The relationship between agricultural land management and gamebird populations has long been realized and the understanding of this relationship was one of the major impetuses for the development of modern wildlife management (Leopold, 1933).

As illustrated in Chapter 3, there have been significant changes throughout western Europe in the management of agricultural lands. These changes include the intensification in the management of arable land, land abandonment, regional specialization stemming from the cessation of traditional mixed management systems, reforestation, and afforestation (Díaz et al., 1997; Eichhorn et al., 2006; Gellrich et al., 2007; Fuller, 1987; Mottet et al., 2006; Pinto-Correia and Vos, 2004; Potter, 1997; Primdahl, 1990; Roura-Pascual et al., 2005; Stoate, 1996; Suárez et al., 1997; Tasser et al., 2007). Combined, these changes in land-use and management have been responsible for population declines and range contractions of gamebirds throughout Europe (Anglestam et al., 2000; Báldi and Faragó, 2007; Calladinde et al., 2002; De Leo et al., 2004; Draycott et al., 2002; Hill and Robertson, 1988; Hudson, 1995; Klansek, 2002; Kurki et al., 2000; Panek, 2005; Pearce-Higgins et al., 2007; Potts, 1980, 1986; Potts and Aebischer, 1994; Vargas et al., 2006).

In Europe, traditional land use practices that incorporated and integrated row crop management, animal husbandry, and forestry practices have been critical in maintaining overall biological diversity in those systems, including abundant gamebird populations, for centuries to millenia (Berendse et al., 2004; Eichhorn et al., 2006; Pinto-Correia and Vos, 2004; Potts, 1986, 1997). In fact, the widespread presence of gray and red-legged partridge in Europe stems from range expansion (facilitated by stocking) due to the 7,000+ year agriculturalization of Europe (Potts, 1986). Additionally, pheasants were introduced into favorable habitat in Europe by the Romans, and their range further expanded by further introductions (Hill and Robertson, 1988). The abandonment of agricultural management that allowed for the expansion and/or introduction of gamebirds in Europe forms the basis of the crisis in the loss of biological diversity throughout Europe, including declining gamebird populations (Berendse et al., 2004; Krebs et al., 1999; Potts, 1986, 1997).

Because of the formerly integrated nature among farming, grazing, and forestry in Europe the changes in land-use that have occurred there over the past half century have impacted gamebirds in both agricultural and forest habitats. For, example, in lowland regions of Europe dominated by row crops and grazing a decrease in coppicing of forest stands for fuel, building materials, and fodder has been detrimental to pheasant populations by reducing shrub cover (Hill and Robertson, 1988) and to gray partridges by allowing the development of woody overstories in residual areas (Potts, 1986). During the same period in montane areas and Fennoscandia the conversion of cropland to pasture, reduced forest grazing, land abandonment, and halting of clear cutting have negatively impacted both gray partridge and grouse populations (Anglestam et al., 2000; Novoa et al., 2002; Pearce-Higgins et al., 2007; Söderström et al., 2001; Swenson and Angelstam, 1993).

The changes in agricultural management in Europe are strongly correlated to declines in farmland birds throughout the continent, including gamebirds (Donald et al., 2001b, 2006). Potts (1980, 1986) showed how these changes have reduced landscape suitability to gray and red-legged partridges, while multiple other studies throughout Europe have confirmed this (Báldi and Faragó, 2007; Klansek, 2002; Moreira et al., 2005; Novoa et al., 2002; Söderström et al., 2001; Vargas et al., 2006; Virkkala et al., 2004). In all these studies a critical factor associated with population declines has been the loss of extensive cereal cultivation, and/or extensive grazing practices, and the landscape heterogeneity they produce.

In North America, agriculture associated with European colonization drove extensive forest clearing in the eastern United States during the 18^{th} through the early 20^{th} centuries and the conversion of the eastern most prairies starting in the 19^{th} century and later for areas further west (Chapter 3). This process of land conversion had positive effects for native species, such as the Northern bobwhite (*Colinus virginianus*) allowing for a large range expansion and population increases (Kabat and Thompson, 1963; Rosene, 1969; Stoddard, 1931). The widespread expansion of agriculture in North America also created favorable conditions for the introduction of the pheasant in 1882 (Hill and Robertson, 1988) and gray partridge in the early 20^{th} century (Potts, 1986).

As in Europe, the intensification of agricultural management in North America has resulted in considerable declines in populations and range for both native and introduced gamebird species (Murphy, 2003). For example, the North American Breeding Bird Survey has shown annual range-wide population declines (1966-2005) in the United States of -0.9% for pheasants, -5.9% for Greater prairie-chickens, -3.0% for Northern bobwhite quail, and -1.8% for gray partridge (Sauer et al., 2005). In the case of the greater prairie-chicken (*Tympanuchus cupido*) habitat conversion, combined with intensified farm management, has led to severe population declines, including the extinction of its eastern race (*T. c. cupido*), however, during the initial stages of prairie conversion to row crops this species exhibited an approximate doubling of its range (Schroeder and Robb, 1993). Also, the clearing of forests in the upper-midwestern states and south-central Canada facilitated a similar range expansion for the sharp-tailed grouse (*Tympanuchus phasianellus*) (Johnsgard, 2002).

The positive effect of land conversion from forest and grassland to agriculture on greater-prairie chickens can be seen by comparing the estimated year of maximum abundance to the proportional land area in crops and farms. Using estimates from Schroeder and Robb (1993) for the year of maximum abundance for 13 states for which the majority of the state area was occupied by greater-prairie chickens, and comparing the proportional area of each state in agriculture [farmland by state (USDA, 2006), area in farms by county by state (U.S. Census Bureau, 1850-2000)] (Table 5.1), suggests that a mean optimal area in cropland and farmland is between ∼30% and ∼50% (Fig. 5.5). These values are consistent with observations of the expansion and decline of greater-prairie chickens in North Dakota (Johnson and Knue, 1989) and Canada (Houston, 2002).

Subsequent research has illustrated the strong influence that landscape structure has on the presence and abundance of both Greater prairie-chickens and Lesser prairie-chickens (*Tympanuchus pallidicinctus*), as well as Sharp-tailed grouse (*Tympanuchus phasianellus*). In these studies the presence or stability of populations was associated with a decreased dominance of rowcrops, decreased landscape heterogeneity and continuity of grasslands (Ammann, 1963; Fuhlendorf et al., 2002; McDonald and Reese, 1998; Niemuth, 2000; Ryan et al., 1998; Yeatter, 1963). Similarly, declines in the abundance of Northern bobwhite quail have been correlated to large-scale trends in land use in the United States, although these analysis also suggest that farm-scale management is equally influential in determining populations (Brady et al., 1998; Roseberry and Sudkamp, 1989).

Similarly, in South Africa the introduction of European agriculture has both favored some gamebird species and facilitated range expansions [Helmeted guineafowl (*Numida meleagris*), Swainson's spurfowl (*Pternistis swainsonii*)] (Little et al., 2000). As in Europe and North America, the increasing intensification in row crop and pasture management discussed in Chapter 3 can have significant negative effects on multiple species including Helmeted guineafowl (Malan and Benn, 1999; Ratcliffe and Crowe, 2001a), Swainson's spurfowl (Jansen and Crowe, 2002), and Red-winged (*Francolinus levaillantii*) and Gray-winged francolins (*Francolinus africanus*) (Jansen et al., 1999; Mentis and Bigalke, 1981).

The landscape heterogeneity that row crop cultivation introduced into savanna woodlands, and the associated increase in food resources benefits helmeted guineafowl, however, the effect of cultivation becomes negative as row crops become increasingly dominant (Malan and Benn, 1999; Pero and Crowe, 1996; Little et al., 2000; Ratcliffe and Crowe, 2001a,b,c). Pero and Crowe (1996) found declining populations of helmeted guineafowl in KwaZulu-Natal, South Africa to be associated with a higher proportion of

Table 5.1: Estimated year of peak abundance of greater prairie-chickens by states where the majority of the state was occupied (Schroeder and Baydack, 2001) and proportion of each state in crops (USDA, 2006) and in farms (assessed by county) (U.S. Census Bureau, 1850-2000) for that time period. Differences in the two metrics are due to differences in defining farms between the two data sets, as well as the averaging of county values from the U.S. Census data.

State	*Year of peak abundance*	*% of state in farmland (USDA)*	*% of state in farmland (US Census)*
Illinois	1860	56.4	32.4
Indiana	1860[a]	54.9	22.7
Iowa	1880	68.7	45.6
Kansas	1877	40.7	19.2
Michigan	1930	27.3	28.7
Minnesota	1900	47.1	41.0
Missouri	1865	48.6	17.5
Nebraska	1875	20.1	19.0
North Dakota	1920	80.0	46.1
Ohio	1850[b]	62.7	37.2
Oklahoma	1910	64.5	33.1
South Dakota	1890[c]	23.1	14.7
Wisconsin	1890	40.0	27.9
mean(se)		48.8(5.1)	29.6(3.0)

[a]The year of peak abundance was inferred to be similar to Illinois
[b]The year of peak abundance was inferred to be in the decade prior to that in Illinois and Indiana
[c]Schroeder and Baydack (2001) give 1880 as the date of maximum abundance, however, since this preceeded statehood for South Dakota (1889) comparable agricultural data are not available. For this reason data from 1890 were used.

farmland in row crops ($\sim$24%) compared to stable populations ($\sim$13%), while extirpated populations were associated with higher proportions of afforested land ($\sim$29%). Moreover, extirpated populations were associated with less area dedicated to livestock grazing ($\sim$53%) compared to stable populations ($\sim$76%) (Pero and Crowe, 1996).

Also in farmland of KwaZulu-Natal, Malan and Benn (1999) found a significant relationship between guineafowl populations and land use. An important factor correlated with low populations numbers was the amount of pesticides used on farms, however, equally important was the diversity of land uses within a farm, particularly the area of savanna woodlands. This relationship is evident in the relationship between guineafowl populations and the ratio of the area of savanna woodlands to cereal fields (Fig. 5.6). In general, guineafowl populations increased as the area of savanna woodland increased in relation the area planted in cereals.

As reviewed above, agricultural land management influences avian diversity and abundance at multiple temporal and spatial scales, however, these impacts inherently stem from the effects that land use has on survival during the breeding and non-breeding seasons and recruitment. Regardless of the species or the location, an understanding of the effects of land use on gamebirds populations requires an understanding of how land management influences survival and recruitment (Dobson et al., 1988; Potts, 1986).

The management of agricultural lands influences gamebird populations by determining the amount and quality of required habitats over the course of a year, the abundance, availability, and quality of food, and the populations and hunting efficiencies of predators. Based upon the life history of gamebirds, the annual life cycle can be divided into the context of the non-breeding, nesting, and brood rearing periods. Outside of the breeding season, cover for thermoregulation and predator avoidance, and food abundance and availability can be critical for survival. During the breeding season the amount of nesting habitat and its quality influences territory size and determines the number of breeding females, while combined with predator effects, influences hen and nest survival. Similarly, the amount of foraging habitats for broods and the availability of preferred forage in those habitats are critical for chick survival (Hill and Robertson, 1988; Potts, 1986; Rands, 1988).

5.2.1 Non–breeding season

The period following the fledging of broods until the establishment of courting and/or nesting territories in the spring constitutes the non–breeding season or over-winter period. Mortality in gamebirds during this period is highly dependent upon winter temperature and snow cover and is subsequently variable due to inter-annual variations in winter severity (Carroll, 1990; Church, 1992; Dumke and Pils, 1973; Evrard, 1996; Gabbert et al., 1999; Homan et al., 2000; Panek, 1990; Perkins et al., 1997; Potts and Faragó, 2000; Riley, 1995). The magnitude of those effects, however, can be dependent upon, and mitigated by, the amount and quality of preferred habitats and the availability of food resources. This is particularly relevant because many of the studies of wintering ecology point to the importance of predation as the cause of mortality which emphasizes the importance of the availability of food and habitat in influencing mortality (Carroll, 1989; Church, 1980, 1992; Dumke and Pils, 1973; Perkins et al., 1997; Potts and Faragó, 2000; Smith et al., 1999).

The amount of snow and its persistence is related to mortality in both gray partridges and pheasants. Potts (1986) showed a significant relationship between months of snow cover and winter mortality in gray partridges where regions with 5 months of snow cover had 39% higher mortality compared to areas with no snow cover. For Poland, Panek (1990) illustrated increasing mortality of gray partridges associated with increasing snow depth in the previous half month (Fig. 5.7) and data from Finland (Siivonen, 1953) illustrate the importance of snow depth in gray partridge winter survival, particularly in late winter (Potts, 1980). Also, for pheasants in North Dakota Homan et al. (2000) found that a 2.5 cm increase in snow depth increased mortality by 8% and Kabat and Thompson (1963) found a strong relationship between snow cover and winter mortality of bobwhite quail in Wisconsin (Fig. 5.8).

Temperature can have a negative effect on winter survival through extremes in minimum temperatures. For gray partridges in North Dakota, Carroll (1990) showed a negative influence of days below −17°C, and in Poland, Panek (1990) found decreasing survival with decreasing temperatures (Fig. 5.9) and Potts and Faragó (2000) in Hungary for decreasing temperatures (Fig. 5.10). Evrard (1996) illustrated a similar relationship for pheasants in Wisconsin and Homan et al. (2000) found pheasant survival in North Dakota increased 6% for each degree Celsius increase in mean minimum temperature. The amount and persistence of snow cover can be correlated and subsequently interactive in their effect on winter mortality as seen with pheasants in both Iowa (Gabbert et al., 1999) and South Dakota (Perkins et al., 1997) (Fig. 5.11).

In both Europe and North America studies of winter habitat use in gamebirds indicate a preference for habitats that, through changes in land management practices, have been reduced in area or degraded. Farming practices have reduced the amount of winter stubble, fallow, wetlands, and hedges which are preferred winter habitats for pheasants and gray and red-legged partridges (Aebischer, 1997; Anderson, 2002; Bruner et al., 2005; Carroll, 1989; Church and Porter, 1990; Faragó, 1998; Gabbert et al., 1999; Gates and Hale, 1974; Gatti et al., 1989; Grubešić et al., 2006; Kaiser, 1998; Klansek, 2002; Potts, 1986; Smith et al., 1999; Turtola, 1998; Whiteside and Guthery, 1983). The loss of these preferred wintering habitats, when placed in the context of the mortality associated with extremes in winter weather, suggest that land management practices potentially can be a mitigating factor of winter mortality.

In the case of pheasants in the United Kingdom a reduced frequency of small farm woodlands, coppices, wetlands, and hedgerows (preferred wintering habitats) can lead to higher winter densities and increased mortality due to density dependent factors, however, because of high numbers of released captive raised birds and shooting actually mortality is difficult to determine (Hill and Robertson, 1988). At Seefeld Estate in Austria where an abundance of optimal wintering habitat was available (small woodlands, coppice, wetlands), as well as predator managment and supplemental feeding, winter hen survival was very high (97±3.02%) between November and March (136 days) (Anderson, 2002). Similarly in the USA, wintering pheasants showed a preference for wetlands, shrub cover, and shelterbelts (Gabbert et al., 1999; Gates and Hale, 1974; Gatti et al., 1989; Smith et al., 1999; Whiteside and Guthery, 1983), all habitats that have been reduced by agricultural intensification (Chapter 3).

For gray partridges, Carroll (1989) found greater use of farmsteads when snow depths were greater than 10cm, which is consistent with Church and Porter (1990) and Schulz (1980) who found greater than expected use of farmsteads by gray partridges in winter. These shifts in habitat preferences during extreme weather suggest that in both Europe and North America the effects of increasing winter severity discussed above, may be an overriding factor in both pheasant (Gates and Hale, 1974) and gray partridge (Carroll, 1990; Church, 1992; Potts and Faragó, 2000) population dynamics which, may be exacerbated where sufficient wintering cover is not available.

The increased use of wooded areas during periods of low temperature and deep snow cover is of consequence since the abundance of many species associated with farmland or open habitats is negatively associated with the presence of woodlands. This is particularly relevant to gamebirds since a large proportion of winter mortality in pheasants (Dumke and Pils, 1973; Riley and Schultz, 2001), bobwhite quail (Rollins and Carroll, 2001), prairie grouse (Schroeder and Baydack, 2001), and gray partridge (Carroll, 1989; Church, 1992; Potts, 1986; Schulz, 1980; Weigand, 1980) has been shown to be due to predation by raptors that depend upon trees as roost sites and perches.

In late winter, Snyder (1985) found the majority of late winter mortality of pheasants in Colorado to be by raptors and all birds depredated by raptors to be within 0.4km of wooded areas. Also, Roberston (1997) discussed the relationship between woodlands and pheasant mortality by raptors, attributing tree roosting by pheasants in the United Kingdom to a paucity of raptors. Moreover, depredation of gray partridge by raptors in winter was attributed to use of farmsteads during severe winter weather (Carroll, 1989; Church, 1992; Schulz, 1980) and Panek (1990) illustrated a negative correlation between winter mortality and distance from woodlands (Fig. 5.12).

Although often used less than in relation to their availability, crop fields are still used extensively by gamebirds outside the breeding season in Europe, North America, and South Africa and their management may have important effects on their utility in winter to gamebirds by reducing cover and food resources (Bruner et al., 2005; Carroll, 1989; Church and Porter, 1990; Gabbert et al., 1999; Gatti et al., 1989; Homan et al., 2000; Jansen and Crowe, 2002; Kaiser, 1998; Ratcliffe and Crowe, 2001c). Autumn sown crops, increased harvest efficiency, and post-harvest herbicide can reduce the both the vegetative structure and food availability during the over-winter period. The sowing of autumn cereals and associated use of herbicides removes crop stubble and reduces weeds and their seeds (British Trust for Ornithology, 2002; Draycott et al., 1997; Ewald and Aebischer, 2000; Potts, 1986; Pulliainen, 1984; Rodgers, 2002). Moreover, where crop stubbles are grazed, burned, or tilled post-harvest vegetative cover and seed availability are greatly reduced (Baldassarre et al., 1983; Potts, 1986).

In the USA, an increasing use of dwarf wheat, more powerful combines, and post-harvest herbicides reduces stubble and weed cover and has been shown to reduce pheasant abundance in those field by 80%, whereas tillage of fields reduces abundance by 90% (Rodgers, 1999, 2002). In Europe, autumn sowing, herbicide use, tilling, and burning of stubble reduces the utility of crop fields to gray partridge in a similar manner (Bruner et al., 2005; Potts, 1986; Pulliainen, 1984). Potts (1986) showed that these practices have a relatively small impact on overall survival, however, Pulliainen (1984) expressed concern that such agricultural changes may increase gray partridge mortality where winters are most extreme.

The abundance of waste grain and weed seeds have declined considerably in farm fields in Europe as a function of herbicide use and increased harvest efficiency (Browne and

Aebischer, 2003; Draycott et al., 1997; Potts, 1970, 1986; Pulliainen, 1984); however, in the USA available waste grain has remained relatively constant with increasing yields offsetting increased harvest efficiency (Baldassarre et al., 1983; Krapu et al., 1995; Reinecke and Krapu, 1986; Warner et al., 1985, 1989). In the United Kingdom, by the late 1950's harvest efficiency left approximately 30-45 kg of small cereal grains per hectare (Murton et al., 1963), a number similar to present values (British Trust for Ornithology, 2002). Comparatively, in the Unitd States during the early 1940's approximately 250 kg/ha of small grain cereals were left in crop stubbles (Washington State; Yocum 1943), which dropped to approximately 35 kg/ha by 1975 (North Dakota; Hofman 1978).

By the late 1950's waste corn was in excess of 250 kg/ha in the USA (Nebraska) and remained relatively constant through the 1980's (Iowa, Nebraska, Texas) (Baldassarre et al., 1983; Krapu et al., 1995; Warner et al., 1985, 1989). Losses for corn in Nebraska for 1990 were estimated as >650 kg/ha (Krapu et al., 1995) while by 2002 with the introduction of precision agriculture losses of 160 kg/ha were considered the maximum acceptable in Ontario (OMAFRA Staff, 2002). Trials of harvest efficiency of combines in Argentina indicate a loss of 385 and 135 kg/ha for corn and wheat, respectively, with desired increases in efficiency equating to losses of 308 kg/ha for corn and 108 kg/ha (PRECOP, 2005).

Aside from decreasing loss of grains in harvesting, the availability of waste grain can be significantly reduced by post harvest management of fields. Post harvest disking or reduced tillage practices can reduce waste grain availability by up to 80%, grazing >80%, and deep plowing by 97% (Baldassarre et al., 1983; Warner et al., 1989). In the same manner, in Illinois Roseberry et al. (1979) found that after 1966 there was a large change in post harvest management with harvested corn stubble being nearly 100% tilled or shredded in the fall. In north Texas post-harvest management practices and waterfowl foraging reduced waste corn to 22 kg/ha in late winter, however, due to differences in management many fields maintained waste corn abundance greater than 60 kg/ha (Baldassarre et al., 1983).

5.2.2 Breeding season

The effects of agricultural land management on breeding success of gamebirds is manifested in both nesting and brood-rearing. The amount and quality of nesting habitat determines the number of nests and the success of those nests (hatching success and survival of the incubating adult) while the amount of foraging habitat and the qualities of those habitats (cover and food availability and quality) influences brood survival. The intensification of agriculture has negatively impacted recruitment in gamebirds in several ways; 1) by removing nesting habitat, particularly through the removal of linear habitats along field edges stemming from field amalgamation, 2) altering predator abundance and hunting efficiency 3) destroying nests and killing incubating adults through farm practices (principally mowing), and 4) reducing arthropod populations, directly through insecticide and fungicide use, and indirectly through reduction in weed populations via herbicide use.

Nesting

The intensification of agriculture has an important influence on the number of breeding females, and subsequently the number of nests, by determining the amount of suitable nesting habitat. Although requirements for nesting are species-specific, the general trends of global agriculture have reduced the amount of land that is not in production, while also trying to maintain crop lands in continuous production. The result has been a loss of idle and residual areas, such as hedgerows, fenclines, roadsides, ditches, right-of-ways, wetlands or other areas difficult to cultivate or graze; all which are important for both daily use and reproduction by multiple gamebird species across the globe [gray partridge (Aebischer and Ewald, 2004; Potts, 1980, 1986; Carroll, 1989); francolins (Jansen et al., 2001); pheasants (Anderson, 2002; Bliss, 2004; Hill and Robertson, 1988; Smith et al., 1999; Snyder, 1984; Warner et al., 1987); bobwhite (Roseberry et al., 1979; Taylor et al., 1999); helmeted guineafowl (Prinsloo, 2003)].

Subsequently, the net effect of changes in farmland management is that the area of undisturbed habitat for egg laying and incubation has decreased on farmlands and these changes have forced nesting birds to use less suitable areas such as crop lands that put nests at risk to farm operations. Also, nesting birds in remaining residual areas may be more susceptible to predation due to the size and/or configuration of those habitats. Moreover, farm management can have substantial impacts on the vegetative structure of habitats and subsequently alter the suitability for nesting of, and nest success in, those habitats.

In western Europe agricultural intensification has led to extensive loss and degradation of hedgerows, as well as other residual habitats (Bertacchi and Onnis, 2004; Potts, 1986; Primdahl, 1990; Rands, 1987), reducing nesting and brood-rearing habitat and has been critical to declines in populations of pheasants, gray partridge and red-legged partridge in the region (Aebischer and Ewald, 2004; Bruner et al., 2005; Green, 1984; Hill, 1985; Potts, 1980, 1986; Rands, 1987). Furthermore, the quality of remaining hedgerows and field borders have been reduced in quality as management of those habitats have changed, particularly allowing development of woody vegetation in lieu of shrub cover (Hill and Robertson, 1988; Potts, 1986).

For both gray partridges and red-legged partridges the amount of hedgerow habitats have been shown to be the prinicipal factor determining nest densities (Potts, 1980, 1986; Rands, 1987). For gray partridge, Potts (1986) demonstrated where hedgerows are abundant breeding pairs space themselves out at a minimum distance of 200m and for both gray and red-legged partridge nesting densities reach a maximum where there is approximately 8km of optimal hedgerow for nesting per km^2 (Potts, 1980). The density dependent relationship between gray partridge reproduction and hedgerow density is well illustrated by data from Petrjanos (1990) [in Gossow et al. (1992)] from Lower Austria (Fig. 5.13).

Although not well quantified, the abandonment and intensification of agricultural lands in North America (Chapter 3) has led to considerable loss of hedgerows and field edges which are correlated with decreasing gamebird populations (Potts, 1986). For example, Murtha (1967) illustrated declining covey sizes of gray partridges in New York in relation to the proportional area of the landscape in agriculture (Fig. 5.14) and Kabat and Thompson (1963) found a correlation between declining northern bobwhite populations in Wisconsin with field amalgamation and the loss of hedgerows (Fig. 5.15). Data from Lower Austria (Petrjanos, 1990) and those of Potts (1980, 1986) show that an important aspect of these changes has been to reduce the area of nesting habitat and/or nest success.

Both the aforementioned examples from North America are from regions of deciduous forest that have been cleared for agriculture and analogous to western Europe and highlight the importance of hedgerows in this type of agroecosystem. Comparatively, the agroecosystems of much of the central United States are in regions of converted grasslands and subsequently traditional hedgerow habitats are less common, however, the importance of residual areas, such has roadsides, fencelines, right-of-ways, and wetlands are no less important in these systems for nesting by pheasants (Best et al., 1995; Smith et al., 1999; Snyder, 1984; Warner et al., 1987), gray partridges (Carroll, 1989; Hupp et al., 1980; Potts, 1986; Weigand, 1980), and bobwhite quail (Roseberry et al., 1979; Taylor et al., 1999). For example, Church (1980) found 70% of gray partridge nests were in residual cover which comprised only 2.5% of his study area.

In the United Kingdom, Eastern Europe, and in the prairies of North America, nesting in fenclines, roadsides, and vegetative balks between fields serve as important nesting cover for gray partridges (Carroll, 1989; Potts, 1986). Studies from North America show an increasing dependence by gray partridge upon fencelines and roadsides for nesting since the mid-20th century (Fig.5.16). As discussed by Potts (1986), such idle areas are also important in many areas where spring crop growth is early since they attract breeding pairs and facilitate nesting in adjacent croplands.

For pheasants residual areas are often used for nesting more than in relation to their availability (Hill and Robertson, 1988; Robertson, 1996). Warner et al. (1987) found an increasing importance of roadsides for nesting pheasants in Illinois as the proportion of row crops in the landscape increased. Similarly, in Iowa Clark et al. (1999) found 26.9% of nests in fencelines and roadsides, which combined, only occupied 14.3% of the study area.

How residual areas are managed, or the lack of management, has important implications for determining the value of those habitats for nesting. As alluded to above, in Europe changes in management of hedgerow stemming from reduced coppicing has allowed woody overstories to develop while decreasing the area of shrub and ground cover (Hill and Robertson, 1988; Rands, 1987). The lack of disturbance in idle areas often allows for the encroachment of woody vegetation (Novoa et al., 2002; Roseberry et al., 1979; Rodgers and Hoffman, 2005) or the development of litter and unsuitable vegetation structure (Mckee et al., 1998; Rands, 1987) that reduces the value of those areas for nesting. For both pheasants (Robertson, 1996; Warner et al., 1987) and gray partridges (Carroll, 1989) roadsides can serve as important nesting habitat in areas where row crops are dominant, however, the value of roadsides is dependent upon how those habitats are managed (*i.e.* mowing).

Although croplands may often be used by gamebirds for nesting, they are used less than or equal to their availability and their dynamic nature delays their use until vegetative structure is suitable. A review of the literature on pheasants in North America showed that row crops were the least preferred habitat used for nesting based upon availability (Robertson, 1996), and in studies where the use of row crops by nesting pheasants has been high it has been attributable to a lack of preferred nesting habitat (Anderson, 2002; Roberston, 1997; Snyder, 1984). In Great Britain, Hill and Robertson (1988) while showing a similar pattern, also found that as the growing season continued cereal fields were increasingly selected for nesting by pheasants; a similar temporal pattern in nest site selection is evident in gray partridges (Potts, 1986).

Stubble in crop fields may serve as high quality nesting habitat, however, their selection prior to planting may cause 100% nest loss in such habitats from field preparation and planting (Snyder, 1984). Also, the development of faster developing cultivars and faster harvesting rates puts renesting birds at greater risk towards the end of the nesting season. Despite these threats, nests in cropfields, which are often renests, have a very high success rate. For pheasants in Colorado nest success was 75% in winter wheat and on a farm in Austria managed for pheasants Anderson (2002) and Bliss (2004) found a mean nest success of 66% and 54%, respectively within cereal crops. In both the cases of pheasants in Austria (Anderson, 2002; Bliss, 2004) and gray partridges throughout Europe (Potts, 1986), the use of crops for nesting is limited to regions where sufficient crop growth occurrs prior to the nesting season.

The use of idle lands (including conservation habitats (*e.g.* CRP, Set-aside)), fallows, haylands and grazing lands for nesting is common in many gamebirds (pheasants (Best et al., 1995; Hill and Robertson, 1988; Robertson, 1996; Warner and Etter, 1989); bobwhite quail (Roseberry et al., 1979; Taylor et al., 1999); greater prairie-chickens (Kirsch, 1974; Svedarsky et al., 2003; Yeatter, 1963); gray partridge (Potts, 1980); helmeted guineafowl (Malan, 1998)) and their suitability for nesting and nest success is dependent upon their size, vegetative structure, and management. Of particular importance is the time since last disturbance of idle and fallow lands, the timing and frequency of mowing of hay, and the frequency of burning of and livestock densities on pastures.

The value of idle and fallow land for nesting by gamebirds is species-specific and dependent upon the characteristics of the habitat (vegetative structure, plant diversity, size, shape) and the presence of other necessary habitats within the landscape. Natural succession can reduce habitat quality through the development of excessive vegetative overstories, or densities as shown for northern bobwhite (Roseberry et al., 1979), and gray partridge (Novoa et al., 2002) following land abandonment. Also, for prairie grouse the exclusion of fire and grazing from idle lands can be detrimental by allowing the development of excessively dense vegetation, reduced plant diversity, excessive litter accumulation, and encroachment of woody vegetation (Mckee et al., 1998; Riley et al., 1992; Rodgers and Hoffman, 2005; Svedarsky et al., 2003).

An understanding of the importance of the vegetative characteristics required for nesting by gamebirds is particularly relevant because much of the management directed towards gamebirds concentrates on the creation and management of nesting habitat. By not knowing or not incorporating these requirements has made programs such as Conservation Reserve Program in the United States and set-aside in the European Economic Community largely ineffective for gray partridge (Aebischer and Potts, 1998; Haroldson et al., 2006; Potter, 1997), while regionally successful for bobwhite quail (Roseberry and David, 1994; Riffell et al., 2006), pheasants (Haroldson et al., 2006; Rodgers, 1999) and prairie grouse (Rodgers and Hoffman, 2005).

Northern bobwhite quail have declined in the U.S.A by 3.0% per year since 1966 despite efforts towards stemming the decline and the introduction of the Conservation Reserve Program (Sauer et al., 2005). For example, in the southeastern United States (U.S. Fish and Wildlife Service Region 4), formerly a core area for the Northern bobwhite, populations have declined by 4.1% per year since 1966 and 5.9% since 1986 when the Conservation Reserve Program was initiated (Fig.5.17). Despite over a million hectares of CRP land in the southeastern U.S.A the annual rate of population decline of the bobwhite quail has more than doubled since the period prior to the Conservation Reserve Program (1966-1985, 2.5% per year) (Sauer et al., 2005). Similarly, Roseberry and David (1994) illustrated that despite increasing areas of CRP lands in Illinois there was has been no observable effects on declining populations of Bobwhite quail, however, in Kansas, Nebraska, and Missouri there have been population increases correlated with increases in the area of CRP lands (Riffell et al., 2006).

Both Roseberry and David (1994) and Greenfield et al. (2002, 2003) have discussed that the vegetative characteristics of most CRP, especially in the absence of management, is unsuitable for use by northern bobwhites and likely responsible for the lack of a positive effect on their populations. Riffell et al. (2006) concluded that the management of CRP lands was important in determining its effect on populations. Furthermore, the landscape context in which CRP lands are placed may neutralize any positive effect since adjoining habitats may have negative influences on populations. For example in North Carolina, Palmer et al. (2005) found that the creation of field borders for nesting and brood rearing had a greatly reduced positive effect on bobwhite populations in the absence of predator management.

Although for pheasants the CRP has had benefits in many regions (Iowa (Riley, 1995), Minnesota (Haroldson et al., 2006), South Dakota (Eggebo et al., 2003)), Rodgers (1999) and Doxon (2005) have shown that in western Kansas CRP lands have had no effect on declining populations since nesting habitat is not limiting and the problem lies in the decreased value of croplands during winter and as brood habitat. Similarly, Clark et al. (1999) suggested that the benefits to nesting pheasants of developing grassland areas was greatest in intensive agricultural landscapes where nest habitat was most limiting. Conversely, in the case of gray partridges, due to high vegetative density and large block configurations (as opposed to linear), CRP lands are not suitable for nesting and have not reversed population declines in Minnesota (Haroldson et al., 2006); an analogous phenomena is seen in Europe for gray partridges with set-aside lands (Aebischer and Potts, 1998; Potts, 1997).

In North America, some of the greatest positive effects of CRP lands for gamebirds have occurred for sharp-tailed grouse, and lesser and greater prairie-chickens, facilitating population increases and range expansion (Rodgers and Hoffman, 2005). The success of the creation of habitat for these species, however, has not been universally successful due to management, or lack of management, failing to create suitable habitat (Rodgers and Hoffman, 2005). Monocultural stands of grasses, a lack of forb cover and diversity, litter accumulation, and encroachment of woody cover are all undesirable for nesting prairie grouse (Fuhlendorf et al., 2002; Mckee et al., 1998; Rodgers and Hoffman, 2005; Ryan et al., 1998; Svedarsky et al., 2003) and have all been promoted through the exclusion of fire and seeding practices (Rodgers and Hoffman, 2005). For example, greater prairie-chicken nest success decreases with >25% litter, >5% woody cover, ≤5% forb cover, and grass cover ≤25% (Mckee et al., 1998).

Many idle lands are not permanently idle and are used for grazing or mowed for hay and the issues associated with grazing and hayland management are relevant to such areas. Moreover, haylands and grazing lands often constitute large portions of agricultural landscapes and subsequently their management can be particularly important to gamebird populations. As with crop, idle, and residual lands, how hay and grazing lands are managed are important to gamebird nesting since management determines structural heterogeneity and vegetative diversity. The issues related to the management of haylands and grazing lands center upon the timing and frequency of mowing, stock densities, and the use of fire and the frequency of burning.

The management of haylands, including earlier and more frequent mowing for hay (facilitated by cultivation of hay), fertilization, and increased mechanization, has been one of the greatest changes associated with the intensification in the management of agroecosystems (Chapters 3,4). This is particularly relevant to gamebirds since hayfields are often chosen for nesting by gamebirds as seen in pheasants (Hill and Robertson, 1988; Robertson, 1996), Greater prairie-chickens (Kirsch, 1974; Yeatter, 1963) and gray partridges (Potts, 1980), particularly where other nesting habitats are limited and subsequently the mowing of these fields can potentially cause high levels of nest loss and hen mortality (Gates and Hale, 1975; Potts, 1980; Warner and Etter, 1989). Mowing influences gamebird populations by both destroying nests and killing the incubating adult (usually a hen), subsequently reducing the number of successful clutches and reducing the population of hens available to renest. For example, Church (1980) found 25% loss of gray partridge nests due to farm operations (mostly mowing) and 100% mortality of hens on clutches lost to mowing. For pheasants Warner and Etter (1989) found that in hayfields mowing reduced nest success by 22% and killed >60% of incubating hens.

The dates of first mowing of hay has increasingly become earlier since the mid-20th century while the frequency of mowings have increased (Bollinger et al., 1990; Fuller, 2000; Herkert, 1997; Warner and Etter, 1989; Yeatter, 1963). The implications for nesting gamebirds is that earlier mowing dates leaves less time for hatching of clutches, leading to increased nest lost and hen mortality, while in the same manner second and third mowings put renests at risk (Gates and Hale, 1975; Warner and Etter, 1989; Yeatter, 1963). Collectively, the net result is that fewer nests, either initial or renests, are hatched. Furthermore, in gamebirds clutch size of renests are smaller and hatching success reduced, leading to diminished overall recruitment (Blank and Ash, 1960; Bliss, 2004; Church, 1984a; Clark and Bogenschutz, 1999; McCabe and Hawkins, 1946; Pitman et al., 2006) while, at least in gray partridges, where growing seasons are shorter the loss of first nests may preclude renesting (Carroll, 1989; Gates, 1973).

Although mowing does lead to significant level of nest loss and mortality, in many cases this loss may be compensatory in that if these losses do not occur to mowing, the proportional difference in nest loss will occur due to predation. A review of the literature on gray partridge, by Potts (1980) showed that as nest loss due to mowing increased the proportional loss to predation decreased, which a subsequent study by Carroll (1989) confirmed. Of note is that as losses from mowing decreases, the known losses to predation increases at a disproportionately lower rate than would be expected if there was equal compensation (Fig. 5.18). Although this relationship may be an artifact of sampling inefficiency, it may also suggest that nest success increases nonlinearly as the area mowed decreases as the hunting of predators decreases.

As with haylands, the management of grazing lands can be influential on determining the availability and quality of nesting habitat. An obvious negative management practice is overgrazing, however, their is ample evidence that where grazing is properly managed to increase spatial and temporal heterogeneity in habitat diversity and structure grazing is beneficial and important for ecosystem function (Fuhlendorf and Engel, 2001). Such management includes altering stock densities and the use of fire, both of which are important in determining habitat quality for gamebirds in rangelands (Guthery, 1986; Hagen et al., 2004; Jansen et al., 2000; Jamison et al., 2002; Kirsch, 1974; Malan, 1998; Mentis and Bigalke, 1981; Novoa et al., 1998, 2002; Spears et al., 1993; Stoddard, 1931; Svedarsky et al., 2003).

In general, moderate levels of grazing are shown to create and/or maintain desired habitat structure and diversity for nesting gamebirds in North America (Spears et al., 1993; Svedarsky et al., 2003), Europe (Novoa et al., 2002), and South Africa (Malan, 1998), however, attention to the spatial and temporal patterns in grazing and stock densities is required to create the desired effects. An important aspect of range management is the use of fire, which if applied to too large of an area or too frequently, negatively affects gamebirds through the removal of large blocks of cover and/or alters vegetative structure in a negative manner over both the long and short-term (Hagen et al., 2004; Novoa et al., 1998; Malan, 1998; Mentis and Bigalke, 1981; Rosene, 1969; Stoddard, 1931; Svedarsky et al., 2003). Of principal importance appears to be the development of fine-grained spatial heterogeneity in vegetative structure of rangelands (Mentis and Bigalke, 1981; Novoa et al., 2002; Rodgers and Hoffman, 2005; Stoddard, 1931).

As noted above, aside from the management of idle lands, their configuration and their surrounding habitats determine their utility, is species-specific and subsequently, the interspersion of habitats within the landscape is of consequence. Landscape heterogeneity, interspersion of habitats, and the availability of ” habitats, has long been recognized as an important factor in determining gamebird populations (Leopold, 1933). The importance of the juxtaposition and configuration of habitat types is evident in the responses of pheasants (Hill and Robertson, 1988; Robertson, 1994; Roberston, 1997), prairie grouse (Manzer and Hannon, 2005; Rodgers and Hoffman, 2005; Ryan et al., 1998), guineafowl

(Malan and Benn, 1999; Prinsloo, 2003), francolins (Jansen et al., 2000; Mentis and Bigalke, 1981), spurfowl (Jansen and Crowe, 2002), bobwhite quail (Williams et al., 2000), and gray partridge (Potts, 1986).

Although multiple studies of many species have illustrated relationships between habitat configuration and populations, implying relationships with nesting success, fewer studies have quantified this relationship. For pheasants in Iowa, Clark et al. (1999) showed that larger grassland patches had higher nest success, suggesting that a minimum effective patch is ≥15ha, and ≥60ha is preferable. However, they also stressed that multiple grassland patches within the landscape are preferable to one large patch. The value of interspersed habitats for pheasants, particularly the complexity of edges, was also found for pheasants in the United Kingdom (Robertson, 1994).

In Iowa, Clark and Bogenschutz (1999) found pheasant nest success to be 40% higher in contiguous block habitats compared to linear habitats because of increased depredation of nests in the latter. Moreover, in a landscape with more contiguous grassland renesting occurred 15 days earlier than in more fragmented and intensively managmed landscape (Clark and Bogenschutz, 1999). This is of consequence because clutch sizes decreased later in the season and in the intensified landscape nest success was considerably lower (Clark and Bogenschutz, 1999). Based upon the relationship among grassland area, configuration, and nest success in pheasants, Riley (1995) suggested that in areas dominated by intensified agriculture the addition of grasslands produces the greatest relative benefits for nesting, diminishing with the area of existing grassland habitat, which is consistent with discussions by Robertson (1996) and Rodgers (1999).

Prairie grouse are species dependent upon large areas of grassland habitat, being sensitive to excessive conversion and fragmentation of grassland habitat (Fuhlendorf et al., 2002; Manzer and Hannon, 2005; McDonald and Reese, 1998; Ryan et al., 1998; Svedarsky et al., 2003; Walk and Warner, 1999; Woodward et al., 2001). Recent increases in populations and range expansions of sharp-tailed grouse and lesser and greater prairie-chickens has been attributed to increasing areas of CRP grasslands, particularly where adjoining large blocks of native grasslands (Rodgers and Hoffman, 2005). For example, in Missouri Ryan et al. (1998) found that greater prairie-chicken nest success was more than double in landscapes with more contiguous tracks of natural grasslands and that grassland fragments of <65ha were not used for nesting.

The influence of the juxtaposition and configuration of habitats on nest success is particularly related to the effect of predation. Of particular note is the influence that the proximity to trees and woodlands has on nest success and survival. As discussed above, in gamebirds there is a general relationship between woodlands and survival and this relationship is also evident in nest survival. Also, the configuration of nesting habitats may have an influence on nest survival, with linear habitats potentially having high levels of nest predation because they are more efficiently searched by predators.

In pheasants, gray partridge, red-legged partridge, and prairie grouse egg predation has commonly been associated with corvids and subsequently, an increased proximity of nests to wooded areas increases egg loss to these predators (Hill and Robertson, 1988; Manzer and Hannon, 2005; Potts, 1980, 1986; Potts and Vickerman, 1974; Snyder, 1984). In the United Kingdom, Hill and Robertson (1988) found 23.8% of egg losses in pheasants during laying was due to corvids and 13.3% and 18.7% during incubation from two separate studies. Furthermore, sharp-tailed grouse nest success was lower near woody vegetation where quality nesting habitat was unavailable and 8 times greater in areas of low densities of corvids (<3 corvids/km^2), which was related to landscapes with less than $<10\%$ cropland (Manzer and Hannon, 2005). Where corvid populations are reduced through management recruitment in gamebirds can be significantly increased (Fig. 5.19).

A large portion of depredation of nests and incubating adults is by mammals, particularly canids (Hill and Robertson, 1988; Little and Crowe, 2004; Potts, 1986; Riley and Schultz, 2001; Rollins and Carroll, 2001). In linear habitats gamebird nests and incubating adults may be particularly susceptible to predation since foraging efficiency of mammalian predators is increased. This pattern has been shown in gray partridge, red-legged partridge, and pheasants (Clark and Bogenschutz, 1999; Gates and Hale, 1975; Potts, 1980, 1986; Snyder, 1984; Warner et al., 1987), and the high nest success of pheasants within crop fields discussed above further supports this.

The characteristics of agricultural landscapes may be beneficial to both avian and mammalian predators, which combined with the effects of habitat configuration and juxtaposition, may exacerbate the role predators have on gamebird population dynamics in agroecosystems. For gray partridges in southern England, brood production is density dependent, decreasing non-linear with increasing nest density (Potts, 1980). With the removal of mammalian predators and corvids, however, the magnitude of this density dependent relationship is depressed with brood production decreasing linearly as nest density increases (Potts, 1980, 1986). Using data from Potts (1986) on gray partridge brood production from 3 sites in southern England, with a history of periods with and without predator removal, illustrates the different behavior in the density-dependent relationship between nest density and brood production with and without predator removal (Fig. 5.20).

Based upon this relationship, Potts and Aebischer (1994) showed that some of the negative effects from agricultur on gray partridge recruitment can be mitigated with predator management. Similarly, Palmer et al. (2005) found that the effectiveness of habitat creation for northern bobwhites on farmland was significantly increased through predator management. It should be noted, however, that given the density dependent nature of the relationship between nest success and predation illustrated in Figure 5.20 it is apparent that in landscapes with low gamebird densities that predator control may have little or no effect on overall nest success.

Brood survival

The principal impact that the intensification of agriculture has on recruitment in gamebirds is by decreasing chick survival via reductions in arthropod prey abundance and increasing the dominance of lesser preferred prey species through pesticide use. Also, intensified agricultural management decreases the availabilty of preferred foraging habitats. The importance of arthropod availability to chick survival in Galliformes, and overall population levels, was first extensively discussed by Blank et al. (1967) and Southwood and Cross (1969) for the gray partridge and later expanded upon by Potts (1980, 1986) (Fig. 5.21) and for pheasants by Hill (1985) (Fig. 5.22). The importance of this relationship was further illustrated for gray partridge (Aebischer and Potts, 1998; Boatman and Brockless, 1998; Chiverton, 1999; Green, 1984; Panek, 1992, 2005; Rands, 1985; Sotherton et al., 1993), as well as for pheasants (Boatman and Brockless, 1998; Chiverton, 1999; Hill and Robertson, 1988; Taylor et al., 2006), red-legged partridge (Boatman and Brockless, 1998; Green, 1984; Potts, 1980), capercaillie (Baines et al., 1999; Picozzi et al., 1999), black grouse (Baines, 1996; Baines et al., 1999), red grouse (Park et al., 2001), lesser prairie-chickens (Jones, 1963), greater prairie-chickens (Jones, 1963) and sage grouse (Johnson and Boyce, 1990).

Besides decreasing arthropod availability, the intensification of agricultural management has also decreased chick survival in Galliformes by reducing the area of habitat and crops types preferred for foraging and through changes in the vegetative structure of brood foraging habitat. These changes in habitat availability and quality act synergistically with reduced arthropod availability to forcing chicks to expend more energy foraging over larger areas, which is correlated to lower chick survival and lower populations in pheasants (Fig. 5.23), greater prairie-chickens, and gray partridge (Hill, 1985; Hill and Robertson, 1988; Potts, 1986; Ryan et al., 1998; Warner, 1979, 1984; Warner et al., 1984, 1999). The relationship between brood movements and survival is further influenced by the effects of summer temperature (positively) and precipitation (negatively) on thermoregulation (Green, 1984; Hill, 1985; Potts, 1986; Panek, 1992, 2005).

The increasing application of pesticides reviewed in Chapter 3 has had significant negative impacts upon the availability of arthropods through direct effects of insecticides and fungicides and via indirect effects by reducing weed abundance and diversity in crops fields (Potts, 1986; Rands, 1987; Southwood and Cross, 1969; Sotherton et al., 1989; Taylor et al., 2006). Gamebirds are particularly impacted since they have precocial young, which precludes any influence of parents in influencing food intake. Row crops and pastures are often preferred foraging habitats for gamebird broods, with broods generally showing preferences for edge habitats with relatively sparse vegetation; *i.e.* those areas adjacent to nesting habitat which allow easy movement of chicks (Bliss, 2004; Enck, 1986; Green, 1984; Hill, 1985; Potts, 1986; Warner, 1979). Subsequently, measures designed to mitigate the effects of pesticides on chick survival have concentrated on these edge habitats, such as conservation headlands (field edges unsprayed with herbicides) in Europe or managed CRP land in the U.S.A. that have been effective in increasing chick survival in agroecosystems while not taking cropland out of production (Chiverton, 1999; Doxon, 2005; Haysom et al., 2004; Potts, 1986; Sotherton, 1991; Sotherton et al., 1989, 1993).

A comparison of survival rates and brood size in relation to periods or areas with and without the application of pesticides illustrates the role of pesticides in affecting chick survival (Aebischer and Ewald, 2004; Potts, 1986; Potts and Aebischer, 1994). In the United Kingdom chick survival of gray partridges was considerably higher during the period before the use of pesticides, well above the 35% need for a stable population (Aebischer and Ewald, 2004), compared to the period of their widespread use, which is a pattern also seen in farms and fields where pesticides were used extensively and those where spraying was selective (Fig. 5.24). The role of pesticides in determiming chick survival of gray patridge is illustrated in the dry-land farming regions of North America where pesticide use is considerably lower than in Europe, and chick survival is relatively high (Fig. 5.24). Declining populations of pheasants have also been attributed to reduced chick survival (Chiverton, 1999; Hill, 1985; Hill and Robertson, 1988; Warner et al., 1984, 1999). For example, decreasing chick survival in Illinois since the mid-1940s is not correlated with agricultural land use variables (Warner et al., 1999), suggesting that survival has been reduced through management practices (Fig. 5.25).

The above examples (Fig. 5.24) highlight the effectiveness of conservation headlands in mitigating the negative impacts of pesticides on chick survival in gray partridge, which also has been shown for pheasants (Chiverton, 1999; Hill, 1985) (Fig. 5.26). Aebischer and Ewald (2004) determined that for each 1% increase in the area of conservation headland would increase chick survival of gray partridge by 4%. To maintain a steady population with a mean annual chick survival of 35% thus requires 4% of the arable land in Great Britain to be managed for brood habitat and a chick survival rate of 44%, equal to the period prior to declining populations in Great Britain, would require 6% of arable land to be left unsprayed (Aebischer and Ewald, 2004).

5.2.3 Summary

The effects of agricultural intensification on gamebirds is evident across spatial scales, as well as seasonally. Consolidation of farmland and an emphasis on continual production has decreased the area and structure of preferred wintering, nesting, and brood rearing sites. Equally important, changes in management of those lands, particularly earlier and more frequent mowing and increased use of pesticides, but also increased livestock densities and more frequent burning are associated with many of the population declines of gamebirds observed globally.

In systems dominated by row crops the diversity and configuration of habitats appears most important, with the maintenance of preferred nesting, foraging, and over wintering habitat in close proximity to one another being optimal. Conversely, in rangeland systems small levels of fragmentation by row crops is beneficial; however, that maintenance of large tracts of grasslands are important. The importance of low diversity in cover types within rangeland systems should not be confused with within habitat heterogeneity. The latter is crucial to gamebird conservation. Within grasslands and pastures, a vegetational mosaic of varying heights, densities, species, and litter accumulation is crucial. As discussed in Chapter 3 in both row crop and grazing dominated systems intensification has led to increases in homogeneity within habitats, while increasing landscape homogeneity within row crop dominated systems and increasing fragmentation in rangeland systems.

The changing agricultural landscape associated with intensified land managment, while becoming less hospitable for gamebirds, has increased the area of habitat for both mammalian and avian predators, apparently resulting in increased predation pressures on gamebirds. Some of the ultimate effects of changes in habitat area and characteristics is that they have lead to increased predation through increased predator populations, more efficient hunting by predators, and by placing birds and nests in closer proximity to predators. The review above suggests that the creation of both wintering and nesting habitat may mitigate some of the effects of predation by reducing the density dependent relationship; however, predator management may further reduce the density-dependence of predation and be necessary to obtain the maximum benefits of habitat creation.

Due to high annual mortality the maintenance of abundant populations of many of the Galliformes species, if not all, associated with agroecosystems is dependent upon high levels of recruitment. Subsequently, decreased nest success and brood survival witnessed in relation to agricultural intensification has been the most important factor leading to population crashes of some of the gamebirds reviewed above. Although interannual climatic variation may lead to punctuated periods of high winter mortality from extremes in temperatures and snow fall or low chick survival from drought-related depletions in arthropod abundance, it is apparent that the key factor in global declines of gamebird populations has stemmed from reduced recruitment due to changes in the management of agroecosystems over the last 60 years.

The body of evidence implicating the lack of recruitment for population declines in gamebirds has also allowed for the development of successful mitigation of the effects of agricultural intensification for some species, including creation of nesting and brood foraging habitat, selective application of pesticides, and predator management. For less studied species and regions the exact relationship between agricultural intensification and declining gamebird populations is not as well quantified as in the more intensively studied species and regions; however, there is at the least a strong basis to guide further research and management via inferences.

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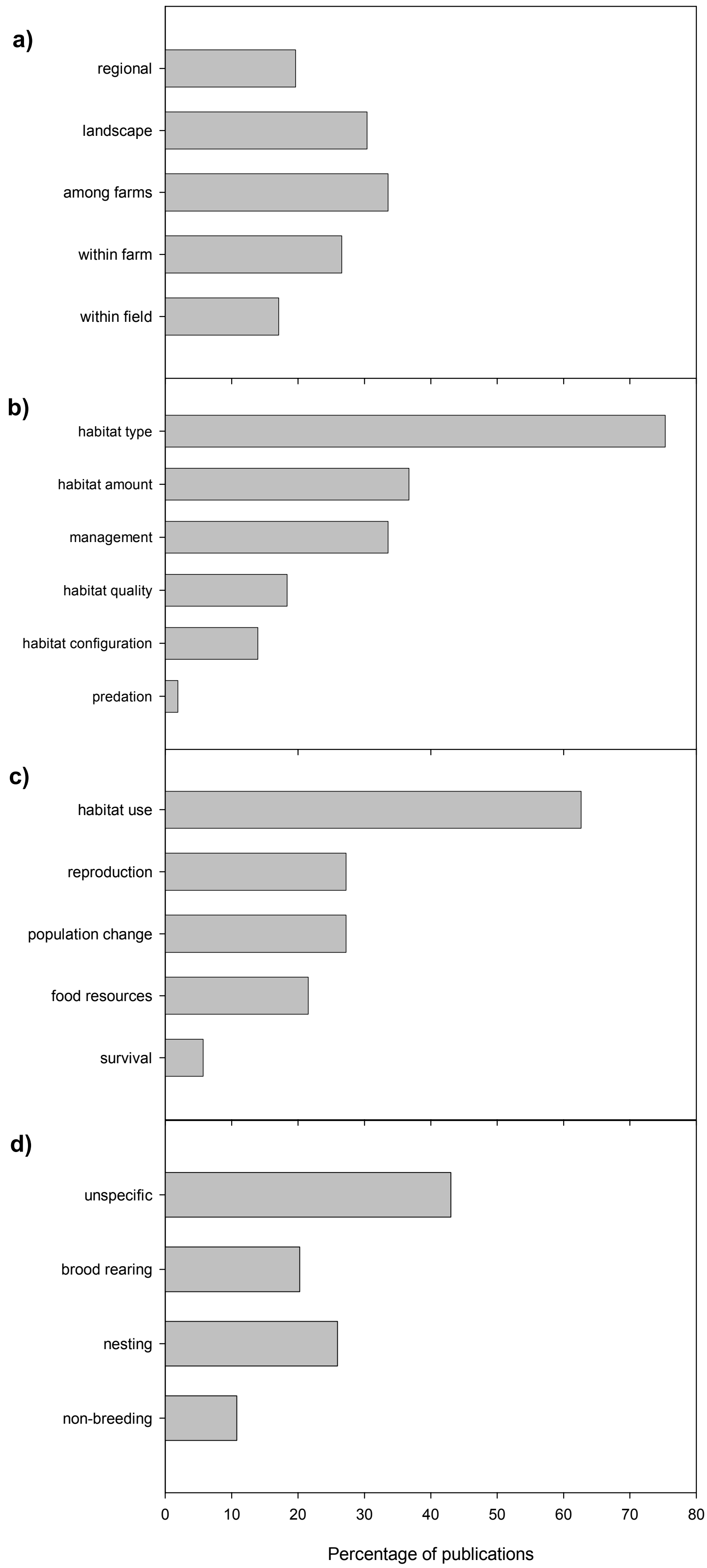

Figure 5.1: Percentage of 158 research articles from 1963-2007 used in the review and synthesis categorized by a) spatial scale, b) study theme, c) ecological parameter of interest, and d) seasonal period covered.

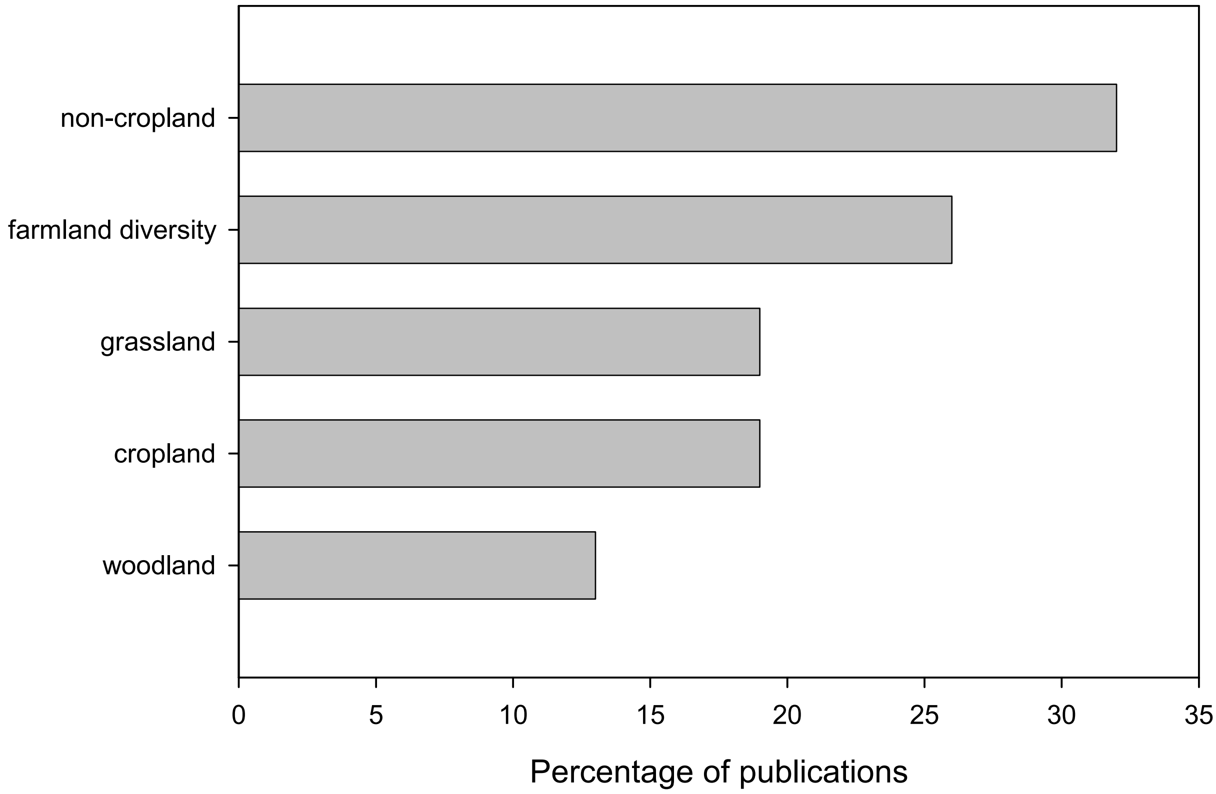

Figure 5.2: Percentage of publications at the regional scale classified by habitat characteristics considered most important for grassland and farmland birds

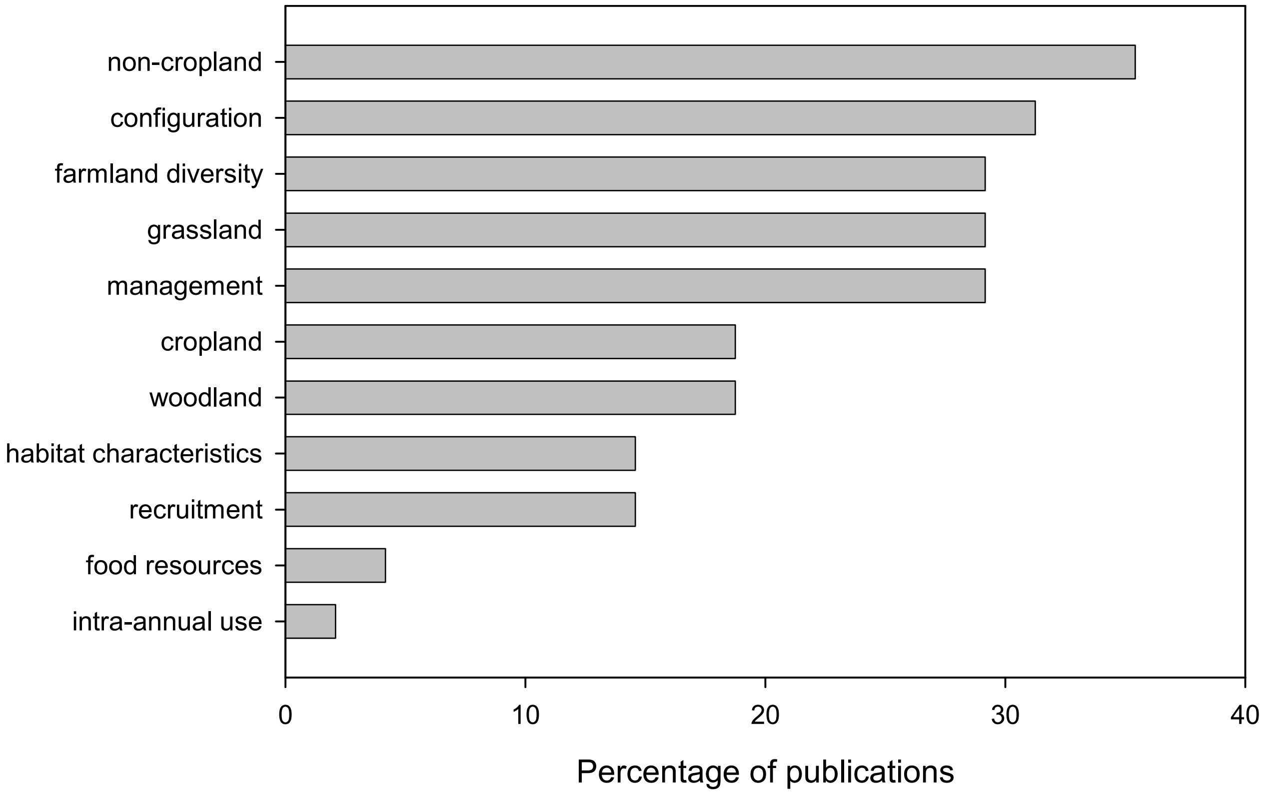

Figure 5.3: Percentage of studies in the analysis conducted at the landscape scale classified by the focus of research topics.

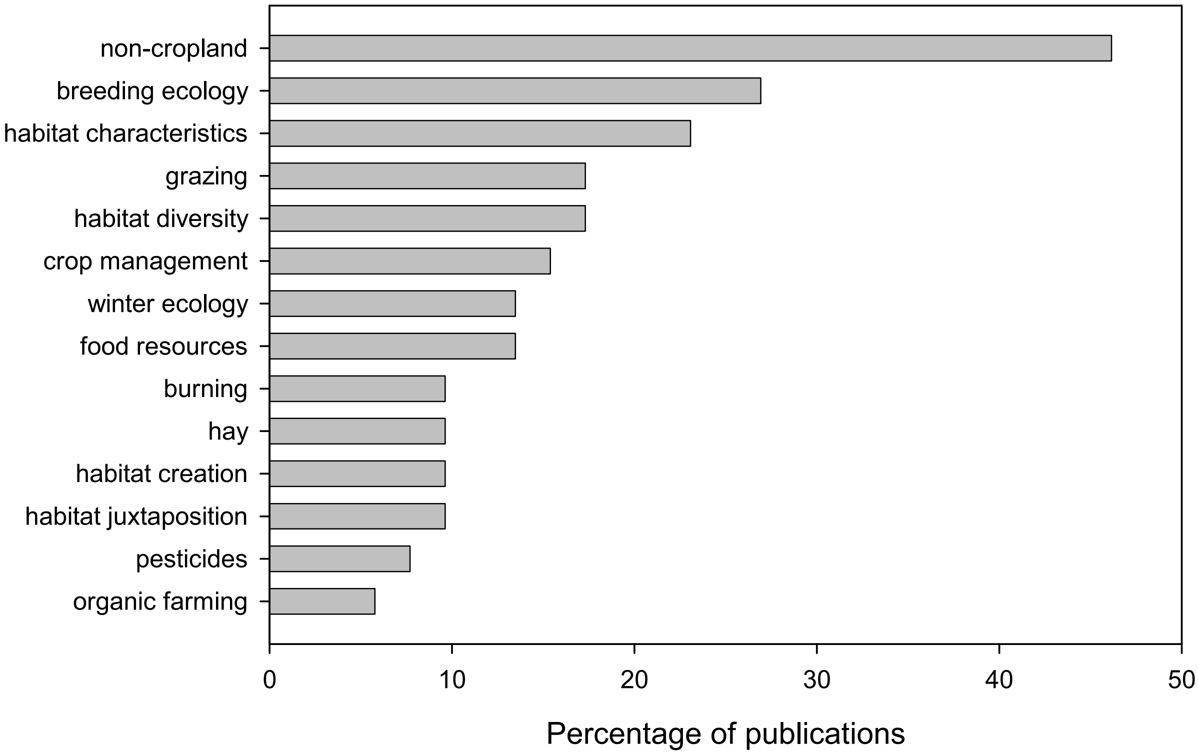

Figure 5.4: Percentage of studies in the analysis making among farm comparisons classified by the focus of research topics.

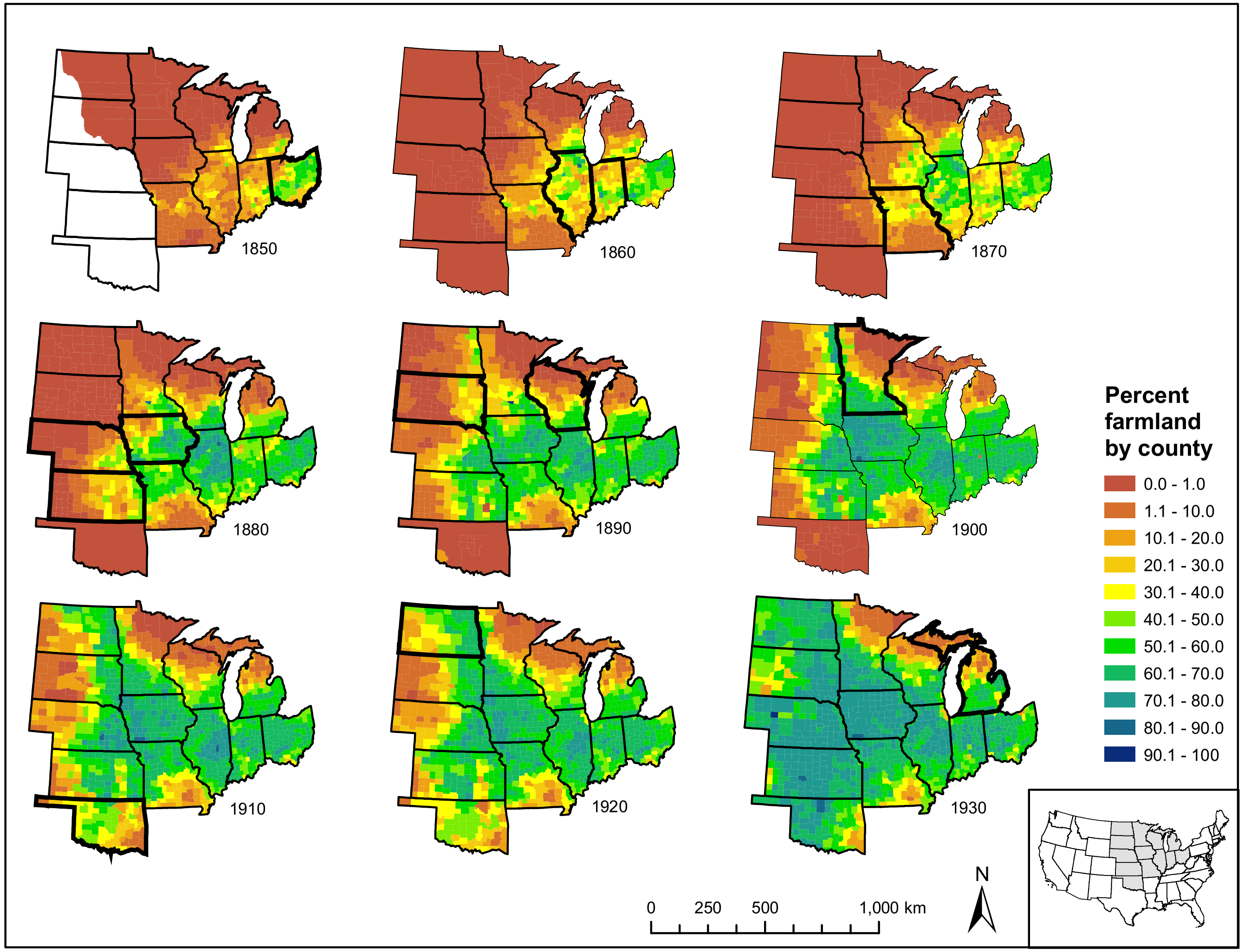

Figure 5.5: Map of Illinois, Indiana, Iowa, Kansas, Michigan, Minnesota, Missouri, Nebraska, North Dakota, Ohio, Oklahoma, South Dakota, and Wisconsin showing percent area in farmland by county for 1850-1930 (U.S. Census Bureau, 1850-2000). Highlighted states indicate the period of maximum abundance of greater prairie-chickens.

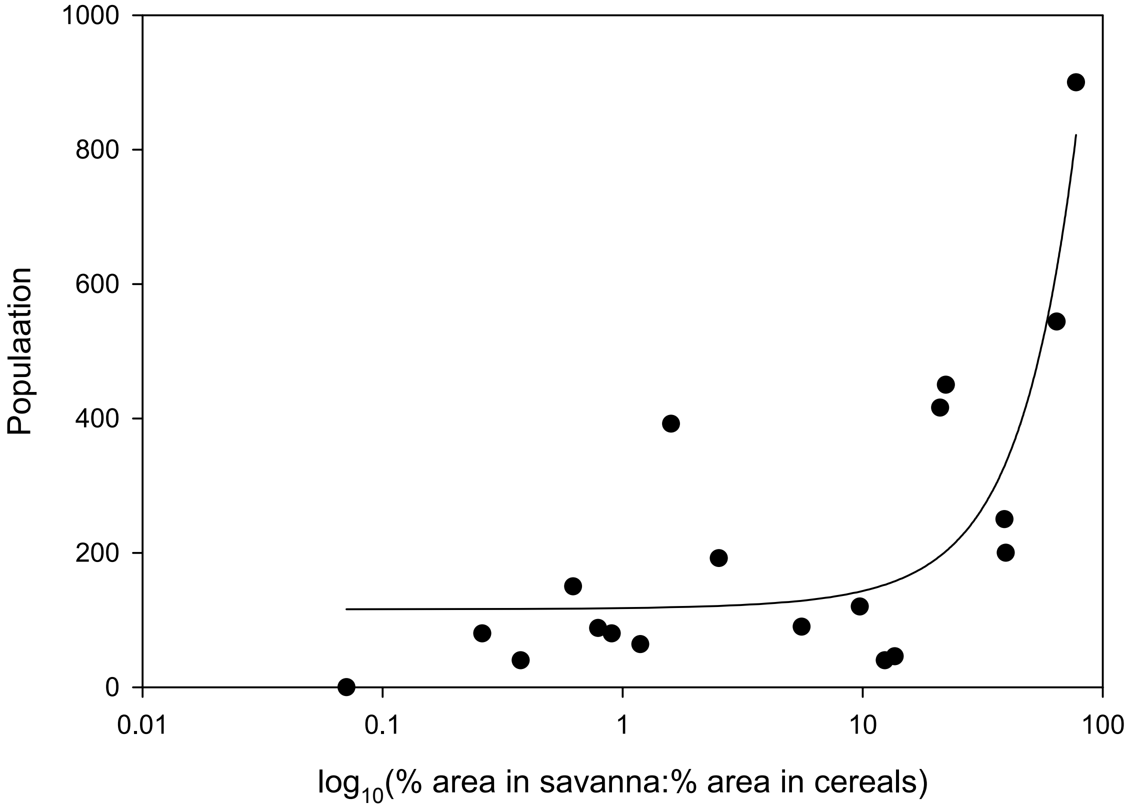

Figure 5.6: Guineafowl population in relation to the ratio of the percent area of savanna woodlands to the percent area of cereals on farmland in KwaZulu-Natal, South Africa (y=115.82+1.78x+$x^{0.0945}$; r^2=0.70). Based upon data from Pero and Crowe (1996)

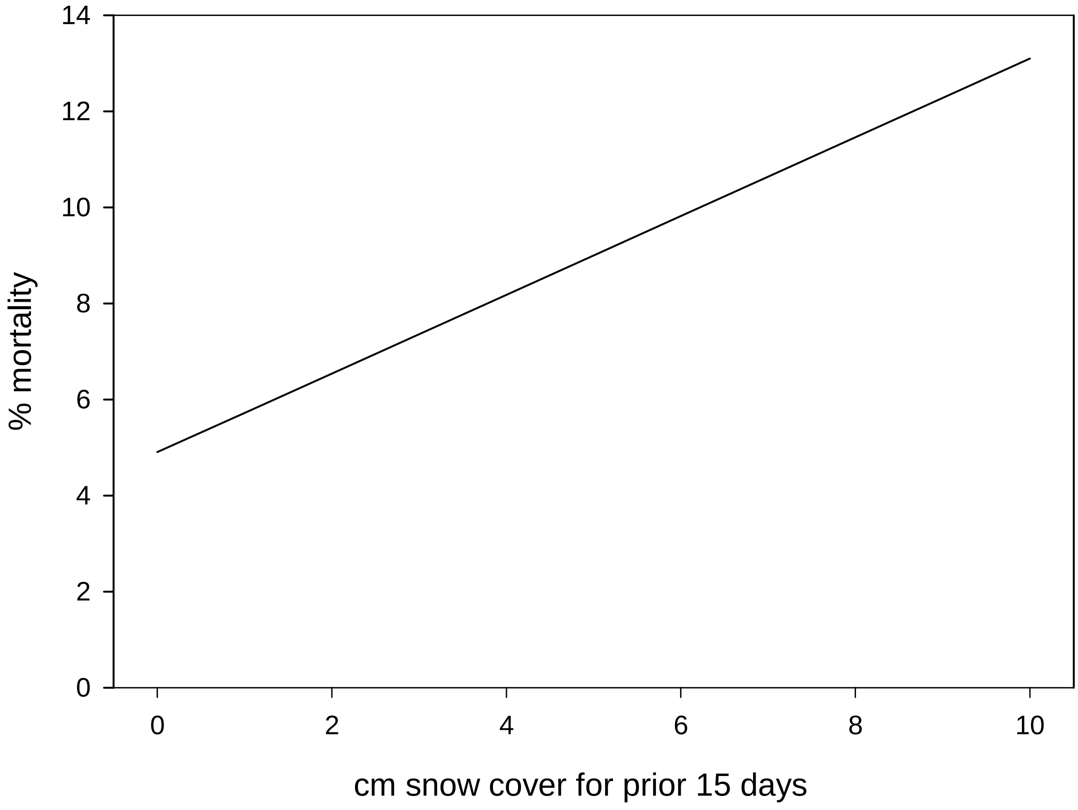

Figure 5.7: Relationship between hard snow cover (December–February) and mortality from Czempiń, Poland from 1985-1988 by half-month periods (y=4.9+0.82x; r^2=0.82) from Panek (1990).

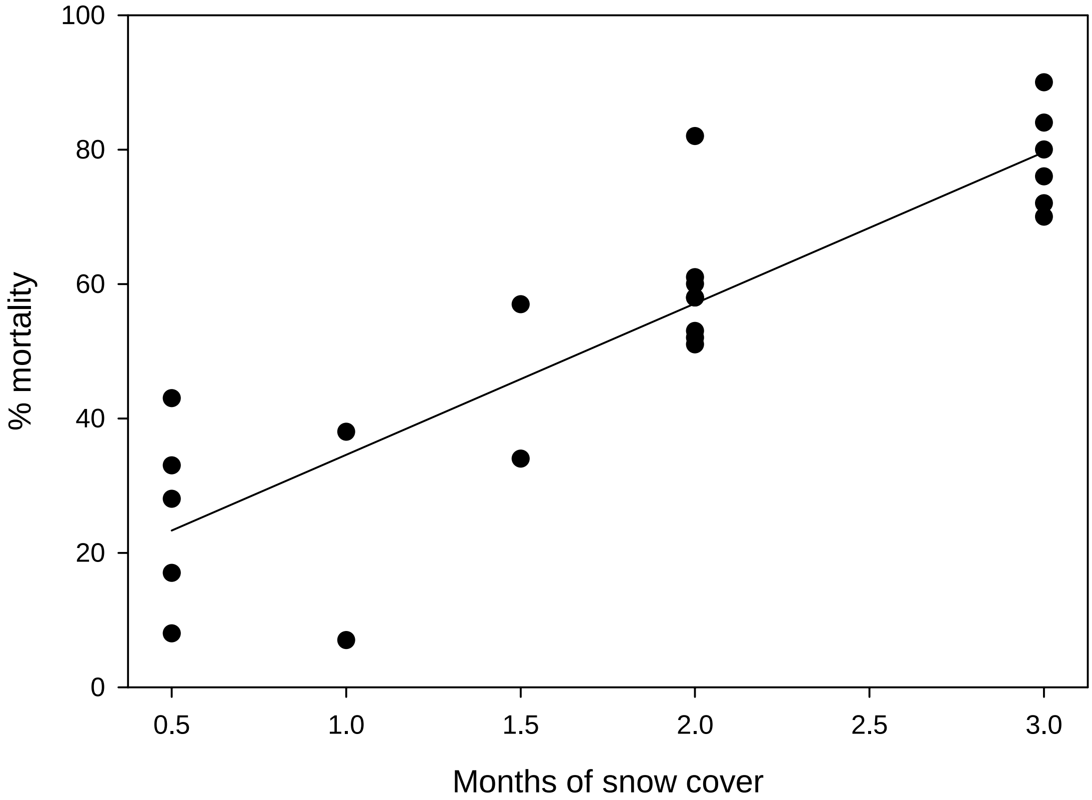

Figure 5.8: Estimated winter mortality of bobwhite quail in Wisconsin, Prairie du Sac in relation to months of snow cover $\geq$ 7.6cm from 1929-1959 (y=12.04+22.53x; r^2=0.77) (Kabat and Thompson, 1963).

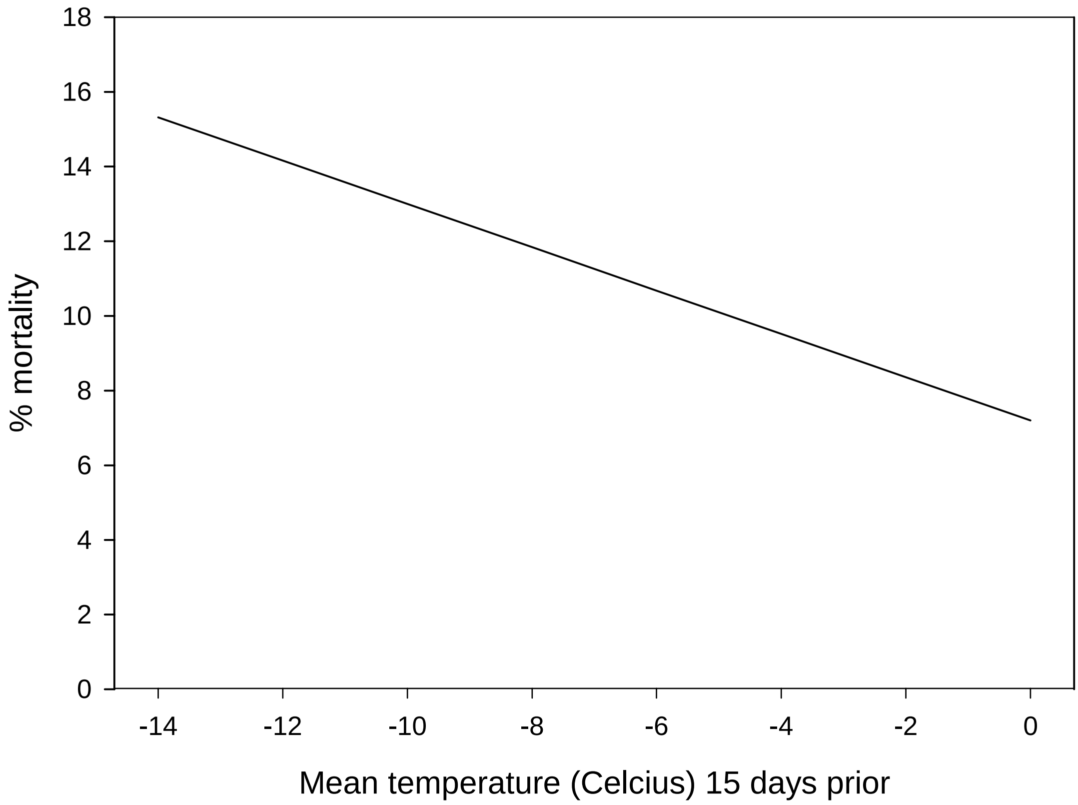

Figure 5.9: Relationship between mean temperature (December–February) and gray partridge mortality from Czempiń, Poland from 1985-1988 by half-month periods (y=7.2-0.58x; r^2=0.94) from Panek (1990).

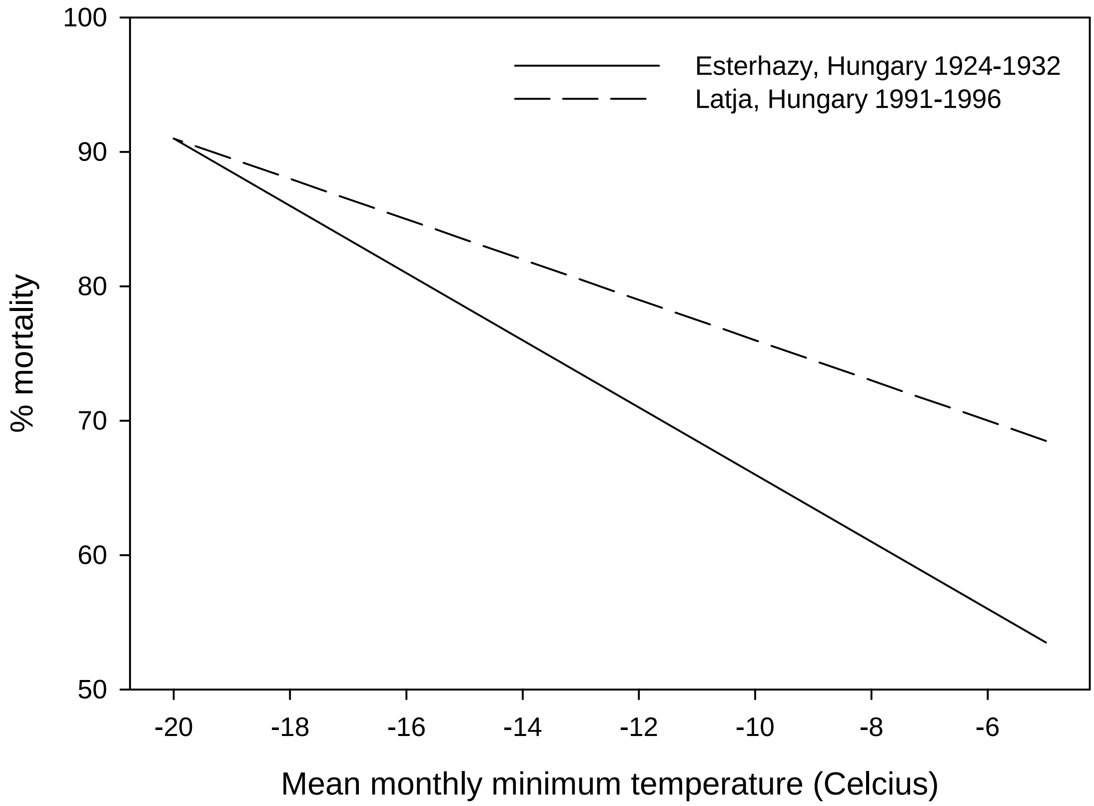

Figure 5.10: Winter mortality of gray partridge in relation to mean monthly minimum temperature (December–March) in Esterhazy, Hungary from 1924-1932, (includes shooting mortality, y=0.59+0.025x, r^2=0.67) and Latja, Hungary from 1991-1997 (y=0.39+0.015x; r^2=0.25) (Potts and Faragó, 2000).

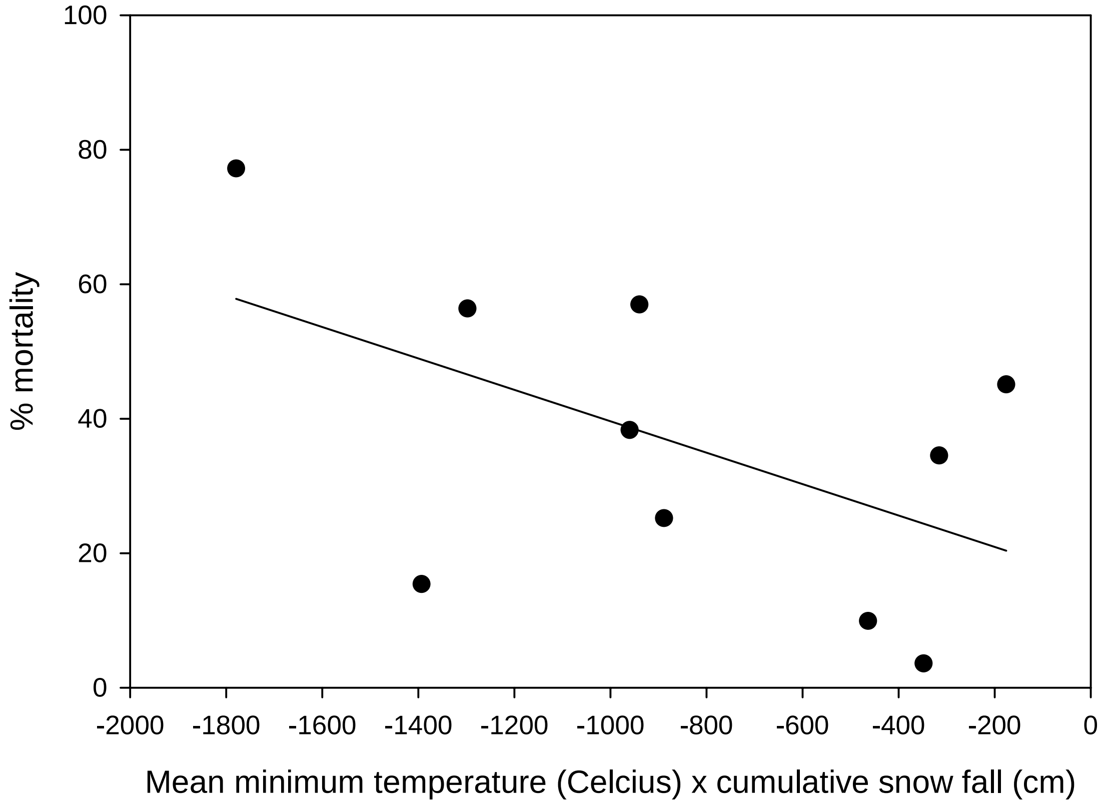

Figure 5.11: Winter pheasant mortality (December–March) from 1990-1994 from two sites in Iowa in relation to the interactive effect between mean minimum temperature and cumulative snowfall (y=16.24-0.02x; r^2=0.28). Data from Perkins et al. (1997).

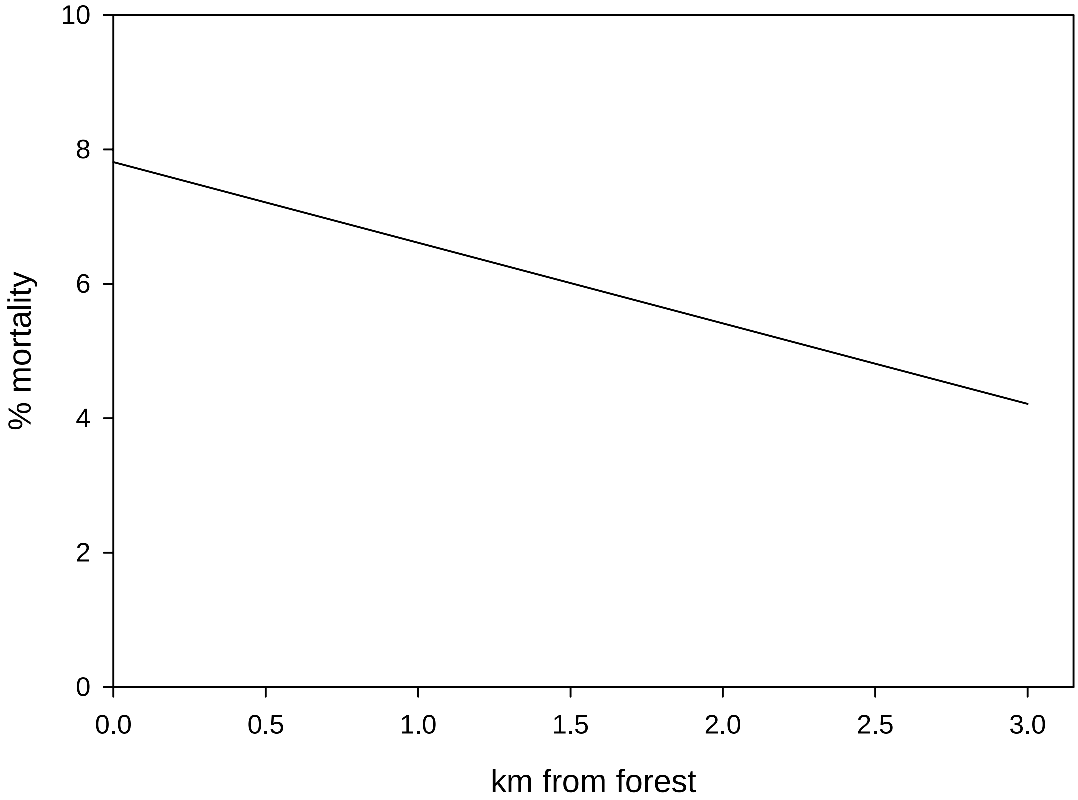

Figure 5.12: Winter mortality (December–February) of gray partridge in Czempiń, Poland from 1985-1988 in relation to covey's distance from woodlands (y=7.8-1.2x; r^2=0.62) from Panek (1990).

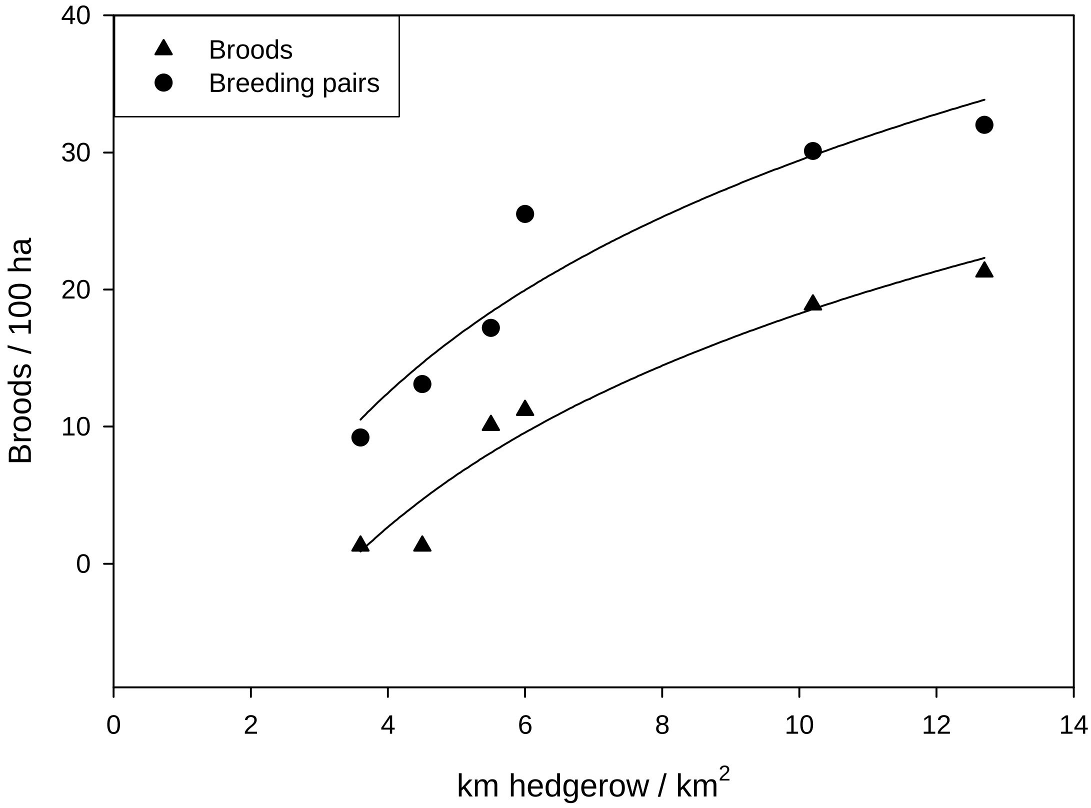

Figure 5.13: Density of breeding pairs (y = 18.506Lnx - 13.193; $r^2 = 0.91$) and broods (y = 16.991Lnx - 20.879 $r^2 = 0.95$) of gray partridge from 12 sites in Lower Austria (Petrjanos (1990) in Gossow et al. (1992))

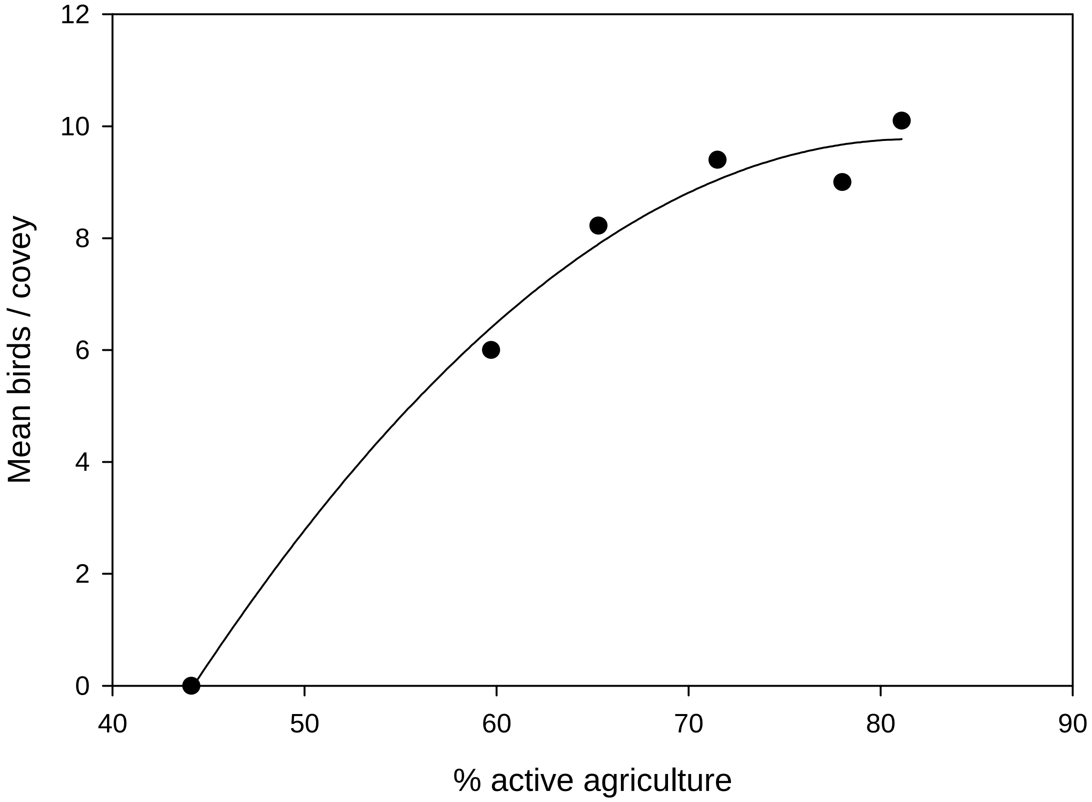

Figure 5.14: Gray partridge covey size in relation to percent area in active agriculture in Jefferson County, New York (Murtha, 1967) (y = -0.0069x^2 + 1.1328x - 36.538; r^2 = 0.99).

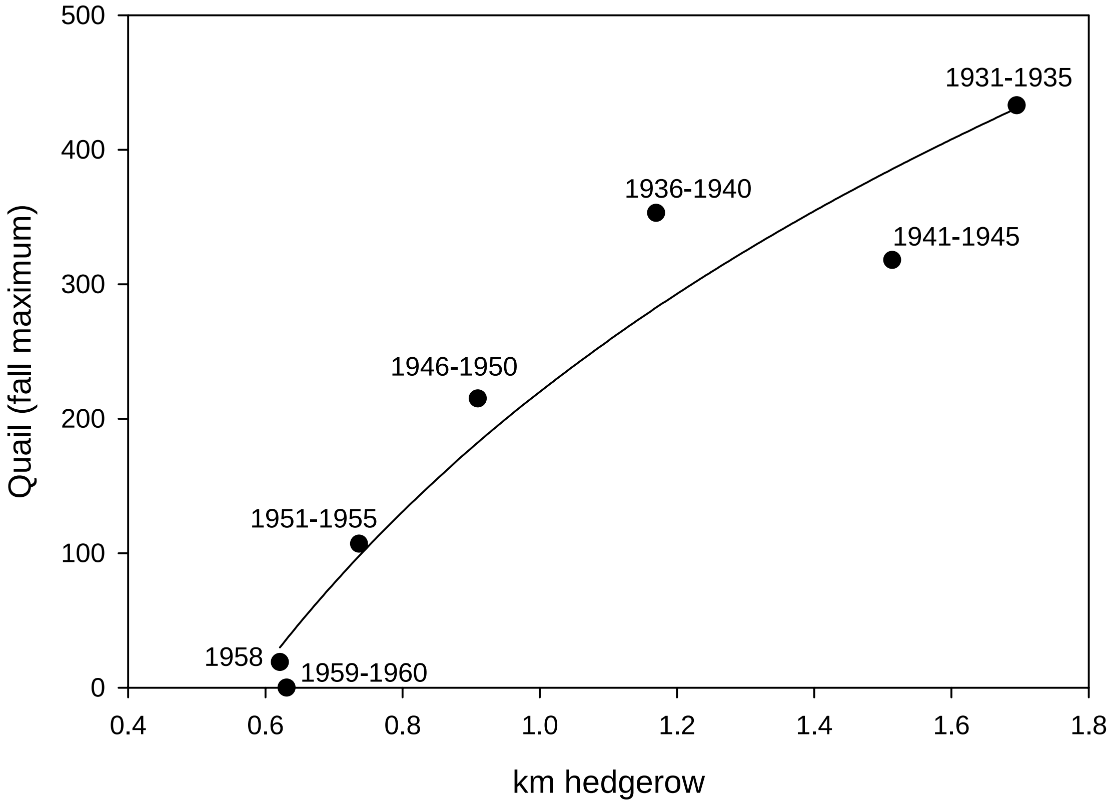

Figure 5.15: Maximum fall quail population by time period for Prairie du Sac, Wisconsin in relation to amount of hedgerow from 1929-1959 (Kabat and Thompson, 1963) (y = 399.4Lnx + 220.03; r^2 = 0.93). The time interval is presented for each point.

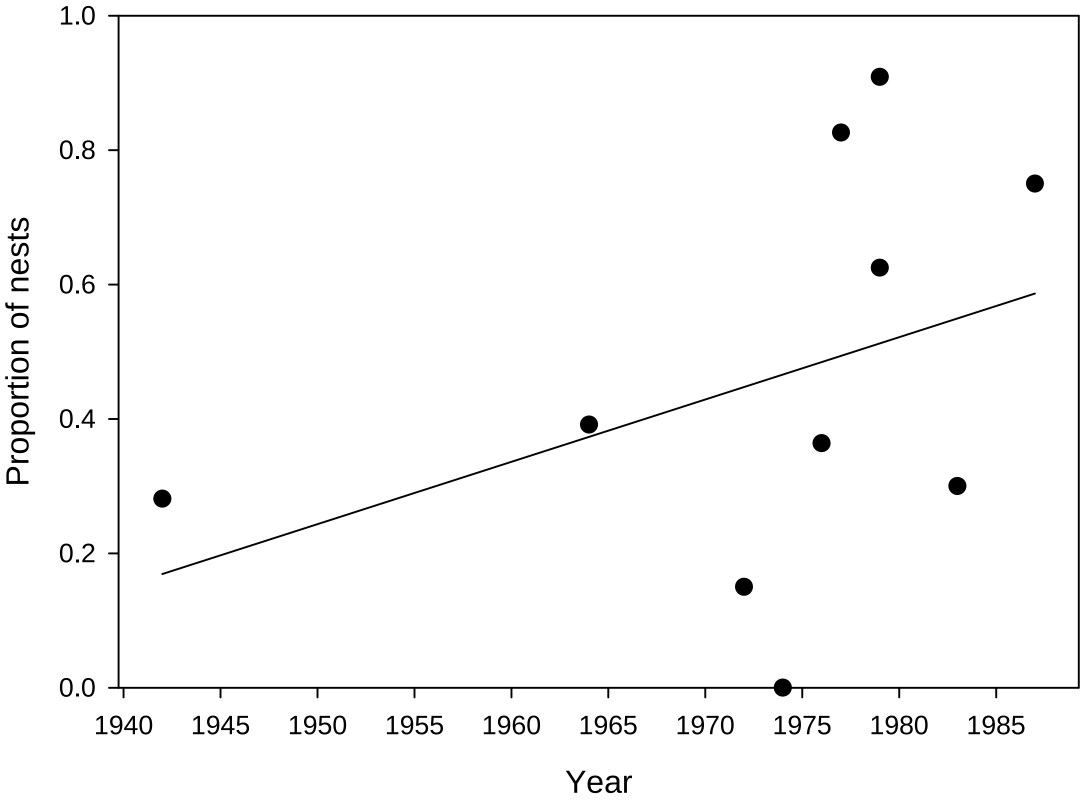

Figure 5.16: Proportion of gray partridge nests in fencelines or roadsides from 10 North American studies from 1942-1987 (year represents the last year of each study) (Carroll, 1989; Church, 1980, 1984b; Gates, 1973; Hunt, 1973; Hupp et al., 1980; Lokemoen and Kruse, 1977; McCabe and Hawkins, 1946; McCrow, 1982; Weigand, 1980) (y = 0.0093x - 17.867; r^2 = 0.15).

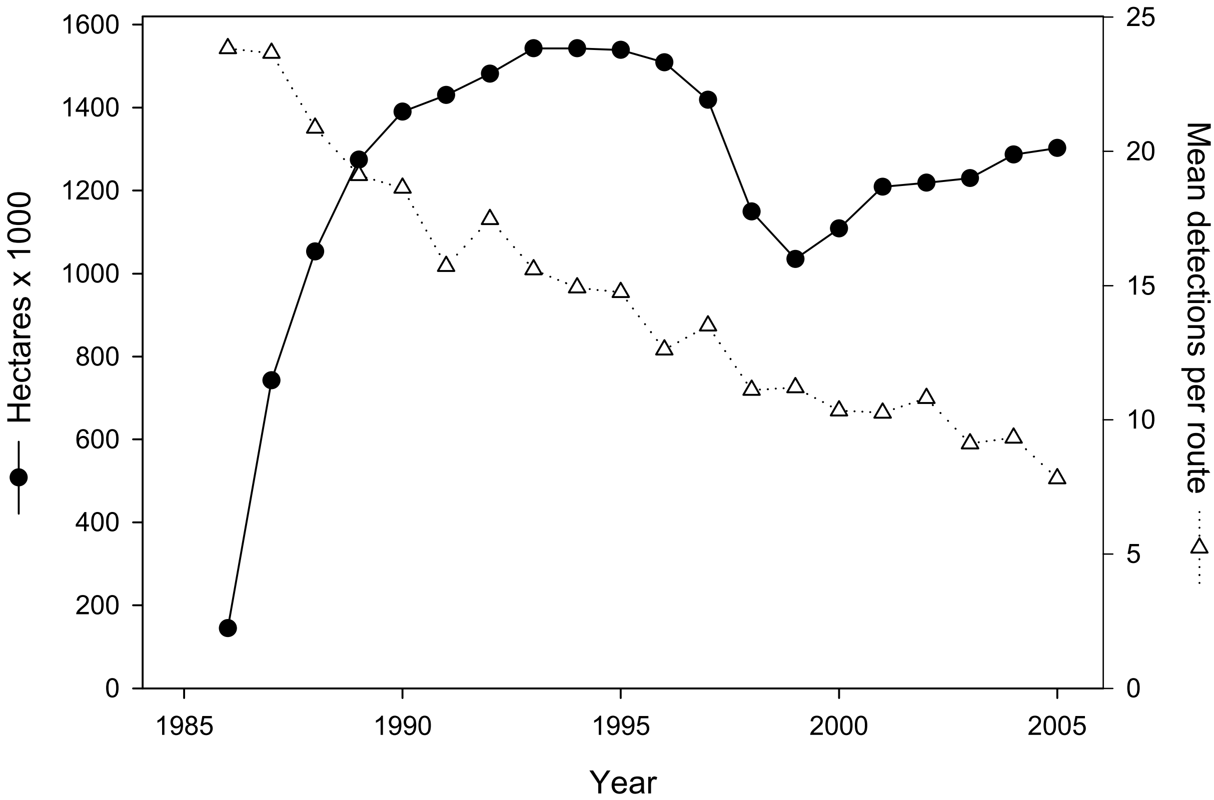

Figure 5.17: Hectares of land enrolled in the Conservation Reserve Program and mean detections of Northern bobwhite on Breeding Bird Survey routes in Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, and Tennessee during 1986-2005.

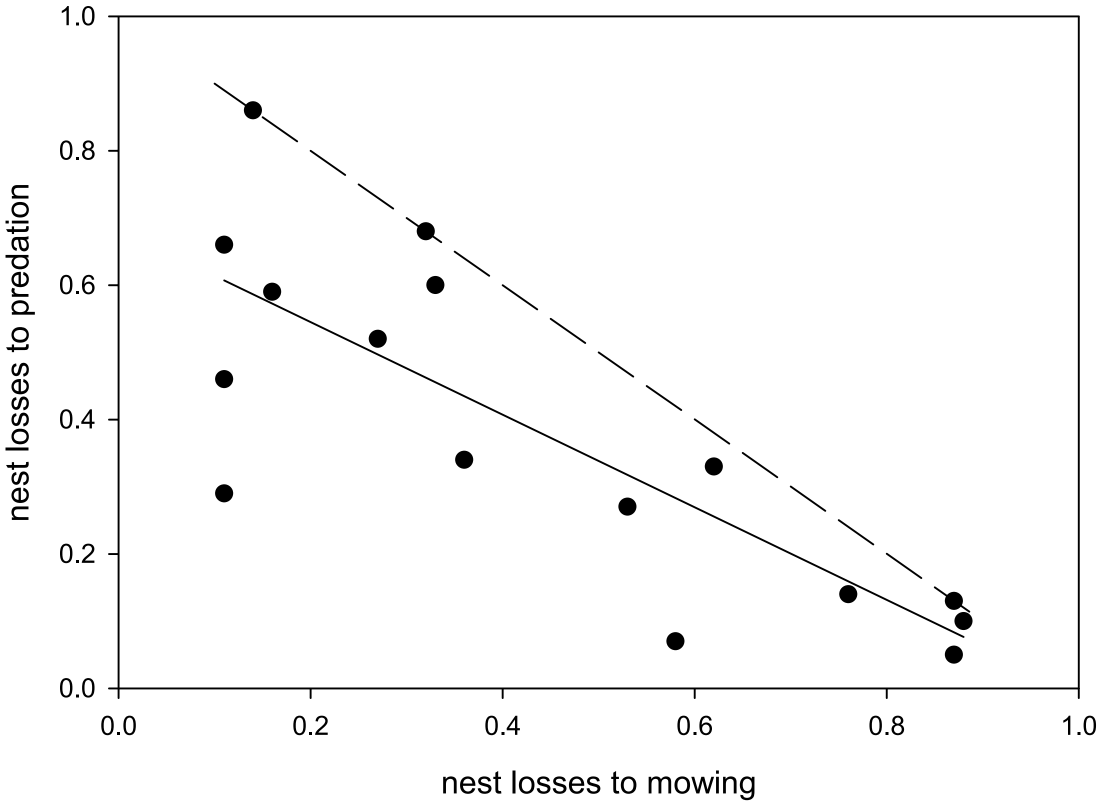

Figure 5.18: Proportions of known nest failures of gray partridges attributed to mowing and predation from 16 studies [values from Potts (1980); Carroll (1989)] (y=0.68-0.69x; r^2=0.65). Dashed line represents complete compensation of known nest failures between mowing and predation.

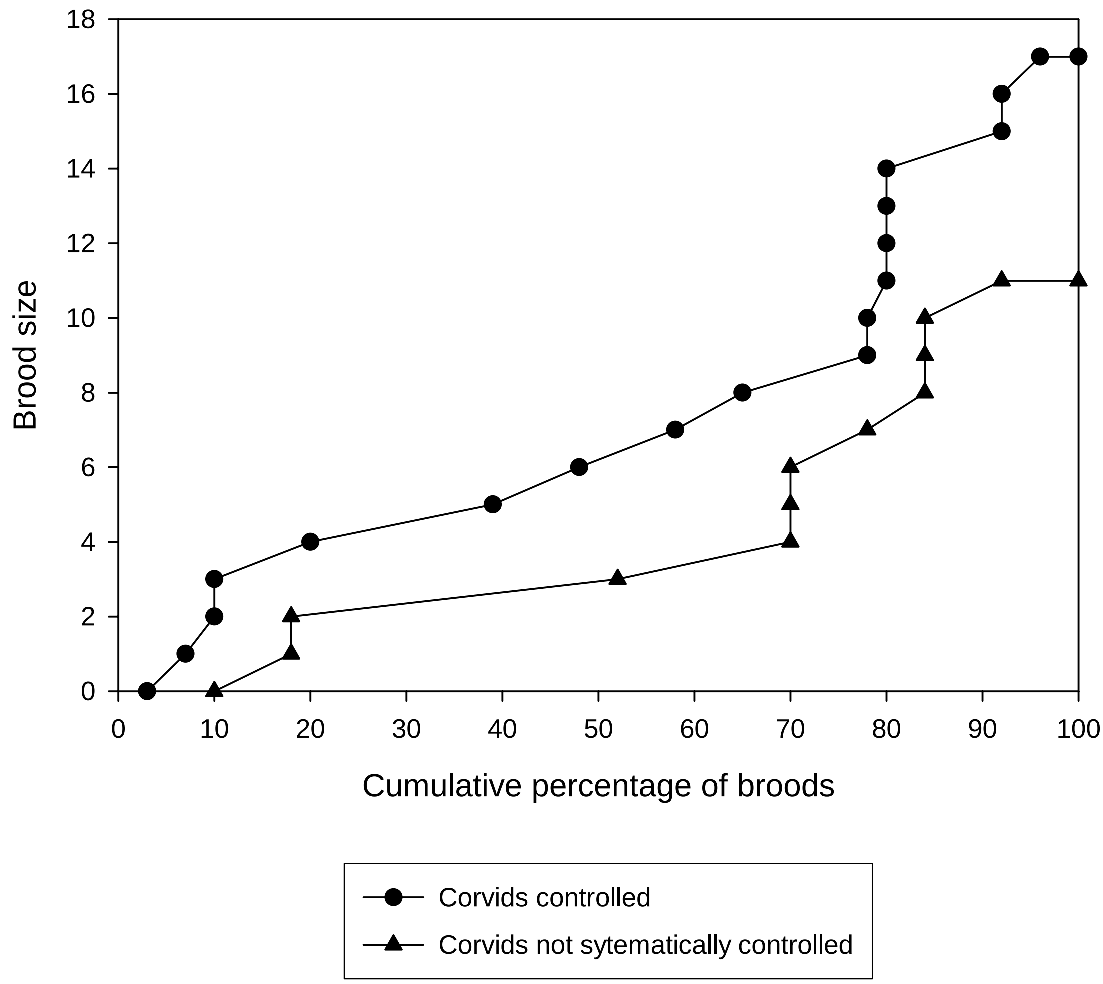

Figure 5.19: Accumulated partridge brood size frequency distributions from two sites in eastern England with and without systematic control of corvids (Potts and Vickerman, 1974).

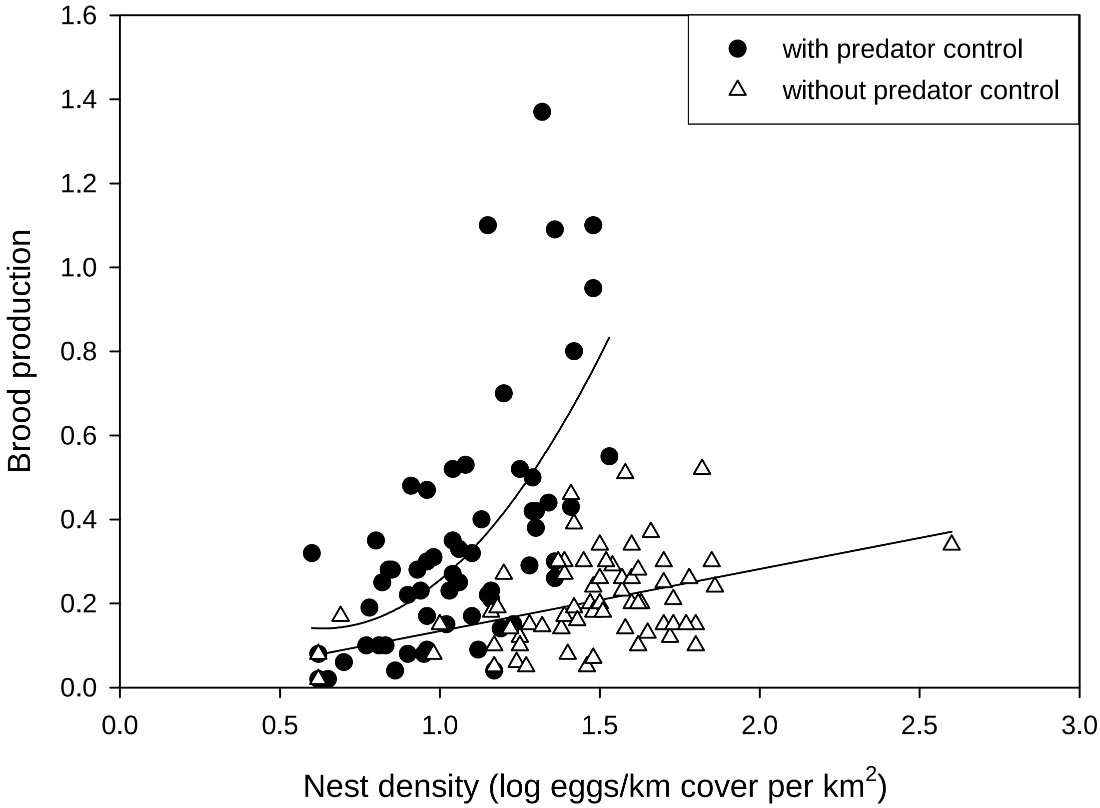

Figure 5.20: Relationship between brood production and nest density from 3 study areas in southern England (Sussex, Damerham, Courtyard Farm) during periods with (y=-0.49-1.09x+0.86x^2; r^2=0.38) and without (y=-0.014+0.15x; r^2=0.17) predator removal. Brood production = $\log(\frac{males}{broods})+\log(\frac{males}{females})$. Data from Potts (1986).

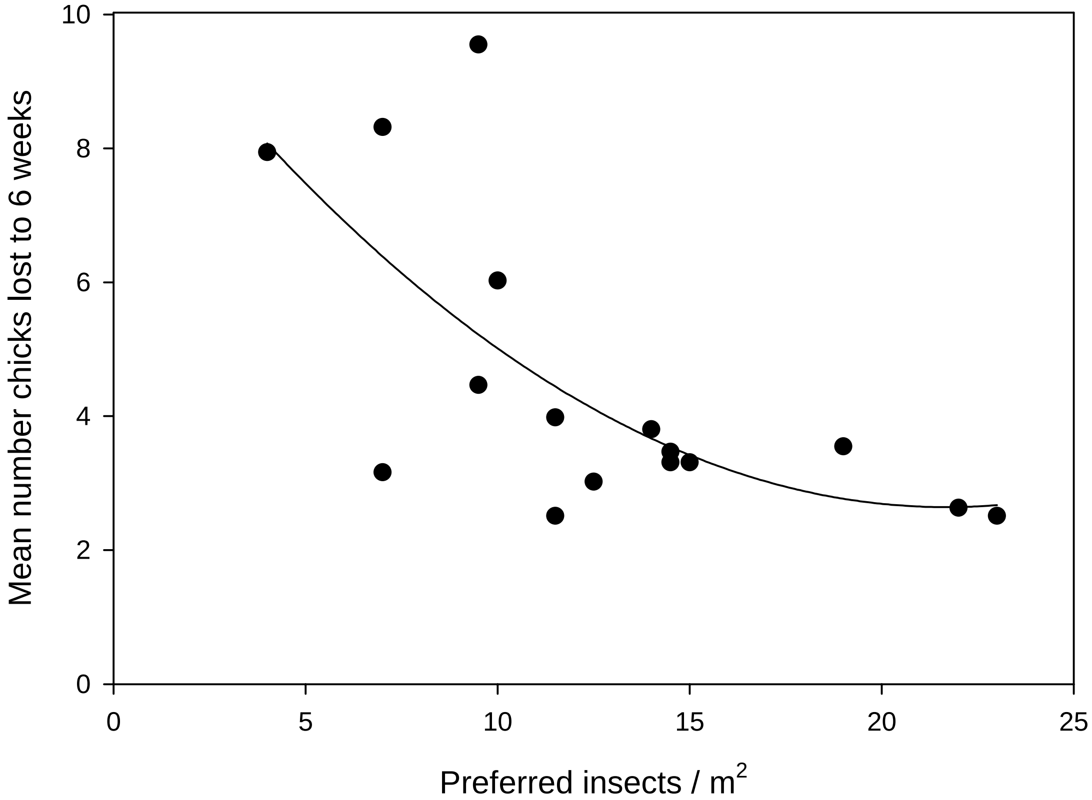

Figure 5.21: Annual chick mortality (chicks hatched/chicks at 6 weeks) in gray partridge in relation to the density of preferred insects in the third week of June in Sussex England, 1969-1985 (y=10.81-0.75x+0.02x^2; r^2=0.46). Data from Potts (1986).

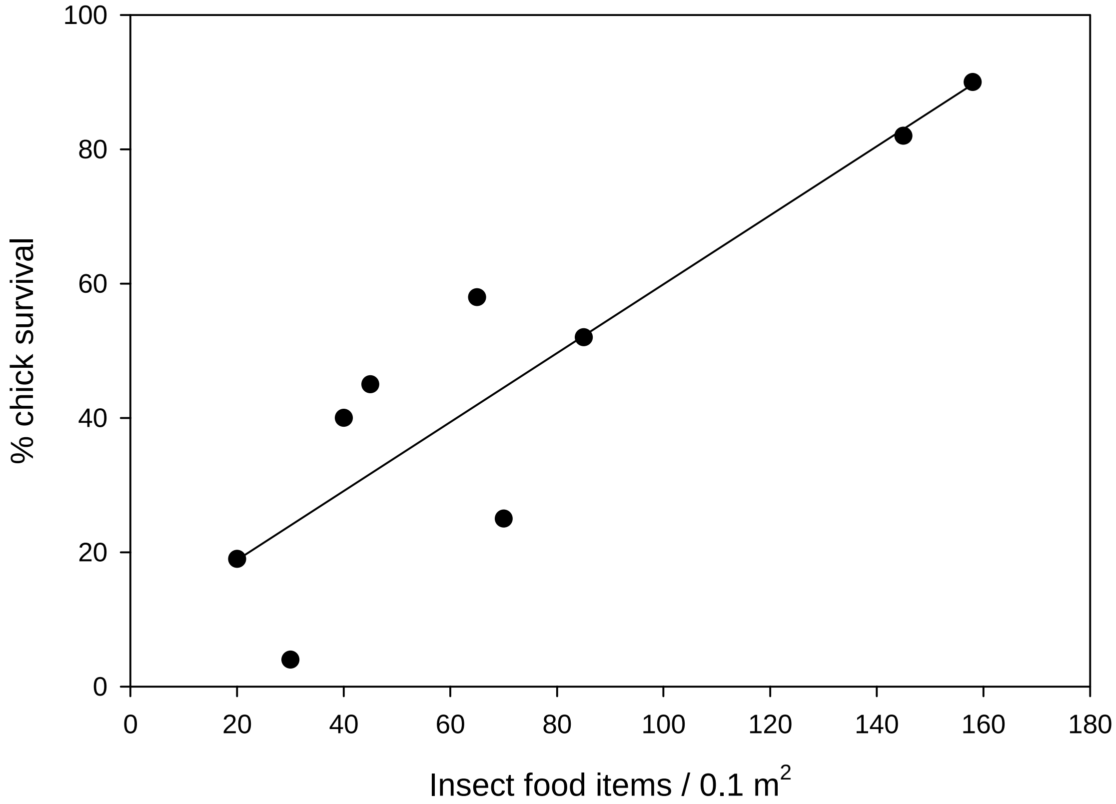

Figure 5.22: Pheasant chick survival to 25 days in relation to the density of insect food items within brood home range (y=8.58+0.51x; r^2=0.79) Hill (1985).

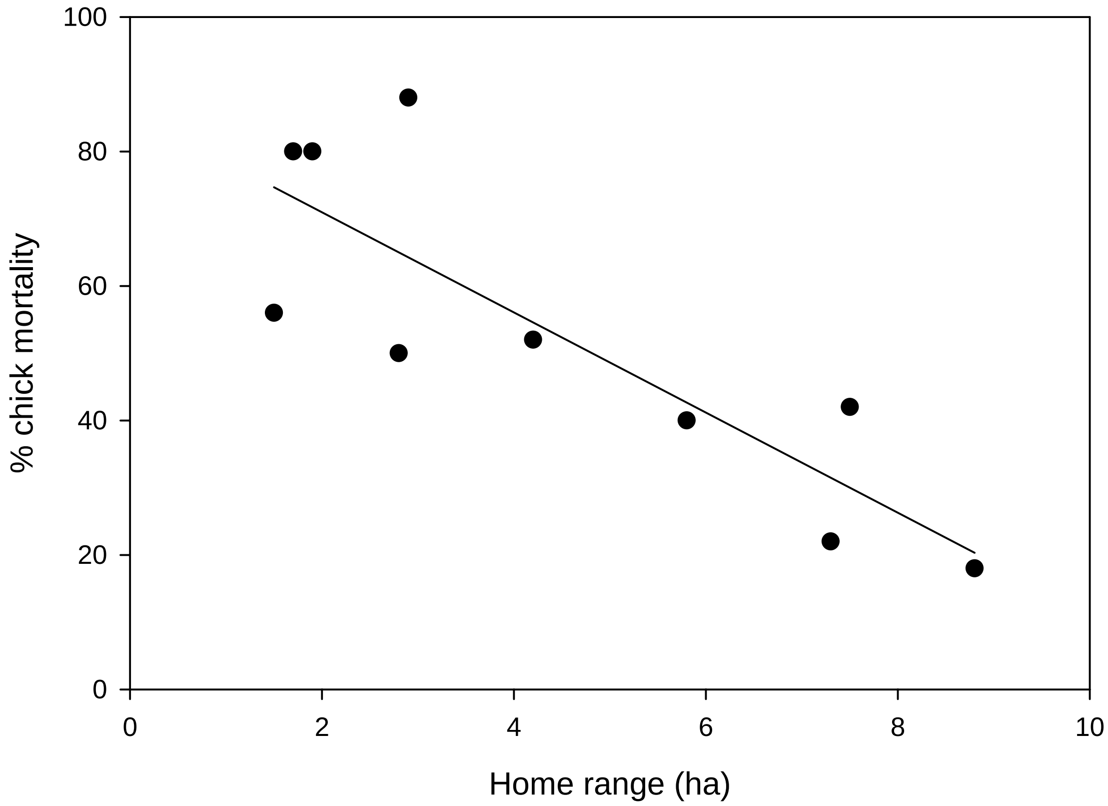

Figure 5.23: Pheasant chick mortality in relation to area of brood home range in England (y=85.88-7.45x; r^2=0.71) Hill (1985).

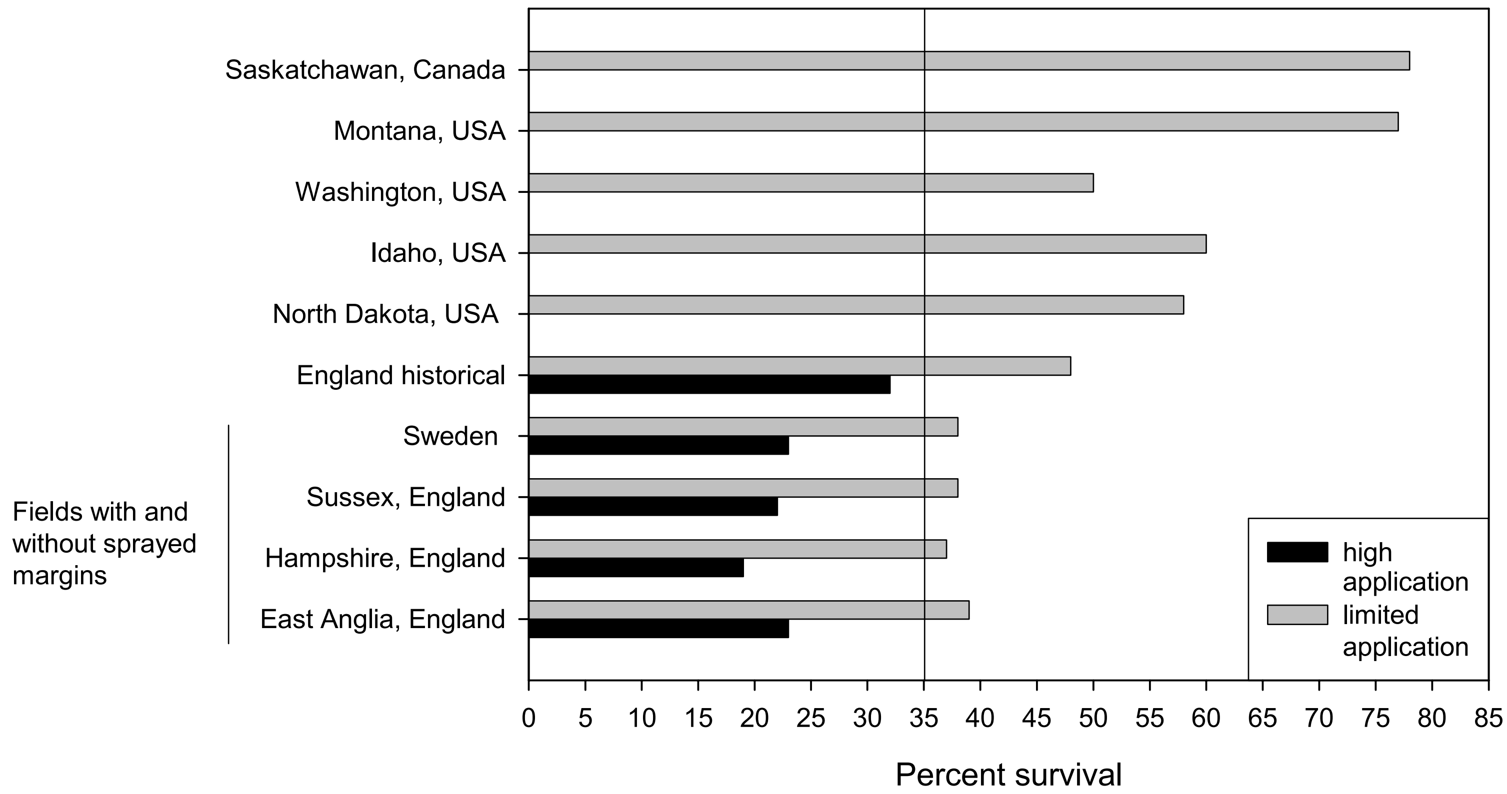

Figure 5.24: Mean chick survival of gray partridges for Saskatchawan, Canada (1971-1972); Montana, USA (1968-1974); Washington, USA (1940-1942); Idaho, USA (1976-1978); and North Dakota, USA (1985-1987) with low pesticide application; England during 1903-1952 (pre-pesticide era) and 1962-1993 (widespread pesticide use); and for Sweden; Sussex, England; Hampshire, England; and East Anglia, England between fields with and without unsprayed cereal field margins (Aebischer and Ewald, 2004; Carroll, 1989; Chiverton, 1999; Hunt, 1973; Mendel and Peterson, 1980; Potts and Aebischer, 1994; Weigand, 1980; Yocum, 1943). Line indicates 35% survival which is require for population stability (Aebischer and Ewald, 2004)

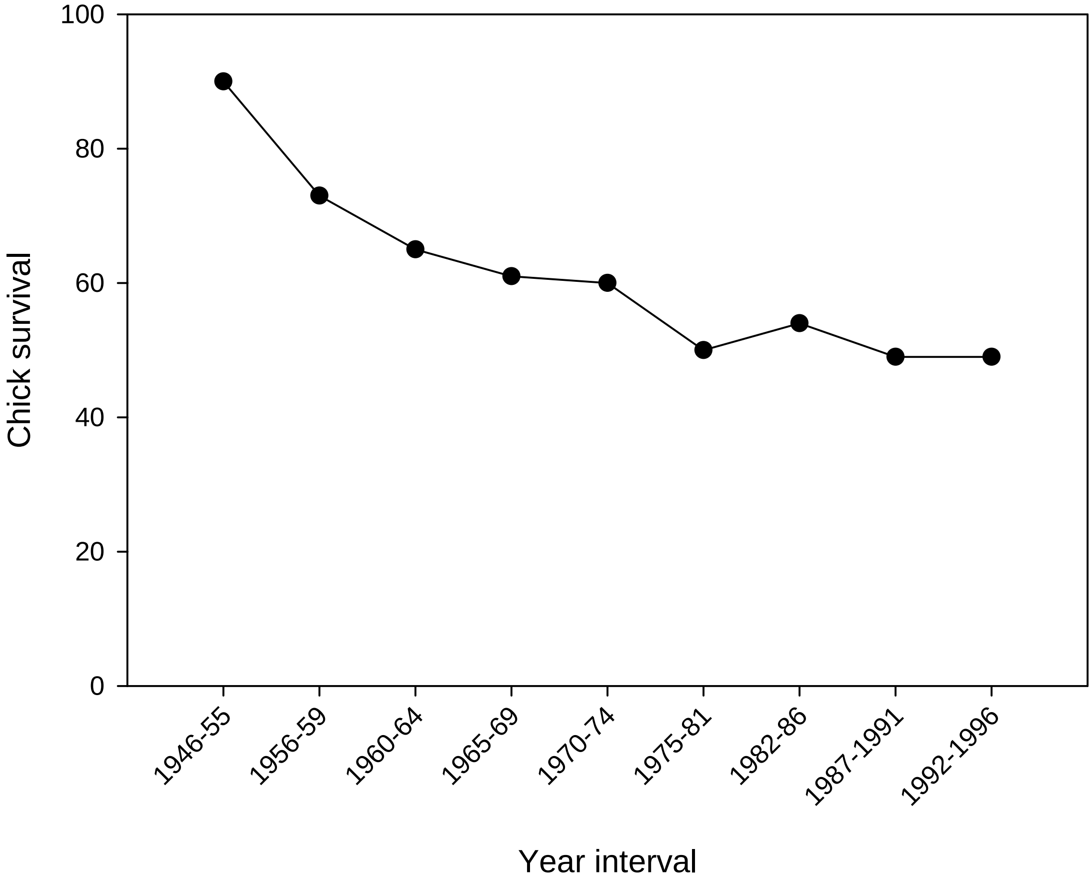

Figure 5.25: Mean pheasant chick survival to 6 weeks for Illinois from 1946-1996 (Warner, 1984; Warner et al., 1999).

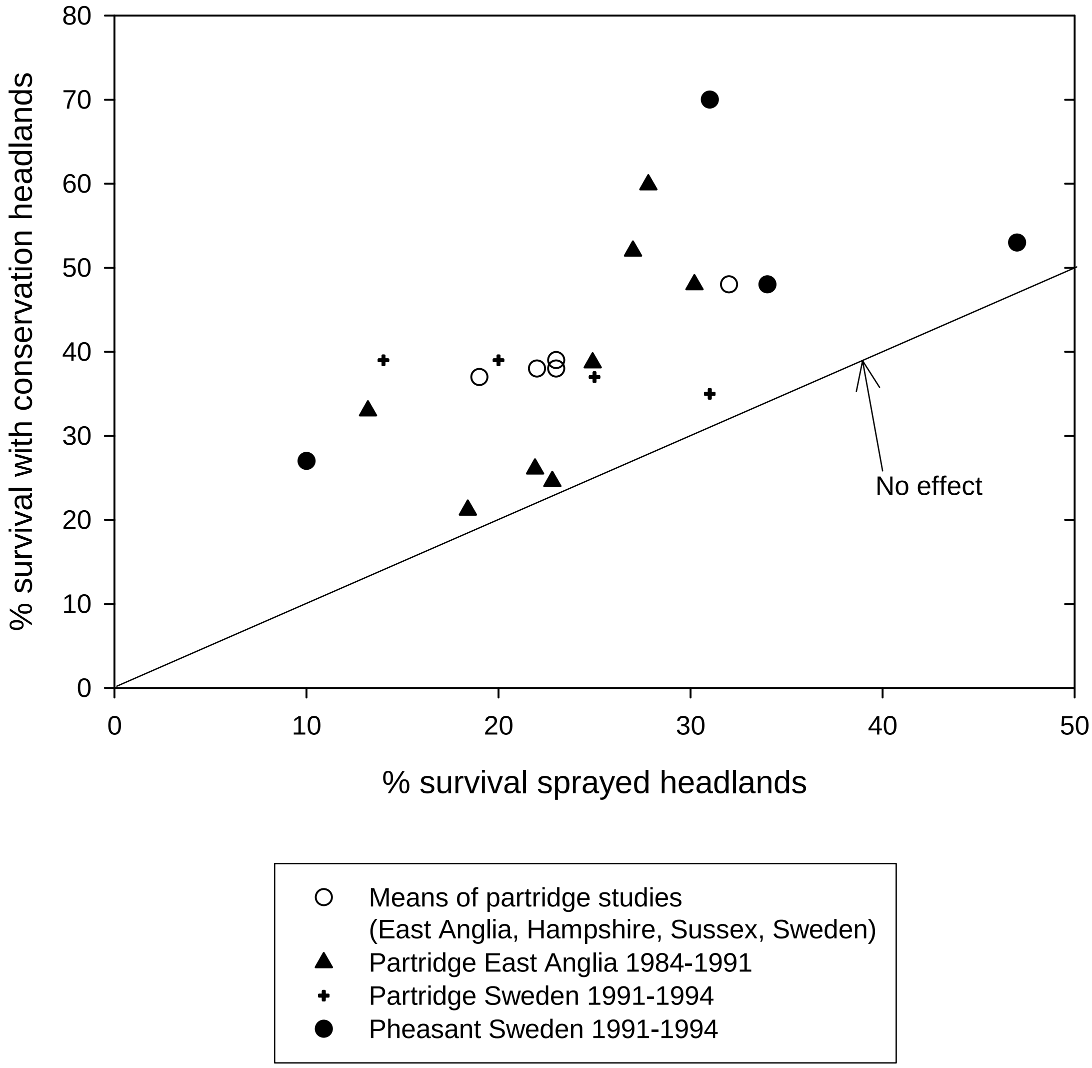

Figure 5.26: Gray partridge and pheasant chick survival in fields with and without conservation headlands (Aebischer and Ewald, 2004; Chiverton, 1999; Hill, 1985). Line represents a neutral effect of conservation headlands on chick survival.

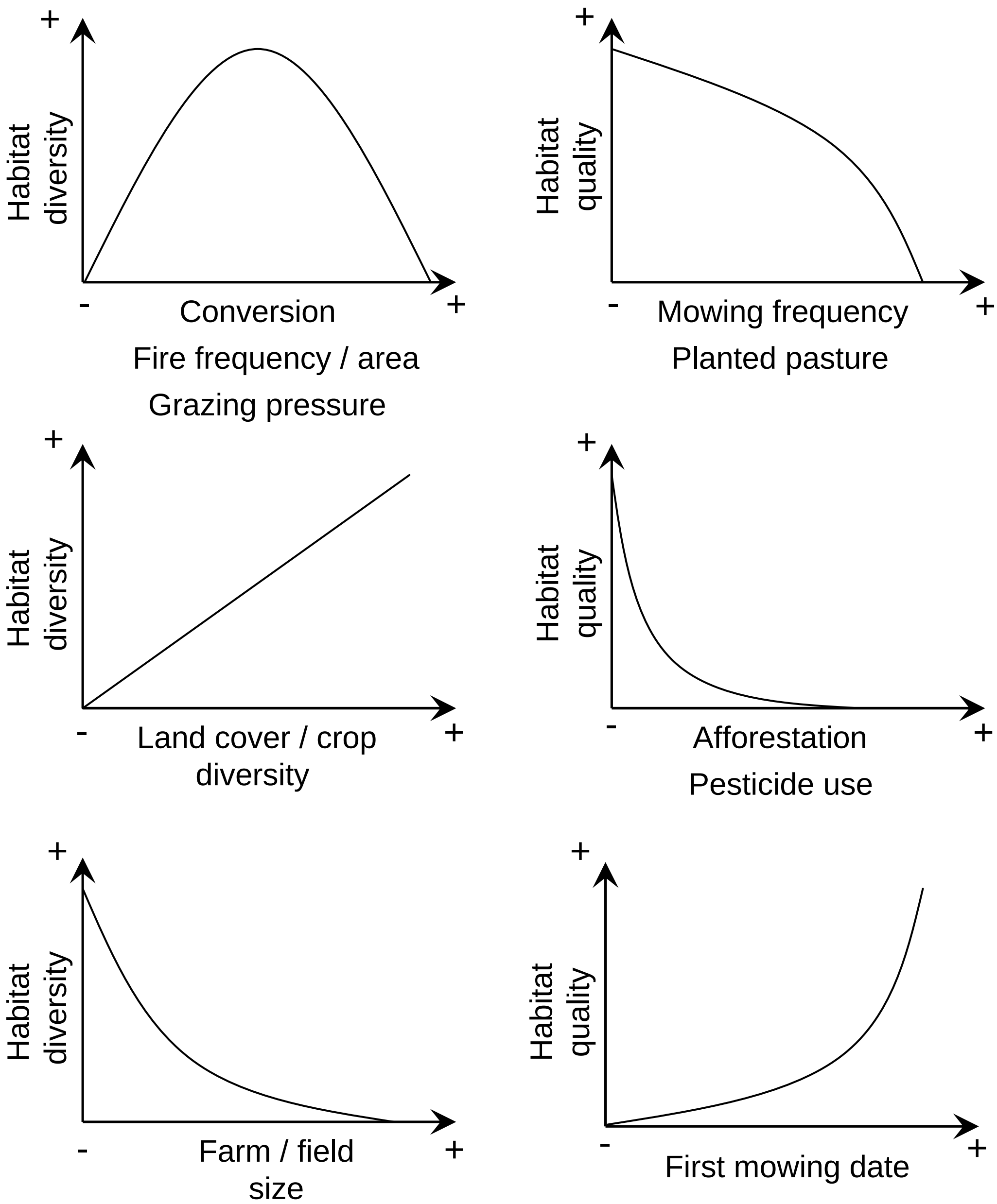

Figure 5.27: Generalized relationships of the effects on habitat diversity by land conversion, frequency and size of fire, grazing pressure, land cover and crop diversity, and farm and field size and the effects on habitat quality by the frequency of pasture mowing, the area in planted pastures, the area afforested, pesticide use, and the date of first mowing.

Chapter 6

Implications of agricultural intensification in the Pampas and northern Argentina for wildlife

In the previous chapters, I have shown the trends and commonalities in the intensification of management of agroecosystems among the principal temperate food producing regions of the world and related those trends to their effects on avian populations and diversity, particularly gamebirds. For Argentina, although there has been evaluations of the overall ecological implications of changes in agroecosystem management (LART-FAUBA, 2004; Ferreyra, 2001, 2006; Ghersa et al., 2002; Manuel-Navarrete et al., 2005; Peiretti, 2001; Qaim and de Janvry, 2005; Qaim and Traxler, 2005; Sattore, 2001; Solbrig and Vera, 2001; Viglizzo et al., 2001, 2003, 2004), there has been little research on, or assessment of, the implications of agroecosystem management on wildlife. This leads to several questions. What have been the historical implications of the introduction of row crop agriculture and livestock husbandry on wildlife in the Pampas and northern Argentina? What have been the effects of rapid intensification over the last 15 years? What is the future outlook?

Despite the dearth of specific research on the effects of agroecosystem management on wildlife in the Pampas and northern Argentina, long-term data sets on agricultural development in these regions (Chapter 4) and others (Chapter 3), research on agricultural management in Argentina, as well as a plethora of studies related to wildlife in agroecosystems (Chapter 5), allow for meaningful comparisons among regions and for strong influences to be made regarding the Pampas and northern Argentina. Moreover, historical accounts aid in determining baseline conditions described by early naturalists prior to large-scale agricultural conversion and (Darwin, 1839; Hudson, 1892).

6.1 The Pampas

In the Pampas, the most significant and most obvious implication of the development of grazing and row crop agriculture has been the large-scale loss, fragmentation, and degradation of grasslands (Bilenca and Miñarro, 2004). The conversion and/or degradation of Pampean grasslands, along with increasing human settlement, led to severe range contractions of or decreased populations of guanacos (*Lama guanicoe*) (Parera, 2002; Roig, 1991), Pampas deer (*Ozotoceros bezoarticus*) (Roig, 1991; Wemmer, 1998), puma (*Felis concolor*) (Parera, 2002), vizcacha (*Lagostomus maximus*) (Parera, 2002), mara (*Dolichotis patagona*) (Parera, 2002), Pampas meadowlark (*Sturnella defilipii*) (Fernandez et al., 2003; Tubaro and Gabelli, 1999), strange-tailed tyrant (*Alectrurus risora*) (DiGiacomo and DiGiacomo, 2004), black-and-white monjitas (*Heteroxolmis dominicana*) (Fraga, 2003), saffron-cowled blackbird (*Xanthropsar flavus*) (Fraga et al., 1998), and greater rheas (*Rhea americana*) (Bucher and Nores, 1988). The contraction in range of all these species was strongly related to increasing settlement of the Pampas, with the greatest effects related to the increasing conversion of grasslands to row crop agriculture starting in the first half of the 20^{th} century.

As would be expected for grassland dependent species, based upon the avian literature (Chapter 5), the remaining populations of the aforementioned bird species are now generally restricted to areas of extensive cattle grazing with little or no row crop agriculture (Fraga, 2003). This pattern is also evident for mammals, particularly larger species. For example, the remaining populations of Pampas deer are confined to the coastal marsh lands and driest areas at the northeastern and western extents of the Pampas, respectively (Demaría et al., 2002; Parera, 2002; Wemmer, 1998). Similarly, puma are now mostly confined to the dry western Pampa and the Sierra de la Ventana; a small mountain range in the austral Pampas, where the few remaining guanacos in the Pampas are also found (Parera, 2002).

The geographic characteristics of the Pampas determined the spatial and temporal pattern of conversion and landuse which continues to the present. *Sensu* the ideas of Ricardo (1817) (theory of decreasing marginal product), the expansion of row crop agriculture and grazing in the Pampas was not homogeneous and proceeded with the most accessible and most suitable land for row crop agriculture being developed first. This was the rolling Pampas (see Fig. 4.12), which is the closest to Buenos Aires and the Río de la Plata, and the best land for row crop agriculture (Morello and Solbrig, 1997; Viglizzo et al., 2003).

As with the western expansion of agriculture in the United States or its inland expansion in South Africa discussed in Chapter 3, the process of agricultural expansion is of significance because it has determined both the temporal and spatial patterns of land use in the Pampas. Although more than a century old, this process continues to the present and determines the ecological characteristics of the Pampas (Chapter 4; Viglizzo et al. 2001, 2003; Baldi et al. 2006). As discussed by Viglizzo et al. (2001), and later by Viglizzo and Frank (2006), Baldi et al. (2006), Sattore (2001), and in Chapter 4, the dominance of row crops in the Pampas is related to soil fertility and drainage, temperature, and precipitation. The result of this relationship has been that the dominance of cultivated land, and subsequently the area of converted grasslands, has been greatest in the rolling pampas, followed by the inland pampas and the austral pampas, while the mesopotamian and flooding pampas have seen the least amount of cultivation (Baldi et al., 2006; Sattore, 2001; Viglizzo et al., 2001; Viglizzo and Frank, 2006).

The implication of this spatial and temporal pattern in land use are multi-faceted. Aside from the total area of grasslands converted, the most productive land for cropping has also been subjected to the greatest amount of grassland fragmentation (Baldi et al., 2006). Since many obligate grassland bird species are area sensitive (Chapter 5), from a conservation perspective, fragmentation of grasslands is as equally important as grassland loss. At the same time consolidation of land holdings, the dominance of several cultivars (particularly soybean and wheat), and increased mechanization have made the converted land more homogeneous, both spatially and temporally (Chapter 4).

The spatial pattern of grassland conversion has lead to a expected result that areas dominated by extensive livestock production now serve as refugia for the species most sensitive to this process. Although this process is not atypical, in the case of the Pampas it is of concern because the recent expansion of row crop cultivation has been greatest in regions formerly considered the least profitable for cultivation (mesopotamian, Western interior, and Flooding Pampa) and contain some of the larger remnants of natural grasslands (Bilenca and Miñarro, 2004). Moreover, grazing practices both in areas of row crop agriculture and in rangelands have been undergoing significant changes (Chapter 4).

The intensification of agriculture in the Pampas has included the expansion of annual row crops at its drier western extent (Chapter 4), where formerly, the high variability of precipitation in the region favored relatively low levels of row crop agriculture (Ghersa et al., 2002; Sattore, 2001; Viglizzo et al., 2001). The introduction of no-till agriculture, which minimizes soil erosion and maintains higher levels of soil moisture, has increased the potential profitability of row crop agriculture, making it more economically feasible in drier regions of the Pampas (Peiretti, 2001; Sattore, 2001). Areas formerly dominated by mixed agriculture and grazing systems have witnessed the conversion of grassland remnants or natural pastures to row crops, more intensive management of forage crops for silage (multiple mowings and fertilizer application), and the concentration of livestock and overgrazing.

Changes in rangeland management in Argentina, as in other regions (Chapter 5), has been observed to have both positive and negative implications on avian abundance and diversity (Comparatore et al., 1996; Herrera et al., 2004; Isacch and Martinez, 2001; Isacch et al., 2005; Zalba and Cozzani, 2004). In general, of greatest importance is the development and maintenance of vegetational heterogeneity in both vertical structure, habitat types, and diversity of plant species. Moderate grazing pressure and infrequent burning of natural pastures has been to shown to maintain an optimum in this respect, however, an effort needs to made to maintain tall grassland habitats (Comparatore et al., 1996; Herrera et al., 2004; Isacch and Martinez, 2001; Isacch et al., 2005; Zalba and Cozzani, 2004).

Additionally, in the rangelands of the western interior Pampas the replacement of natural grasslands with monocultures of exotic forage grasses has become increasingly common, particularly on poorly managed and degraded natural range. Demaría et al. (2002) have discussed this process in relation to Pampas Deer in the province of San Luis, where this process has been promoted by the provincial government with the intention of increasing the carrying capacity of degraded rangelands for livestock; however, large portions of intact and healthy grassland habitat have also been converted. Since well managed natural range is more profitable in the long-term, not only has the conversion of natural grassland been detrimental from a wildlife conservation perspective, it also detrimental in the long-term from an economic perspective (Demaría et al., 2002).

The conversion of lands of marginal agricultural value, but high conservation value also occurs through the promotion of afforestation activities which, given the negative effects on birds associated with the presence of woody vegetation within grassland systems (Chapter 5) is among the more problematic land uses in Argentine grasslands. The small-scale and localized planting of trees as wind breaks and for firewood historically has been, and continues to be, common within the Pampas and the more northerly Campos grasslands of Argentina and Uruguay. The introduction of large-scale tree farming, although limited but increasing in the Pampas, is extensive in the Campos and may be proportionally the greatest threat to grassland wildlife in Argentina. Even tthough plantation forest occupy a relatively small area, their planting is promoted only for marginal lands (*i.e.* extensive grazing land), most of which are of high conservation value and are associated with species of conservation priority that were formerly common in both the pampas and the campos

(DiGiacomo and DiGiacomo, 2004; DiGiacomo and Krapovickas, 2001; Fraga et al., 1998; Fraga, 2003).

The loss of grassland remnants and pastures, combined with continual cropping and overgrazing, has decreased the value of agricultural land to most wildlife, however, their effects are likely further exacerbated by the increased use of agrochemicals that has occurred since the early 1990's. Although incidences of pesticide induced mortality, such as those of Swainson's hawks (*Buteo swansoni*) in the Pampas during the mid-1990's justifiably draw attention (Goldstein et al., 1999), as discussed in Chapter 5 the indirect effects of pesticides are less obvious, but far more significant, through bottom-up trophic effects. Despite their importance these indirect effects have been overlooked or not considered in both research and assessments of ecological effects of agricultural intensification in the Pampas (Ferreyra, 2001, 2006; Ghersa et al., 2002; Manuel-Navarrete et al., 2005; Solbrig and Vera, 2001; Viglizzo and Frank, 2006; Viglizzo et al., 2003, 2004).

The increased use of external inputs, stemming from the globalization of the Argentine economy, has allowed for the development of row crop agriculture in areas of the Pampas where it was formerly rare or absent, while producing throughout the Pampas most, if not all, of the negative effects on wildlife created by agricultural intensification. Importantly, the accessibility of industrialized farming inputs (*e.g.*, machinery, fertilizer, pesticides), combined with the recent lucrative nature of agriculture in Argentina, has not only manifested itself in increasing the area of row crops and the intensification of farm and rangeland management in the Pampas, but has served as the impetus for the expansion of row crop agriculture northwards and westwards into the Chaco and Yungsa forest, while also leading to the intensification of existing agricultural land in these areas.

6.2 The Chaco and Yungas

The long-term degradation and the recent rapid rate of deforestation of the Chaco and Yungas forest in northern Argentina (Chapter 4) presents a significant threat for wildlife conservation, not only in Argentina, but in neighboring Bolivia and Paraguay (Bucher and Huszar, 1999; Grau and Brown, 2000; Grau et al., 2005a,b; Kaimowitz and Smith, 2001; Schofield and Bucher, 1986; Steininger et al., 2001; Zak et al., 2004). Although the extent of habitat degradation and loss is considerable, both the research on the wildlife of the region and global awareness of the situation, is disproportionate to the threat (Ojeda et al., 2003; Redford et al., 1990). At a regional level the Chaco and Yungas forest of northern Argentina still support a nearly complete assemblage of wildlife and maintain high levels of biodiversity and endemism (Beissinger et al., 1996; Brown and Malizia, 2004; Ojeda and Mares, 1989; Ojeda et al., 2003; Redford et al., 1990) and subsequently the conversion and mismanagement of these forest presents a considerable conservation crisis locally, continent-wide, and globally.

The effects on wildlife by the deforestation of both Chaco and Yungas forest for the development of row crop agriculture would obviously be expected to be negative, however, centuries of livestock grazing throughout the region has also greatly affected their vegetational composition and structure (Bucher, 1982; Bucher and Huszar, 1999; Cabral et al., 2003; Reboratti, 1998; Roig, 1991). In the Chaco, which was originally a mosaic of grasslands, savanna, and woodlands, overgrazing and reduced frequency of fire has led to the development of dense scrub ("El Chaco impenetrable") in many areas and in the Yungas overgrazing and trampling from transhumance management of livestock remains a principal threat to the remaining forest (Bucher and Huszar, 1999; Grau and Brown, 2000). Grazing management has also been influential in altering fuel loads for fires, including the introduction of exotic forage grasses, as well as affecting the frequency and intensity of burning (Brown and Malizia, 2004; Bucher, 1982; Bucher and Huszar, 1999; Zak et al., 2004).

As with the Pampas the spatial pattern of land use in the Chaco and the Yungas is dependent upon the factors that determine the suitability of land for different landuses, which in turn dictates the types and severities of threats to wildlife. As discussed in

Chapter 4 the westernmost Chaco and the Yungas forest have a centuries-long history of land conversion for agriculture and the eastern Humid Chaco saw extensive development beginning in the early 20^{th} century. This pattern is principally due to sufficiently high annual precipitation making row crop agriculture economically feasible in these regions and, similarly, the recent expansion of row crop agriculture is partly a function of increasing annual precipitation and crop field management increasing soil moisture (Chapter 4).

The spatial pattern of historic and recent deforestation of Chaco and Yungas forest is of particular relevance to wildlife conservation because areas of highest biodiversity are the regions that have and continue to be under the highest developmental pressures. For example, the interface of low elevation Yungas and Chaco forests supports greater mammalian diversity compared to either higher elevation Yungas or the adjoining semi-arid Chaco (Ojeda and Mares, 1989) but is little protected and is subjected to some of the greatest developmental pressures (Brown and Malizia, 2004; Gasparri and Parmuchi, 2003; Montenegro et al., 2003). Moreover, aside from habitat loss, agricultural management in those regions have undergone rapid intensification in management (Chapter 4).

Although the deforestation of the Chaco and Yungas presents a considerable threat to wildlife, the degradation of forests and other anthropogenic impacts, such as harvest, are similarly of concern. In the case of the Chaco forest degradation stemming from grazing has been shown to affect both forest structure and species composition (Adamoli et al., 1990; Boletta et al., 2006; Bucher, 1982; Bucher and Huszar, 1999; Cabral et al., 2003), which in turn has been illustrated to affect avian and arthropod distributions (de Casenave et al., 1998; Gardner et al., 1995). Although much of the Chaco was not exploited for livestock production until the early 20^{th} century, grazing of domestic livestock had already produced noticeable large-scale changes in land cover by this time (Adamoli et al., 1990; Bucher and Huszar, 1999; Schofield and Bucher, 1986). Writing of the Argentine Chaco, Grubb (1919) stated "*Some thirty years ago great plains of luxuriant grass extended along the banks of the upper Bermejo* [River] *and in other parts. Owing, however, to* *over-stocking and other causes, these grass plains have become to a great extent covered with low scrub, and many large stretches during the dry season are perfectly bare, so that clouds of dust follow the traveller* [sic] *as he journeys*", illustrating that by the turn of the 20^{th} century the vegetative mosaic of the Chaco was already compromised in some regions.

The long-term effects of introduced livestock in Argentine Chaco has produced direct influences on wildlife through impacts on landscape configuration, vegetative structure and composition, and by direct competition with native species. Combined with deforestation and direct human exploitation or persecution, grazing effects have been responsible for the extirpation of the guanaco and Pampas deer from the Argentine Chaco (Bucher, 1987; Parera, 2002; Roig, 1991; Terán, 2000) and the constriction in range and population reductions of Lowland tapir (*Tapirus terrestris*), peccaries (*Tayassu tajacu, Tayassu pecari, Catagonus wagneri*), Giant armadillo (*Priodontes maximus*), jaguar (*Panthera onca*), Maned wolf (*Chrysocyon brachyurus*), Vampire bats (*Desmodus rotundus*), and Greater rheas and other bird species (Altrichter, 2005b; Altrichter and Boaglio, 2004; Bucher and Nores, 1988; Chalukian, 2003; Parera, 2002; Roig, 1991).

At the same time, the loss of large native herbivores, the introduction of exotic herbivores, and poor range management facilitated population increases in two rodent species, the Chacoan mara (*Dolichotis salinicola*) and the vizcacha (Bucher, 1987). The presence of large populations of these species maintain the vegetative structure and composition of degraded Chaco forest through herbivory and inhibiting fire by reducing fuel loads, thus promoting dense shrub growth even when livestock are removed (Bucher, 1987).

In northern Argentina the harvest of wildlife for food and commercial products can be important supplements to the incomes of rural people (Altrichter, 2005a; Banchs and Moschione, 2006; Barbarán and Saravia Toledo, 2000; Porini, 2006) The exploitation or persecution of wildlife in northern Argentina is highly correlated with both the spatial and temporal distributions of human settlements. Harvest for commercial products, food, or the pet trade is greatest in relation to human habitation and subsequently, populations of exploited species are reduced as well (Altrichter, 2005b; Altrichter and Boaglio, 2004; Banchs and Moschione, 2006; Bucher and Nores, 1988). For persecuted species such as jaguars, Altrichter et al. (2006) found presence to be correlated with both low human habitation, low road densities, and time since the founding of settlements and, similarly, the extirpation of the Spectacled bear (*Tremarctos ornatus*) in the Argentine Yungas can be attributed to human persecution (Peyton, 1999).

The role of harvest in the conservation of wildlife in northern Argentina is of importance because, even though degradation of the regions forest is extensive, they still do, or can support most, if not all of the historical biodiversity. In the case of Chaco forest, wildlife has been shown to tolerate moderate fragmentation from deforestation (Areskoug, 2001; de Casenave et al., 1998), however, deforestation is associated with increased human habitation and subsequently harvest. This relationship implies that the ecological effects of low or moderate levels of landuse are less significant for wildlife conservation than the economic and social drivers associated with process of deforestation. The success of management programs for exploited species in the Chaco and Yungas, such as the blue-fronted parrot (*Amazona aestiva*) and tegu lizards (*Tupinambis* spp.), that reduce harvest while increasing overall profits suggests that this is indeed the case (Banchs and Moschione, 2006; Brown and Malizia, 2004; Mieres and Fitzgerald, 2006; Porini, 2006).

The economic well-being of rural people appears to be a key factor intertwined with both the expansion of the agricultural frontier, grazing management, and wildlife conservation. For example in the case of the Chaco, Bucher and Huszar (1999) showed that over the long-term the sound management of grazing, integrated with controlled extraction of natural resources, is most profitable. However, overgrazing reduces the carrying capacity of the range, overexploitation of natural resources reduces profits from this source, and minimal returns for the extracted natural resources due to excessive numbers of middlemen, ultimately negate much of the potential economic profits from Chaco forest (Bucher and Huszar, 1999; Banchs and Moschione, 2006) and leads to land abandonment or sales at reduced prices, which has been exploited by agribusiness for the development of soybean cultivation (Kaimowitz and Smith, 2001).

6.3 Conservation outlook

The recent changes in the agricultural and grazing sectors in the Pampas, Chaco, and Yungas of Argentina have important conservation implications both locally and globally. The continued conversion and fragmentation of temperate grasslands remnants in the Pampas for row crop agriculture, and the intensification in the management of new and existing crop and grazing lands, presents both a biological and economic crisis. Globally, temperate grasslands are the most endangered biome with the Pampas being considered critically endangered based upon proportional areas converted and protected (Hoekstra et al., 2005), but is not emphasized at the international level because of relatively low biodiversity and endemism (Kareiva and Marvier, 2003). Similarly the Chaco forest are overshadowed in South America by issues related to the Amazon or "biological hotspots" of the tropical Andes despite supporting high biodiversity and rates of deforestation that are greater than these regions (LART-FAUBA, 2004; Gasparri, 2004; Gasparri and Parmuchi, 2003; Redford et al., 1990). Even though the Argentine Yungas is included in the tropical Andes hotspot, the relatively small area that it occupies within Argentina and its location at the southern extent of the tropical Andes, appears to reduce the focus on this forest compared to that of more northerly countries.

In the Pampas, as in the prairies of North America, the relatively long-term history of conversion and utilization shrouds the extent of changes that have occurred since European colonization. However, the former mixed cropping and grazing management system utilized until late into the 20^{th} century created and maintained sufficient habitat, both in area and quality, to support a large portion of the faunal community. Moreover, these former systems, because of low amounts of external inputs were more sustainable than the present highly intensified systems (Ferreyra, 2001, 2006). Along the same lines the long history of degradation of rangelands in northern Argentina masks the large ecological changes that have occurred on these lands, which is further exacerbated by the ongoing conversion of these lands to agriculture.

The continued expansion of agriculture into grassland and woodlands, and conversion of grassland remnants and natural pastures to row crops or for intensive grazing, continues to reduce habitat availability while increasing landscape and farm heterogeneity. The loss of remaining habitat and changes in land management are of particular importance within the Pampas since both row crops and pastures are, and have historically been, the dominant landcover and responsible for maintaining much of the original biodiversity of the original system. Subsequently, the effects of changes in agricultural land management on wildlife discussed in Chapter 5 is of relevance to wildlife conservation in the Pampas, but despite this importance research into the ecological impacts of the recent changes of agricultural management in the Pampas has not included the effects upon wildlife or flora conservation (Ferreyra, 2001, 2006; Ghersa et al., 2002; Qaim and de Janvry, 2005; Qaim and Traxler, 2005; Qaim et al., 2003; Solbrig, 1997; Solbrig and Viglizzo, 2000; Solbrig and Vera, 2001; Viglizzo et al., 2003, 2004, 2005).

Although some studies address the ecological impacts of increased use of agrochemicals in Argentina, they all do so in the context of soil and water contamination and potential direct effects (Qaim and de Janvry, 2005; Qaim and Traxler, 2005; Qaim et al., 2003; Solbrig and Viglizzo, 2000; Viglizzo et al., 2004, 2005). As discussed in Chapter 5 the use of pesticides, even those that are relatively benign, has important bottom-up effects upon ecosystem function and to ignore or to discount these effects upon ecosystem function can lead to an underestimation of their impact. Given the global commonalities observed in relation to pesticide use and the conservation of agroecosystem wildlife (Chapter 5, 6), efforts towards researching the recent large increases in agrochemical applications on agroecosystem function and wildlife in Argentina should be a priority, as well as research into management actions to mitigate these effects.

In comparison to the Pampas, the switch from cotton cultivation, which is a very pesticide intensive crop (Qaim and de Janvry, 2005), to soybeans in the eastern Chaco has likely led to decreased pesticide application in volume, frequency of applications, and efficacy. Within the areas of preexisting row crop agriculture, however, the switch to soybean cultivation has also accompanied with the same management as seen in the Pampas, including larger fields, continual cropping, and judicious pesticide use, even if at lower levels than when cotton was dominant. Subsequently, the effect on wildlife within in the eastern Chaco agroecosytems via management would be expected to be analogous to those observed in the Pampas.

With the introduction of genetically modified crops comes concerns of development of resistance of animal and plant pests to pesticides or toxins produced by genetically engineered crops (*e.g.*, *Bt* cotton) (Gould, 1998; Moyer et al., 1994). To minimize the development of resistance it is important to maintain sufficient genetic diversity of pest species so that resistant genotypes do not become dominant. In the case of *Bt* cotton, Qaim and de Janvry (2005) illustrated the effectiveness of leaving at least 20% of cultivated land in unsprayed non-*Bt* cotton, which serves as a refugia for genotypes of pest species and maximizes profits in the long-term. The incorporation of unsprayed crop areas into agricultural management potentially may serve a dual purpose as refugia for wildlife. Considering the large area cultivated in glyphosate resistant soybeans in Argentina, and concerns over the increasing weed resistance to herbicides, promoting unsprayed areas of cropland with Argentina may serve as both a measure to insure long-term agricultural productivity, while creating habitat for wildlife.

Although the introduction of no-till agriculture has had positive impacts for reducing soil erosion and maintaining soil moisture (Peiretti, 2001; Salvador, 2001) it should not be considered the only method available, especially considering the relatively high cost of machinery and inputs associated with it. Other practices, such as contour plowing, are effective for controlling soil loss and may be more economical, as well as effective in maintaining biodiversity. Also, crop residue and standing stubble are potentially beneficial for soil moisture and for wildlife (Chapter 5, 6; pers. obs) compared to fields where crop residues are harvested and their conservation and economic value needs to be assessed.

Row crop agriculture in Argentina is increasingly expanding onto more marginal and less productive land (Chapter 4), while increased external inputs appear to have increased yields little (Fig. 4.9), a process that is evident globally (Cassman et al., 2003). Although from a conservation perspective it is theoretically more viable to maintain only the most productive land under cultivation and maximize yields from those lands (Green et al., 2005), yield stagnation as inputs increase suggests that such a scenario may not be feasible (Chapter 4, Cassman et al. 2003). Moreover, increasing crop yields facilitates the cultivation of increasingly marginal lands in Argentina (Chapter 4), as well as globally, aside from increasing negative ecological impacts associated with intensive agricultural management (Matson and Vitousek, 2006).

The effects of the “improvement” of lands unsuitable for agricultural production on wildlife in Argentina needs to be better assessed, including long-term economic gains from such practices. As previously discussed, the conversion of natural habitats to monocultural pastures is increasingly common and widespread, as are the planting of tree plantations, both of which pose particular threats to wildlife in the Pampas and northern Argentina since these practices generally occur on marginal lands that are also of high conservation value. These changes are occurring on areas of extensive grazing and are reflective of the larger changes in livestock husbandry that have occurred in Argentina.

Although livestock production, particularly cattle, has remained an important component of agricultural production in Argentina, the growing importance of row crops, new technologies, and reduced profitability margins have facilitated changes in livestock management over the last 15 years (Chapter 4). Moreover, the past and continued conversion and degradation of natural rangeland in the Pampas and northern Argentina has resulted in negative effects for wildlife and should be expected to increase given the increasing developmental pressures in these systems. The intensification or mismanagement of grazing in Argentina reduces both the long-term profitability and ecological integrity of these systems (Bucher and Huszar, 1999; Ferreyra, 2001, 2006; Ghersa et al., 2002) and subsequently, the integration of ecologically-based grazing management would be expected to not only have beneficial wildlife benefits but also maintain the long-term productivity of rangelands (Fuhlendorf and Engel, 2001) and should be an emphasis of future research on livestock production and wildlife conservation.

Considering that little land in Argentina is under public ownership, particularly in the Pampas, integrating wildlife conservation with economically viable food production systems are of primary concern. The increasing dominance of row crop agriculture, and its associated levels of high external inputs, combined with the relatively low priority and support given to wildlife conservation at the national and provincial levels makes the feasibility of such an integration difficult. In this situation, specific programs directed towards wildlife conservation are not possible, and subsequently the most plausible and effective approach will be the development and promotion of economically viable production systems that are also beneficial to wildlife.

An important consideration for wildlife conservation in Argentina is the system of concentrated land tenure, and the socio-economic forces associated with it, that have produced a land managerial system that promotes poor land management due to a lack of oversight. Land is often not directly exploited by the owner, who is often absentee, but rather rented by a second party, who in the case of grazing then stock the land. In the case of row crops these second parties also include *pooles de siembra* (sowing pools), which are speculative investment funds that rent land to third parties. At the third or fourth level are the *contratistas agrícolas* (contracted laborer) that handle the farm operations from planting to harvest. Subsequently, in Argentina the land owner is often at least three levels removed from the actual management of the farm and, dependent upon the length of their commitment, the actual producers may have little or no interest in the long-term well being of the land. This has important implications for all aspects of conservation, including wildlife, because there is little or no involvement of a party with a commitment to long-term productivity or integrity of the land.

In relative terms the cost of agricultural inputs in Argentina is at an all-time low, which combined with reduced export tariffs and foreign demand, have been the driving forces behind the increasing dominance and expansion of row crops in Argentina. However, given the present and projected increases in petroleum prices (Campbell and Laherrére, 1998) a shift towards more traditional mixed crop systems may present economically viable and more sustainable alternatives to the present industrialized row crop systems (Ferreyra, 2001, 2006; Ghersa et al., 2002; Viglizzo et al., 2001). Moreover, promotion of traditional extensive livestock management, that emphasizes long-term gains based upon ecologically-based management of rangelands, perhaps holds some of the greatest potential for slowing the expansion of row crop agriculture and the intensification of livestock management, while conserving wildlife in Argentine agroecosystems.

The development of programs that integrate food production, ecological integrity, and wildlife conservation will require both research, education, and promotion of relevant systems that greatly reduce the discounting of the long-term production. Such an integration is a key to improving both agricultural production and conservation at the global level, as well as in Argentina (Banks, 2004; Robertson and Swinton, 2005). Within Argentina the development of such programs will be highly dependent upon both national and provincial support that given the present economic and political climate and trends in Argentina is not likely to be forthcoming.

The development of adequate wildlife conservation programs is hindered by multiple factors, many of which are the same as outlined by Mares (1986) for South America over 20 years ago. These include the availability of adequate funding, sufficient reliable data, and governmental infrastructures to support research and successfully apply management plans, including law enforcement. Although Argentina has an extensive network of universities and several dedicate wildlife related non-governmental organizations, the overall scope of environmental problems, combined with insufficient funding, governmental impotence, and decentralized rural populations, are often overwhelming to the implementation of wildlife conservation actions. Despite this, however, there are some approaches that may be useful and others that have shown some success.

Particular issues related to the management of wildlife in Argentina is the often opportunistic but selective nature of wildlife use, a decentralized rural population, and little regulatory enforcement. Because of this, exploitation of wildlife by rural inhabitants is often random with little accountability to others in a community or legally. The unregulated harvest of species for food, sport, or for commercial purpose is common throughout Argentina with little concern for the legality of it or to ethical considerations (J.J. Thompson pers. obs.; Altrichter 2005a, 2006). Since much of the enforcement of harvest is done via vehicle searches at road blocks, harvest is generally unregulated, and the movement of harvested animals or products from those animals can easily go unchecked.

Some of the most successful wildlife management in Argentina has centered on commercially marketed species in the north of the country which have centered on the development of controls on harvest and marketing at distribution points and incentives for sustainable practices by maximizing profits to the harvesters (Banchs and Moschione, 2006; Porini, 2006). The management of the harvest and trade of blue-fronted Amazon parrots has met with sufficient success that it is now serving as a model for other psittacines in Argentina (Moschione and Banchs, 2006). Such an approach has been successful for commercially harvested species in Paraguay as well (Mieres and Fitzgerald, 2006) and is a planned approach for other commercially exploited species in Argentina (Bolkovic and Ramadori, 2006).

The keys behind this approach are an increased centralization of trade in extracted resources, which shortens the chain of dealers and allows for a better quantification of trade and determination of the origin of products. The advantages gained are several and have additional indirect benefits. By centralizing trade, the sale, use, and/or export of wildlife products can be tracked and regulated, which allows for better controls of exploitation indirectly by controlling trade. Moreover, centralization of trade and tighter controls has shortened the chain of dealers involved, allowing harvesters to receive a greater proportion of realized profits (6-7 times greater) from the trade of wildlife products that they harvest (Banchs and Moschione, 2006).

By receiving greater profits, and developing a stable system of trade, this process has made the sustainable harvest of wildlife more feasible since the harvesters now have greater interest in the long-term viability of the resource. In effect the degree that resources are discounted in the long-term is greatly reduced, which then serves as a motivation for management of the resource. Furthermore, in the case of the extraction of Blue-fronted Amazon chicks the increased profitability gained has allowed for the purchase of land by rural inhabitants, or averted it abandonment, and serves as an affront to deforestation from agriculture and the overexploitation of new areas (Banchs and Moschione, 2006).

Although this approach is effective, its utility is limited in that it useful only for high value commercialized species. For species in northern Argentina and the Pampas that are subjected to harvest for subsistence and/or sport the ability to quantify, regulate, or manage harvest is limited under the present system. For example, species such as the greater rhea and lesser rhea (*Rhea pennata*) are commonly harvested illegally by rural people, in this instance primarily for meat, and is typically consumed locally and goes undetected (J.J. Thompson pers. obs.; Martella and Navarro 2006). Such a scenario is typical of most wildlife harvest in Argentina regardless if for subsistence, commercialiation, or sport and stems from a decentralized rural populations and lack of adequate enforcement, combined with poverty, cultural traditions, and the opportunistic nature of much of the wildlife harvest (J.J. Thompson pers. obs.; Altrichter 2005a, 2006).

An important aspect of the success of management of the harvest of Blue-fronted Amazons has been the redirection of a portion of funds from sales into research on the resource and the incorporation of data from that research into the management of the species (Banchs and Moschione, 2006). Besides collecting required data and improving management decisions, the funding of research from sales gives harvesters a vested interest in the resource and includes them in the decision-making process. This facet of the program, besides being effective, serves as a model for future programs that will not only improve management but also will aid in amassing data on the little studied wildlife of Argentina.

An aspect of wildlife management in Argentina that holds potential for improvement is related to sport harvest, including the use of license fees for research and better enforcement, the development of scientifically-based management, and improved managerial infrastructure within and among provinces. Since Argentina is an important destination for foreign hunters this could potentially could include additional fees for such individuals that are solely for research purposes. Additionally, better organization and accountability of hunting outfitters should help in maintaining mostly reputable operators within the industry that have a vested interest in the long-term well-being of the resources base. Moreover, developing programs for research and management of sport harvested species should develop increased confidence in such actions, and hopefully lead to better cooperation of hunters and outfitters in regards to data collection and management actions.

The development of an effective infrastructure is imperative to not only conduct research and make management-decisions, but to implement management and enforce managerial regulations. Furthermore, such infrastructures need to be coordinated with neighboring provinces since the distributions of harvested species often include multiple provinces. The issues related to migratory species, or those species of conservation concern whose distributions cross international boundaries (*e.g.*, jaguar, tapir) illustrate an additional need for to internationalize wildlife management through trans-boundary approaches.

Although wildlife in the Pampas and northern Argentina have been under pressures from habitat conversion, land degradation, and overexploitation for centuries, from the latter part of the 20^{th} century until the present these threats have escalated. The extent of these threats are increasingly being recognized, however, responses to address these issues have been comparatively few as funding and a sufficient provincial and federal infrastructure are lacking. Despite this there have been some successes in the conservation of wildlife in Argentina, and given increasingly public awareness both nationally and internationally, there exists the potential to make improvements to, and increase the effectiveness of, wildlife conservation and management in the Pampas and northern Argentina, as well as, the country as a whole.

6.3.1 Tinamous and agricultural changes in the Pampas

Collectively, the tinamous constitute the most important gamebirds in Argentina, both for sport and subsistence hunting. Of the 16 species of tinamou found in Argentina 6 are most commonly harvested, all of which are associated with grassland, steppes, and savanna ecosystems; Andean tinamou (*Nothoprocta pentlandii*), brushland tinamou (*Nothoprocta cinerascens*), Darwin's tinamou (*Nothura darwinii*), elegant-crested tinamou (*Eudromia elegans*), red-winged tinamou (*Rhynchotus rufescens*), and spotted tinamou (*Nothura maculosa*). Of these, the spotted and red-winged tinamous in the Pampas, due to their distribution in relation the majority of human population in Argentina, are the most exploited.

Historically, Darwin's, elegant-crested, spotted, and red-winged tinamous were considered common species within their distributions in the Pampas (Bohl, 1970; Bump and Bump, 1969; Darwin, 1839; Hudson, 1892). In the austral Pampas, in the south of the province of Buenos Aires during 1833, Darwin (1839) noted that "*The plains abound with three kinds of partridge* [tinamous], *two of which are as large as hen pheasants*"; the two larger species referred to being the elegant-crested and red-winged tinamou, and the third the Darwin's tinamou.

Even though the introduction of extensive agriculture was particularly beneficial to spotted tinamous (Bump and Bump, 1969), as early as the late 19^{th} century Hudson (1892) noted the negative impact that excessive grazing had on the distribution of red-winged tinamous in settled areas of the Pampas, and Bump and Bump (1969) and Bohl (1970) noted that such practices were also associated with lower abundances of Darwin's, spotted, and elegant-crested tinamous. Although often heavily exploited for their meat and for sport, spotted and red-wing tinamou remained abundant throughout most of their distribution in the Pampas up until the latter part of the 20^{th} century (Bump and Bump, 1969).

Grazing has important influences on the abundance and distribution of spotted and red-winged tinamous, with the smaller spotted tinamou tolerating moderate grazing and preferring shorter and sparser vegetative structure in comparison to the red-winged tinamou (J.J. Thompson, pers. obs.; Bump and Bump 1969; Comparatore et al. 1996; Isacch and Martinez 2001; Leveau and Leveau 2004). Subsequently, red-winged tinamous are often confined to areas of extensive grazing or agricultural areas with abundant refugia, such as wetlands (J.J. Thompson, pers. obs.). The preference for habitats with shorter vegetation explains the historically high abundance of spotted tinamous (up to $\sim$5 birds/ha) in mixed agricultural systems in the Pampas (Bump and Bump, 1969).

Interestingly, there also appears to be some segregation in habitat use between the spotted and Darwin's tinamou where the two species' distributions overlap, despite having similar habitat preferences (J.J. Thompson, pers. obs.; Bump and Bump 1969; Vickery et al. 2003). Although Darwins' tinamous often utilize crop fields (J.J. Thompson, pers. obs.; Bump and Bump 1969; Mosa 2004), where they co-occur with spotted tinamous they tend to remain mostly confined to grassland regions, whereas spotted tinamous utilize both crop and grassland habitats (J.J. Thompson, pers. obs.; Bump and Bump 1969; Vickery et al. 2003).

The changes in agricultural management that have occurred in Argentina since the 1990's (Chapter 4) have coincided with notable declines in the abundance of spotted, Darwin's, and red-winged tinamous in the Pampas. Based upon the effects that agricultural intensification has had on the distribution and abundance of the Galliformes in temperate agroecosystems (Chapter 5), and the result of my research on the survival and habitat use of spotted tinamous in agricultural systems in the Pampas (Chapter 6), it is likely that the underlying factors leading to declines in Galliformes attributable to changes in land management affect the Tinamiformes in a similar manner (Thompson, 2004).

Aside from the Pampas, a similar relationship among land use intensity and abundance of spotted and red-winged tinamou has been observed in Brazil (Menegheti, 1985; Pinheiro and López, 1999), as well as in Uruguay based on accounts of residents. The role that small-scale, low intensity, agriculture plays in maintaining tinamou populations is evident in the Lerma Valley in the province of Salta in northwestern Argentina where farm sizes are mostly <50 ha (S. Mosa, unpubl. data) and among-farm diversity is very high with forage crops and fallows common (J.J. Thompson, pers. obs.). In this area abundances of Darwin's tinamou and Andean tinamou are high, and as in the case of the spotted and red-winged tinamou in the Pampas, the larger con-specific Andean tinamou, shows a preference for habitats with taller vegetative structure compared to the smaller Darwin's tinamou (Mosa, 2004).

Within extensive grazing areas in the Pampas, mostly in the eastern flooding Pampas, and portions of the austral and interior Pampas, healthy populations of spotted, Darwin's, and red-winged tinamous still exist, but these areas are increasingly more restricted as row crop agriculture increasingly expands and intensifies (J.J. Thompson, pers. obs.). In the more intensively managed areas of the Pampas, particularly the rolling Pampas, tinamous have become conspicuous in their absence as both row crop and grazing management have intensified.

My research (see Appendix) suggests that the loss of refugia within agricultural areas in the Pampas resulting from intensification in land management may be factor in decreasing tinamou populations in these systems through increased predation. This effect would be consistent with observations on the Galliformes (Chapter 5) and is plausible given the size and diversity of the avian and mammalian predator community in Pampas (J.J. Thompson pers. obs; de la Peña and Rumboll 1999; Parera 2002).

The over-winter survival I documented in the spotted tinamou related to the intensity of land use is feasibly a factor in decreasing abundances, however, it is not excessive compared to Galliformes (Chapter 5). This suggests that, as with the Galliformes, a reduction in recruitment may be the greatest cause for observed decreases in abundance. This is especially plausible in light of the large increases in pesticide use, the loss of nesting habitat in the Pampas, and the role that these factors have had in decreasing abundances of Galliformes throughout the world (Chapter 5).

As discussed above, the lack of infrastructure and the relative low priority given wildlife conservation preclude the adoption management options that could possibly be incorporated into agroecosystem management in the Pampas to increase habitat availability and quality for tinamous and other wildlife dependent upon these systems. Moreover, despite a passionate wing-shooting and field trial community, their positive influence on conservation is minimal, especially in the context of larger economic and political forces within the country. Combined with a relative dearth of research into the effects of agricultural management on wildlife, and specifically on tinamous, these factors limit the potential development and implementation of management actions for tinamous in the Pampas and Argentina as a whole.

6.3.2 Potential management actions

Although at the present there are limitations to the application of habitat and/or harvest management in agroecosytems for Argentine wildlife in general, and tinamous in particular, based upon observed effects of agricultural management on birds in other regions, and successful mitigation of those effects (Chapter 5), there are potentially multiple approaches that can be used to offset some of the negative effects of agroecosytem management on wildlife in Argentina. A key component in implementing such actions is the development and integration of economically viable agricultural practices that are also beneficial to wildlife in agroecosytems by employing grazing and farming practices that maximize the benefits from the ecological services provided by agroecosystems (Banks, 2004; Fuhlendorf and Engel, 2001; Robertson and Swinton, 2005).

Based upon the review and synthesis of research on avian ecology in agroecosystems (Chapter 5), the diversity and quality of habitats across scales are the most important factors influencing avian populations in agricultural systems and are principally driven by decisions related to land management and allocation (Fig. 5.27). Although these relationships act in combination and in a species-specific manner, they can be utilized to conceptualize approaches towards land use that meet both economic and conservation needs.

Increasing habitat heterogeneity across spatial scales within areas devoted to extensive grazing is critical for the conservation of biodiversity in these areas, as is the promotion of mixed cropping systems in regions dominated by row crop agriculture. To best accomplish this requires innovative inter-disciplinary research that incorporates an ecological perspective to maximizing long-term agricultural production that is secondarily beneficial to wildlife. Key to the acceptance of such practices is a recognition of the long-term costs and benefits accrued under different management scenarios (*i.e.* reducing discount rates).

In the case of extensive grazing, an illustration of the long-term profitability and integrity of rangelands is imperative. By maintaining sustainable stocking rates where grazing pressure is managed from an ecological perspective, including fire, will not only present a viable alternative to conversion to row crops in areas where this is possible, but will also maintain the conservation value of rangelands by mimicking natural ecosystem function (Fuhlendorf and Engel, 2001). A potentially useful tool in developing and promoting economically and ecologically sound grazing management over conversion to row crops is to exploit the significant role grazing has played, and continues to play, in Argentine history and culture.

Promoting sustainable and economically viable grazing on natural pasture will serve as a disincentive to the introduction of perennial forage crops or to afforestation, which will furthermore maximize the conservation value of rangelands. For the Argentine Chaco such an approach has been shown to be economically viable but difficult to implement due to high discounting of the future (Bucher and Huszar, 1999). The usefulness of diversification in serving as a deterrent to land conversion is illustrated by the success of increasing the value of Blue-fronted Amazon parrots to harvesters, allowing for the purchase of land by rural people rather than agrobusiness concerns (see above).

In areas dominated by row crops the promotion of mixed cropping systems can have multiple long-term economic benefits, largely through reducing the cost of external inputs and diversifying the economic base. This would also increase spatial and temporal heterogeneity, producing suitable habitats and refugia within these systems. However, given the present profitability of row crop production in Argentina, particularly soybean, and the shift to finishing cattle with supplemental feeding, a return to former mixed-cropping systems may not be feasible. Despite this there are several areas to be investigated to improve the wildlife conservation value of row crop systems and the economic productivity of those systems.

As discussed in Chapter 4, the principal changes in row crop agriculture include the widespread adoption of genetically modified crops, no-till sowing, and high levels of pesticide application. Within in this cropping system there are options that potentially can increase long-term profitability and that should produce significant conservation benefits. Of these, the recognition that the genetic diversity of arthropod and plant pests needs to be maintained to maximize long-term crop productivity of the dominant row cropping systems, via maintaining areas out of production, will have significant benefits by creating biodiversity refugia. Furthermore, such areas can be integrated with buffer strips designed for soil conservation and contamination management.

The widespread use of no-till sowing, although associated with high levels of pesticide use, presents several opportunities for the development of crop management that can reduce pesticide use, reduce expenses, maximize profits, and greatly increase the conservation value of those lands. An obvious, although likely unpopular option, is to tax pesticide sales towards the goal of promoting more responsible application with revenues going towards research into natural resource issues associated with pesticide use.

As illustrated in Chapter 5 a reduction in the application of pesticides has significant potential in increasing habitat quality. Within that context, management systems that reduce pesticide use by relying more on ecosystem services within the present no-till systems should be examined more closely. The undercropping system studied by Cederbaum et al. (2004) is an excellent example of a system that is both economically viable and increases the conservation value of crop fields. In combination, for row crop regions, the promotion of mixed management systems with the incorporation of fallow areas and buffers, and reduced pesticide use, could collectively make significant improvements to the overall quality and quantity of wildlife habitat in these systems.

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Appendix

Habitat Use and Survival of the Spotted Tinamou (*Nothura maculosa*) in Agroecosystems in the Province of Buenos Aires, Argentina

.1 Introduction

Globally, populations of grassland and shrubland birds have been declining due to habitat conversion and agricultural intensification (Askins, 2000; Goriup, 1988; Murphy, 2003; Pain and Pienkowski, 1997; Vickery and Herkert, 1995). In agroecosystems of austral South America habitat loss and the intensification in management have been extensive and rapid, particularly in the pampas of Argentina starting in the early 1990's, typified by the increased use of external inputs, increased yields, and a shift towards agricultural production for export markets (Ferreyra, 2001; Ghersa et al., 2002; Hall et al., 2001; Solbrig and Vera, 2001; Viglizzo et al., 2001).

The spotted tinamou (*Nothura maculosa*) is a common bird of grasslands and agroecosystems in eastern austral South America, one of the most important terrestrial gamebirds in the region, and formerly common in agricultural systems (Bucher and Nores, 1988; Bump and Bump, 1969; Cabot, 1992; Davies, 2002; Menegheti, 1985). In recent years, within the pampas of Argentina, the spotted tinamou has become increasingly conspicuous by their absence apparently stemming from the expansion and intensification of grazing and row crop practices.

All tinamous are relatively poorly studied; however, in austral South American grasslands the tinamous replace the Galliformes and are their ecological equivalent, which allows for inferences to be drawn among the Galliformes and the Tinamiformes in regard to tinamou ecology (Thompson, 2004). I used radio telemetry to investigate my hypothesis of ecological equivalence. Based upon the observed effects of agricultural intensification on Galliformes and existing knowledge of the spotted tinamou that within pampean ecosystems I predicted that survival of spotted tinamous would be negatively correlated, and home range size positively correlated, with increasing land use intensity while habitat selection would favor areas most similar to natural grassland in vegetative structure. (Bump and Bump, 1969; Thompson, 2004).

.2 Study area

My study sites were located in the district of San Miguel del Monte in the province of Buenos Aires, Argentina (Fig.1). San Miguel del Monte is located in the flooding pampa, a regional subdivision of the ~760,000 km^2 Río de la Plata grassland system that covers northeastern Argentina, Uruguay and southeastern Brazil (Soriano et al., 1991). Traditionally the flooding pampa has been used principally for extensive livestock production (Hall et al., 1988), however, since the early 1990's row crop agriculture has become an increasingly important land use.

I selected two study sites; one dedicated to row crops and the other used for used for a mix of row crops and grazing. The row crop site was 160 hectares, of which 85% was used for soybean, corn, and winter wheat production, and the remaining area comprised of wetlands or field borders. The site with mixed row crop and grazing uses was 230 hectares, 50% of its area used for soybean, corn, and winter wheat production, and the remainder, including wetlands, used for cattle grazing.

.3 Methods

During July 2003, I fitted 4 birds with pendant-style transmitters (6.0 g, 2.2-2.3% of body mass) equipped with an activity switch (Holohil Systems Ltd., Ontario, Canada) at the row crop dominated site and 14 birds in June 2004 at the mixed use site. In 2004, no birds were radio-tagged at the row crop site because none were detected over a 2 month search in the autumn of that year. All birds were captured at night using spotlights and hand nets. Due to uncertainties in sexing birds related to age (Bump and Bump, 1969), sexual differences were not included in the study. In both years birds were located 3 times per week from the date of capture until October 23 (mid-winter to early spring) dependent upon accessibility to the sites.

Due to mortality, insufficient number of radio locations, radio failure, or radio loss I used 3 birds from the row crop site and 8 from the mixed use site for analysis. Locations were entered into a geographic information system (GIS) for each site using ArcGIS software (Environmental Systems Research Institute, Inc.). Minimum convex polygons (MCP) (Mohr, 1947) were calculated for each individual using the adehabitat package version 1.4 (Calenge et al., 2006) in R 2.3.1 (R Development Core Team, 2006) and the proportion of radio locations and MCP in different habitat types determined using the GIS.

Within the row crop site I defined 6 habitat types; winter wheat, fallow, wetlands, corn stubble, tilled land, and field edges. For the mixed use site I identified 5 habitat types; winter wheat, fallow, wetlands, mowed fallow, and grazed pasture. I used compositional analysis (Aebischer et al., 1993) at the 2^{nd} order, based upon radio locations and MCP, to evaluate habitat preferences within the study sites and at the 3^{rd} order for habitat selection within home ranges (Johnson, 1980).

The compositional analysis was performed using BYCOMP.SAS (Ott and Hoovey, 1997) and, to obtain sufficient sample size and facilitate comparison between sites, I combined data from both sites and aggregated habitat types into 5 categories; winter wheat, fallow, wetlands, edge, and short herbaceous (corn stubble, tilled land, mowed fallow, and grazed pasture). Additionally, survival was estimated using Kaplan-Meier staggered entry design (Kaplan and Meier, 1958; Pollock et al., 1989). The overlap of standard errors were used to determine statistically significance differences in survival and mean home range size.

.4 Results

The mean 100% MCP from the row crop site was larger (19.0 ha, SE = 10.4 ha) than that from the mixed use site (15.9 ha, SE = 7.3 ha), although differences were not significant due to high variance. Survival ($\hat{s}$ = 0.73, SE = 0.19) was higher in the mixed use site over 20 weeks compared to the row crop site ($\hat{s}$ = 0.33, SE = 0.19) over 15 weeks (Fig. 2). Mortality of the radio-tagged birds from both sites was attributed mainly to predation (91%).

At the row crop site winter wheat, wetlands, and field edges were used less, and corn stubble more, than their availability based upon both the mean proportions of MCP and radio locations within those habitat types (Fig. 3). In tilled land the mean proportion of MCP and radio locations indicate approximately equal use in relation to availability, while in fallow, based on the mean MCP use was equal to availability, but considerably higher than its availability based upon the mean proportion of radio locations (Fig. 3). As at the row crop site about half of the area of the mixed use site was in winter wheat, which was utilized less than its availability (Fig. 3). Fallow, mowed fallow, and wetlands were all used more than their availability, while based upon the mean proportion of MCP, grazed pasture was used equal to its availability, and less than its availability based upon the mean proportion of radio locations (Fig. 3).

The 2^{nd} order compositional analysis using the aggregated data from both sites, and based upon MCP, ranked short herbaceous habitat as the most utilized habitat in relation to availability, with fallow, winter wheat, and edge ranked equally as second, followed by wetlands (Table 1A). No habitats were used significantly more than others (p=0.05) but fallow and short herbaceous habitat were preferred over wheat, fallow over wetlands, short herbaceous over fallow, and wetlands over short herbaceous (Table 1A).

The same analysis using radio locations ranked fallow and short herbaceous habitats as the first and second most utilized habitats, respectively, in relation to availability, followed by wetlands (Table 1B). Winter wheat and edge were the least used in relation to availability (Table 1B). Fallow was utilized significantly more than wheat (p=0.05) while fallow, short herbaceous, and wetlands were preferred over wheat, fallow and short herbaceous were preferred over wetlands, and fallow over short herbaceous (Table 1B).

At the 3^{rd} order, the compositional analysis ranked fallow as most preferred, while short herbaceous habitats, wetlands, and wheat were all used equally, and edge habitats were avoided (Table 2). Within home ranges fallow was utilized significantly more than short herbaceous habitats (p=0.05) and was preferred over wheat (Table 2). Wheat was preferred over short herbaceous habitats, which were preferred over wetlands (Table 2).

.5 Discussion

The mean home range size of spotted tinamous at both sites was affected by movements related to changing habitat amounts and characteristics and cattle disturbance. At the row crop site, as winter wheat reached ~10cm in height birds began to utilize those areas, often exclusively, and as the wheat matured to ~25cm in height those areas were abandoned for areas with shorter vegetation. Within the mixed use site the largest movements by birds were related to disturbance by cattle.

The lower surival in the row crop dominated site is consistent with observations of Pinheiro and López (1999) who found lower abundances of spotted tinamous in agricultural land in southern Brazil compared to natural grasslands. Additionally, for the Galliformes there are multiple cases where increased intensification in land use has led to lower survival and declining populations (*e.g.* Berner, 1988; Hill and Robertson, 1988; Jansen et al., 2000; Malan and Benn, 1999; Potts, 1986). Based upon this, the observed differences in survival between the two sites are expected if the spotted tinamou is viewed as an ecological equivalent to the Galliformes. Admittedly, sample sizes are small, particularly for the row crop site; however, the rarity of spotted tinamous at the row crop site in 2003 and their absence from the site in 2004 suggest a real process rather than a statistical artefact.

Table 1: Table 1. Results of 2^{nd} order compositional analysis of habitat preferences within the study sites based on A) minimum convex polygon (MCP) home ranges and B) proportion of radio locations. Higher ranking indicates greater use compared to availability. Within the matrix, (+) signifies that the row habitat is preferred over the column habitat, whereas a (−) signifies the opposite. Significant difference between habitats (p <0.05) is indicated by (+++) or (− − −).

A

Habitat	*wheat*	*wetlands*	*edge*	*fallow*	*short herbaceous*	*rank*
wheat	.	.	.	−	−	2
wetlands	.	.	.	−	+	1
edge	.	.	.	.	.	2
fallow	+	+	.	.	−	2
short herbaceous	+	-	.	+	-	3

B

Habitat	*wheat*	*wetlands*	*edge*	*fallow*	*short herbaceous*	*rank*
wheat	.	−	.	− − −	−	0
wetlands	+	.	.	−	−	1
edge	.	.	.	.	.	0
fallow	+++	+	.	.	+	3
short herbaceous	+	+	.	−	.	2

Note: Because of low or no use a P value for edge habitat could not be computed.

Table 2: Table 2. Results of 3^{rd} order compositional analysis of habitat preferences within home ranges. Higher ranking indicates greater use compared to availability. Within the matrix, (+) signifies that the row habitat is preferred over the column habitat, whereas a (-) signifies the opposite. Significant difference between habitats (p <0.05) is indicated by (+++) or (− − −).

A

Habitat	*wheat*	*wetlands*	*edge*	*fallow*	*short herbaceous*	*rank*
wheat	.	.	.	−	+	1
wetland	.	.	.	.	−	1
edge	.	.	.	.	.	0
fallow	+	.	.	.	+++	2
short herbaceous	−	+	.	− − −	.	1

Note: Because of no use a P value for edge habitat could not be computed.

Habitat preferences by the spotted tinamou, and the closely related Darwin's Tinamou (*Nothura darwinii*), within both natural and agricultural habitats, favor areas with relatively low (10-30 cm) and sparse vegetation (pers. obs., Bump and Bump, 1969; Isacch and Martinez, 2001; Leveau and Leveau, 2004; Mosa, 2004) and explains the pattern of habitat use at both sites. For example, use of winter wheat was most frequent when plants were 10-25 cm tall. Although wheat was generally avoided once it reached >25cm in height, birds then used it as escape cover.

The most preferred habitats; fallow, mowed fallow, and corn stubble all shared in common a well developed ground cover of herbaceous vegetation, both living and dead, that was not in excess of 50 cm and with little or no emergent vegetation above that level. Tilled land was used more as it was colonized by herbaceous vegetation, particularly clover (*Trifolium* spp.), and vegetative cover increased.

Spotted tinamous are often common in pastureland (pers. obs., Bump and Bump, 1969; Menegheti, 1985; Pinheiro and López, 1999), as are Darwin's tinamou (pers. obs., Bump and Bump, 1969; Mosa, 2004), due to the low vegetative structure that is maintained through moderate grazing. At the mixed use site, however, pastureland was overgrazed so that ground vegetation was cropped near to ground level, which explains a lower than expected preference for grazed areas. The preference for relatively short vegetation also explains the avoidance of field edges in the row crop site. Field edges consisted of tall (>1m) and dense grass and also contained woody vegetation, which were avoided by the birds.

The difference in the use of wetlands among the sites appeared to be a function of the water levels within wetlands at each site. At the row crop site wetlands contained water and were avoided, where as at the mixed use site, wetlands were dry and contained suitable herbaceous cover along their perimeter that was utilized by the birds. It should be noted that although wetlands were not used by individuals at the row crop site, much of the fallow areas were not put into production due to their proximity to wetlands, subsequently wetlands were indirectly responsible for the availability of preferred habitats.

The preferences and differences in habitat use within and between sites are consistent with the results of the 2^{nd} order compositional analysis since fallow areas and the habitats comprising the short herbaceous category, while more variable, are the habitats most similar in structure to natural grasslands. Similarly, the quality of wetlands varies annually dependent upon precipitation, reducing interannual use, while row crop fields and edge were avoided or used considerably less in relation to their availability.

Similarly the within home range preference for fallow shown by the 3^{rd} order compositional analysis is consistent with habitat preferences at the site level. Although within home ranges wetlands, wheat, and short herbaceous habitats were rank equally as the second most preferred habitats, wheat was preferred over short herbaceous, and short herbaceous over wetlands. The selection of wheat over short herbaceous habitats likely stems from the use of young wheat fields by some individuals, while the preference for short herbaceous habitats over wetlands may be due to structural aspects of vegetation, as well as the inundation of some wetlands.

The preferences in habitat, size of home ranges, and survival that I observed were consistent with expectations based upon existing knowledge of tinamou ecology, and the response of Galliformes and other bird species to the intensification of agricultural land use (Thompson, 2004). From this study, and others (Canavelli et al., 2003; Bellis et al., 2004; Demaría et al., 2002; Fernandez et al., 2003), it is apparent that the intensification of agriculture, as with gamebirds in other regions, is negatively affecting the spotted tinamou in Argentine agroecosystems.

The continued expansion and intensification of agriculture in Argentina suggests that pampean agroecosystems will continue to be degraded, with the most ecologically valuable systems being maintained in areas only suitable for extensive livestock production. Moreover, within intensively managed systems, fallow and areas unsuitable for production (i.e. wetlands) will increasingly become critical for biodiversity conservation.

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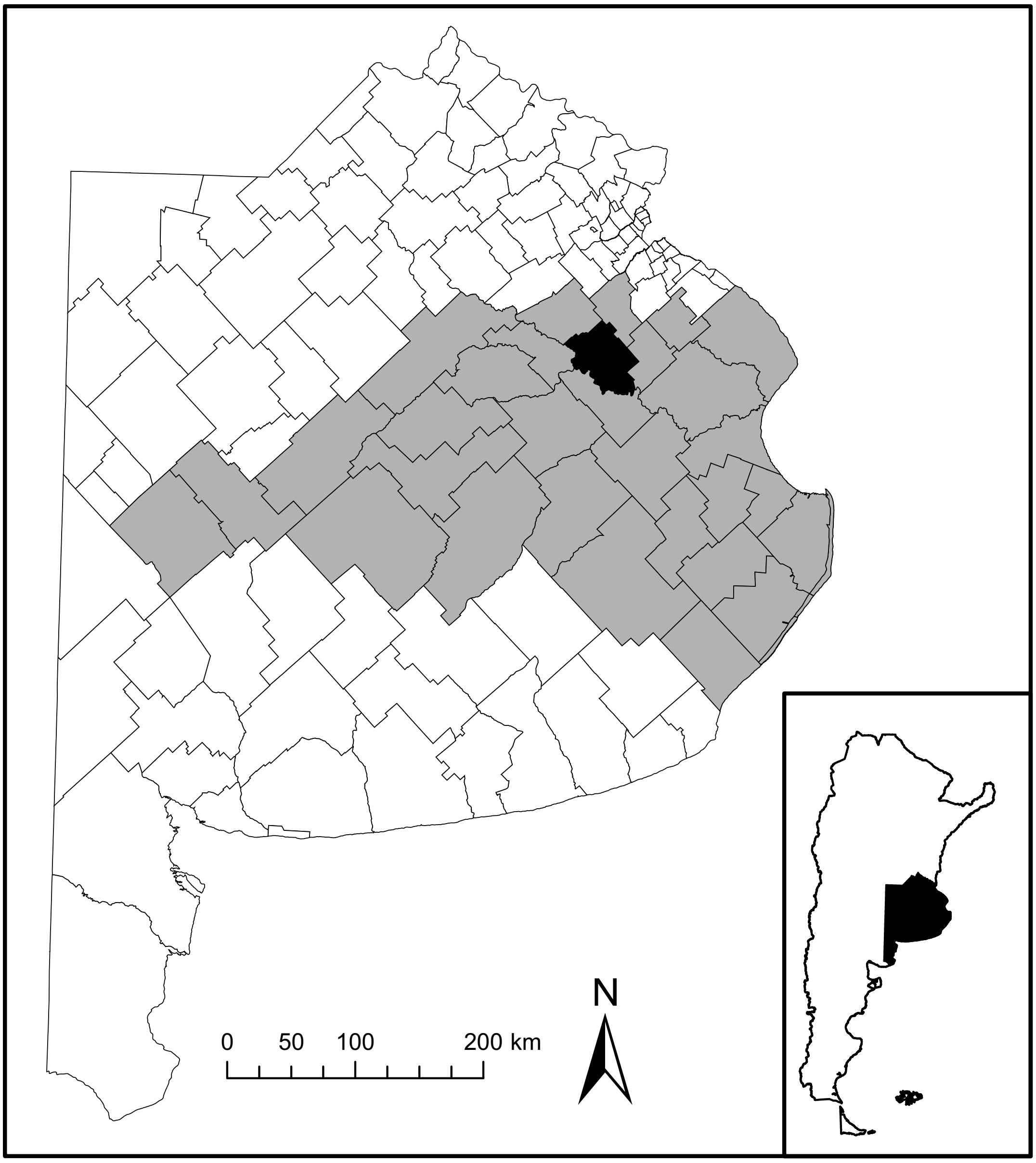

Figure 1: Map showing the location of the district of San Miguel del Monte in black and the distribution of the flooding pampa (by district) in gray within the Province of Buenos Aires, Argentina

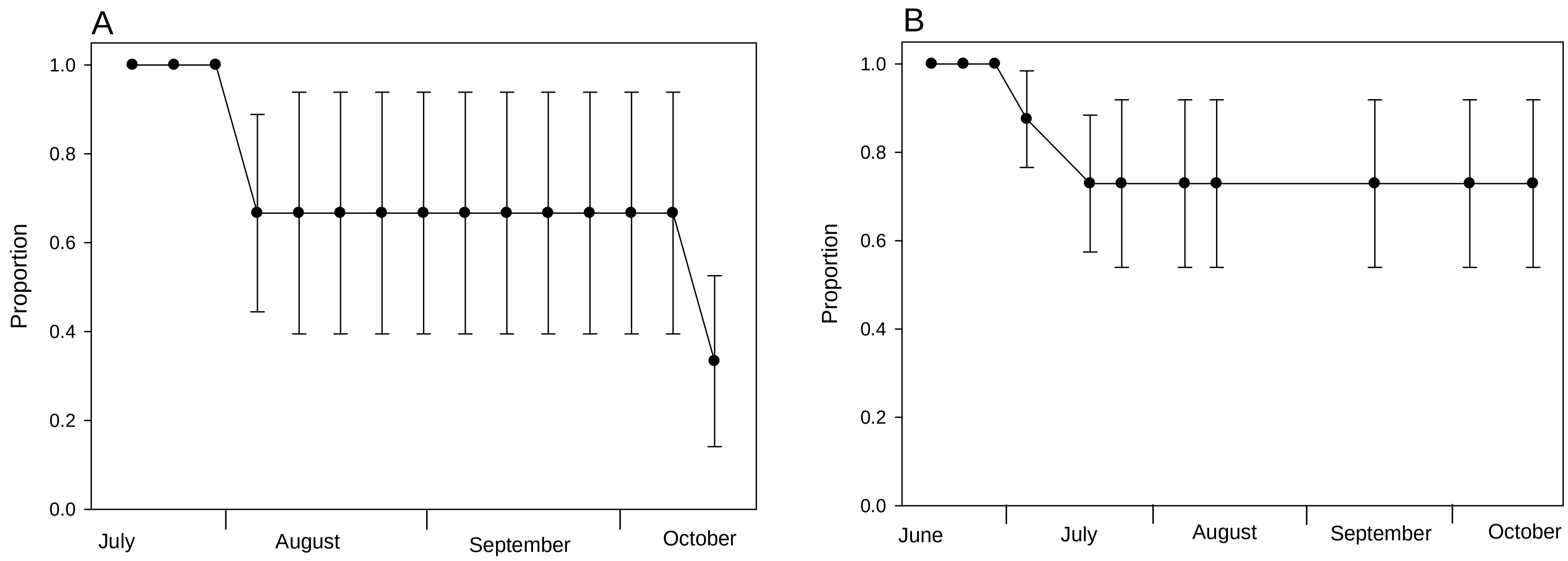

Figure 2: Estimated survival and standard error for spotted tinamous in the A) site dominated by row crops ($n = 3$) and B) the mixed grazing and row crop site ($n = 8$)

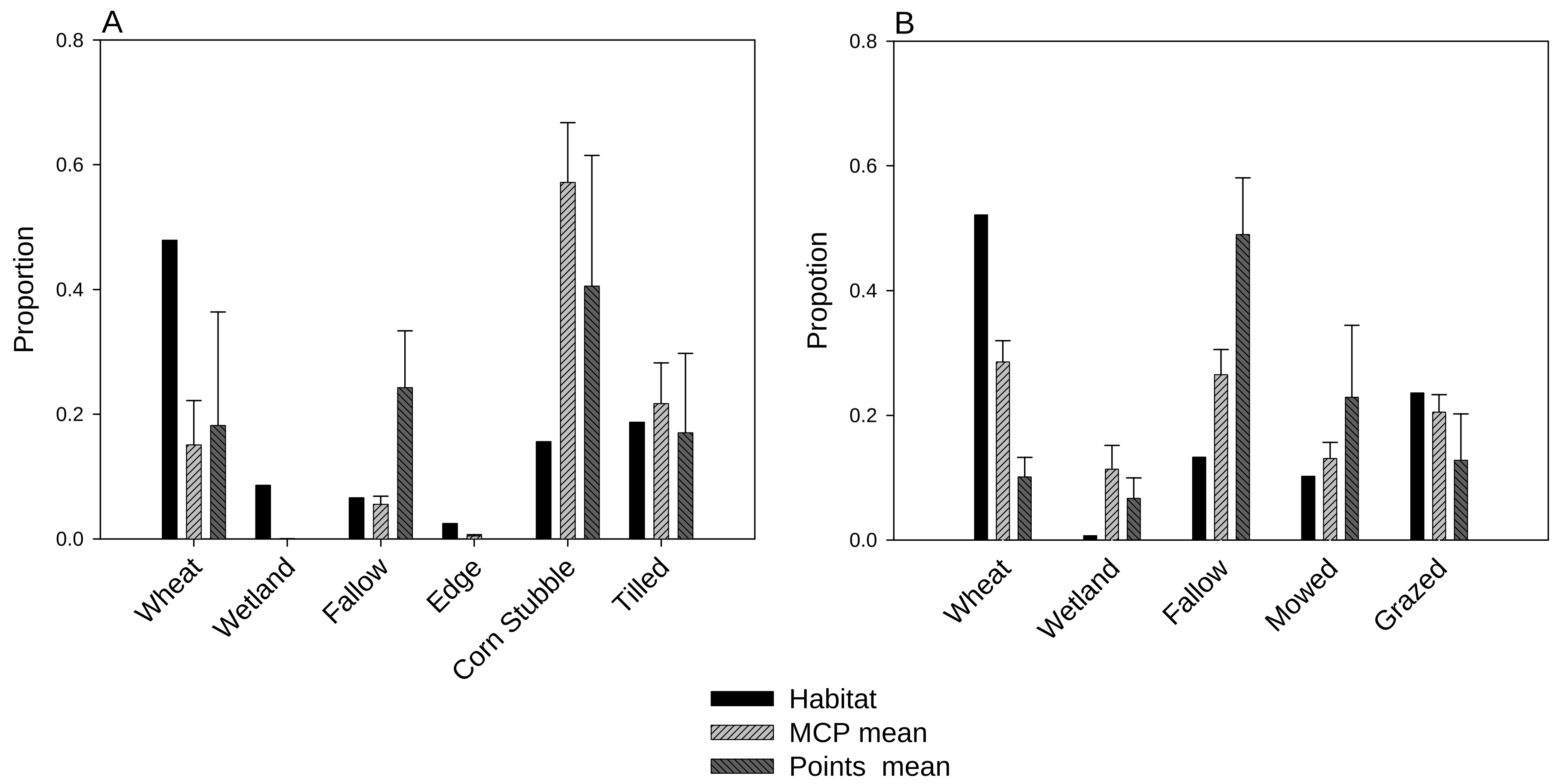

Figure 3: Proportional habitat use by spotted tinamous based on mean area of minimum convex polygon (MCP) and mean number of radio locations (Points) in relation to proportional availability of habitat types for A) the row crop site and B) the mixed use site. Error bars represent standard error.