Universitat de Lleida

In search of balance: Agricultural economic value and bird conservation in the Lleida plain

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TESI DOCTORAL

In search of balance: Agricultural economic value and bird conservation in the Lleida plain

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GERMÁN SÁNCHEZ ARCE
DOCTOR OF PHILOSOPHY IN ECONOMICS

IN SEARCH OF BALANCE: AGRICULTURAL ECONOMIC VALUE AND BIRD CONSERVATION IN THE LLEIDA PLAIN

Motivated by the desire to establish the relationships that exist between the development of the economy and the capacity of the natural environment to satisfy and coexist with this growth, this thesis is composed of three chapters which common threat is to balance agricultural production and biodiversity conservation in a changing world. In these chapters, I try to quantify, through simulations and in a real scenario, optimal land use configurations under various assumptions and environmental constraints. These chapters contain a common base methodology by which we have obtained data on spatially heterogeneous potential productivities for various crops and levels of intensification in the area through a crop growth simulator. We have moreover given an economic value to these productivities using price and cost data. Finally, we have used an algorithm to solve these problems. We have thus obtained different results. In Chapter 2, we evaluate the economic implications of bird community conservation through a multi-objective spatial optimization problem. The maximum economic value of agriculture, in the absence of biodiversity constraints, is estimated at €135.1 M, and €90.7 M when all species are preserved. We further investigate how this curve fluctuates based on whether the existing irrigation limitation zones were optimally chosen or removed. Moreover, we compare the effectiveness of a conservation policy grounded on triage approaches with those that aim to safeguard specific yet significant species. Our findings reveal intriguing outcomes, such as the fact that Steppe specialists are neither the last to be conserved nor the most expensive. Another critical discovery is the existence of a trade-off between conservation and economic development, but this is only applicable to certain species. However, we note that there is substantial potential for improvement if land uses were optimally allocated. In Chapter 3 we examine the acceptance of conservation policies, taking into account the presence of negatively affected farmers, whom we use as a proxy for the risk of opposition. To address this issue, we introduce a range of mechanisms, including transfer systems and both homogeneous and heterogeneous subsidies, and assess their effectiveness. We expand upon these results by presenting additional findings on the assumption of incomplete information on the part of the regulator and alternative social welfare measures. Our results suggest that, when conservation objectives are sufficiently high, optimal solutions results in the existence of losers, potentially complicating their implementation. We demonstrate that agreements between farmers or regulators may offer a viable solution to this issue without disincentivizing economic activity, as is shown to occur with standalone homogeneous payments. Heterogeneous payments are much more efficient and we argue that even with incomplete information could be preferred to a homogeneous payment. Lastly, in Chapter 4 we focus on the impact of climate change and biodiversity on representative crops in the area. To do this, we again use the crop growth simulator, introducing data from climate predictions in the medium (2040-2070) and long term (2070-2100) under two different emission scenarios (RCP 4.5 and 8.5). With this, our objective is to observe, under various assumptions of farmers' reaction to climate change, what the economic consequences of changes in climate variables are. Our findings indicate that high-emission scenarios can end up in the long term with significant losses, especially if farmers do not cope by selecting new land uses.

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Motivado por el deseo de establecer las relaciones que existen entre el desarrollo de la economía y la capacidad del entorno natural para satisfacer y coexistir con este crecimiento, esta tesis se compone de tres capítulos que intentan cuantificar, a través de simulaciones y en un escenario real, configuraciones óptimas de uso del suelo bajo diversos supuestos y restricciones ambientales. Estos capítulos contienen una metodología común en la que hemos obtenido datos sobre productividades potenciales espacialmente heterogéneas para diversos cultivos y niveles de intensificación en el área a través de un simulador de crecimiento de cultivos. Además, hemos dado un valor económico a estas productividades utilizando datos de precios y costes. Finalmente, hemos utilizado un algoritmo para resolver estos problemas. Así, hemos obtenido diferentes resultados. En el Capítulo 2, evaluamos las implicaciones económicas de la conservación de la comunidad de aves a través de un problema de optimización espacial multiobjetivo. El valor económico máximo de la agricultura, en ausencia de restricciones de biodiversidad, se estima en €135.1 M, y €90.7 M cuando se conservan todas las especies. Investigamos más a fondo cómo fluctúa esta curva en función de si las zonas de limitación de riego existentes se eligieron o se eliminaron de forma óptima. Además, comparamos la efectividad de una política de conservación basada en enfoques de triaje con aquellas que tienen como objetivo salvaguardar especies específicas pero significativas. Nuestros resultados revelan resultados interesantes, como el hecho de que las especies esteparias especialistas no son las más costosas de conservar. Otro resultado crítico es la existencia de un compromiso entre la conservación y el desarrollo económico, pero esto solo es aplicable a ciertas especies. Sin embargo, observamos que existe un gran potencial de mejora si los usos del suelo se asignaran de forma óptima. En el Capítulo 3 examinamos la aceptación de las políticas de conservación, teniendo en cuenta la presencia de agricultores afectados negativamente, a quienes utilizamos como proxy del riesgo de oposición. Para abordar este problema, introducimos una serie de mecanismos, incluyendo sistemas de transferencias y subsidios tanto homogéneos como heterogéneos, y evaluamos su eficacia. Ampliamos estos resultados presentando hallazgos adicionales sobre la suposición de información incompleta por parte del regulador y medidas alternativas del bienestar social. Nuestros resultados sugieren que, cuando los objetivos de conservación son suficientemente altos, las soluciones óptimas necesitan la presencia de estos mecanismos, lo que potencialmente complica su implementación. Demostramos que los acuerdos entre agricultores o reguladores pueden ofrecer una solución viable a este problema sin desincentivar la actividad económica, como se muestra que ocurre con los pagos homogéneos. Los pagos heterogéneos son mucho más eficientes y argumentamos que incluso con información incompleta podrían ser preferidos a un pago homogéneo. Por último, en el Capítulo 4 nos centramos en el impacto del cambio climático y la biodiversidad en cultivos representativos del área. Para ello, volvemos a utilizar el simulador de crecimiento de cultivos, introduciendo datos a partir de predicciones climáticas a medio (2040-2070) y largo plazo (2070-2100) bajo dos escenarios diferentes de emisiones (RCP 4.5 y RCP 8.5). Con esto, nuestro objetivo es observar, bajo diversos supuestos sobre la reacción de los agricultores al cambio climático, cuáles son las consecuencias económicas del cambio en las variables climáticas. Nuestros hallazgos indican que los escenarios con altas emisiones pueden terminar a largo plazo con pérdidas significativas, especialmente si los agricultores no hacen frente seleccionando nuevos usos del suelo.

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Motivat pel desig d'establir les relacions existents entre el desenvolupament de l'economia i la capacitat del medi natural per coexistir amb aquest creixement, aquesta tesi es compon de tres capítols que intenten quantificar, a través de simulacions i en un escenari real, configuracions òptimes d'ús del sòl sota diverses suposicions i restriccions ambientals. Aquests capítols contenen una metodologia comuna en la qual hem obtingut dades sobre productivitats potencials espacialment heterogènies per a diversos cultius i nivells d'intensificació en l'àrea a través d'un simulador de creixement de cultius. A més, hem donat un valor econòmic a aquestes productivitats utilitzant dades de preus i costos. Finalment, hem utilitzat un algoritme per obtenir aquestes configuracions óptimes. Així, hem obtingut diferents resultats. En el Capítol 2, avaluem les implicacions econòmiques de la conservació de la comunitat d'aus a través d'un problema d'optimització espacial multiobjectiu. El valor econòmic màxim de l'agricultura, en absència de restriccions de biodiversitat, s'estima en €135.1 M, i en €90.7 M quan es conserven totes les espècies. Investiguem més a fons com fluctua aquesta corba en funció de si les zones de limitació de regadiu existents es van triar de forma òptima o es van eliminar. A més, comparem l'efectivitat d'una política de conservació basada en enfocaments de triatge amb aquelles que tenen com a objectiu salvaguardar espècies específiques però significatives. Els nostres resultats revelen resultats interessants, com el fet que les aus estepàries especialistes no són ni les últimes a conservar-se ni les més cares. Un altre descobriment crític és l'existència d'un trade-off entre la conservació i el valor econòmic agrícola, però això només és aplicable a certes espècies. No obstant això, observem que hi ha un gran potencial de millora si els usos del sòl s'assignessin de forma òptima. En el Capítol 3 examinem l'acceptació de les polítiques de conservació, tenint en compte la presència d'agricultors afectats, als quals utilitzem com a proxy del risc d'oposició. Per abordar aquest problema, introduïm una sèrie de mecanismes, incloent sistemes de transferències i subvencions tant homogènies com heterogènies, i avaluem la seva eficàcia. Ampliem els nostres resultats suposant que existeix informació incompleta per part del regulador i utilitzant diverses mesures alternatives del benestar social. Els resultats suggereixen que, quan els objectius de conservació són prou alts, les solucions òptimes necessiten la presència d'aquests mecanismes de compensació, cosa que potencialment complica la seva implementació. Demostrem que els acords entre agricultors o reguladors poden oferir una solució viable a aquest problema sense desincentivar l'activitat econòmica, com es mostra que ocorre amb els pagaments homogènies independents. Els pagaments heterogènies són molt més eficients i argumentem que fins i tot amb informació incompleta podrien ser preferits a un pagament homogeni. Finalment, en el Capítol 4 ens centrem en l'impacte del canvi climàtic i dela biodiversitat en cultius representatius de l'àrea. Per fer-ho, tornem a utilitzar el simulador de creixement de cultius, introduint dades a partir de prediccions climàtiques a mitjà (2040-2070) i llarg termini (2070-2100) sota dos escenaris diferents d'emissions (RCP 4.5 i RCP 8.5). Amb això, el nostre objectiu és observar, sota diverses suposicions sobre la reacció dels agricultors al canvi climàtic, quines són les conseqüències econòmiques del canvi climàtic. Els nostres resultats indiquen que els escenaris amb altes emissions poden acabar a llarg termini amb pèrdues significatives, especialment si els agricultors no fan front seleccionant nous usos del sòl.

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Glossary

Crop growth simulation model Mathematical representation of the processes that govern the growth and development of crops. These models use scientific principles and mathematical relationships to simulate the effects of environmental and management factors on crop growth, development, and yield. 3, 15, 22, 70, 104, 167, 173

Ecosystem services Are the many and varied benefits to humans provided by the natural environment and healthy ecosystems. 1, 97

Habitat Suitability Index Numerical index that represents the capacity of a given habitat to support a selected species . 4, 16, 33, 38, 70, 72, 84–86, 103

Multi-objective optimization Also known as Pareto optimization, is an area of multiple-criteria decision making that is concerned with mathematical optimization problems involving more than one objective function to be optimized simultaneously. 3, 70, 98, 198

Steppe birds Bird species adapted to extensive farming systems and traditional cultivation of cereals, legumes and other crops, together with fallow land. 4, 10, 32, 34, 39, 64, 65, 68, 70, 103, 107, 109, 115, 116, 122, 124, 126, 127, 142, 169, 184

Trade-off Situational decision that involves diminishing or losing one quality, quantity, or property of a set or design in return for gains in other aspects . 2, 4, 9, 15, 26, 31, 47, 64, 67, 68, 70–72, 97, 101, 108, 154, 156, 191–194, 197, 198

Winter cereals Winter cereals are biennial cereal crops sown in the autumn that germinate before winter and continue their life cycle in spring. They have a different plant structure than spring cereals, as they need to survive low temperatures and snow cover. Examples of winter cereals include wheat, barley and rye. 25, 42, 45, 53, 71, 83, 100, 105, 127, 140, 160, 182, 185, 188, 193

Acronyms

CAP Common Agricultural Policy. 30, 118, 121, 143, 145

DUN Declaración Única. 4, 38, 49, 105, 130, 170, 182

ESYRCE Encuesta sobre Superficies y Rendimientos de Cultivos. 44, 45

EV Economic value.

ICGC Institut Cartogràfic i Geològic de Catalunya. 23, 24, 81

INE Instituto Nacional de Estadística. 116

MAPA Ministerio de Agricultura, Pesca y Alimentación. 21, 27, 28, 44

MR Management regime. 22–30, 35–37, 44, 46, 59, 61, 83, 84, 168

NBS Nash Bargaining Solution. 137, 138, 140, 143, 161

RCP Representative Concentration Pathway. 5, 168, 170, 176–180, 182–185, 187–189, 191–194

SPAs Special Protection Areas. 4, 18, 38, 58, 59, 102, 124, 149–151, 162

SQ Status quo.

STICS Simulateur mulTIdisciplinaire pour les Cultures Standard. 3, 4, 15, 22–25, 44, 49, 70, 73, 81–83, 87, 94–96, 104, 128, 129, 167, 168, 173–175, 192

Chapter 1

Introduction

During the last century, we have experienced a transformation of the landscape through human alteration (World Bank, 2020). This has had implications not only for economic outcomes, but also for the natural environment and its ability to provide ecosystem services. Within agricultural environments, this process has been characterized not only by an expansion of agricultural land, but also by its intensification (Giralt et al., 2021; Kragt and Robertsonm, 2014). This intensification development is based not only on the use of different management regimes, but also on the introduction of different crop types. As a result, the habitat structure has been dramatically changed, with direct implications for biodiversity, as many species have not been able to adapt to the new human-driven landscape schemes (Dasgupta, 2021; Millennium Ecosystem Assessment, 2005). But, in spite of this, we cannot forget that agricultural production is essential to guarantee food security and economic development. Biodiversity conservation policy goals are rarely achieved in isolation and must take into account their socioeconomic implications. With this in mind, conservationist policies must not neglect the consequences of their measures and must select those that can guarantee agricultural production and economic development while assuring the resilience of the natural environment. Moreover, land is usually in the hands of private owners, so biodiversity conservation policies must be supported by them in order to be successful. Consequently, it is crucial to consider not only what occurs inside conservation areas but also outside their boundaries (Polasky et al., 2007).

While the scientific literature has explored this topic in depth (e.g. Barraquand Martinet, 2011; Bateman et al., 2014; Polasky et al. 2007), the lack of research quantifying biodiversity conservation measures within agricultural areas in real scenarios motivated me to study the trade-offs between biodiversity conservation and agricultural economic output in a specific area. To this end, I build a linear land use optimization model tailored to a real case study: the Lleida Plain (Catalonia, Spain), an agriculturally based region that has experienced an intensification of agriculture through the construction of several irrigation channels during the last century, leading to an ongoing conflict between agricultural expansion and the sustainability of a bird community. Some species have been found to be poorly adapted to these new conditions and their populations have experienced a significant decline (Giralt *et al.*, 2021). The first goal of my thesis is to quantify the inefficiency levels in the current landscape configuration regarding the attainment of these two objectives, economic agricultural value and biodiversity preservation. The second objective, which derives directly from the first, is the evaluation of the opportunity cost in terms of the value of agricultural production due to the conservation of different species. These objectives were intended to be accompanied by a better understanding of actual conditions in agricultural regions through the use of realistic data.

In addition, it is essential to ensure that conservation policies are accepted by farmers and other stake-holders. Strategies should not be developed without considering political and economic factors as they may generate more opposition and fewer chances of implementation (Polasky et al., 2007). At the macro level, countries may be less willing to support global conservation strategies if they disregard their economic interests (Dobrovolski et al., 2014; Hurrell & Kingsbury, 1992). At the micro level, a vocal minority opposed to new agricultural policies can have significant policy outcomes, and consequently, policies should also account for the resistance of those farmers (Arbuckle, 2013). Unfortunately, conservationist policies often give rise to winners and losers, in which case implementing this policy may be difficult since a major opposition to these rearrangements comes from the fact that the losing farmers are worse off. My third aim is to study how conservation goals can create a dichotomy between winners and losers in the study area and to explore mechanisms that can reduce these losses and facilitate the viability of both biodiversity conservation and economic sustainability. Finally, I explore the implications of climate change in the relationship between agricultural economic value and the preservation of biodiversity in the study area.

Throughout the dissertation, I use a multi-objective, spatially explicit optimization model to identify the optimal landscape configurations considering agricultural land uses. I divided the region into minimum decision-making units (representative farmers) that present the most representative crops and management regimes in the area. The optimization relies on the spatial nature of the problem, where the same decisions may have different impacts depending on location. To account for this, I use the crop growth simulation model STICS (Simulateur multidisciplinaire pour les cultures standard) completed with spatially-specific values of climate and soil conditions to simulate crop yield values, which are complemented with economic values. I consider realistic land use choices in the region, accounting for 4 crops - wheat, barley, corn, and vineyards - with different management regimes, considering not only irrigation types and amounts but also the use of fertilizers, tillage options, etc. As a result, I depart from previous studies that generally contrast productive (e.g., agriculture) with non-productive options (e.g., grasslands) (e.g. Polasky et al., 2008; Barraquand & Martinet, 2011) by accounting for different productive options with different levels of economic return and impact on nature. To link economic value and biodiversity conservation I use bird species as an indicator. I build a habitat suitability index that relies in the previous works of Estrada et al. (2004) and Cardador et al. (2014).

The preservation of biodiversity within agricultural environments presents a multifaceted challenge and constitutes the primary focus of this dissertation. Next, we present how each chapter contributes to this overarching goal. All chapters share the common idea of how to balance agricultural production and biodiversity conservation, especially in a changing world. Although each chapter focuses on a different aspect, this integral approach is essential to develop sustainable, realistic and adaptable solutions to the challenges of reconciling economic development and sustainability. They also share a common methodology that allows for the comparison of many of the different scenarios proposed.

In Chapter 2, I present the core of the dissertation by explaining in more detail the procedure to generate our own database, which will be used in each chapter. Furthermore, I address our first two goals and hence I measure the trade-off between biodiversity and agricultural economic value and quantify the opportunity cost of preserving a community of 83 birds - with varying habitat preferences - in Lleida. I focus in particular on steppe birds, which have high conservation interest and have experienced the most significant declines during recent decades. To do this, I constructed an index of habitat preferences for each of these 83 species as regards each of the land uses considered, accounting for birds' requirements regarding nesting and foraging, and measuring the capacity of each land use to provide the resources to satisfy those needs. Then, by combining the crop yields generated by STICS and the habitat suitability index, I constructed a Pareto frontier representing the maximum possible economic value for a given number of species or group of species to be protected. I require a given species to be at least as adapted to the landscape as it is today. I use digital maps to check for current land uses from DUN (Declaración Única Agraria). This method allows us to observe the order of protection in terms of opportunity cost and can be useful for policymaking.

I included in the spatial configuration irrigation constraints regarding infrastructure limitations or conservation areas denominated as SPAs (Special protection areas). Consequently, I also take advantage of the methodology used to compare our results not only with respect to land use optimization but also with the optimization of conservation areas to check if they are optimally located. Finally, I depart from the triage approach - where all species are considered equally important in terms of conservation - and focus on steppe bird conservation to check what a target conservation policy may implicate in terms of reducing opportunity cost.

However, as I pointed out previously, the implementation of these optimal landscapes could be difficult for a policymaker since an important opposition to these rearrangements comes from farmers being worse off. Few studies have analyzed this issue in depth in the spatial optimization literature (one example is Pitafi et al., 2009), so my intention in Chapter 3 was to use our database to expand our analysis with special emphasis on the acceptance of the optimal landscape patterns obtained, using the number of losers - farmers who are earning lower economic profits compared to the status quo - as a proxy for opposition and therefore fulfilling my third goal. I propose two scenarios that involve the existence of agreements by which this problem could be solved. Firstly, a solution related to a transfer system where winners are able to compensate losers. On the other hand, a solution proposed by the regulator where there are no losers but there are no transfers. Our interest is to observe how these solutions without losers can affect the agricultural economic value of the region. Further, I compare the performance of these mechanism with a more broadly used and inefficient type of policy mechanism, a homogeneous per hectare payment scheme to promote agricultural practices aimed at protecting biodiversity and the environment. Finally, for measuring the gains in efficiency that could be obtained by tailoring payments I propose a heterogeneous payment per hectare that can be made to each farmer individually, so that it can be more efficient and achieve the same biodiversity objectives but with a much lower budget. To add realism, I present an alternative solution where the regulator estimates these compensations with incomplete information, and a solution considering the inequality between farmers.

Finally, in Chapter 4, I analyze for the same case study the consequences of climate change on agricultural economic value and biodiversity through habitat transformations. I assume that if climate conditions change, farmers will be motivated to change their decisions and this can have implications for biodiversity. To carry out this task, I use forecast climate data until 2100 and simulate crop yields under future conditions with two different greenhouse emissions scenarios (Representative Concentration Pathway (RCP) 4.5 and 8.5). Then, I simulate different scenarios based on farmers' decision strategies: continue with current land uses or cope with new climate conditions to maximize their economic profits. I also add a centralized solution where a regulator maximizes economic value subject to total irrigation constraints, and also with respect to biodiversity conservation goals. I find this especially relevant in the context of future climate conditions, where water is expected to be scarce.

The structure of the remainder of this thesis is thus organized into three chapters containing the essays described above, as well as a conclusion that summarizes the key points and contains suggestions for future research.

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Chapter 2

Farmland economic value and bird sustainability: A trade-off quantification for the Lleida plain

2.1 Introduction

The change from natural habitat into agricultural systems is one of the main reasons for the loss of terrestrial biodiversity. During the 20th Century, from 40-50% of the terrestrial natural areas have been transformed into agricultural or urban uses (Western, 2001). This massive conversion has lead to the current situation, where 36.5% of the land is devoted to agriculture globally (World Bank, 2020). This increase in land used for food production has been driven by two powerful forces: the population increase - the number of inhabitants on Earth has gone from 1.65 billion in 1900 to 8 billion in 2022, with 9.7 billion forecast in 2050 - and the improvement of human living conditions that has taken place during the past century. Although the requirements for agricultural expansion to feed humanity are uncertain and depend on many factors - such as changes in diet composition - they are expected to be significant. FAO projections (FAO, 2007; United Nations, n.a.) indicate that the increase in arable land and permanent crops worldwide between 2005 and 2050 implies an increase of more than 69 million of hectares - an increase of about 4.7% of arable land -. This is not, by far, the greatest magnitude estimated, since other predicted scenarios estimate increases of more than 310 million of hectares from 2006 to 2050 - 20% of arable land - (Hertel et al.

2016). Tilman *et al.* (2011) also estimated that an additional billion hectares of agricultural land will be necessary to satisfy human demand for food, fiber, and fuel in the future.

However, land-use decisions focused on a single output - such as agricultural economic benefit - have higher chances of resulting in the decline of the provision of other services or damage to the natural environment (Millennium Ecosystem Assessment, 2005). Agriculture faces a double challenge: supplying enough food to meet future demand and simultaneously reducing the environmental impacts that the production of this food may require. Consequently, if both goals are to be satisfied, avoiding potentially irreversible and non-desirable effects, the inherent trade-offs between satisfying human needs and maintaining the ecosystems in the long run should be considered (Rodríguez *et al.*, 2006). If we refer to biodiversity, the objective is to satisfy the economic objectives while ensuring the preservation of the species. However, since species have heterogeneous preferences for different land uses, there is an ongoing debate of conservation priority. Those favoring triage suggest that the management of species should be based on cost-efficiency reasons - although this may lead to low investment in threatened species -. The other point of view suggests that focusing on the most urgent species is crucial since this may also increase the survival rates of many other species in the future (Wilson et al., 2011).

This trade-off appears at different levels of intensity all over the world and the *Iberian Peninsula* is not an exception. In Southern Europe, land intensification through irrigation has favored crop yields increase in many dry areas and even, allowing the introduction of new crops in previously low productive areas. One of these places is the *Lleida* plain (*Catalonia, Spain)*. This area has been typically devoted to field crops, with an original steppe-related habitat. Steppes are defined as those environments dominated by herbaceous plant formations with low height, typical of continental temperate zones, which in Spain occupy more than half of the cultivated zones - mainly in the Castilian plateau, Extremadura and the basins of the Guadalquivir and Ebro rivers -. They can also be found in southern Portugal, as is the case of *Baixo Alentejo* (Leitão et al., 2011) or *Beira Baixa* (Catry et al., 2011).

In Catalonia, the steppe environments correspond with the large interior plains occupied by extensive farmland areas which alternate with the rest of the steppe lands. Although they are not true steppes (usually referred to as *pseudo-steppes*), they have a similar vegetation structure characterized by an undulating landscape, mostly occupied by rainfed crops such as wheat or barley, with their associated fallow land (Sainz, 2013). The traditional management of these farms, together with their adaptation to the dry Mediterranean climate (no more than 500 mm of annual precipitation levels), has managed to create a friendly landscape for fauna, mainly for granivorous and also herbivorous birds. However, the extension of irrigation projects in recent decades and the corresponding intensification of crop management have led to the cultivation of crops totally unsuited to a climate characterized by water scarcity and jeopardized the sustainability of a large number of steppe bird species that cannot thrive on irrigated crops (Cabodevilla et al., 2022).

Considering this, the *Lleida* plain has traditionally given refuge to a rich and diverse steppe bird community, which has its principal populations in Western Europe in the Iberian Peninsula.[1] Most of these birds are scarce in Europe and have their last refuge in the Iberian Peninsula, sometimes with populations representing very high percentages in comparison with the world breeding population. An important proportion of steppe birds are exclusively from these habitats. Species like *Tetrax Tetrax*, *Pterocles orientalis*, *Pterocles alchata*, *Melanocorypha calandra* and *Chersophilu duponti* have strict ecological needs which mean that they cannot be found outside these areas. Consequently, this has led the birds associated with these habitats to be threatened and with high risks of extinction. Furthermore, species such as *Circus aeruginosus* and *Hieraaetus fasciatus*, which are not characteristic from steppes, also depend on these types of habitats (Estrada, 2004a).

From the economic perspective, the main characteristics of these rainfed lands is their seasonality and low productivity. This occurs due to the low precipitation levels which are

[1]This is the case of the *Little Bustard (Tetrax Tetrax)* or *Coracias Garrulus*, which have in this plain the main community in the north-eastern part of the Iberian Peninsula, exceeding the total population of whole countries like France or Italy.

concentrated during autumn and spring, while winter is extremely dry, and due to the occurrence of high air temperatures during grain filling, both causing terminal drought. This limits the plant production and with that the trophic resources available for the wildlife. This also results in low densities for most of the species, and a low crop yield. Although land intensification has been employed to increase crop yields under these circumstances, it also implies a drastic, fast, and large-scale change with demonstrated negative effects on biodiversity, and particularly on birds adapted to rainfed farmlands and steppes (Brotons et al., 2004; González-Estébanez et al., 2011; Traba et al., 2013; Cardador et al., 2015; De Frutos et al., 2015, as cited in Giralt *et al.*, 2021). In the Iberian Peninsula, extensive cereal areas have been replaced or fragmented with other crop systems like irrigated fruit trees or herbaceous crops (such as corn or alfalfa), resulting in a drop in numerous species of European fauna and flora during the last 40 years (Benton et al., 2002; Donald, 2001; Krebs et al., 1999; Robinson and Sutherland, 2002; Siriwardena et al., 1998, as cited in Barraquand & Martinet, 2011). The agricultural intensification in the *Lleida* plain has not benefited these species. In particular, the abundance of some birds, which were adapted to rainfed lands associated with specific types of vegetation, has been affected negatively by landscape transformation (Estrada et al., 2004a). The creation of irrigation channels has allowed the establishment of new crops which are more productive, but at the same time has modified the natural environment. In fact, irrigated and rainfed crops are radically different in terms of yields. To see this, in 2017 the surface devoted to rainfed crops was twice that of irrigated crops. However, the former only produced one quarter of the latter in terms of production (Reguant, 2017).

Regarding this issue, the EU has been concerned about the preservation of natural resources and biodiversity in farmland in recent decades, resulting in both the Birds Directive and the Habitats Directive, and the creation of the *Natura 2000* network. Clearly, establishing formal protected areas — such as in the *Natura 2000* network - is one means of conserving habitat, but socioeconomic and food requirement constraints limit the amount of land with such a status, making conservation necessary outside their boundaries (Polasky et al., 2007). In fact, farmland also hosts relevant biodiversity, including both common and emblematic species (Chamberlain et al., 2000; Söderström and Pärt, 2000, as cited in Bamière *et al*., 2013). This relates to the concept of High Nature Value (HNV) [2], referring to the farming systems and the farmland that support high levels of biodiversity or species and habitats of European conservation concern (Beaufoy and Cooper, 2009). However, developing indicators of HNV farmland has been challenging due to data limitations, privacy issues, and methodological diversity (Keenleyside et al., 2014). Moreover, the trade-offs between agricultural production and conservation are not well understood, and the role of policy instruments such as Natura 2000 or the Common Agricultural Policy (CAP) in maintaining HNV farming systems is unclear (Mikulcak et al., 2013).

The problem is, however, that the increase in food production has been achieved by intensifying agricultural practices that have resulted in great environmental impacts. This has threatened the survival of many species (Dasgupta, 2021; Millennium Ecosystem Assessment, 2005) not only because the extension of farmland has increased but also because the expansion of farmland has been accompanied by land use intensification through the increase of irrigation water, fertilizers, soil contamination and erosion, among others (Giralt *et al.*, 2021; Kragt & Robertson, 2014). Consequences of this concern can be found in EU environmental legislation that, although not specific to agriculture, has had important implications for agricultural land management. We will thus focus on the reliability of policy instruments that spatially limit the set of land management regimes that can apply intensification techniques that are non-biodiversity friendly.

The selected methodology for analyzing these trade-offs between production and conservation in agricultural landscapes is based on multi-objective optimization processes, which have been extensively applied (Drechsler et al., 2020; Rosenstock et al., 2016). These models account for the multiple usually competing demands for biodiversity conservation and

[2]The term High Nature Value (HNV) farming was coined in the 1990s to raise awareness for the declining farmland biodiversity in Europe (Baldock et al., 1993). HNV farming soon became a key concept in the European Union's (EU) environmental and agricultural policy.

agricultural productivity (Strauch et al., 2019). They have aided in locating areas where trade-offs, such as those between biodiversity and crop yields are low, thus enabling the identification of eco-efficient solutions (Kaim et al., 2021; Todman et al., 2019). Although in their articles many authors use predefined scenario lists (e.g. business as usual, specific land use expansion, etc.), this type of analysis can be approached from a model-based design coupled with land use models and optimization techniques. Decision-makers can thus explore a vast variety of solutions depending on the conditions introduced in the model. These models can become very complex through the addition of spatial components, stakeholders, variables, objective functions, etc. (Liu et al. 2013). Consequently, and differently from the predefined scenarios approach, potential thresholds are not avoided (Kennedy *et al.*, 2016), and potential optimal solutions are detected.

Agricultural land usually performs various functions, such as providing food and hosting a variety of species. These functions often conflict, requiring compromises to optimize the provision of both goods. Multi-objective spatially explicit models are suitable for addressing this challenge, which involves either producing more output at the same environmental cost or the same output at a lower environmental cost. Spatial optimization algorithms, which rely on accurate approaches such as linear programming (Sadeghi et al., 2009) or heuristic approaches such as genetic algorithms (Memmah et al., 2015) can facilitate this achievement. The resulting Pareto or efficiency frontiers enable the characterization of inefficiencies and the quantitative evaluation of trade-offs between competing objectives. Additionally, these methods enable the identification of land use configurations that provide optimal solutions to mitigate trade-offs (Law et al., 2021). Modeling results can also facilitate the comparison of observed land use with optimized land use solutions.

Within the multi-objective optimization literature, alternative methodologies have been carried out. For example, Barraquand and Martinet (2011) analyze these trade-offs between biodiversity and agricultural intensification with a dynamic approach. Even recognizing the inherent advantages of this methodology, such as the inclusion of species population growth components or dispersion or the consideration of variables that have a marked temporal character such as prices, they must leave aside the complexity in other aspects included in this thesis such as the use of crop simulators or the use of habitat suitability indices, which have a non-dynamic character. In practice this can be done, but the complexity that would entail doing it for a temporal scale that is compatible with the effects at the biodiversity level would make it computationally very demanding.

Furthermore, any statistical or econometrics approach requires very rich and large databases that were not available (Rosenstock et al., 2016). In our case, crucial information about the irrigation practices performed in each cell was not available and we could not have estimated the productivity from different irrigation regimes. In particular, the data should be able to inform about the performance of the system under different conditions. Without this wealth of data, the estimates we obtain from a specific case are difficult to extrapolate to a different one. This provides limited flexibility for conducting analyses that may arise when certain regulatory characteristics change, among other factors. On the other hand, many species have minimal presence in each cell. Therefore, relating land uses to species that are absent in most observations complicates the estimation of the relationship. Other methods such as Data Envelopment Analysis (DEA) techniques have been used to measure the trade-offs between different competing land uses (Picazo-Tadeo et al., 2011; Perez Urdiales et al., 2016). However, in our case, we need to estimate the Pareto frontier given current production conditions and approximate it when climatic and/or regulatory characteristics change. This lack of data also made it difficult to apply another widely used method, Stochastic Frontier Analysis (Orea and Wall, 2016).

On the other hand, the participatory approaches literature, since Pretty's seminal paper (1995), has stressed the importance of involving farmers and local communities in decision-making and conservation efforts, and has explored the role of social capital in biodiversity conservation in agricultural areas (Pretty and Ward, 2001), emphasizing the importance of community engagement and cooperation. Moreover, participatory research has been conducted within the agriculture and biodiversity trade-offs literature, providing interesting approaches to identify actor-relevant objectives (Rosenstock et al., 2016) and supplying relevant information to support quantitative tools (DeFries and Rosenzweig, 2010). This literature has shown that involving stakeholders in agri-environmental problems, detecting factors influencing farmers' adoption of sustainable practices can result in the identification of policies that are more congruent with real-world conditions (Dessart et al., 2019). However, the focus of this study was to quantify the agriculture and biodiversity trade-offs and to offer a flexible approach that could be adapted to different and hypothetical scenarios. The identification of the behavioral factors that influence farmers' decisions to adopt conservationist practices requires field work that was out of our expertise and beyond our resources.

Considering this, the goal in this chapter is to study the set of trade-offs between agricultural economic value and biodiversity protection in the *Lleida* plain and design the best land configurations and strategies able to guarantee agricultural productivity and biodiversity conservation of the area - from a *Social Planner* perspective -. In other words, we quantify the economic value which has to be set aside so as to maintain wildlife in the region. Due to the advantages stated above, we choose to apply a spatially explicit multiobjective optimization by designing a bioeconomic land use model together with a computational optimization solver. Our results are repeated for different biodiversity goals - depending on how ambitious or targeted they are - to highlight the economic implications of different conservation strategies. Moreover, we explore the implications of reallocating or removing the current conservation areas in the trade-off analysis. For all this, we need to assess crop yield differences under different agricultural management types - catalogued depending on irrigation quantities and types - but also locations, and to analyze the impacts on biodiversity of such practices. We apply the theoretical framework to a realistic farmland scenario, using the *Lleida* plain as our case study. To estimate potential crop yields for each location of our area of study we use the crop growth simulation model (STICS) (Brisson et al., 2008), fed with climatic, soil, management and plant feature data. On the other hand, we approximated the impact of the previous choices on biodiversity. For that, we used a habitat preference index considering species-specific requirements. Considering the implications of each land use within each specific location, we maximize the economic value subject to different biodiversity constraint levels - number of species protected -.

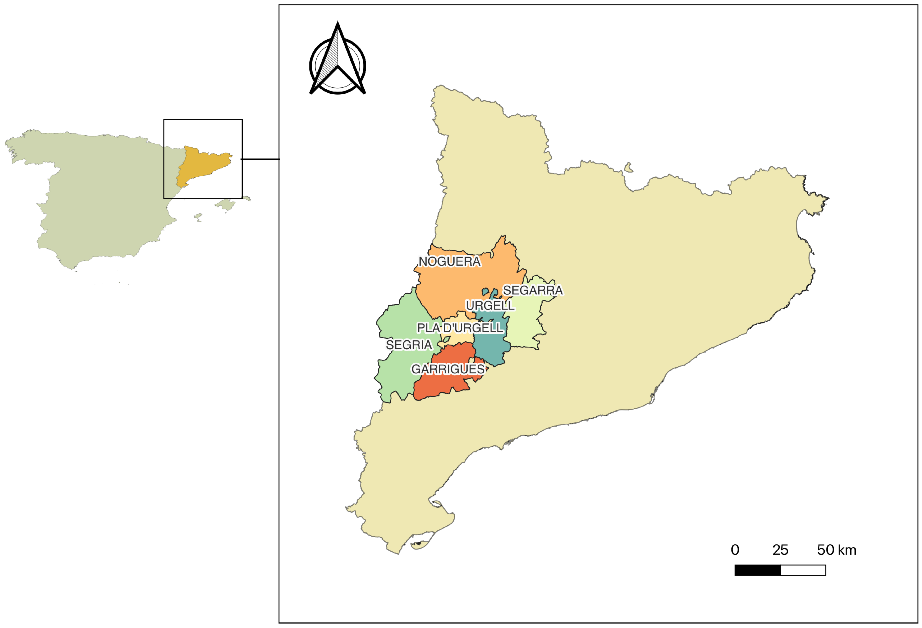
The structure of the chapter is organized as follows. First, we detail our case study and set out the methodology employed to spatially quantify the crop yields attributed to different agricultural land uses (section 2.2). Subsequently, we describe the theoretical model including the optimization problem and construction of the biodiversity habitat suitability index and variations with respect of the model assumptions (section 2.3). We continue with the results (section 2.4) and conclusions (section 2.5).

2.2 Case study and economic data elaboration

2.2.1 Description of the case study and spatial data framework

The analyzed region comprises the counties of Segrià, Urgell, Pla d'Urgell, Segarra, Garrigues, and Noguera, located in the province of Lleida, Catalonia, Spain (as shown in Figure 2.1). It covers an area of 5,582 square kilometers in a vast plain with relatively uniform height and climate across locations. The northeastern areas experience higher precipitation levels, with annual values reaching up to 562 mm annually, while the central and southwestern areas receive less than 400 mm annually, resulting in more limited biomass productivity without irrigation. Annual maximum mean temperatures range from 23°C to 25°C. These lands are located at a relatively low altitude, with elevations ranging from 120 to 500 meters above sea level (PTP, 2007). Forested areas are mainly situated at the periphery of the study region.

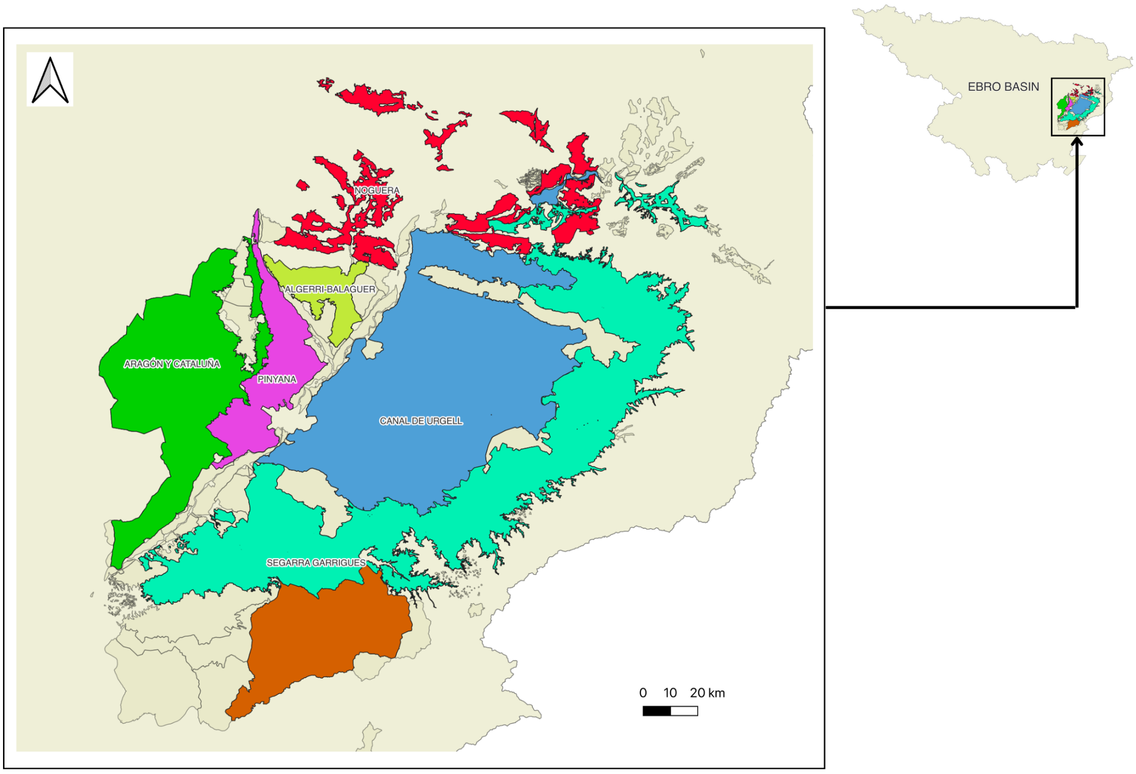
Figure (2.1) CASE OF STUDY MAP



Notes: Six colored counties are those considered for the study (*Lleida* plain). Map composition elaborated using *QGIS 3.24*. Author: Own elaboration.

In the Lleida plain, agricultural profitability has historically been hindered by aridity and irregular precipitation patterns. At the end of the 18th century, approximately 75% of the area of the Lleida Plain, including the current counties of Segarra, Garrigues, Segrià, Noguera, Urgell, and Pla d'Urgell, was almost barren. However, this area has since decreased. The tradition of irrigation in Catalonia dates back to the 12th century with the Pinyana channel in Lleida. However, significant irrigation projects emerged in the second half of the 19th century, the Urgell channel being the most important in the Lleida region. Other notable channels, such as Aragón y Cataluña, were built in the early 20th century. In the 1980s, modernization efforts were initiated with the construction of Algerri-Balaguer and Segrià Sud channels. In 2004, the Segarra-Garrigues channel project began, aiming to irrigate 68,645 hectares, promote economic growth, and offer new product alternatives. While designed to be environmentally conscious, the project necessitated significant landscape transformation. These irrigation projects have progressively transformed agricultural practices in the lower regions of the Lleida plain, dominated largely by intensive agriculture (Reguant, 2017). Some of the most important irrigation systems of the Lleida plain are represented in Figure 2.2.

Figure (2.2) MOST IMPORTANT IRRIGATION SYSTEMS IN THE LLEIDA PLAIN



Notes: QGIS 3.24. Author: Own elaboration.

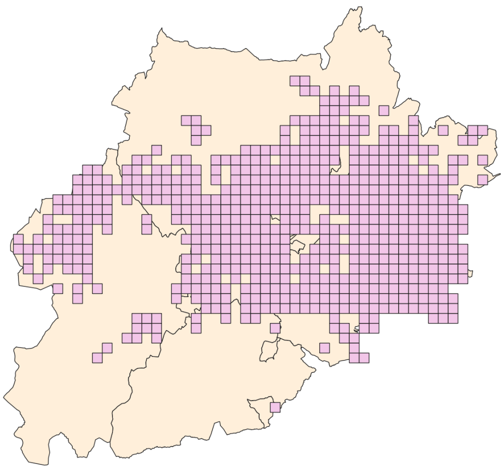
As expected, the development of the *Segarra-Garrigues Channel* project has not been exempt from controversy among different stakeholders. One of the most significant issues is that 42% of the cultivated land within the project perimeter was designated as *Special Protection Areas (SPAs)* for birds, which imposes strong limitations on irrigation. Consequently, different irrigation regimes have been established regarding the maximum amounts of water allowed per hectare. Specifically, in the core of agricultural production in the *Lleida* plain, *SPA* areas occupy 64,261 ha, of which 34,183 are potentially irrigated territories of the *Segarra-Garrigues* project (Reguant, 2017). These changes in *Lleida* can be generalized to the rest of the Iberian Peninsula. From 1990 to 2012, the area occupied by irrigated crops grew from 22,989 km^2 to 28,239 km^2 (Δ 22.8%), while the area of rainfed crops decreased from 113,174 km^2 to 107,709 km^2 (Δ -4.8%). However, the overall agricultural land use has experienced a decline from 292,836 km^2 to 275,556 km^2 (Δ -5.9%), while artificial surfaces have increased by 83% (Fernández-Nogueira & Corbelle-Rico, 2018).

For our model, and since it allows for the singling out of highly homogeneous soil and climate units, the study area has been divided into 2×2 km squared cells (Figure 2.3). A total of 602 cells - which we could interpret as representative farmers - were included in an artificial grid, only including those with at least 30% of their area dedicated to the crops considered in this work. [3] The selection of these crops was based on four criteria: firstly, they are predominant in the region; secondly, their yield could be reliably estimated; thirdly, land uses must be matched from the economic and conservationist perspective, and consequently the impact of a given land use must be quantifiable on both sides; and fourthly, a limit was imposed on the number of crops to avoid excessive computational efforts. The selected crops were wheat, barley, corn, and vineyard. In fact, barley and wheat are the most relevant crops in the study area, representing 55.25% and 25.7% of the sown area, respectively, followed by corn that represents 12.9%. These crops can be linked to two of the habitats included in Estrada *et al.*, 2004b (Table 2.1): *Cereal crops and Irrigated herbaceous crops*, which includes barley and wheat, and corn, respectively. We also included vineyard, representing 1.5% of the study area, which was matched with its analogous category. Consequently, we used three of the five habitat categories included in the bird atlas. [4] Fruit trees were excluded due to difficulties in simulating their yields with process-based models, even though they are relevant in the region. Cells with significant proportions of land devoted to urban, forest or other land use types were excluded from our analysis and were considered fixed.

[3] For a more detailed of the spatial grid construction see subsection 2.A.

[4] Irrigated herbaceous crops, irrigated fruit trees, non-irrigated fruit trees, vineyards and cereal crops.

Figure (2.3) GRID COMPOSITION



Notes: Squares represented in pink are those cells considered for the optimization process. Each is a minimal decision unit and can be considered as representative farmers. The background map shows the administrative area of the counties associated with the Lleida plain. Map composition elaborated using *QGIS 3.24*. Source: compiled by author.

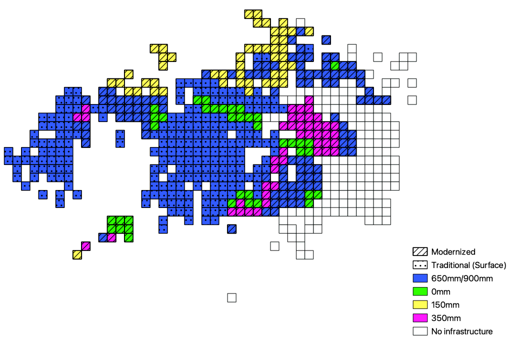
Table (2.1) MATCH BETWEEN BIRDS HABITAT AND CROP SELECTION

Habitat (*Estrada et al.,2004b)*	**Crop selection**
Cereal crops	Barley Wheat
Irrigated herbaceous crops	Corn
Vineyards	Vineyards

Notes: Habitat categories specified by *Estrada et al. (2004b)*.

As previously mentioned, the region is subject to local management restrictions. Hence, we take into account spatial limitations on irrigation for each cell, in order to approximate our solutions to the current situation and also to enable us to work with various assumptions about their optimal allocation. The management of irrigation is constrained by two factors: firstly, the requirement for infrastructure to irrigate, for example, the eastern parts of the Segarra region are rainfed without irrigation infrastructure. Additionally, not all irrigation areas are modernized, but rather rely on surface/traditional irrigation methods, such as the Urgell Channel. Secondly, legal actions are being applied in specific areas by restricting the maximum quantities of supplied irrigation (SPAs). To spatially identify these limitations, we use the Irrigation Plan 2008-2020 (Dirección General de Desarrollo Rural, 2017), which distinguishes between modern and traditional irrigation areas, as well as other non-irrigated territories. The Ministry of Agriculture, Fisheries and Food (MAPA) provided us with a digital map that specifies the maximum irrigation amount, distinguishing between 0mm, 150mm, 350mm, and 650mm per year. [5] For areas under surface irrigation, we assume that the maximum irrigation volume per year is 900mm. [6] As we will explain later, these restrictions will define the set of land uses considered. We merged both maps to obtain a combination of infrastructure and maximum irrigation amounts and types (see Figure 2.4). [7]

Figure (2.4) LOCATION OF IRRIGATION CONSTRAINTS IN THE STUDY AREA



Notes: Values in each cell have been selected according to the maximum area criteria. Description of color and symbol in the legend. Source: Compiled by author.

[5]: 1 mm = 0.001 m^3 of water per m^2.

[6] This is the amount of water used for corn. In the case of wheat, it would be 258mm, while in the case of barley, it would be 129mm.

[7] Some cells overlapped with more than one irrigation systems with different constraints. In that case, we considered the one with the highest percentage of the area in each cell as representative.

As observed, the areas without access to irrigation are located in the eastern parts, while the central and western zones can potentially be irrigated. Traditionally mostly irrigated lands are mostly located in the central parts of the region, specifically in Urgell and Pla d'Urgell, but also in western zones such as Segrià. On the other hand, modernized irrigation territories can be found to the east of Urgell, corresponding to the Segarra-Garrigues irrigation system, with a significant proportion of land under irrigation constraints. Additionally, in northern territories such as Noguera, there are areas with access to modern irrigation.

2.2.2 Crop yield simulation

We simulated potential crop yields for the selected crops in the study area for each cell under different management regimes (hereafter referred to as MR). Since no data on potential yields were available at this resolution scale, we utilized a soil-crop process-based model. Due to its sensitivity and versatility, we employed the Multidisciplinary Simulator for Standard Crops (STICS), developed by the French National Research Institute for Agriculture, Food and Environment (INRAE) (Brisson et al., 2008). STICS simulates the behavior of the soil-crop system at a daily scale over one or several crop cycles and is organized into modules composed of sub-models dealing about specific mechanisms. There are three modules that deal with the ecophysiology of above-ground plant parts (e.g., yield formation), four modules that deal about root and below-ground crop interactions (e.g., root growth, water and N acquisition), another module that deals with the interactions between the management practices and the soil-crop system, and a microclimate module that simulates the combined effects of climate, water balance, temperature, and air humidity within the canopy. The model can be extended to various crop systems. Unlike other popular crop growth simulation models, it simulates processes such as frost or heat stress and provides more detailed information on phenology stages. Consequently, it requires high-quality input data and some expertise to be used properly (Constantin et al., 2015). Specifically, this model demands data from four main categories: soils, climate, management, and plant type, which are represented in Table 4.2. STICS has already been used to simulate scenarios in

Mediterranean environments (e.g. Plaza-Bonilla et al., 2018).

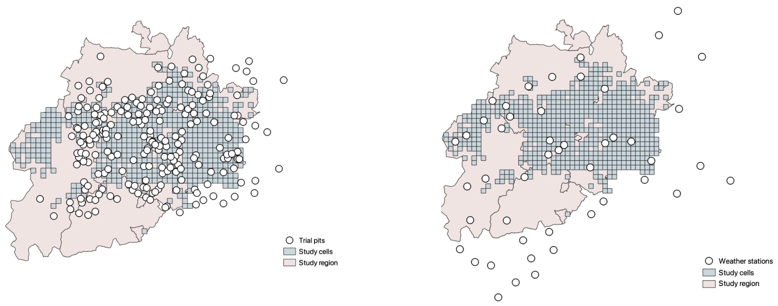
Table (2.2) STICS INPUT DATA

Categories	Source	Input names	Nº of sample points
Soil	Institut Cartogràfic i Geològic de Catalunya	pH, calcium carbonate content, water content at field capacity and wilting point, Clay, Organic carbon, rooting depth, Organic Nitrogen	216
Climate	Servei Metereòlogic de Catalunya (Meteocat)	Precipitation, max. temperature, min. temperature, vapor pressure, global radiation, windspeed at 2m, evotranspiration, CO2.	47
MR	Expertise	Supply of organic residues, tillage, sowing, N fertilization, irrigation, harvest.	-
Plant	STICS default	Barley, wheat, corn, vineyard.	-

Notes: The crop growth simulation model *STICS* required four input categories to simulate crop yields: soil, climate, management and plant information. Each category is composed of different variables represented in the third column. The last column corresponds to the number of spatially located sources of information (trial pits and weather stations).

Soil data were collected from 216 soil sampling pits located in different areas of *Catalonia* (Figure 2.5.a) and provided by the *Institut Cartogràfic i Geològic de Catalunya (ICGC)* through the *Geoindex display*. To obtain climate data (weather stations represented in Figure 2.5.b), we used daily information provided by the *Servei Metereòlogic de Catalunya (Meteocat)* from 47 weather stations. We considered eight climate variables: *Precipitation (mm/day), Maximum temperature (^oC), Minimum temperature (^oC), CO2 concentration (Ppm), Vapor pressure (Mbar), Global radiation (Mj/m^2), Wind speed at 2 m height (m/s)*, and *Penman Evotranspiration (mm/day)*. The *STICS* model required daily information for each of these climate variables for a given year. Instead of using information from a single recent year, we created a synthetic year by averaging climate data for each day and station over the past 20 years.[8] The main purpose of this procedure was to avoid non-representative years that could potentially distort our results. Therefore, for each weather station, the value for each day and variable was the average of all the years with data availability. [9] As weather stations or soil pits were not available for all cells, we carried out dot geostatistical interpolation processes (*Kriging* method) in the region to obtain crop yield values for each cell. This resulted in a raster layer with values for each pixel and variable, and we subsequently computed the average pixel value within each cell.

Figure (2.5) TRIAL PITS AND WEATHER STATIONS LOCATIONS



a. Trial pits

b. Weather stations

Notes: (a) Trial pits location. White dots represent trial pits (*ICGC*). (b) Weather stations (Meteocat) location. White dots represent weather stations location. | Dots variable values were afterwards interpolated and averaged for each cell. Green squared cells are those selected for the optimization process. Background map shows the administrative area of the case study.

On the one hand, we selected five distinct crop intensification regimes (MR) that correspond to varying amounts of irrigation supplied ($mm/year$) along with specific practices related to N fertilizer use, tillage, plowing, sowing, and harvesting. These regimes include rainfed lands (0 mm), modern pressurized irrigation with three different levels of irrigation

[8]We used data for a given year and weather station only when information was available for every day of a single year for that variable.

[9]For more detailed information about the *STICS* parameterization, see subsection 2.B.

(150mm, 350mm, 650 mm), and traditional surface irrigation. [10] The information was obtained mainly from *Plaza-Bonilla et al.* (2018), data sheets, and expertise when needed. In STICS, we not only considered the amount of irrigation water supplied but also its efficiency. For cereals and corn, we considered surface and sprinkler irrigation, while for vineyards, we simulated drip irrigation.

However, not all regimes were considered for each crop due to agronomic reasons (Table 2.3). For instance, winter cereal crops such as barley and wheat do not require high irrigation volumes to achieve their potential yield in this region since their development coincides with the periods with greater precipitation. Specifically, we considered rainfed, traditional irrigation and 150/350 mm modern irrigation for barley and wheat; no irrigation and 229 mm for vineyards (this amount was selected following DARP (2013). Higher amounts of water were found to be excessive); we considered all categories for corn.

Table (2.3) COMBINATIONS OF CROPS AND IRRIGATION SCHEDULES

MR/Crop	**Wheat**	**Corn**	**Barley**	**Vineyard**
Rainfed	*Yes*	*Yes*	*Yes*	*Yes*
Surface	*Yes*	*Yes*	*Yes*	*No*
150mm	*Yes*	*Yes*	*Yes*	*Yes**
350mm	*Yes*	*Yes*	*Yes*	*No*
650mm	*No*	*Yes*	*No*	*No*

Notes: Yes=considered; *No*=Not considered. (*) Vineyard irrigation amount was 229mm.

Finally, with respect to plant parameters, we used the variety *Biensur* for durum wheat, *Escourgeon* for barley and *Cecilia* for corn. For vineyards, we used *Chardonnay* as representative. They were selected for being close to common varieties of this region, although they were parameterized originally for *France*. However, we modified some parameters to

[10] These amounts were intentionally designed to coincide with current limitations on the area and commonly applied techniques and agricultural inputs usage.

improve the accuracy of simulations to observed values in *Lleida* (Plaza-Bonilla et al., 2018).

Due to the complexity of the variables used when applied to real scenarios, the model is based on some assumptions to make it tractable. Firstly, management decisions are discrete. This implies that only a limited set of practices are considered, rather than continuous (e.g., amount of water, fertilizers, etc.). In second place, the nature of the model is static. This means that results should be interpreted as potentially reached scenarios, ignoring dynamics and transitions from the current situation. Finally, we do not consider heterogeneity in the farmer's ability. Consequently, crop yields should be interpreted as potential, and therefore those obtained if the farmer always applies the practices efficiently. Still, the results shown in this study should be useful to observe and track dynamics and inherent trade-offs between presumably antagonistic objectives.

2.2.3 Agricultural economic outcomes valuation

In our study, we consider estimated crop yields to approximate the potential benefits of various land-use decisions in each cell. This is crucial since farmers' land-use decisions are predominantly influenced by profitability, rather than productivity. We utilize the net margin as an economic value measure and calculate the annual economic net margin EV_{ijr}, defined as follows:

$$EV_{ijr} = a_i \times (Q_{ijr} \times P_j - C_{jr} - W_{ijr})$$

Here, Q_{ijr} denotes the potential crop yield (in tons/ha) for a given crop j and management regime (MR) r combination in each location i, which varies based on soil and climate characteristics. a_i represents the area in hectares corresponding to location i, while P_j (in €/ton) is the market price for the crop. C_{jr} represents the cost (excluding water cost) of the crop under different MRs (in €/ha), and W_{ijr} denotes the water cost (in €/ha).

We assumed a single value for the prices of corn, wheat, barley, and vineyards, as prices differ only between crop varieties. We obtained price data from the Spanish Ministry of Agriculture, Fisheries and Food through ”Índices y precios percibidos agrarios” (MAPA, n.d.), which measures monthly and annual national price trends received by farmers for the sale of agricultural products. Since these crops are highly competitive commodities and their prices are set in world markets, we assumed that the prices received by farmers in Catalonia were the same as those in the rest of Spain. To avoid non-representative years, we used average prices for the period 2010-2020 for each crop, with grape from vineyards being the most expensive crop among those examined, with an average value of € 0.35/kg. The prices of other crops are comparable, with barley, corn, and wheat having average values of € 0.18/kg, € 0.17/kg, and € 0.19/kg, respectively.

Due to the unavailability of detailed information on the costs associated with each MR, we employed general data for each crop and subtracted the costs of water and fertilization specific to each crop and management category combination. In Table 2.4, we present a summary of the costs per hectare per year in euros for all crop and MR combinations, excluding the cost of water. The cost of water varies depending on the irrigation system, with higher water usage resulting in higher costs, as well as the specific irrigation systems used in each location, which is associated with unique payment schemes and prices. Typically, traditional irrigation systems incur annual independent-water usage payments, while modernized systems based on pressurized irrigation comprise a fixed and variable cost contingent upon the volume of water utilized.

Accounting for water costs, there are a total of 66 irrigation systems in the *Lleida* plain. The larger ones include the *Urgell Channel* (75,000 ha with traditional irrigation methods), the *Segarra-Garrigues Channel* (81,376 ha with modern irrigation methods), the *Pinyana Channel* (13,500 ha with traditional irrigation methods), and the *Algerri-Balager* (7,395 ha with modern irrigation methods) (Reguant, 2017). Due to the small size of most systems and the lack of public cost data, we used the cost of the main irrigation systems for

Table (2.4) COSTS FOR EACH COMBINATION OF CROP AND IRRIGATION MANAGEMENT (EXCLUDING WATER COSTS)

	Crop			
MR	*Wheat*	*Corn*	*Barley*	*Vineyard*
Rainfed	707	1616	633	3155
Modern 150mm	724	1624	650	3165
Modern 350mm	750	1650	675	-
Modern 650mm	-	1701	-	-
Surface	750	1701	685	-

Notes: Source: *IDESCAT, 2021*. Unit of measure: €/ha/year. Water costs are excluded. Cells with no data correspond to non-feasible options in the study area and consequently are not included in the optimization problem.

the other areas. Specifically, the irrigation costs of the *Urgell Channel* were generalized to other non-modernized irrigation systems, while *Algerri-Balager Channel* was used as a reference for modernized channels.[11,12] In addition to this generalization, we used channel-specific costs for the *Pinyana* and *Torres de Segre - Carrassumada* systems. Except for the *Segarra-Garrigues Channel*, for which we obtained updated data from its website,[13] and the *Algerri-Balaguer Channel*, for which the data were provided by email, the rest of the costs are shown in Reguant (2017). The irrigation cost data for the main irrigation systems are described in Table 2.5.

The costs associated with each crop, such as salaries, seeds, leasing, herbicides, mechanization, electricity, fuels, insurance, and amortization, were obtained from the *Xarxa Comptable Agrària de Catalunya (IDESCAT, 2021)*, excluding family labor cost. The average prices during the last decade were used, while the cost of fertilizers was obtained from *Estudios de costes de explotaciones agrícolas* (MAPA, n.d.). In order to consider all costs and revenues of farmer i we included in C_{jr}, the *Common Agricultural Policy (CAP)* sub-

[11] We used the latest cost as the reference for modernized channels because the *Segarra-Garrigues Channel* was recently constructed, and prices are likely higher and not representative.

[12] The other minority channels in the cells of study are: 1. Non-modernized: Séquia D'Albesa, Pla de Corbins, Séquia de Torrelameu, Hortes de Térmens, Séquia del cup de Menàrgues i Balaguer; 2: Modernized: Llobregós, Torreblanca, Pla del Sas, C. de R. castello de Farfanya, Reg de Suport de la Noguera, Pla de Camarasa, Baldomar, Baronia de Rialb, Tiurana and Bassella.

[13] https://aiguessegarragarrigues.cat/es/precios-vigentes/

Table (2.5) WATER COST FOR EACH IRRIGATION PLAN AND SYSTEM

		Cost €			**Total cost**
Irrigation System	**WL**	*Fixed (ha)*	*Var. (m^3)*	MR	*(€/ha)*
SEGARRA-GARRIGUES (M)				Rainfed	0.0
	150mm	91.9	0.104	150mm	248.5
	350mm	102.8	0.104	350mm	468.2
	650mm	118.9	0.104	650mm	797.6
ARAGÓN Y CATALUÑA (T)	Any	27.2	0.0017	Rainfed	0.0
				900mm	42.3
CANAL D'URGELL (T)	Any	143.0	No	Rainfed	0.0
				900mm	143.0
CANAL DE PINYANA (T)	Any	92.0	No	Rainfed	0.0
				900mm	92.0
ALGERRI BALAGUER (M)	Any	40.0	0.0457	Rainfed	0.0
				150mm	108.6
				350mm	200.0
				650mm	337.5

Notes: T=Traditional Irrigation System; M=Modern Irrigation System. Var.:Variable irrigation cost depending on the amount of water used (in m^3). Fixed: Fixed water cost per hectare. MR: Management regime. WL: Water limitation. We considered *Canal de Pinyana* as a traditionally irrigated system, although also modernized irrigation is commonly carried out. Cost data is annual.

sidies. The calculation of these subsidies was based on the procedures described by *FEGA (Fondo de Garantía Agraria)*, taking into account the category of the crop (irrigated, rainfed, or permanent), the region, and the payment for the *Greening* practices. The region in our study area is almost entirely the same, so the same payment value is used for each crop. The annual payment per hectare is € 126 for rainfed wheat, barley, and corn, € 137 for vineyard, and € 320 for irrigated wheat, barley, and corn. The *Greening* payment currently represents 52% of the basic payment, and both payments are added to obtain the final CAP payment for each crop and MR combination.

2.3 Bio-economic optimization model

In our model, we consider a *regulator* who determines the optimal crop and MR for each location to maximize economic value under environmental constraints. These decisions are also subject to infrastructure and legal restrictions that may apply in a given place - such as conservation areas or irrigation technologies -.

2.3.1 Maximization problem

We divide the landscape into I equally sized areas that serve as minimal decision units. [14] In each location $i = 1, 2, \ldots, I$, the regulator chooses which crop $j = 1, 2, \ldots J$ to cultivate and which management plan $r = 1, 2, \ldots R$ to implement. The vector $x = [x_{1jr}; x_{2jr}; \ldots x_{Ijr}]$ of dimension $1 \times I$ defines the set of crop and MR selected. The set of possible and feasible choices is $X : x \in X$. This set accounts for spatial restrictions on infrastructure or regulation affecting land use choices as previously described. Note that, due to the spatial nature of the problem, the number of possible outcomes increases exponentially as the number of cells and land uses increases. Each variable $x_{ijr} \in [0, 1]$ indicates the proportion of available land [15] devoted to crop j and MR r planted in cell i. If it is not planted,

[14] In this study, they correspond to the cells or representative farmers described in section 2.2.

[15] The area included in our case study. That is, the arable land which is currently being used for the crops selected.

$x_{ijr} = 0$. Therefore, we have a set of decision variables that indicate which land uses are selected in each location. We do not impose the choice to be homogeneous inside each cell - there can be different land use choices within the same location -.[16] However, all current arable land must be planted - no land abandonment - and therefore we need to satisfy $\sum_{j=1}^{J}\sum_{r=1}^{R} x_{irj} = 1$. The economic objective function is therefore a function of x, and can be described as $EV(x) = \sum_{I=1}^{I}\sum_{j=1}^{J}\sum_{r=1}^{R} EV_{ijr}x_{ijr}$, where $EV(x)$ is a matrix of parameters including the economic value of each choice for each location. Note that the objective is to maximize the economic value at landscape level, and therefore we add up the economic value of each cell. The *Benchmark* maximization problem is described as:

$$
\begin{aligned}
&\max_{x \in X} \; EV(x) \\
&s.t. \sum_{j=1}^{J}\sum_{r=1}^{R} x_{irj} = 1 \quad \forall \quad i \\
&x_{irj} \in [0,1] \quad \forall \quad i,j,r
\end{aligned}
\tag{2.1}
$$

where the two constraints indicate the total land usage and boundaries of the decision variables, as indicated previously. Note that Equation 2.1 does not include biodiversity conservation constraints. Therefore, it could be interpreted as the decentralized solution where representative farmers individually maximize their economic outcomes. In the following section we extend this configuration with an ecological model with the aim of calculating the trade-offs between economy and biodiversity under scenarios that differ on the restrictions regarding management and biodiversity. In subsection 2.C a brief summary of the variables and parameters included in this chapter is provided.

[16] In practice, most of the cells in the optimal solutions are associated with a single land use choice. This occurs because crop yields are correlated spatially and similar choices are grouped in areas. Moreover, relaxing this assumption reduces the computational effort significantly.

2.3.2 The biodiversity habitat suitability model: indexes and constraints

After defining the economic maximization problem, we describe the habitat suitability model. In this section, we propose a method to assess the effects of agricultural land-use decisions on biodiversity. To incorporate biodiversity into the optimization problem, we introduce a new - at this stage conceptual - constraint as follows:

$$\sum_{s=1}^{S} Z_s \geq Z^* \tag{2.2}$$

where:

$$Z_s = \begin{cases} 1 & \text{if} \quad \sum_{i=1}^{I} \sum_{j=1}^{J} \sum_{r=1}^{R} B_{jrs} x_{ijr} \geq B_s^* \\ 0 & \text{otherwise} \end{cases} \tag{2.3}$$

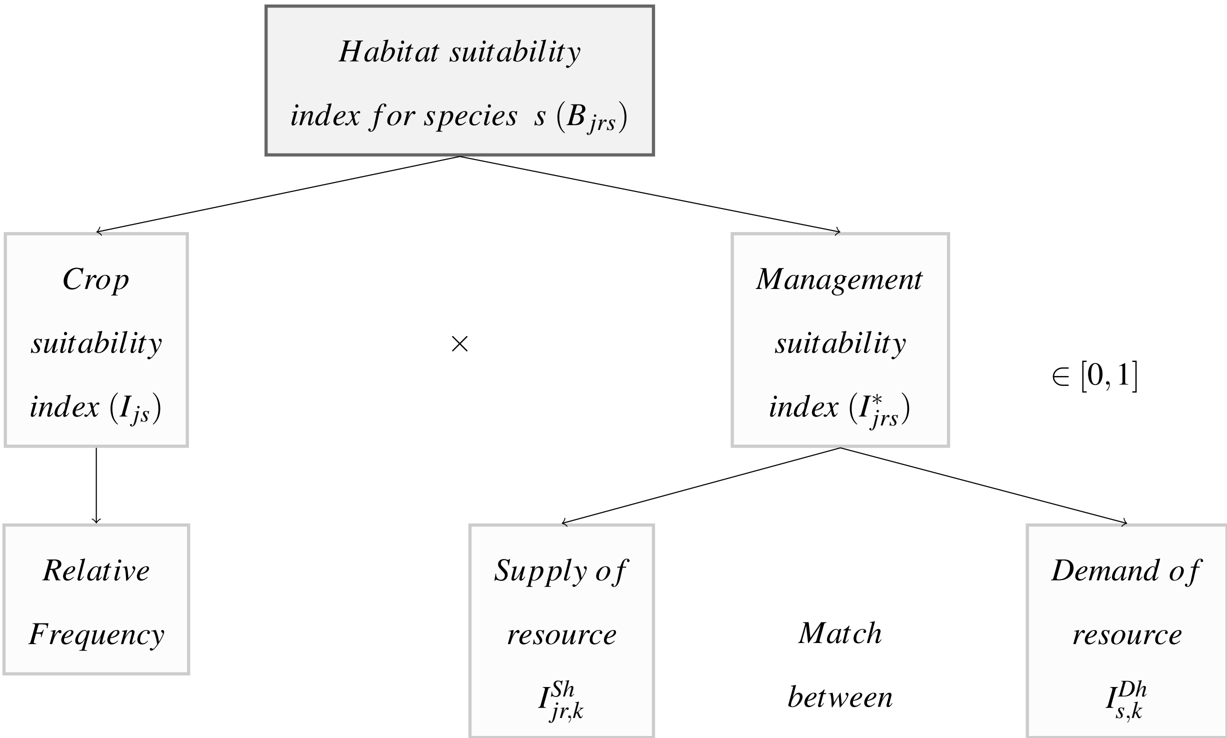
where Z_s is a binary variable that takes value 1 for species s if the sum over locations of the suitability index B_{jrs} for species s is equal to or greater than a given threshold B_s^*, (Equation 2.3). The final goal is to conserve at least Z^* species (Equation 2.2). [17] This model can be used with any number S of species.

Within this framework, we use birds as an indicator of the quality and biodiversity richness of the habitats in our study area, since they are - especially steppe birds - the most representative, emblematic and protected elements of biodiversity of this territory. Furthermore, birds are one of the most studied and monitored groups, especially at large spatial scales where their distribution has been reliably estimated as in our case (Gibbons et al., 2007). Moreover, although not all bird species react similarly since there are habitat specialists - adapted to specific habitats and therefore more susceptible to habitat modification - and generalists - which are adapted to most habitats and consequently more resilient -, they

[17] From now on, when we refer to conservation goal/level we mean the term Z^*.

are all good indicators of changes in the natural environment. Therefore, measuring specific indices that rely on bird populations is helpful to understand biodiversity changes at the community level. [18] Our two main goals when designing our biodiversity habitat suitability measure is to accurately represent the situation of the bird community and to be sensitive to the habitat, food and nesting preferences of each bird species.

Figure (2.6) DIAGRAM OF HABITAT SUITABILITY INDEX



The habitat suitability index B_{jrs} was constructed using two components. The first, I_{js} referred to crop suitability, and the second to adjust for the impacts of intensification on each crop (see Figure 2.6 for a scheme of the index components). We obtained the index of crop preference I_{js}, for each species and habitat, from the *Atles dels ocells nidificants de Catalunya 1999-2002* (Estrada et al., 2004b).[19] It was calculated as the ratio between the

[18] This index is designed focused on bird preferences and traits, although it could be adapted to other taxa.

[19] We use the breeding atlas rather than wintering birds atlas since it better reflects the influence on birds

frequency of habitat j occurrence in the 1×1 km *Universal Transverse Mercator* squares where species s was observed and the frequency of habitat j occurrence in all Mercator squares in *Catalonia (Spain)*, using this formula:

$$I_{js} = \frac{\text{Frequency of habitat } j \text{ cells where species } s \text{ was observed}}{\text{Frequency of habitat } j \text{ in the whole territory}} \tag{2.4}$$

Note that it accounts for how specialized a given species is for a given habitat. For example, a rare species usually observed in a rare habitat would lead to a very high index of preference for species s for habitat j. On the other hand, even if a given habitat is uncommon, widespread species that are observed in many habitats will have a lower preference index for that habitat. The habitat classification comprised 22 categories.[20]

However, the habitat preference index described in Equation 2.4 does not account for the limitations imposed on important ecological functions such as nesting or dietary needs by agricultural land intensification. In the previous index for example non-irrigated cereal crops would have the same impact on habitat suitability as irrigated ones. To include this we followed *Cardardor et al. (2014)*, who created a habitat preference index based on plant characteristics and dietary and nesting bird needs. Although they did it only for four emblematic steppe birds we expanded the number of species to represent the whole bird community of the area. For that we used species traits data from *Storchová & Hořák (2018)*. However broadening the number of species considered forced us to simplify the index since the species trait information was more limited.

during the period when crops are planted. Through their projects, especially *SOCC (Seguiment d'Ocells Comuns a Catalunya)*, they monitored data from bird observations since 1997 based on transects covered mainly by volunteers observing and/or hearing birds annually.

[20]These categories are: Beaches, Wetlands, Suburban, Urban, Irrigated herbaceous crops, Irrigated fruit trees, Rainfed fruit trees, Vineyards, Cereal crops, Rocky places, Alpine and subalpine meadows, Bushes and Mediterranean meadows, Scrubs and mountain meadows, Beeches and riverside forests, Oak groves, Cork oaks, Holm oaks, Pine groves or exotics, Firs and black pine groves, Black pine, Red pine, White pine .

To construct this second component of bird-crop-intensification compatibility (I^{*}_{jrs}), we define a variable $I^{h}_{jr,s,k}$ representing the degree of match between the bird species' required resource k and the ability of crop j and MR regime r to provide such resource that meets need h. We consider that birds have to satisfy two essential needs: nesting and nourishment. In our case, resources k are those that help meet dietary and nesting needs required by bird species. We represent the satisfaction level provided by resource k associated with crop j and MR regime r to meet need h of species s as the product of habitat resource k availability ($I^{Sh}_{jr,k}$) and species s demand for resource k ($I^{Dh}_{s,k}$). All these computations have been carried out for two different seasons: spring and summer. The final value has been obtained by averaging them.

Next, we present the indicator that corresponds to the dietary needs by $I^{f}_{jr,s,k}$ where $h = f$. [21] In the case of dietary needs k represents different possible sources of food such as seed availability, plant availability, invertebrate availability or vertebrate availability. Note that $I^{Sf}_{jr,k}$ is a real number that represents the supply of resource k associated with crop j and MR regime r and following *Cardador et al. (2014)* we assume that $I^{Sf}_{jr,k}$ is inversely related with the level of agricultural intensification:

$$I^{Sf}_{jr,k} = \frac{1}{1 + \Sigma^{M}_{m=1} I_m \times g_m} \tag{2.5}$$

where $m = \{1,2,3\}$ is the number of practices that negatively affect the provision of resource k. We identified three practices ($M = 3$) that can have a negative impact on each resource k: $m = 1$*: Agrochemical inputs used,* $m = 2$*: irrigation level and* $m = 3$*: plow.* I_m takes value 1 if practice m is carried out by farmers in the considered area, and zero otherwise. Further, g_m represents the intensification level of each practice. We assume that the more intensive those practices are, the lower the provision of resources k. For the two practices agrochemical in-

[21] The term f is given in reference to 'foraging'.

puts and water application, the corresponding g_m is calculated proportionally depending on the amount of agrochemical inputs and water used, with $g_m = 1$ assigned to the crop and MR with the lowest use of agrochemicals and water (reference point). For the third practice, we assume $g_3 = 1$ if plow negatively affected the provision of k and zero otherwise. Note that $I^{Sf}_{jr,k} = 1$ if crop j and MR regime r do not negatively affect the provision of resource k .

On the other hand, $I^{Df}_{s,k}$ represents species s demand for resource k. We assume $I^{Df}_{s,k} = 1$ if species s needs food type k , and 0 otherwise. Next, we match the supply of these resources with the corresponding species demand. In the case of dietary resources we do this matching following the requirements set out in *Storchová & Hořák (2018)*.[22] That is, for each k then $I^{f}_{jr,s,k} = I^{Sf}_{jr,k} \times I^{Df}_{s,k}$. Below, we obtain a unique indicator of food adequacy of crop j and MR regime r to the requirements of species s adding up the different resources k:

$$I^{f}_{jrs} = \sum_{k} I^{f}_{jr,s,k} = \sum_{k} (I^{Sf}_{jr,k} \times I^{Df}_{s,k}) \tag{2.6}$$

where I^{f}_{jrs} is the habitat suitability adjustment component with respect to the dietary needs for species s of crop j and MR r.

For the nesting requirement (now $h = n$) we differentiated between species nesting on the ground and those nesting above ground (trees, etc.). $I^{Dn}_{s,k} = 1$ if species s needs nesting type k, and 0 otherwise. On the other hand, $I^{Sn}_{jr,k} = 1$ if crop j and MR regime r do not negatively affect nesting type k , and 0 otherwise. We consider, for example, that surface irrigation more negatively affects ground-nesting bird species (Elas et al., 2023). Next, as in the case of dietary needs, we calculate an indicator of nesting suitability of species s: $I^{n}_{jrs} = I^{Sn}_{jr,k} \times I^{Dn}_{s,k}$.

Taking into account that the species need both dietary and nesting requirements, we calculate the geometric mean of these requirements to obtain a final measure of the match

[22] There are 4 categories regarding diet: *Granivores* if the species' diet is composed of seeds, *Folivore* if the species' diet is composed of plants, species whose diet is composed of vertebrates and species whose diet is composed of invertebrates.

between bird species needs and habitat provision of those needs: [23]

$$I^*_{jrs} = (\prod_{h=f,n} I^h_{jr,s})^{1/2} \tag{2.7}$$

Finally, we incorporate the index of habitat preference and define our biodiversity index for species s, crop j and MR regime r (B_{jrs}) as:

$$B_{jrs} = I^*_{jrs} \times I_{js} \tag{2.8}$$

Summarizing, it is the original index calculated by *Estrada et al., (2004b)* I_{js}, adjusted by a set of ecological requirements I^*_{jrs} based on nesting and dietary suitability, which is affected by intensification. More in-depth information is detailed in subsection 2.D.

Note that each species s is affected by crop j and MR choices r. Therefore, for each location there will be a matrix B_{jrs} of unique parameters representing suitability between species and habitats[24]. To be able to capture the biodiversity of the whole area we need to measure a biodiversity index at a landscape level. We define our biodiversity index B_s at a landscape level for species s as:

$$B_s = \sum_{i=1}^{I} \sum_{j=1}^{J} \sum_{r=1}^{R} B_{jrs} x_{ijr} \tag{2.9}$$

that gives an estimate of the situation of species s in the whole area.

A species is considered protected in a given landscape if its current situation under existing land-uses (Equation 2.3) is not worse than our species-specific threshold. We define

[23] Since some species may have zero values on the satisfaction of one of the two basic needs, we replace zero values with 0.0001 to avoid mathematical errors.

[24] We do not consider location influence, which may be related with altitude and climatic conditions, as the area considered is relatively homogeneous. However, more precise analysis should consider this.

this threshold as the status quo. Previous studies have mainly used biological thresholds that distinguish between survival and extinction of population dynamics (e.g. Polasky et al., 2008). However, these thresholds are often poorly estimated (especially for a large number of species) since they require data which is not usually available (González-Suárez et al., 2012). The status quo situation, although it may seem less relevant, allows us to use a more specific and practical indicator for conservation. We calculate each specie-specific biological threshold B_s^* by adding up the product of the habitat suitability index and the proportion of each land use jr in the status quo (x_{jr}^{SQ}). [25] as:

$$B_s^* = \sum_{i=1}^{I} \sum_{j=1}^{J} \sum_{r=1}^{R} B_{jrs} \times x_{ijr}^{SQ} \tag{2.10}$$

A species s is considered to be on the correct path towards preservation if, in the land-use scenario resulting from the optimization process, the biodiversity index at the landscape level B_s is at least equal to the status quo index, that is, if $B_s \geq B_s^*$. This approach enables us to define our biodiversity measure as the number of species that exceed a particular value (which can be computed directly from Equation 2.3), rather than by aggregating indices across species.

The species included in this study were selected based on various criteria. First, we incorporated all birds associated with this region by the ICO (Institut Català d'Ornitologia). We ensured that species protected by Special Protection Areas (SPAs) were also included. We excluded certain species that were not directly related to farmland decisions, such as those that are aquatic or mountain specialists.[26]

[25] We used a digital map from *Declaración Única Agraria (DUN, 2019)* to account for the proportion of each crop j and management r. This map included the spatial location, size and crop planted for each parcel in Catalonia.

[26] The removed species were Aquila chrysaetos, Ardeola ralloides, Bubo bubo, Dryocopus martius, Emberiza citrinella, Falco peregrinus, Gypaetus barbatus, Gyps fulvus, Lanius collurio, Milvus migrans, Perdix perdix, Pernis apivorus, Prunella modularis, Pyrrhocorax pyrrhocorax, and Tetrao urogallus

Our study focused primarily on farmland, shrubland, and steppe birds, comprising a set of 83 species with specific habitat preferences. We conducted our analysis by differentiating between subsets of species: Generalist species, Steppe species, and Farmland species. Generalist species are those adapted to at least two different habitats. Within the Steppe species, we further distinguish between Steppe and Steppe specialist species. The former represents all species associated with steppes, but can also be found in other habitats. The latter includes only four species that are exclusively associated with steppes or pseudo-steppes.[27] We also examined how our results would change if we considered specific threatened bird species listed as Vulnerable, Endangered, Critically Endangered, or Near Threatened in Catalonia (SIOC, 2012), based on IUCN criteria. We used the latest available information, which dates back to 2012. All of the species considered can be found in subsection 2.E. The results associated with the issue of target conservation strategies are included and explained in subsection 2.4.5. Until that section, the analysis is primarily carried out for the whole bird community (83 species).

2.3.3 Scenarios, Pareto Frontier and solution methods

We propose different scenarios regarding restrictions on intensification or biodiversity constraints (Table 2.6). Consequently, the scenario varying features are: (a) Intensification constraints (as shown in Figure 2.4); (b) Biodiversity targets (i.e., number of species or subset of species). The intensification constraints - understood as locations where certain practices are limited - are analyzed considering three cases: (a.1) Conservation areas are placed as nowadays (benchmark). (a.2) Reallocation of conservation areas but maintaining total area intact (a.3) Complete removal of conservation areas. The biodiversity targets are differentiated by the number of species aimed to be protected Z^*, which ranges from 0 to S.

[27] Steppe specialists are also included in the Steppe category. Farmland species correspond to those associated with agricultural environments, although not necessarily to steppes or pseudo-steppes. This classification is based on the categories included in Estrada et al. (2004b).

Table (2.6) SCENARIOS CONSIDERED

Conservation areas assumption	Z^* 0	1	2	3	...	S
(a.1) Benchmark (as nowadays)	→	→	→	→	→	→
(a.2) Optimal conservation areas allocation	→	→	→	→	→	→
(a.3) Non conservation areas	→	→	→	→	→	→

Notes: Z^* is the number of species aimed to be protected. S is the total number of species considered. $Z^* = S$ implies that under this scenario all the species must be protected. Arrows are intended to represent from less to more ambitious conservation goals.

For each assumption regarding conservation areas we solve the problem for each Z^*, and with that we construct a *Pareto Frontier*, which represents the maximum economic value obtained under each Z^*. We will focus firstly and more deeply on the scenario where the conservation areas are assumed to remain as nowadays. We denote this scenario (Equation 2.11) - which is obtained by combining Equation 2.1 and Equation 2.2 - as *Benchmark* scenario:

$$\begin{aligned} \max_{x \in X} EV(x) \\ s.t. \sum_{j=1}^{J} \sum_{r=1}^{R} x_{irj} = 1 \quad \forall \quad i \\ x_{ijr} \in [0,1] \quad \forall \quad i,j,r \\ \sum_{s=1}^{S} Z_s \geq Z^* \end{aligned} \tag{2.11}$$

The other two assumptions about the conservation areas (a.2 & a.3) will be considered as an extension.

The solution to each problem of land-use optimization described is complex because it involves not only determining what actions to take and at what scale but also where to allocate each land-use. The optimization process becomes exponentially more complex with the addition of new locations or decision variables and even more so when non-linear objectives or constraints are added. As a result, a wide variety of optimization algorithms have been developed to address this issue, including heuristic algorithms based on genetic concepts and simulated annealing as well as deterministic algorithms such as the simplex method. In recent years, optimization solvers have gained attention from research communities due to their ability to handle large numbers of constraints (Anand et al., 2017).

After defining the theoretical framework, we model it using a specific language. Once the decision variables and objective functions have been identified, the problem is solved using a specific algorithmic process that produces optimal solutions. We use Gurobi from GuRoBi Optimization Inc., which generates optimal solutions and enables optimal decision-making through a fast and powerful mathematical optimization solver. Gurobi is currently used to tackle complex real-world problems and can solve all major problem types (both convex and non-convex), including linear programming, mixed-integer linear programming, quadratic programming, etc. Our code has been implemented using the Python API.

We run the optimization process by using the concurrent solver, implying that a set of algorithms are applied simultaneously and the fastest to obtain a solution is chosen, mixing between deterministic and heuristic methods. The optimization time analysis and algorithm selection is beyond the scope of our work; therefore, we use this method and consider the fastest solution provided. Nevertheless, we run the optimization process for different time limits to confirm that optimal or near optimal solutions are found.

2.4 Results

2.4.1 Simulated potential crop yields

Our simulations exhibit a positive correlation between crop yield and intensification, which is consistent with the expectations (Table 2.7). Corn is found to be the most productive crop, with an average annual yield of over 19 ton/ha for high rate irrigated fields. In contrast, barley and wheat exhibit lower yields of 7.58 ton/ha and 9.11 ton/ha, respectively. However, corn yield drops to 2.6 ton/ha in the absence of irrigation, while the yield reduction in cereals is not as significant. This results in yield multipliers - defined as the number of times increase in crop yield achieved using the most effective yield-enhancing technique - of 7.6, 3.3, and 5.1 for corn, barley, and wheat, respectively.

Table (2.7) STATISTIC OF CROP YIELDS FOR DIFFERENT COMBINATIONS OF CROP AND MANAGEMENT LAND USES

Land use		**Statistics**			
Crop	**Management**	**Mean**	**SD**	**Min.**	**Max.**
Barley	Rainfed	2.30	1.29	0.44	6.46
	150mm	6.49	0.88	4.36	8.47
	350mm	7.58	1.17	4.97	10.12
	Traditional	4.53	2.40	1.50	9.50
Corn	Rainfed	2.60	2.60	0.00	7.89
	150mm	6.32	2.24	0.20	12.79
	350mm	13.92	2.10	9.70	18.72
	650mm	19.70	0.90	16.72	23.23
	Traditional	19.29	0.88	16.42	20.83
Wheat	Rainfed	1.78	1.17	0.62	5.50
	150mm	5.80	0.61	4.91	9.36
	350mm	9.11	0.65	5.74	11.23
	Traditional	5.74	1.39	3.71	10.72
Vineyard	Rainfed	4.48	0.37	3.36	5.36
	Irrigated	6.52	3.40	4.20	15.75

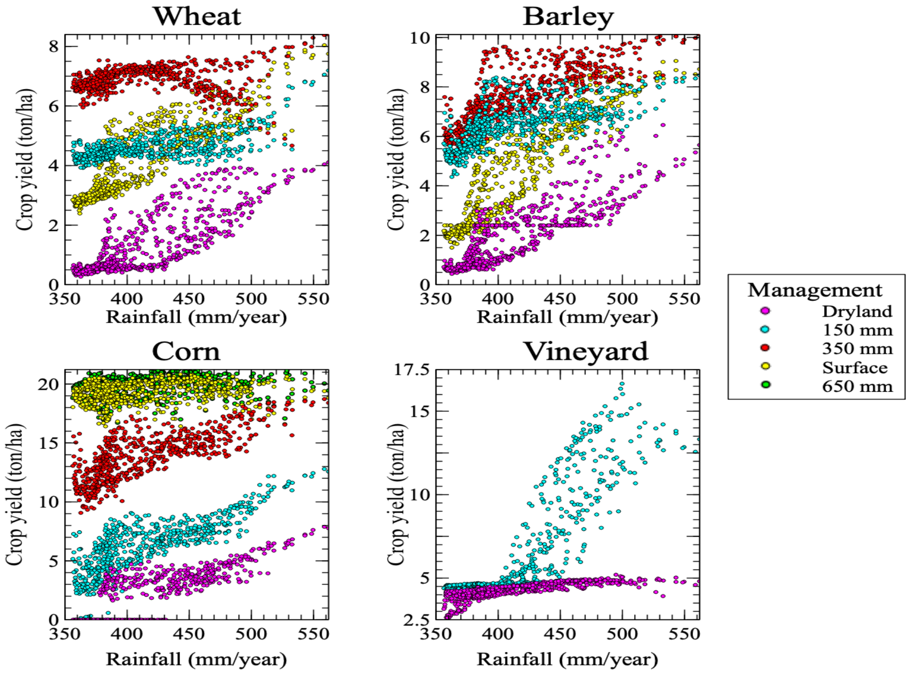
Notes: Mean unit: Annual *ton/ha*. SD=Standard deviation. Min.=minimum value. Max.=maximum value. mm=millimeters of water per year.

Our findings suggest that corn is more sensitive to irrigation constraints than winter cereals (barley and wheat), and thus, conservation areas may negatively impact aggregate production, given that corn is the most productive crop under study. Conversely, vineyards exhibit smaller yield variations between irrigated and non-irrigated, with a yield multiplier of 1.44. Also, note that for winter cereals and corn, traditional irrigation - although using higher water quantities - produces less than modern irrigation - 650*mm* in the case of corn and 150*mm*/350*mm* in the case of barley and wheat -. These results indicate that modern irrigation techniques could lead to more efficient water use and be a viable option for land use planning. For potential crop yield distributions - by *MR* and crop - see Figure A1.

Regarding the spatial analysis, our findings reveal that there are significant differences in soil properties across different locations for all the variables studied, while climate variables show higher spatial correlation (subsection 2.H). In general, central and western regions exhibit higher maximum temperatures, evapotranspiration, and global radiation compared to the eastern and north-eastern areas, where precipitation levels are higher. Notably, precipitation levels are strongly and positively correlated with crop yields, particularly for crops that are less irrigated (Figure 2.7). Moreover, we find that the relationship between crop yield and precipitation for each agro-climatic region is concave, implying decreasing marginal effects. [28] This suggests that the specific location, especially in the absence of access to irrigation due to infrastructure, legal, or other constraints, plays a critical role in determining crop production. On the other hand, intensification through irrigation could act as a homogenizing factor, reducing location-specific effects on crop yields.

[28]For vineyards, there seems to be a threshold below which a minimum amount of water is required to ensure productivity.

Figure (2.7) RELATIONSHIP BETWEEN PRECIPITATION LEVELS AND CROP YIELD FOR DIFFERENT MANAGEMENT TYPES AND CROPS



Notes: Each dot represents one cell. Each dot color represents a specific MR.

Additionally, our results reveal that, while corn yields show a ceiling constraint with both surface and 650*mm* irrigation, traditionally irrigated barley and wheat benefit more from precipitation than the corresponding crops under modern irrigation techniques (150*mm* 350*mm*). This could be because sprinkler irrigation is more uniformly distributed over time than surface irrigation, and therefore higher precipitation levels may help satisfy the crops' water requirements during periods when water is not applied, thereby closing the yield gap between the two irrigation methods. This finding highlights the importance of considering irrigation methods and precipitation patterns in optimizing crop production.

Our previously presented results are consistently higher than the yields currently reported by the *Encuesta sobre Superficies y Rendimientos Cultivos* (ESYRCE) (MAPA, n.d.). This is because STICS does not simulate the reduction in yields due to biotic effects (weeds, pests, diseases, etc.). In addition, variations in management practice skills among farmers are not taken into consideration. Therefore, our results should be viewed as the maximum potential yields achievable under ideal conditions. It would be ideal to validate our results by comparing them with observed data. However, the ESYRCE data are aggregated at a coarser level and based on a limited number of surveys, making it difficult to perform a proper comparison. Nevertheless, based on expertise and considering the yield gap (i.e., the difference between potential and actual yield), our simulated values closely align with the actual results for all crops and management practices.

2.4.2 Habitat suitability indexes

We present here (Table 2.8) the landscape suitability index values obtained by applying Equation 2.8 to the different species. For simplicity, we report aggregate data by species subgroups. Overall, if we consider all species, their preferences are spread among different land uses, although with a higher average preference for barley and wheat. Within each group, intensification negatively affects bird preferences, with birds showing a clear preference for rainfed land. [29] If we focus on species associated with steppes (*Steppe*), their preferences lean clearly towards winter cereals rather than corn or vineyards. This level of specialization is even more evident if we consider *Steppe specialists*, who exhibit similar preferences for winter cereals, but with lower adaptation to corn or vineyard-dominated habitats (e.g., for rainfed corn, 0.22 as opposed to 0.44 for *Steppe* or 0.62 for *All species*). *Farmland* species show similar results to the *All species* group, but with a higher adaptation to barley and wheat.

[29] This derives from Equation 2.5.

Table (2.8) AVERAGE HABITAT SUITABILITY INDEX BY GROUP OF BIRD SPECIES

		Barley & Wheat				Corn					Vineyards	
Group	*S*	0	150	350	Trad.	0	150	350	650	Trad.	0	Irr.
All species	83	0.71	0.57	0.55	0.38	0.62	0.55	0.52	0.48	0.36	0.65	0.52
Steppe	11	0.87	0.62	0.59	0.24	0.44	0.33	0.31	0.28	0.09	0.34	0.27
Steppe specialists	4	0.82	0.51	0.48	0.01	0.22	0.15	0.14	0.12	0.00	0.26	0.21
Farmland	37	0.81	0.62	0.59	0.33	0.6	0.51	0.49	0.45	0.3	0.65	0.52

Notes: Values have been normalized for all species to be comparable. We assign for each species the value 1 for the preferred habitat, and the rest are computed in relative terms. Values range between 0 and 1. Trad.=Surface irrigation. Irr.=Irrigated. *S* is the number of species in each group.

In general, this table provides a useful overview of habitat suitability patterns of bird species in different crop types and irrigation systems. However, it is important to note that these values are only an approximation and habitat suitability may vary depending on additional factors such as geographic location and time of year.

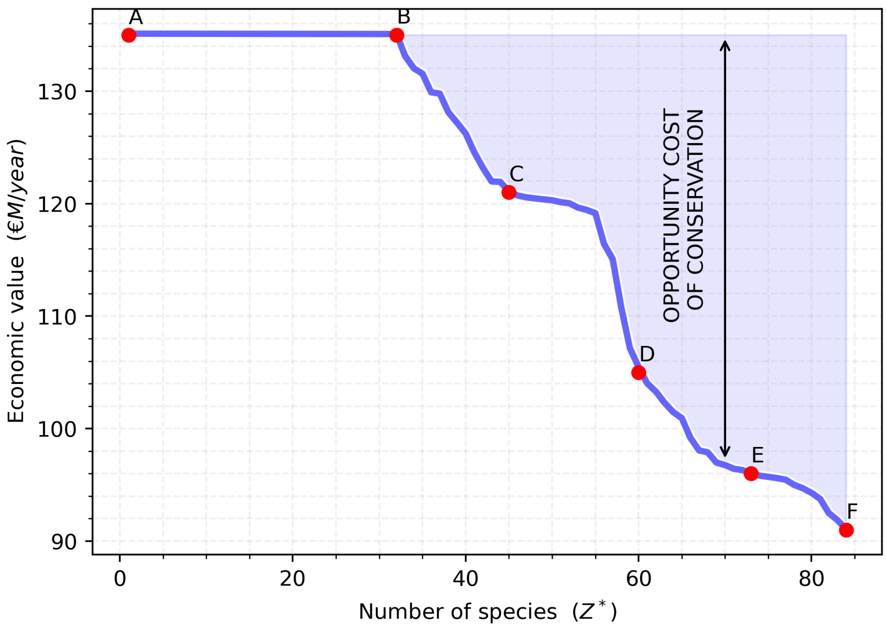
2.4.3 Pareto frontier

We have solved the optimization problem formulated in Equation 2.1 subject to the constraints presented in Equation 2.2 for multiple levels of Z^* using the algorithm outlined in subsection 2.3.3. Specifically, for each cell i, we have identified the crop j and MR r that maximize the total economic value while maintaining a given number of sustained species, Z^*, subject to habitat suitability constraints and the characteristics of the case study. The set of feasible choices, denoted as $X : x \in X$, is determined by the intensification constraints at each location, which we assume are pre-defined and remain constant. Furthermore, we have established ecological constraints at the landscape level, requiring the preservation of a specific number of species in the region, without necessitating their presence in individual cells. This approach ensures that each species is conserved in the areas where the opportunity cost of conservation is the lowest. This principle guides the trade-offs between economic yield and biodiversity preservation, and the Pareto frontier illustrating these trade-offs is presented in Figure 2.8. Data from specific points along the Pareto frontier are presented in Table 2.9.

Each point on the *Pareto* frontier corresponds to a particular conservation status, which is defined as the number of species that can be sustained at the landscape level. When no species preservation is required ($Z^* = 0$), the maximum economic value achievable in the area is € 135.1 M, while preserving all 83 species reduces the value to € 90.7 M. This implies that enforcing all species preservation constraints incurs an annual implicit cost of € 44.3 M for farmers, which represents the opportunity cost of conserving these species in the region.

The negative slope of the *Pareto* frontier indicates a trade-off between species conservation and economic value, implying that biodiversity conservation imposes costs on agricultural production. However, some species do not create trade-offs, indicating that they can easily adapt to the conditions of the maximizing economic value scenario. This can be visually observed in Figure 2.8, where economic value remains constant for the first 31 species (from point A to B), resulting in zero conservation opportunity cost. Beyond point B, sustaining additional species requires a modification in land use with respect to the scenarios associated with $Z^* \leq 31$, which incurs a conservation cost.

Figure (2.8) PARETO FRONTIER



Notes: The Pareto curves depict the relationship between annual biodiversity and economic values (€M). Each point on the curve represents the maximum economic value achievable under a given biodiversity constraint, as indicated on the x-axis. The letters on the curve denote specific points along the Pareto frontier.

Table (2.9) HIGHLIGHTED SOLUTIONS ON THE PARETO FRONTIER: DATA AND COMPARISON WITH THE STATUS QUO

		Values		**%**		**Δ%EV**	
	Point	**EV (€M)**	Z^*	**EV (€M)**	Z^*	$100 \times [\frac{EV(x^*)}{EV(x^{SQ})} - 1]$	
Highlighted solutions :						**M.1**	**M.2**
	A	135.1	0	100	14.5	94.9	91.2
	B	135.1	31	100	14.5	91.9	88.2
	C	120.6	45	89.3	54.2	74.1	70.8
	D	105.0	59	77.8	71.1	51.6	48.7
	E	95.7	72	70.9	86.7	38.2	35.5
	F	90.7	83	67.2	100	31.0	28.4
Status Quo:							
	M.1	69.3	83	51.3	100	-	-
	M.2	70.6	83	52.3	100	-	-

Notes: Maximum economic value for each level of biodiversity constraint. In the second and third columns, EV and Z^* respectively represent the total economic value and level of biodiversity constraint at the corresponding point of the Pareto Frontier. In the fourth and fifth columns, EV and Z^* respectively represent their percentage reduction with respect to their corresponding maximum values. In the last column, $\Delta\%EV$ represents the percent increase in EV with respect to the status quo. $EV(x^{SQ})$: Economic value under the status quo land configuration. $EV(x^*)$: Economic value under the optimal land configuration.

The slope of the *Pareto* frontier reflects how changes in conservation goals affect agricultural economic value. Although the function is quasi-linear, it is not strictly concave, which suggests that there are points along the curve where the opportunity cost of conserving an additional species in terms of economic value does not necessarily increase at an increasing rate. Instead, it may increase at a constant or decreasing rate due to differences in the preferences of bird species and the objective being related to the total number of species. As a result, a marginal increase in Z^* may be associated with a different species target objective with different preferences than the previous set of birds being conserved, which leads to changes in the land use pattern. For example, to increase the number of protected species from Z^* to $Z^* + 1$, it could be optimal to protect two not previously conserved species with similar landscape preferences - and remove the protection from a previously protected one with different preferences - rather than preserving only one new species and still maintaining the conservation of the previously protected one. In both cases the number of protected species will be $Z^* + 1$, but can lead to points where areas devoted to specific land uses change use abruptly although maintaining the general trend. Therefore, our methodology enables the identification of conservation thresholds on the Pareto frontier, delineating regions where the conservation of an additional species would result in a notable decline in agricultural economic value. This phenomenon is evident in the Pareto curve, especially between points B and C, where the curve exhibits a steep drop and significant negative marginal effects of Z^* on EV.

To compare our optimization results with the status quo, we used data from DUN (DARP, 2019) to estimate the current economic value of the study area. This dataset provides information on the crops planted in each plot during 2019, as well as whether or not they were irrigated. We estimated the economic value of each crop using simulated yields from STICS, as observed data at the municipality and province level may not provide a good approximation of the overall economic value in the selected cells. Additionally, using simulated data allows for a direct comparison with the Pareto frontier points, thereby eliminating possible methodological biases. However, due to the lack of information about the level of intensification for each plot, we developed two measures (M.1 and M.2) based on independent assumptions. For M.1, we assumed that the level of intensification for each plot corresponded to the maximum irrigation amount allowed in the cell containing that plot. For M.2, we assumed that the level of intensification applied was the one that maximized simulated economic value. Both measures produced similar results, with M.1 and M.2 yielding economic values of 69.3 €M and 70.6 €M, respectively, (see Table 2.9 for the conservation and economic values of the Pareto frontier).

Henceforth, while preserving all conservation objectives ($Z^* = 83$), the optimization of land utilization to allocate them in areas where they are most productive may result in an annual increase of € 21.4 M [30] (+31.0%) in potential economic value. This finding leads to two conclusions: firstly, there exists significant inefficiency in decisions regarding agricultural land use; and secondly, an increase in economic value does not necessarily imply a reduction in conservation objectives. This difference would rise to € 65.8 M (+94.9 %) in case of $Z^* = 0$.

In Table 2.10 we show the number of farmers who have transitioned from one land use to another by comparing their status quo choices with the optimal solution. Note that in both the status quo and optimal solution, different choices can be simultaneously selected for the same cell. To make this manageable, we consider the option representing the largest size within each cell. For simplicity's sake, we have chosen two opposing conservation goals: $Z^* = 0$ and $Z^* = 83$. This allows us to determine which options must be altered in order to transition from the current configuration to the optimal one if conservation goals are either non-existent or high.

The results indicate that if $Z^* = 0$, most choices result in planting highly irrigated corn - either traditional or modern, depending on location. Barley appears to be particularly favored for low irrigation regimes as it performs well under these conditions. Additionally, most cells where rainfed barley is currently planted remain unchanged (87%). Wheat is

[30] Utilizing the M.1 definition.

also selected but primarily associated with moderate intensification regimes and areas with current restrictions allowing for some irrigation - performing better than barley - but still constrained - avoiding planting highly irrigated corn. Under this conservation goal, vineyards do not appear to be a preferred option in any cell.

Table (2.10) LAND USE TRANSITIONS

Panel A: $Z^* = 0$

	B0	B150	B350	BT	C0	C150	C350	C650	CT	W0	W150	W350	WT	V0	V Irr.
B0	120	-	-	-	-	-	-	-	-	17	-	-	-	-	-
B150	-	63	-	-	-	-	-	-	-	-	11	-	-	-	-
B350	-	-	-	-	-	-	1	140	-	-	-	16	-	-	-
BT	-	-	-	-	-	-	-	-	59	-	-	-	-	-	-
C0	6	-	-	-	-	-	-	-	-	-	-	-	-	-	-
C150	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
C350	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-
C650	-	-	-	-	-	-	-	7	-	-	-	-	-	-	-
CT	-	-	-	-	-	-	-	-	87	-	-	-	-	-	-
W0	16	-	-	-	-	-	-	-	-	9	-	-	-	-	-
W150	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W350	-	-	-	-	-	-	-	9	-	-	-	-	-	-	-
WT	-	-	-	-	-	-	-	-	25	-	-	-	-	-	-
V0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
V Irr.	-	-	-	-	-	-	-	3	11	-	-	-	-	-	-

Panel B: $Z^* = 83$

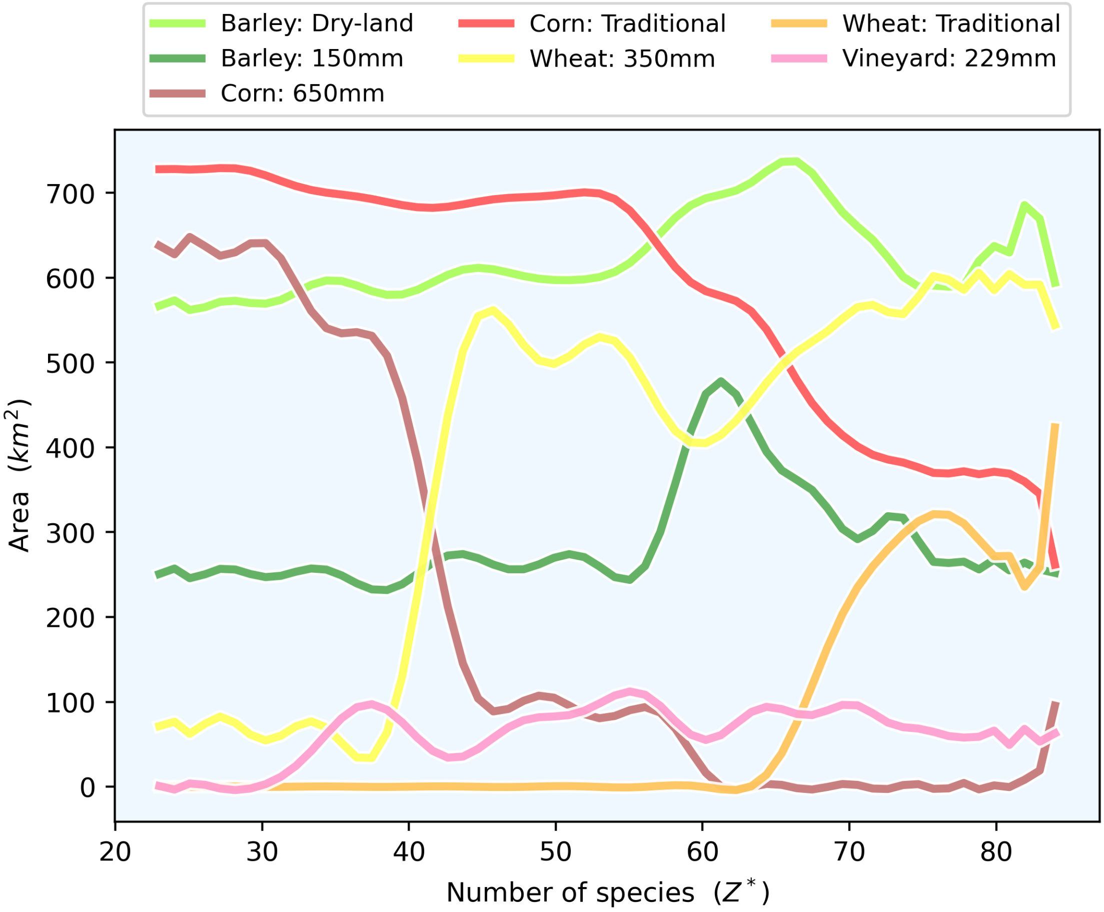
	B0	B150	B350	BT	C0	C150	C350	C650	CT	W0	W150	W350	WT	V0	V Irr.
B0	116	-	-	-	-	-	-	-	-	17	-	-	-	4	-
B150	-	63	-	-	-	-	-	-	-	-	11	-	-	-	-
B350	-	-	2	-	-	-	-	23	-	-	-	118	-	-	14
BT	3	-	-	-	-	-	-	-	9	-	-	-	47	-	-
C0	6	-	-	-	-	-	-	-	-	-	-	-	-	-	-
C150	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
C350	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-
C650	-	-	-	-	-	-	-	-	-	-	-	6	-	-	1
CT	7	-	-	-	-	-	-	-	39	-	-	-	41	-	-
W0	16	-	-	-	-	-	-	-	-	9	-	-	-	-	-
W150	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W350	-	-	-	-	-	-	-	1	-	-	-	7	-	-	1
WT	1	-	-	-	-	-	-	-	16	-	-	-	8	-	-
V0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
V Irr.	-	-	-	-	-	-	-	-	1	-	-	3	1-	-	-

Notes: Values represent the number of representative farmers who have changed from a given land use to another. B=Barley, C=Corn; W=Wheat; V=Vineyards. Numbers following crop abbreviations indicate irrigation regime in *mm*. T=Traditional irrigation. Irr. = Irrigated vineyard. Land use labels located in the rows indicate land use in origin. Land use labels located in columns indicate ending land use.

If the conservation goal is $Z^* = 83$, the scenario changes significantly. The resulting land uses through optimization are more evenly distributed due to bird habitat requirements. Barley still appears to be preferentially selected under low irrigation regimes; however, this more diverse habitat through territories devoted to highly irrigated wheat or vineyards is mainly at the expense of corn selection.

The previous comparison allowed us comparing the optimal solutions with respect to the status quo. Now, we examine the manner in which the level of biodiversity constraint influences land use decisions and landscape configuration (Figure 2.9) from less to more ambitious conservation goals. For low conservation objectives (i.e., low Z^*), irrigated corn (either 650*mm* or traditional) emerges as the dominant choice due to its general profitability (Table 2.7). A critical observation is that modern and traditional irrigation methods are mutually exclusive within a given location, as they rely on distinct types of infrastructure (or lack thereof). Consequently, 650*mm* corn prevails in areas equipped with modern irrigation infrastructure, while surface irrigated corn dominates in those possessing traditional irrigation infrastructure. Nevertheless, there are considerable regions where non-irrigated or low-irrigated (150mm) barley crops are selected in the optimal solution. Barley is the most profitable crop under low irrigation regimes, which are prevalent in numerous cells of our case study. Henceforth, it is the chosen option for these cells. The same reasoning applies to barley with 150mm management. Wheat, on the other hand, appears to be more profitable under higher irrigation schemes; thus 350mm is a preferred option in certain locations. Other choices - such as 350mm corn or non-irrigated wheat - possess less relevance and hence are not displayed but still planted when conservation objectives are low.

Figure (2.9) AREA ALLOCATED TO EACH LAND USE ACCORDING TO THE BIODIVERSITY REQUIREMENT



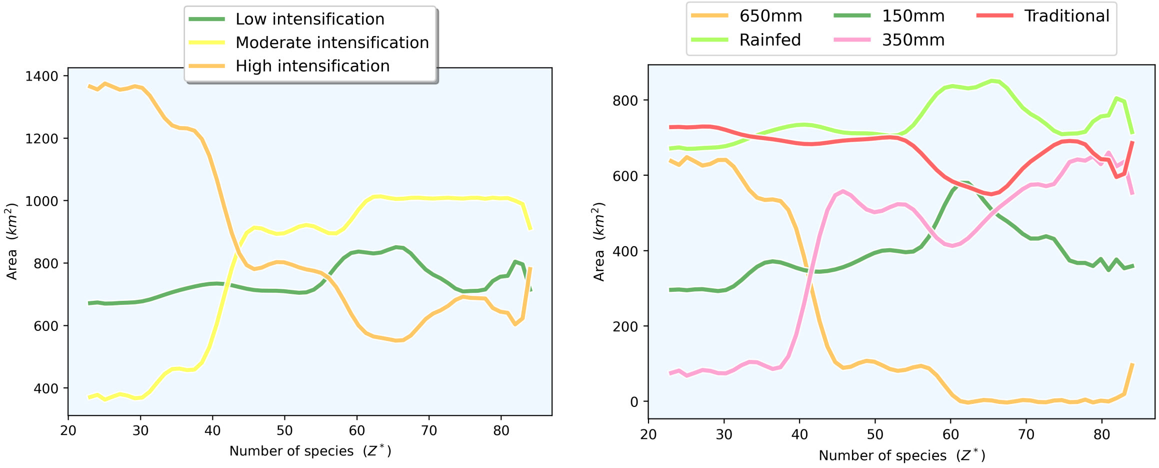
Notes: Area (in km^2) allocated to each land use according to the biodiversity requirement (x-axis). For visualization purposes, some land uses with negligible area are not shown. The horizontal axis is truncated at $Z^* < 20$ because y values do not change.

When $Z^* \geq 31$, economic and conservation objectives begin to conflict, resulting in changes to land use patterns and total economic value - as depicted in Figure 2.8. The most significant shift is observed with the replacement of 650mm corn with 350mm wheat (approximately at the point $Z^* = 40$ in Figure 2.9). While other land uses remain relatively stable, the optimal means of protecting additional species at this level appears to be a switch from corn to winter cereals - in this case wheat, as it performs better under high irrigation schemes than barley. Additionally, it can be observed that 650mm corn is also offset by an increase in irrigated vineyards. This reveals that although the latter are systematically less profitable than corn - without considering added value - the initial species to be conserved will benefit from this new crop. It should also be noted that traditionally irrigated corn is not affected at this stage, indicating that the cost of crop replacement to satisfy conservation goals is higher in non-modernized irrigation systems than in modernized ones and therefore switched earlier. When $Z^* \geq 50$, non-modernized areas previously dominated by corn crops are replaced. This change is primarily explained by a gradual increase in rainfed barley but also by an increase in surface irrigated wheat devoted area. [31]

Although general land patterns are clearly identified, marginal changes are not monotonic along the x-axis (i.e. Z^*). This means that for specific levels of biodiversity constraints, some increasing (or decreasing) land use trends must be reversed to maximize the economic objective. As mentioned, these non-monotonic changes arise from the complex variety of habitat preferences among species and the model formulation (i.e. the habitat suitability model equations). These trends can be better comprehended by categorizing land uses into distinct groups (Figure 2.10 & Figure 2.11). It becomes evident that all previously explained processes culminate in a situation where landscapes dominated by crops utilizing high irrigation schemes (650mm & traditional) are supplanted by those with moderate intensification schemes (150mm & 350mm). Rainfed lands increase in size but remain relatively stable (Figure 2.10a). More specifically, rainfed territories experience a slight increase as they substitute surface irrigated crops, while 650mm corn is primarily replaced with 350mm crops (Figure 2.10b).

[31] These changes are not always monotonically associated with Z^*. For instance, when $Z^* \in (55, 65)$ the trends of 150mm barley and 350mm wheat are temporarily interrupted. When preservation objectives are high ($Z^* \geq 70$), trends stabilize and a landscape dominated by non-irrigated barley and 350mm wheat can be observed. Additionally, a significant proportion is devoted to traditionally irrigated wheat and corn.

Figure (2.10) RELATIONSHIP BETWEEN CONSERVATION GOALS AND INTENSIFICATION



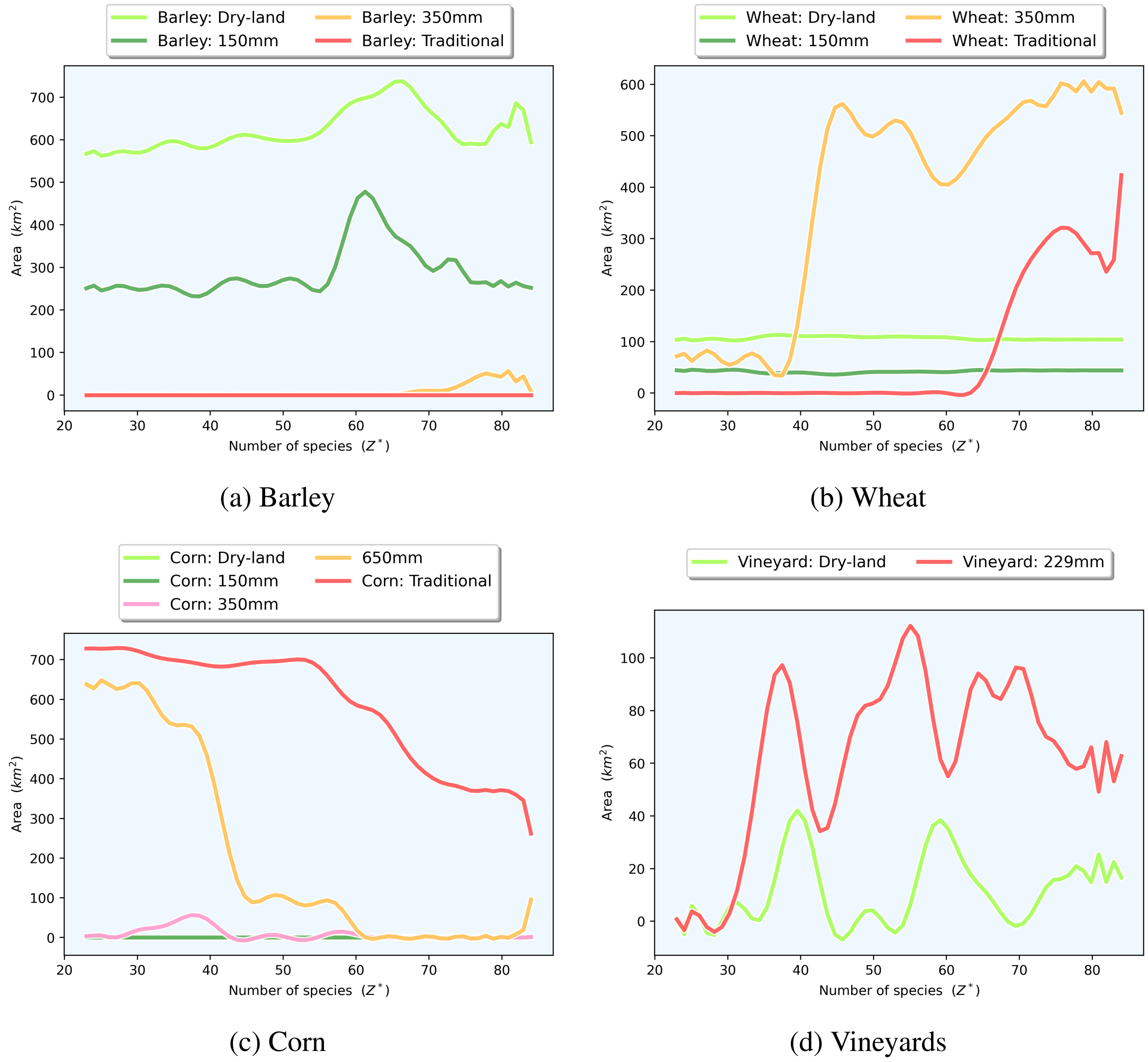
(a) Low/Mid/High intensification

(b) 0/150/350/650/900 mm

Notes: Low intensification includes rainfed lands. Moderate intensification includes 150mm & 350mm irrigation schemes. High intensification includes 650mm & traditional (900m) irrigation schemes.

We also examine these findings from the standpoint of crop selection (Figure 2.11). While some results are derived directly from previous graphs, this facilitates comparison of how farmers may choose different crops beyond intensification levels under varying conservation objectives. As noted, the expansion of barley crops remains relatively constant. Only when conservation objectives are high ($Z^* \geq 70$) are 350mm schemes implemented - albeit not very significantly. Furthermore, it is not anticipated that traditionally irrigated barley will be planted since wheat or corn are expected to be more profitable. As a result, barley is primarily allocated to 150mm schemes and particularly rainfed lands where it is found to be most profitable (Figure 2.11a). With regard to wheat, it appears to be the most competitive alternative to irrigated corn when biodiversity objectives are expected to be met; first in modernized regions and later in traditionally irrigated lands (Figure 2.11b). Corn farmlands experience the opposite effect. Additionally, we can observe how neither non-irrigated nor 150mm corn is planted at any point on the Pareto frontier while corn at 350mm is scarcely considered and only occurs when biodiversity goals are low. This happens because although 350mm corn may still be more profitable in some locations than 350mm wheat, more diverse species preservation requirements will compel them to change from 650mm corn to 350mm wheat - even more evident considering that in some places irrigation is limited to a water volume of 350 mm (refer to Figure 2.11c). The case of vineyards is less significant since their planting is primarily motivated by biodiversity requirements rather than economic interest. When biodiversity goals are low, private interest in planting is negligible and the size of the area allocated for such crops approaches zero. However, as biodiversity goals increase, so does the private interest in planting and the size of the area allocated for such crops. This is because vineyards are considered to be a crop that contributes positively to biodiversity conservation of some species.

Figure (2.11) RELATIONSHIP BETWEEN CONSERVATION GOALS AND LAND USE CHOICES BY CROP



(a) Barley

(b) Wheat

(c) Corn

(d) Vineyards

The same solution can be approached from a spatial perspective, which is useful for decision-making as it illustrates which territories will be impacted and to what extent. We map the optimal spatial solutions under four different biodiversity goals ($Z^* \leq 31, \quad Z^* = 40, \quad Z^* = 60, \quad Z^* = 80$) in Figure 2.12.[32]

By comparing Figure 2.12c and Figure 2.12d, we can see that as the conservation constraints increase, some modern irrigated territories located in the east within the *Segarra-Garrigues* irrigation area shift from corn to irrigated wheat. However, surface-irrigated corn in the central plains remains unchanged under this conservation goal. This is because the most productive alternative to surface-irrigated corn is surface-irrigated barley or wheat. Since the relative crop yield loss from modern irrigation is smaller than traditional irrigation[33], when conservation goals are higher - implying coexistence of different crop types - it is more efficient to first shift from corn to other crops within irrigation-modernized territories[34].

When the majority of species must be preserved (Figure 2.12a & Figure 2.12b), the area occupied by corn is significantly reduced in a landscape dominated by wheat and barley, both irrigated and non-irrigated. However, corn still constitutes a considerable area within the region. The eastern region, corresponding to the *Segarra* county, remains unchanged by biodiversity goals due to the lack of irrigation infrastructure availability. This results in the planting of barley, which is more profitable than wheat - and even more so when compared to corn and vineyards - under *no-irrigation* conditions. In practice, since this difference is not very large - although consistently higher between locations - many farmers may alternate or choose between these crops without obvious preferences. Additionally, if we examine Figure 2.4 carefully, we can see how conservation areas shape landscapes through irrigation constraints by encouraging not only different management types but also different crops. As a result, it seems clear that any scenario designed to conserve most species should comprise

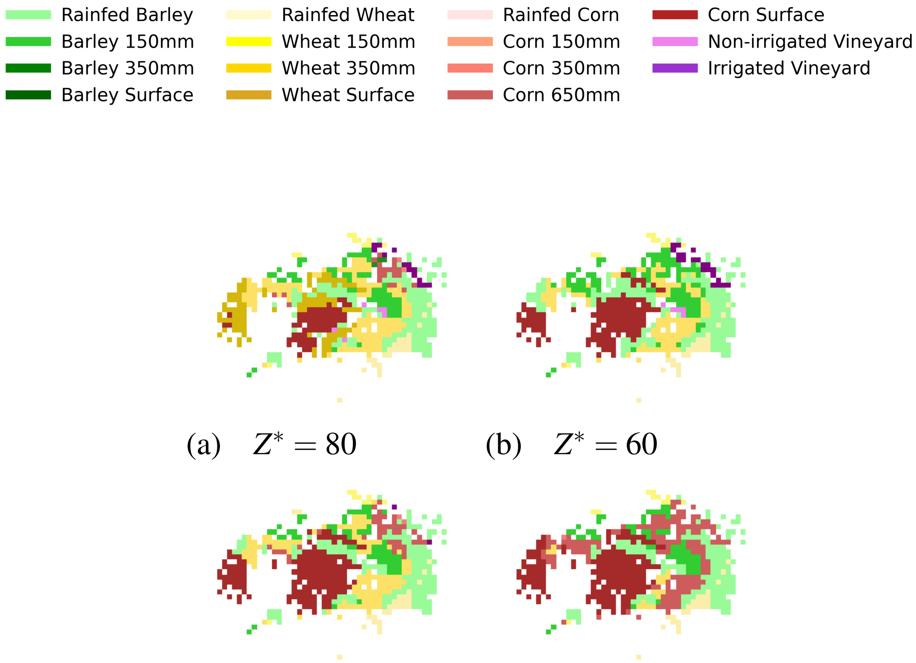
[32] $Z^* \leq 31$ is selected because the solution does not vary when fewer than 32 species are protected.

[33] See Table 2.7.

[34] The alternatives proposed are a simplification of reality. In practice, a farmer may decide to reduce water supply in non-modernized areas rather than shift to non-irrigated crops or other crops. Constrained by computational costs, this approach allows us to track general trends.

diverse habitats that satisfy all species' requirements.

Figure (2.12) LAND USE SPATIAL PATTERN UNDER DIFFERENT BIODIVERSITY GOALS



(a) $Z^* = 80$ (b) $Z^* = 60$

(c) $Z^* = 40$ (d) $Z^* \leq 31$

Notes: Each color represents a different land use (combination of crop and management regime). Legend on the top.

2.4.4 Optimization analysis with modifiable conservation areas

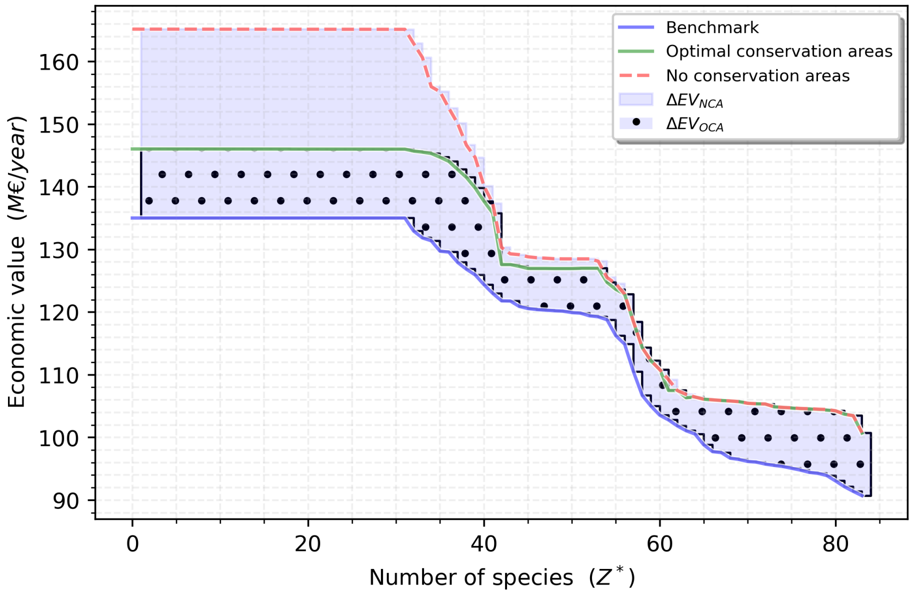
A great proportion of the study zone is located in or next to SPAs, as has been shown in Figure 2.4. So far we have assumed that the extension and spatial distribution of these areas has been predetermined and has remained fixed. Now, we depart from this assumption and also optimize about efficient conservation areas allocation.

To investigate any potential gains resulting from changes in the spatial distribution of these preservation areas, we permit their spatial distribution to vary. Our objective is to determine whether introducing changes in the spatial distribution can improve both biodiversity conservation and agricultural economic value. In other words, we aim to measure the extent to which they can shift the Pareto Frontier. Specifically, we propose two distinct scenarios (as outlined in subsection 2.3.3): first, the *Optimal Conservation Areas (OCA)* scenario, where we allow the spatial distribution of protection areas to change location while keeping the total area of protected surface equal to the number of hectares protected in the benchmark scenario. [35] That is, areas that were previously designated as SPAs and had limited MR choices can now choose from a range of crops and MRs that are physically possible within their cell. This allows us to identify potential areas for improvement in reallocation; that is, to identify the optimal distribution of conservation sites without reducing their size that maximizes agricultural economic value while maintaining bird conservation constraints.

In the second scenario, the *No conservation areas (NCA)*, we assume that all conservation areas are fully removed - not reallocated - and the set of land use limitations respond only to physical or technological restrictions such as the availability of irrigation infrastructure within each location. We solve our optimization problem under the assumption that there are no biodiversity protected areas - although we are still imposing the biodiversity constraints required to assure the sustainability of the species -. This allows us to maximize the agricultural economic value taking into account biodiversity preservation restrictions but non-land use restrictions.

[35] Note that conservation areas are required to be located outside territories without irrigation infrastructure. This is done to avoid ineffective conservation effects in places where irrigation would not have occurred due to limitations. Additionally, the total extent of the conservation area is determined by category. This means that the total extent restricted to 0mm must be equal to that of the status quo. This also applies to the 150mm and 350mm categories.

Figure (2.13) MAXIMUM ECONOMIC VALUE FOR DIFFERENT CONSERVATION GOALS IF LOCATIONS OF CONSERVATION AREAS ARE FIXED, REMOVED OR OPTIMIZABLE



Notes: Pareto curves between number of species preserved and economic value per year (€M). The red line represents economic value which could be obtained if irrigation constraints could be placed to optimize the trade-off between economic value and biodiversity. Blue line represents the case where conservation areas are placed as nowadays (Benchmark). The green line represents the case where conservation areas would be removed completely. Each point of the curve represents the maximum economic value which can be obtained subject to a biodiversity constraint, which is defined in the x-axis.

Table (2.11) MAXIMUM ECONOMIC VALUE FOR DIFFERENT CONSERVATION GOALS IF LOCATIONS OF CONSERVATION AREAS ARE FIXED, REMOVED OR OPTIMIZABLE

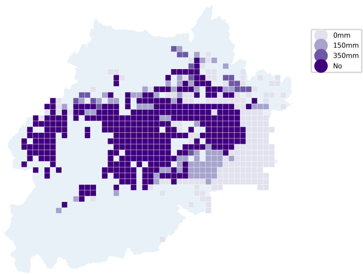
	Z^*									
Scenario	*0*	*10*	*20*	*30*	*40*	*50*	*60*	*70*	*80*	*83*
Benchmark	135.0	135.0	135.0	135.0	124.4	119.9	103.55	96.2	93.1	91.0
OCA	146.1	146.1	146.1	146.1	137.1	126.5	107.5	105.3	104.2	100.7
NCA	165.1	165.1	165.1	165.1	140.1	128.5	110.8	105.5	104.2	100.8
ΔEV_{OCA}	11.0	11.0	11.0	11.0	12.7	6.6	3.9	9.2	11.2	9.7
ΔEV_{NCA}	30.1	30.1	30.1	30.1	15.7	8.6	7.3	9.3	11.2	9.7

Notes: $\Delta EV_{OCA} = EV_{OCA} - EV_{fixed}$. $\Delta EV_{NCA} = EV_{NCA} - EV_{fixed}$. OCA: Optimal conservation areas; NCA: No conservation areas. Values are in €M.

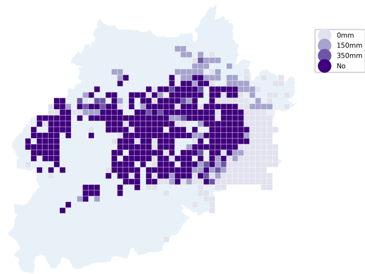
Comparing these two scenarios with the *Benchmark* scenario (as shown in Figure 2.13 & Table 2.11), the economic value associated with the OCA is at least as high as the NCA for each conservation goal ($EV_{NCA} \geq EV_{OCA} \quad \forall \quad Z^*$). The reasoning is straightforward and arises from the fact that applying additional restrictions will always limit the set of possible options and consequently the set of potential solutions. For the *OCA* scenario, it is a subset of the potential solutions for *NCA*. This explains why the maximum economic value of the NCA scenario is always larger then the maximum economic value of the OCA scenario. Interestingly, we can see how the differences between both scenarios diminish as the conservation goals become more rigorous. When conservation goals become highly demanding, all conservation efforts - in terms of limiting intensification in some areas - should be implemented and this leads to similar solutions in both cases. On one side, when Z^* increases, most of the different land uses should be present to satisfy the requirements of all bird species, including areas with low productive crops and MR combinations. Consequently conservation areas would be placed in those territories without a low impact of the economic value in both scenarios.

The results of our optimization problem thus indicate that the economic value in the Optimal Conservation Area (OCA) scenario exceeds those obtained in the benchmark scenario. More specifically, there is an annual increase of €9.7 M if all species are protected ($Z^* = 83$) and an increase of € 11.0 M if $Z^* = 0$. On the other hand, in the NCA scenario, the EV increase when there are no conservation goals would be € 30.1 M - which implies an additional increase of € 20.4 M with respect to OCA - while if all species must be preserved it is € 9.7 M - coinciding with OCA solution -. In Figure 2.14 we show the allocation of optimal conservation areas (under the *OCA* scenario). It seems that when conservation goals are low (for example when conservation does not interfere with economic maximization ($Z^* \leq 31$), conserved locations beyond zones without infrastructure are mainly located in the eastern territories, showing that limiting irrigation water there - where precipitation levels are higher and temperatures lower -, would not have as big an impact on economic value as in other places. When conservation goals are more ambitious, these conservation areas are displaced to more centered ones. As seen in Figure 2.9, the replacement of a region dominated by corn fields with wheat implies that other crops should be planted, also in more centered areas. This of course encourages the use of conservation areas in these places since wheat is much more resilient to low irrigation than corn.

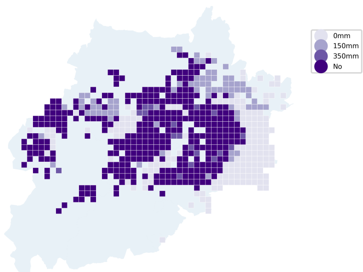
Figure (2.14) CONSERVATION ZONES UNDER THE OPTIMAL CONSERVATION AREA (*OCA*) SCENARIO



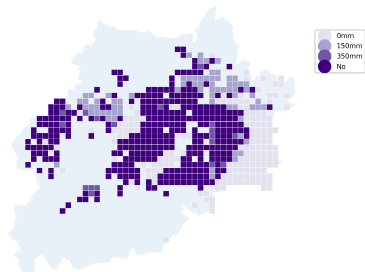
(a) $Z^* \leq 31$



(b) $Z^* = 40$



(c) $Z^* = 60$



(d) $Z^* = 83$

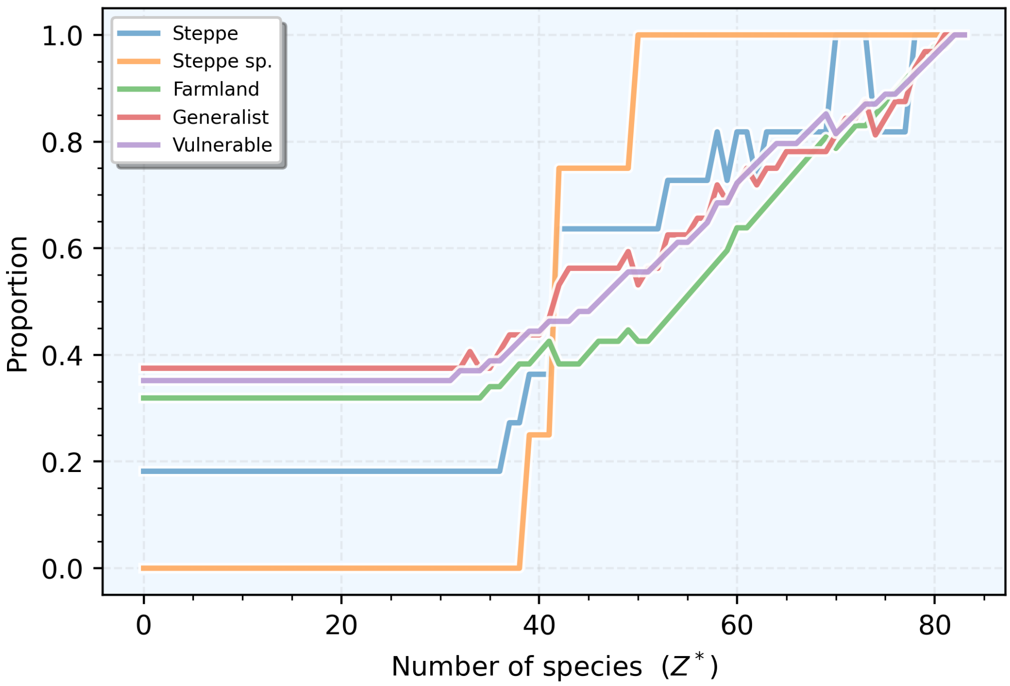
2.4.5 Triage vs. target conservation: optimizing economic value considering conservation of species with heterogeneous relevance

Until now, we have considered species as equally important (triage approach). However, conservation policy often prioritizes the preservation of specific endangered species rather than the entire bird community. In this section, we address this issue by solving the problem defined in Equation 2.1 subject to Equation 2.2 for different groups of species. Specifically, a *regulator* optimizes expected income subject to a set of sustainability constraints that include the conservation of each group individually, without introducing additional constraints involving other bird types. The conservation areas remain constant.

A starting point to check the relevance of target conservation strategies is to observe which group of species - according to their preferences or conservation interest - is protected from less to more ambitious conservation goals (i.e.: from low to high Z^*). We can later compare these results with the optimal solutions maximizing EV if instead of considering all species as equally important we impose the conservation of specific species. Consequently, we identify the proportion of each subgroup of species as described above which is conserved for each Z^* (Figure 2.15). In this case, we differentiate among the subgroups referred to in subsection 2.3.2: *Steppe, Steppe specialists, Farmland, Generalist*, and *Vulnerable* species. We observe that generalist species are conserved on average earlier than other subgroups. The last subgroup of species to begin being preserved corresponds to the *Steppe specialists*. Interestingly, *Steppe specialists* are not the last to be conserved and therefore not the most costly under this approach. As Z^* increases and more species must be conserved, the differences between groups become smaller.

The conservation of all the steppe specialists birds would be reached when $Z^* = 50$. In this case, the maximum economic value of the agricultural production in the area is € 120.1 M. Moreover, if all steppe species must be conserved it is firstly reached at $Z^* = 69$, with a maximum economic value of € 96.7 M. Finally, if the conservation goal is oriented to all the vulnerable species, it would be achieved at $Z^* = 82$, which practically occurs when all species are protected. In this case, the maximum economic value would be € 91.8 M.

Figure (2.15) CONSERVATION ANALYSIS BY GROUP OF SPECIES



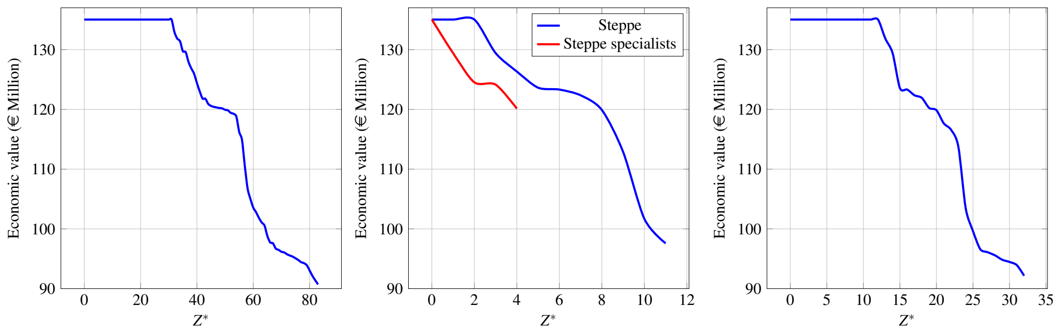
Notes: Proportion of species conserved for each group of species with respect to Z^*. The vertical axis values represent the proportion (between 0 and 1). Horizontal axis represents Z^*. For each Z^*, higher values represent a higher proportion of species of that group that are preserved, and consequently that with that given conservation goal Z^* this group is selected early to be preserved. Groups are not necessarily mutually exclusive. For example, Steppe specialists are also included in the Steppe category, and vulnerable species can be included in every category.

Once we have identified the order of conservation for different groups, we construct a *Pareto frontier* for three distinct subsets of endangered bird species: *Steppe* (11 species), *Steppe specialists* (4 species), and *Vulnerable* (33 species) to illustrate the trade-off between conservation goals and economic value in each case (see Figure 2.16.a-c). [36] As expected, the slopes of the three *Pareto frontiers* decrease, indicating a trade-off between conservation and economic value. However, for both steppe and vulnerable species, there are a few species that do not generate trade-offs. For steppe birds, this is the case for *Burnihus oedicnemus* and *Calandrella brachydactyla*. Moreover, when we target only steppe birds, the maximum economic value obtained is € 97.6 M which is slightly higher than the value obtained when preserving all 83 species (recall that the maximum *EV* conserving all species is € 90.7 M). This means that a steppe targeted conservation strategy would be able to increase

[36]In Figure 2.16.a, we present the Pareto curve for all species for the purpose of comparison.

the economic value by € 28.6 M (+41.3%) with respect to the status quo. Furthermore, by enforcing protection for these 11 steppe species, an additional 49 species (60 including the 11) would be protected. [37] With this, 70.2% of the generalist species, 75% of the vulnerable species, and 72% of the farmland species are protected. Additionally, if we consider only the four steppe specialists, the maximum economic value obtained is € 120 M. Then, preserving *Steppe specialists* would have significantly lower opportunity costs than preserving steppe birds due to their limited number - four species - and aligned interests. This reinforces the importance of conservation strategies focused on few species but with special conservation interest. An additional unexpected result is that by focusing on the conservation of steppe specialists, the conservation of 31 additional species is achieved, although none of them are steppe species. [38]

Figure (2.16) PARETO FRONTIERS BETWEEN CONSERVATION GOALS AND ECONOMIC VALUE BY GROUP OF SPECIES



a. All species   b. Steppe species   c. Vulnerable species

While analyzing the conservation of groups of species can be intriguing, it is also possible to rank individual species based on their conservation difficulty - the cost associated with preserving them. To accomplish this, we calculate the species-specific cost of conservation using three distinct measures. Firstly, under current conditions, we evaluate the

[37] This number has been computed by obtaining the total area size of each land use choice and computing a landscape suitability index for each species and comparing them with the status quo suitability indexes.

[38] In fact, by conserving the four specialist steppe species, the rest of the steppe species are also very close to being conserved. Specifically, the level of habitat suitability of these species decreases by an average of 4.63%.

economic conservation difficulty of species s as the maximum economic value obtained if the goal is to protect that specific species. Secondly, we determine the difficulty of conserving species s (Rank) by identifying which Z^* corresponds to the scenario in which species s is first conserved (the higher this number, the more effort required to conserve a species).[39]

Table (2.12) STEPPE BIRD SPECIES ECONOMIC CONSERVATION DIFFICULTY MEASURES

Species	**EV**	**Rank**	**Diff.**	**Species**	**EV**	**Rank**	**Diff.**
Burhinus oedicnemus	135.1	6	0.393	*Falco naumanni*	135.1	14	0.304
Calandrella brachydactyla	135.1	6	0.393	*Melanocorypha calandra*	122.0	42	0.748
Calandrella rufescens	120.5	47	0.804	*Pterocles alchata*	122.0	42	0.748
Chersophilus duponti	131.6	34	0.678	*Sylvia conspicillata*	135.1	14	0.304
Circus pygargus	135.1	31	0.781	*Tetrax Tetrax*	122.0	42	0.748
Coracias garrulus	135.1	21	0.487				

Notes: EV refers to the maximum economic value obtained if the objective is to protect only that specific species (€ M). Rank refers to the conservation goal Z^* where the species s is firstly conserved (if the rank is higher it takes longer more to be conserved and consequently is more costly). Diff. is a measure of economic conservation difficulty.

Although the conservation difficulty of a species can be calculated by determining under which conservation goal level Z^* it is first preserved, in some cases a species may be conserved under a certain conservation goal Z^* but not under more stringent goals. As a result, we design a third index of species conservation difficulty Diff_s as follows:

[39] Refer to subsection 2.G to see which species are conserved under each conservation level Z^*.

$$\text{Diff}_s = \frac{\sum_{l=1}^{l=83} l - \sum_{l=1}^{l=83} l \times Z^*_{83-l,s}}{\sum_{l=1}^{l=83} l}$$

where subscript l represents the preservation goal and is defined as the number of species intended to be conserved. Z^*_{ls} indicates whether or not a species is conserved under conservation goal l. Z^*_{ls} takes on a value of 1 if species s is preserved under a given preservation goal l, and 0 otherwise. There is a variable Z^*_{ls} for each preservation goal l. For example, species s may be conserved when the conservation goal is to preserve 50 or more species ($Z^*_{50,s} = 1$), but not when it is only to preserve 40 or more ($Z^*_{40,s} = 0$). For a summary of these results, see Table 2.12 for values of steppe species, as well as their conservation rank and associated economic value/cost. The index Diff_s takes on values between 0 and 1. If a species is always conserved, $\text{Diff}_s = 0$, and if it is never preserved, $\text{Diff}_s = 1$. Note that, for a given species, this index decreases with the number of conservation goals where that species is conserved and if it is conserved when conservation goals are lower - or "earlier". From an economic perspective this makes sense since, if it is possible to conserve a given species when conservation goals are lower, the economic costs of conservation are also lower.

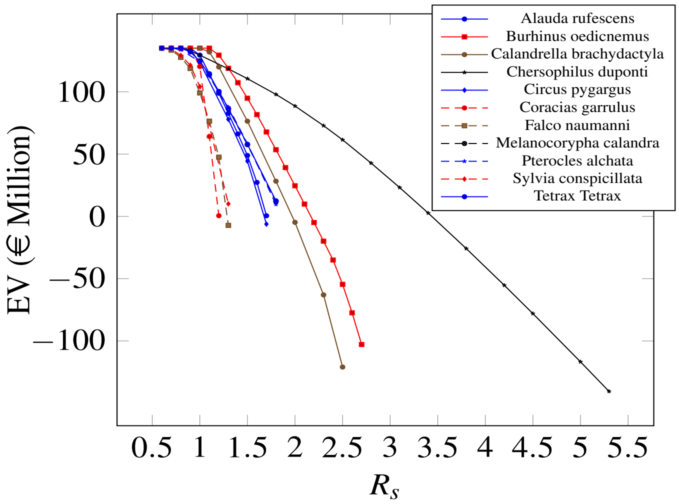
Interestingly, we observe that 6 out of 11 steppe species [40] do not generate trade-offs with economic value if the conservation strategy is targeted towards them, since the *EV* corresponds to the scenario with $Z^* = 0$ (€ 135.1 M). With respect to the other species, a trade-off exists but it is relatively low compared to the strategy of conserving all species or even conserving the entire steppe-bird community. The difficulty index is found to be highly correlated with the rank measurement and thus provides similar insights. Additionally, it is worth noting that three species (Melanocorypha Calandra, Pterocles Alchata and Tetrax Tetrax) have identical values indicating that they have similar conservation opportunity costs - and therefore very similar habitat preferences.

[40] Burnihus oedicnemus, Calandrella Brachydactyla, Circus pygargus, Coracias garrulus, Falco Naumanni and Sylvia Conspicillata.

As we have observed, focusing on individual species has significant implications in terms of the economic cost of conservation. In addition to this, we examine the implications of not only maintaining but also improving the current status of each of the 11 species classified as steppe birds. Our primary objective is to determine whether it is possible to increase landscape suitability indexes for individual species and their associated costs. For each species, we assume that conservation measures were implemented for a single target species - without considering the status of other species. To do this, we calculate this trade-off for different biodiversity threshold levels determined by the multiplier R_s, where $B_s^+ = B_s^* \times R_s$. B_s^+ represents the new conservation threshold for species s, which is interpreted as the original threshold multiplied by R_s. Notice that $R_s = 1$ is the value used throughout the chapter to determine whether a species is conserved or not. Following standard procedures, we compute the maximum economic value for each species and level of R_s (refer to Figure 2.17 for a graphical representation and Table 2.13 for values).

The EV decreases as long as the level of R_s imposed is higher. Note that the maximum EV without biological constraints ($EV =$ € 135 M) can be achieved for most species when the level of R_s is 0.7 . On the other hand, the species cannot mathematically obtain unlimited levels of R_s. When $R_s = 1$ (i.e., the default value), most of the steppe species require a cost for their conservation. From among them, only, *Burnihus oedicnemus* and *Calandrella brachydactyla* can be above the current threshold without reducing economic output. Note also that the species do not react equally to changes in R_s. Species with feasible higher levels of R_s indicate that their current situation could improve with respect to their potential. This is the case of *Chersophilus duponti*, which could increase its habitat index at the landscape level more than 5 times. On the contrary, species with the steepest curves present lower potential improvements (for example: *Coracias garrulus*).

Figure (2.17) RESILIENCE TRADE-OFF FOR EMBLEMATIC SPECIES FOR DIFFERENT CONSERVATION THRESHOLDS



Notes: Maximum economic value (€ M) for different levels of R_s for specific species. $R_s = 1$ is the default value. Values which are not represented for different species imply a non-mathematical possible result.

Table (2.13) MAXIMUM ECONOMIC VALUE FOR DIFFERENT CONSERVATION THRESHOLD LEVELS FOR SPECIFIC SPECIES

	R_s							
Species	**0.7**	**0.8**	**0.9**	**1.0**	**1.3**	**1.5**	**1.8**	**2**
Alaudala rufescens	135.0	135.0	135.0	129.4	82.7	48.9	-	-
Burhinus oedicnemus	135.0	135.0	135.0	135.0	118.9	94.8	53.5	24.6
Calandrella brachydactyla	135.0	135.0	135.0	135.0	106.4	76.4	28.2	-4.8
Chersophilus duponti	135.0	135.0	132.8	129.4	118.4	110.6	97.9	88.6
Circus pygargus	135.0	135.0	133.5	123.7	77.9	44.38	-	-
Coracias garrulus	135.0	135.0	135.0	120.2	-	-	-	-
Falco naumanni	133.4	127.6	118.7	99.1	-7.23	-	-	-
Melanocorypha calandra	135.0	135.0	132.9	124.9	86.3	57.5	11.9	-
Pterocles alchata	135.0	133.7	129.7	124.1	87.7	58.4	9.6	-
Sylvia conspicillata	134.3	129.2	103.1	104.1	10.1	-	-	-
Tetrax Tetrax	135.0	135.0	132.0	124.5	86.6	57.9	12.5	-

Notes: Missing value (-) represents not feasible solutions.

2.5 Conclusion & discussion

In this study, we present a framework for assessing the trade-offs between agricultural economic value and environmental conservation goals. To this end, we solve a linear land use spatial optimization model in a real landscape, the *Lleida* plain in Spain, which is significant both for agricultural economic value and for preservation of bird species. This is particularly important as there is an ongoing conflict between land intensification and biodiversity conservation, with steppe birds being the most threatened by agricultural intensification (Mañosa et al., 2020). Our methodology is easily adaptable to other cases and provides enough flexibility to obtain diverse results that can inform conservation policy.

Our method is based on multi-objective optimization and is adapted to land use selection with location-specific effects on both agricultural economic value and number of species conserved. It allows for valuing specific-species conservation difficulties by introducing a habitat suitability index for each species from a community of 83 bird species and is therefore extremely useful and flexible for policy design. We solve the model with aggregate agricultural economic value as the objective function for different conservation goal levels assuming the current characteristics of the area (such as irrigation constraints and SAP areas). We construct a *Pareto* frontier that shows the current trade-off between economic value and biodiversity preservation. Additionally, we extend this methodology to other scenarios by exploring the implications of removing or optimally allocating conservation areas, as well as the economic consequences of targeted conservation policies.

One of the main challenges we faced was obtaining realistic potential agricultural production in locations without sufficient data or where even those data did not exist. To address this problem, we used a crop growth simulation model (*STICS*) sensitive to management and crop type, as well as climate and soil variables that are location-specific. A second challenge was measuring biodiversity sustainability, which was adapted using two resource-based habitat suitability indexes based on *Estrada et al., (2004)* and *Cardador et al., (2014)* that account for crop selection and intensification level effects for each species.

The simulated crop yields showed significant disparities within the *Lleida* plain due to soil and climatic conditions, with intensification being a key determinant for agricultural productivity and food security. Between crops, corn was found to be the most profitable with high irrigation, while winter cereals were more productive under low or no irrigation. The biodiversity section displayed differences between species' habitat preferences, with some being generalists, others being specialists of low-productive land uses, and others being well-aligned with economic interests. Furthermore, not all species have the same conservation interest and there is no trivial solution even considering conservation goals exclusively.

The optimization model, solved with the algorithm *Gurobi*, showed that strategies focused on maximizing economic return involve an important negative trade-off with the biodiversity objective. Policies should, therefore, be oriented towards balancing both targets. Additionally, we found that conserving 31 species does not generate a trade-off with economic value, mainly corresponding to generalist or specialist species of land-uses correlated with high economic value crops. None of these 31 species is a steppe specialist bird. The agricultural opportunity cost of preserving all species was estimated to be € 44.3 M euros annually when optimizing resources (along the Pareto frontier). Furthermore, without reducing the conservation status of any species, the annual current crop economic value of the area could increase by € 21.4 M simply by optimally reallocating land uses.

We expand on these results by departing from the assumption that location of SPAs remains fixed. First, we showed that sustainability for all species considered could be achieved while also increasing economic value by an additional € 9.7 M $(+10.6\%)$ if we were to optimize the spatial allocation of conservation areas - while maintaining their size. Also, we optimize our model without conservation areas but imposing the preservation of all bird species; in this case, the gains over the benchmark model were similar. The optimal allocation of conservation areas does not negatively impact economic value when compared to the total removal of conservation areas when conservation goals are demanding. This is due to the fact that conserving all species requires the existence of low intensified land uses, which will tend to be located in the lands with lower opportunity cost under both assumptions.

Alternatively, realistic conservation policies may be designed to target specific species or subgroups. Our optimization process revealed that conserving all steppe specialist species, which are emblematic of the region, generates a trade-off with economic value and they are not prioritized under low conservation goals. In general, generalist species and those aligned with economic interests are more easily conserved due to their lower opportunity cost of preservation. However, when a triage approach based on opportunity cost is used, steppe specialists are not the most costly to conserve. Furthermore, our findings indicate that strategies focused on steppe species, particularly steppe specialists, result in a substantial increase in economic value in comparison with aiming to protect all species, while also preserving a high proportion of other species.

Our results are limited by a set of assumptions that were necessary to make the analysis feasible. Firstly, simulated crop yields are potential and may therefore be subject to upward biases. Ideally, crop yields should be validated in the region using observed field data; however, this would be unrealistic at this scale. Instead, we used expert knowledge to validate the fact that our results were close to those obtained in practice. On the other hand, habitat suitability indexes are a simplification of birds' general preferences and are intended to reflect their real preferences. Despite this, bird population dynamics depend not only on static associations but also on current populations, distributions and connectivity issues associated with colonization and migration processes. Additionally, our results strongly depend on how the habitat suitability index was constructed and on the definition of the conservation threshold - status quo - which is more related to policy matters than extinction processes.

Another limitation is related to the level of realism of the policies mentioned since they require a social planner imposing choices to maximize social welfare. Although the *CAP* and other related policies have been designed to incentivize farmers in this direction, decisions are usually made at an individual level rather than based on cooperative strategies. Our future work should also focus on incentive schemes that favor the implementation of land use policies at the landscape level. Some of these issues are explored in Chapter 3. We could also expect that relative prices of crops may fluctuate in the future, which could have implications for determining optimal landscape-level solutions. Even when using average price data from recent years to reduce bias, there is no guarantee that changes in prices will not have a long-term effect.

Although not discussed in this work, one possible criticism may be the existence of spillovers from the policy regarding price shifts due to changes in supply or input slippage. In the first case, we assume that the region is not large enough to consider prices as endogenous, although crop supply may change. In the second case, we consider that farmers may adapt to these limitations by using more inputs in other regions outside the study area (spillover effects). However, this is usually difficult to forecast (Pfaff et al., 2017), and it is not clear that conservation policies can always have a significant impact (Alpízar et al., 2013). Moreover, we could also expect positive spillovers through knowledge transfer, where farmers could be forced to become more efficient with less input usage.

In addition to enhancing the research by addressing the aforementioned assumptions, our future endeavors aim to incorporate climate change scenarios into this investigation. There are several reasons for this. Firstly, it is easily adaptable to climate change effects since *STICS* would only require new inputs - particularly forecast climate variables - which are currently available through several regional climate models such as those retrieved from *Cordex*. Additionally, it would allow us to identify how landscapes will be modified under different assumptions, considering that yields may decrease for some crops but increase for others. This will incentivize farmers to make new decisions regarding water use and crop selection. Designing new policies to address this scenario would involve a challenge that requires rigorous analysis. This issue will be explored in Chapter 4.

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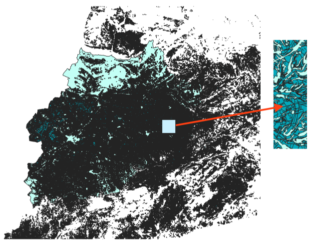
Appendices

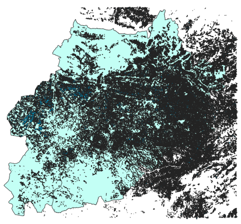
2.A Grid construction

Here we detail the process carried out to identify and display the study area as an artificial grid. We firstly use the digital maps of the *Declaración Única Agraria*, which provides plot-level information about crops planted (Figure A.1a). Afterwards, we exclude those plots where there is no currently planted (2019) wheat, barley, corn or vineyards (Figure A.1b). The third step is to construct a grid on the area. This GRID is based on $2 \times 2km^2$ squares (Figure A.1c). Once we have defined the grid, we overlap the plot map with the grid, and select those squares with at least 30% of their area (i.e. $\geq 120ha$) devoted to the plantation of the selected crops (Figure A.1d). Finally, we identify those squares in the grid and create the final map (Figure A.1e).

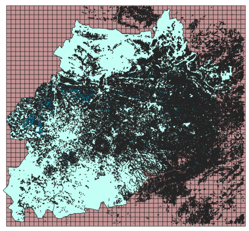
Figure (A.1) CELL GRID CONSTRUCTION



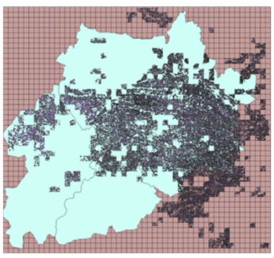
(a)



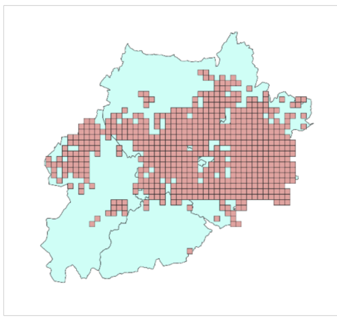
(b)



(c)



(d)



(e)

2.B STICS parameterization

Soil data have been obtained from 216 soil sampling pits made since 1991 in different locations of *Catalonia* (Figure 2.5) and provided by *Institut Cartogràfic i Geològic de Catalunya (ICGC)* through the *Geoindex display*. [1] Although *STICS* demands a lot of detailed information, we handled only that which has significant implications for crop yield. In the case of soil properties, we considered *Initial soil pH, Total Carbonate Content, Gravimetric water content at field capacity of each soil layer (%), Gravimetric water content at wilting point of each soil layer (%), Clay content after decarbonation (%), Soil Organic Nitrogen content in the soil layer (dry soil), Thickness of each soil layer (cm) and the initial soil Carbon to Nitrogen ratio*.

The sampling points were selected from the ICGC database to cover the maximum area of our study region. We assumed that soil properties do not change substantially over time, and therefore we selected observations irrespective of the sampling year. In addition, only areas on preexisting agricultural soils have been included in our sample, since farming tends to modify soil properties and adds organic material, and considering that other non-agricultural soils would distort the results. We also account for two soil layers' depths. The first, corresponding to the plough layer was considered from 0-20 cm depth (based on average values of each observation), while the second was obtained by spatial interpolation using observations of different depths (larger than 20cm). We consider the depth of the second layer that could affect plant growth and yield formation, i.e. rooting depth.

To obtain the available water holding capacity (i.e., the soil's ability to retain water and make it sufficiently available for plant use (USDA, 2008)) we use the soil water characteristic equations from *Saxton & Rawls (2006)*. [2] On the other hand, since no data on *Organic Nitrogen* were available, we estimated it by using the *Carbon to Nitrogen ratio*. The *Organic*

[1] See more information on https://www.icgc.cat/en/Public-Administration-and-Enterprises/Services/Soils/Geoindex-Sols

[2] They allow us to obtain a sufficiently accurate estimation of these variables without needing field or laboratory measurements, which are often difficult, costly or impractical. For that, we use data of proportions of sand, clay and organic matter, which were available as well at *ICGC* website.

carbon data were obtained from the *Organic C reservoir map of agricultural soils* (*IRTA et al.*, 2018). This consists of a raster map covering *Catalonia* with a 180 m pixel resolution published in 2018 and obtained through covariates related to environmental, topographic, management and edaphic variables. It was measured in mass per surface unit (kg C m^2), and therefore we transformed it into *Organic Carbon concentration* (%) [3], which is the variable used in *STICS*. Since there were no sampling data on soil for every cell, we used geostatistical methods to interpolate the characteristics of the soils for locations with missing data. We used *ArcMap v10.8* software to perform dot geostatistical interpolation processes (*Kriging* method) for each variable required by *STICS* with respect to soil properties. As a result, we obtained a raster layer with continuous variable values over the region, which we discretized computing the average value for each variable in each cell.

For climate variables, we used information provided by *Servei Metereòlogic de Catalunya (Meteocat)*. These data were obtained from weather stations located in the area. Since we also perform spatial interpolations, we additionally included other stations from adjacent regions to obtain more accurate values for the border areas. [4] We considered 8 climate variables: *Precipitation (mm/day), Maximum temperature (^oC), Minimum temperature (^oC), CO2 (Ppm), Vapour pressure (Mbar), Global radiation (MJ/m^2), Wind speed at 2 m height (m/s), Penman Evotranspiration (mm/day).* From them, only CO_2 concentration was not available, so we give a value of 300 parts per million (*Ppm*) for all cells, since we expect it not to have significant spatial variance.

STICS required daily information of each climatic variable mentioned for a given year. Instead of using information from a single recent year, we create a synthetic one averaging climate data for each day and station. Therefore, the value for each day, variable and station was the average of previous years. Not all the stations presented the same amount of data,

[3]See *Estimació dels estocs de carboni als sòls, a escala de país: dificultats i reptes*, available at https://www.icgc.cat/L-ICGC/Agenda/Jornada-El-Carboni-Organic-dels-sols-agricoles-una-eina-per-a-la-mitigacio-del-canvi-climatic-a-Catalunya. We used *Apparent density (g/cm^3=1.45* and *Thickness of the layer (cm)* = 30 since this was the one used on the original map of organic carbon reservoir.

[4]Specifically: Ribera d'Ebre, Priorat, Conca de Barberà, Anoia and Solsonès.

variables and starting date. Consequently, we considered only those stations with at least 10 years of continuous data and whose last year ended after 2015, so as to avoid stations with a low number of observations and long-term climate change effects, respectively. As with the soil variables, we ran a *Kriging* process. Nevertheless, since climatic variables are daily, we did it for each variable and day of the year. As a result, we generated a raster map for each climatic variable and day of the year [5], whose values were averaged for each cell.

We defined 5 different crop management regimes (MR), which are identified with varying irrigation amount ($mm/year$), although they are also accompanied by specific practices regarding use of N fertilizers, tillage, plowing, sowing and harvesting, specifically, rainfed (0 mm), modern irrigation with 3 different irrigation amounts (150, 350, 650 mm) and traditional surface irrigation (900 mm). [6] Not all the regimes were considered for each crop for agronomic reasons. For example, winter cereal crops do not need high amounts of irrigation to be close to their actual potential yield in these regions. Specifically, we considered rainfed, traditional irrigation and 150/350 mm modern irrigation for barley and wheat; rainfed and 229 mm for vineyards; and all the categories for corn. The information was obtained mainly from *Plaza-Bonilla et al. (2018)*, data sheets and expertise when needed. STICS required amounts, varieties and application dates on the variables described above during a year, which were adapted to the 5 categories defined, representing the most accurate real practices. In STICS we did not only consider amount of water supplied but also its efficiency [7]. The parameter *effirr* is the STICS parameter which indicated the efficiency per amount of irrigation water used. For cereals and corn, we considered surface and sprinkler irrigation. For vineyards, we simulated drip irrigation.

[5] In total, 2921 raster maps were created.

[6] These amounts were designed intentionally to coincide with current limitations in the area.

[7] Since traditional irrigation methods are less efficient in terms of yield per unit of water *effirr*= {*traditional=0.65; modern=0.9; drip irrigation=1* }

2.C Summary of model parameters

Table (A.1) VARIABLE AND PARAMETERS SUMMARY

Symbol	Name	Set of values
Land use choice:		
x	Land use indicator	x$\in [0,1]$
i	Location	i$\in \{1,2, ,602\}$
j	Crop	*j* ∈*(Barley, Wheat, Corn & Vineyard)*
r	Management (MR)	*r* ∈ *(Rainfed, 150mm, 350mm, 650mm, surface)*
Ecological model:		
s	Species	s$\in \{1, ,83\}$
B_{jrs}	Habitat preference index	$\mathrm{B}_{jrs} \geq 0$
B_s	Landscape species-specific ecological average index	$\mathrm{B}^* \geq 0$
$I^*_{jr,s}$	Habitat intensification adjustment component	$\mathrm{I}^*_{jr,s} \geq 0$
$I_{j,s}$	Crop habitat-preference index	$\mathrm{I}_{j,s} \geq 0$
k	Habitat resource	Described in subsection 2.3.2
h	Ecological need	h$\in$ {Dietary,Nesting}
$I^D_{s,k}$	Species demand for resource k	$\mathrm{I}^{\mathrm{D}}_{s,k} \in \{0,1\}$
$I^S_{jr,k}$	Habitat supply of resource k	$\mathrm{I}^{\mathrm{S}}_{jr,k} \in [0,1]$
m	Agricultural practices affecting biodiversity	m$\in$ {Agrochemical inputs, irrigation, plow}
g	Intensification level	g≥ 0
B^*_s	Habitat threshold of a given species	$\mathrm{B}_{s,threshold} \geq 0$
Z_s	Species conservation indicator function	$\mathrm{Z}_s \in \{0,1\}$
Z^*	Number of species conserved	$\mathrm{Z}^* \in [0,83]$
Economic model:		
EV	Aggregated economic value	EV$\in \mathbb{R}$
Q	Potential crop yield	Q≥ 0
P_j	Crop price	$\mathrm{P}_j \geq 0$
C	Agricultural production costs	C≥ 0
W	Water cost	W≥ 0

2.D Habitat suitability index model details

Some considerations about the habitat suitability index construction:

- The sum of resources matching dietary needs is truncated to 1 following *Cardador et* *al. (2014)*.

- Additionally, the index based on *Cardador et al. (2014)* that accounts for intensification is normalized to 1 for each species and crop, where 1 represents the maximum score (corresponding to non-irrigated crops since the index is negatively related to intensification measures). As a result, $I^*_{jrs} \in [0,1]$. This was done in this manner to represent proportional losses rather than absolute - albeit abstract - adjustments to the crop selection index I_{js}.

- The crop intensification index I^*_{jrs} was calculated for barley/wheat and corn. We did not have sufficient data for vineyards so we relied on the plausible assumption that irrigation in vineyards would not significantly affect birds (considering that it is drip irrigation).

- The parameter g_m (intensification level of practice m) was bounded between 1 and 2 for the effects of fertilizers and irrigation - the effects of plowing did not depend on the degree of intensification but only considered whether or not plowing was carried out. Among all crops and levels of intensification, the practice m with lower usage was set to 1 (thus it does not reduce the index I^*_{jrs} since it is in the denominator). For each practice, the maximum level was set to 2 corresponding with the land-use choice with higher levels of that practice. This was designed to avoid different - and uncontrolled - effects of irrigation or fertilizers on habitat suitability indexes. As a result, using maximum amounts of fertilizers will have an equal negative impact to using maximum amounts of irrigation (on food supply). Similarly, using minimum amounts of fertilizers will have an equal positive impact to using minimum amounts of irrigation. These relationships can be explored in Table A.4. The food supply indexes for each land-use choice are shown in Table A.5.

- Additionally, we had data on how practices affected food supply depending on the season (Spring / Summer). Ideally, the value would be obtained based on the breeding season of each species. However, given the static nature of the index, we simplified it and averaged the food supply for each resource k between seasons. In *Cardador* *et al. (2014)*, they also considered how vegetation height affected habitat suitability indexes. Unfortunately, we did not have sufficient data for all species considered regarding preferences for vegetation structure. Even so, we could expect that higher vegetation structures are generally detrimental to birds' basic nourishment needs and therefore the effects of intensification on birds might be worse than those shown in this study.

2.E Selection of species

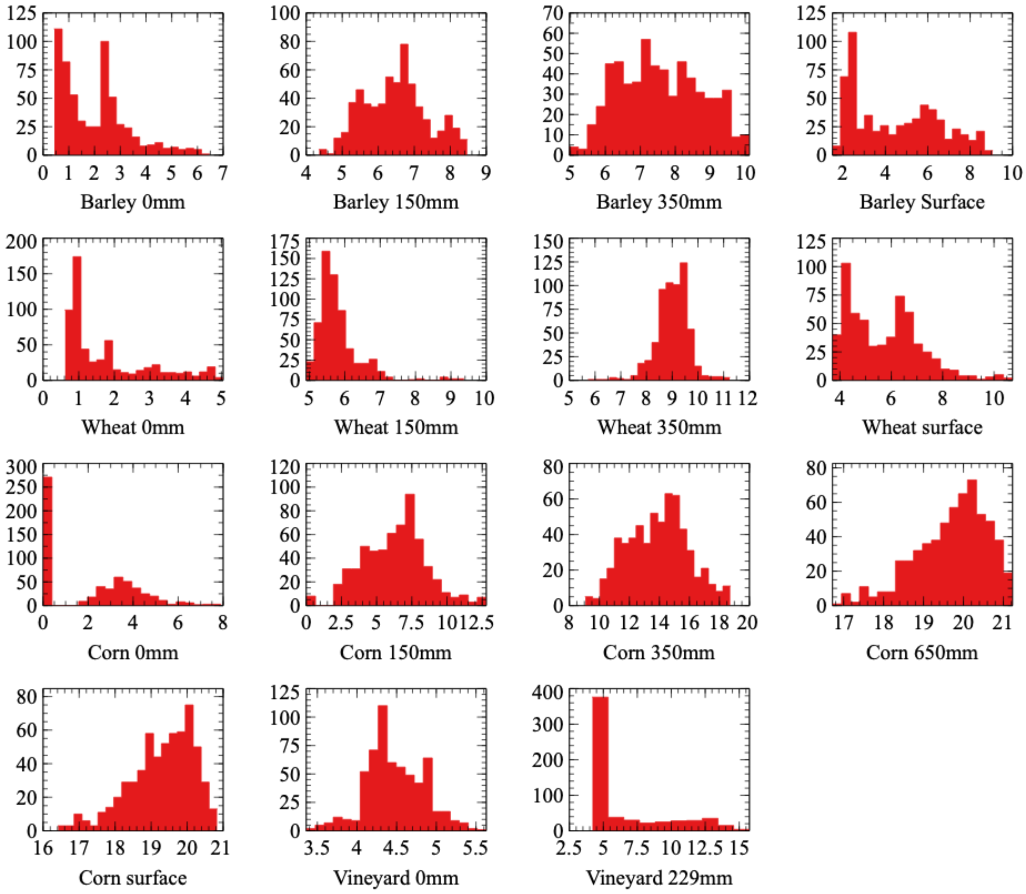
Table (A.2) SPECIES CONSIDERED IN THE STUDY

Scientific name			
Alauda arvensis	Circus pygargus*+	Himantopus himantopus	Passer domesticus+
Alcedo atthis	Cisticola juncidis	Hippolais polyglotta	Passer montanus
Alectoris rufa	Clamator glandarius	Hirundo daurica+	Petronia petronia
Anthus campestris	Columba oenas	Hirundo rustica	Phasianus colchicus+
Ardea purpurea+	Columba palumbus	Ixobrychus minutus	Pica pica
Asio otus+	Coracias garrulus*+	Jynx torquilla	Picus viridis
Athene noctua	Corvus corone	Lanius meridionalis+	Pterocles alchata*+
Burhinus oedicnemus*+	Corvus monedula	Lanius senator	Saxicola torquatum+
Buteo Buteo	Coturnix coturnix	Lullula arborea	Serinus serinus+
Calandrella brachydactyla*+	Cuculus canorus	Luscinia megarhynchos	Streptopelia turtur
Calandrella rufescens*+	Egretta garzetta	Melanocorypha calandra*	Sturnus unicolor
Caprimulgus europaeus	Emberiza calandra	Merops apiaster	Sturnus vulgaris
Caprimulgus ruficollis+	Emberiza cia+	Milvus milvus+	Sylvia borin
Carduelis cannabina+	Emberiza cirlus	Monticola solitarius	Sylvia communis
Carduelis carduelis+	Emberiza hortulana	Motacilla alba	Sylvia conspicillata*+
Carduelis chloris+	Falco naumanni*+	Neophron percnopterus+	Sylvia undata
Chersophilus duponti*	Falco subbuteo+	Nycticorax nycticorax	Tetrax tetrax*+
Ciconia ciconia+	Falco tinnunculus	Oenanthe hipsanica+	Turdus viscivorus
Circaetus gallicus+	Galerida theklae	Oenanthe leucura	Tyto alba+
Circus aeruginosus+	Galerida cristata	Otus scops+	Upupa epops
Circus cyaneus+	Hieraaetus fasciatus+	Parus major	

Notes: (*) Steppe habitat adapted species. (+) species considered as vulnerable by *IUCN*.

2.F Simulated crop distributions

Figure (A1) CROP POTENTIAL YIELD DISTRIBUTIONS FOR DIFFERENT COMBINATIONS OF CROP AND MANAGEMENT REGIMES



Notes: Data were obtained through simulations carried out by the crop growth simulation model *STICS*. Horizontal axis represents crop yield in ton/ha and year. Vertical axis represents cell frequency. Surface: traditional irrigation methods

2.G Species conservation status for each point of the Pareto Frontier

Table (A.3) SPECIES PROTECTION STATUS BY CONSERVATION GOAL

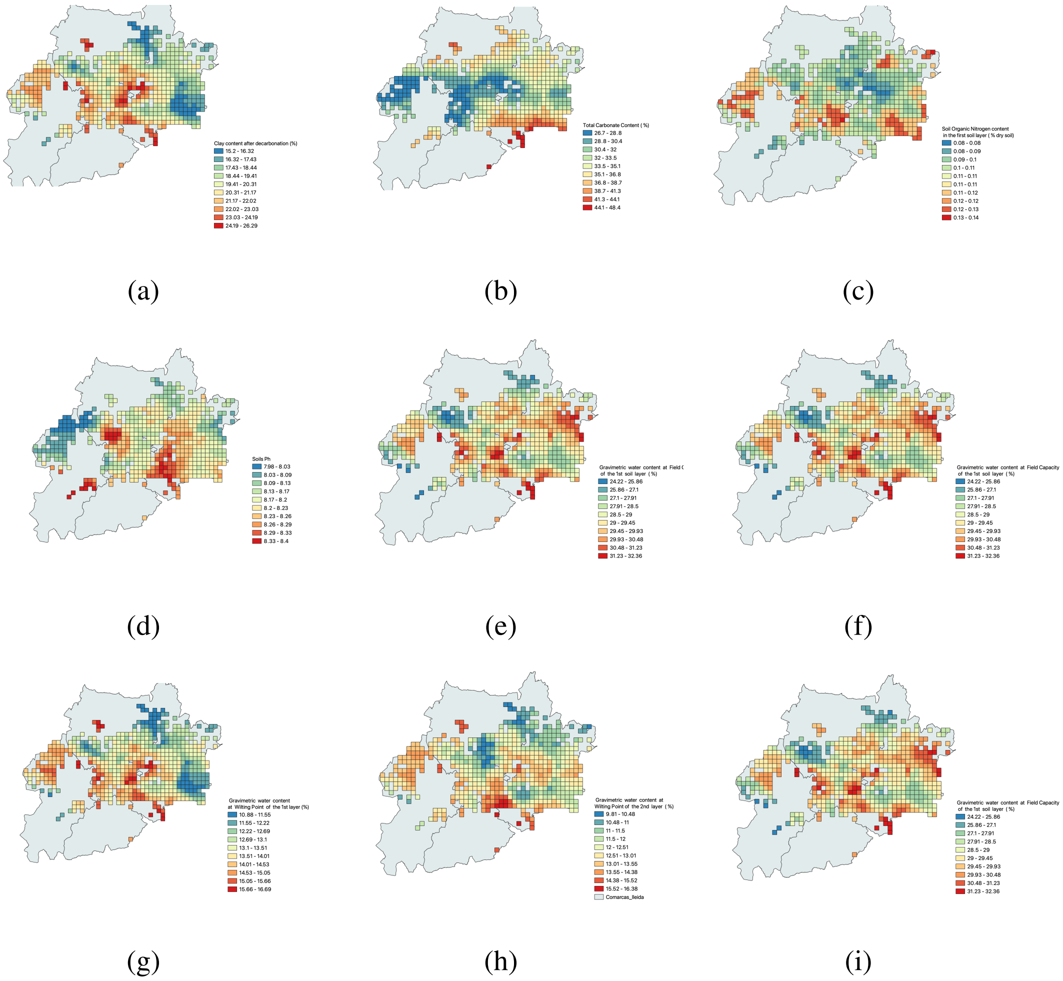
Species	0	5	10	15	20	25	30	35	40	45	50	55	60	65	70	75	80	83
Alauda arvensis	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1
Alcedo atthis	0	0	0	0	0	1	0	1	1	0	1	1	1	1	1	1	1	1
Alectoris rufa	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1
Anthus campestris	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1
Ardea purpurea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Asio otus	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Athene noctua	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Burhinus oedicnemus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Buteo Buteo	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Calandrella brachydactyla	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Calandrella rufescens	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1
Caprimulgus europaeus	0	0	0	0	0	0	0	1	1	0	1	1	1	1	1	1	1	1
Caprimulgus ruficollis	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Carduelis cannabina	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Carduelis carduelis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Carduelis chloris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Chersophilus duponti	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1
Ciconia ciconia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Circaetus gallicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Circus aeruginosus	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1
Circus cyaneus	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1
Circus pygargus	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
Cisticola juncidis	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Clamator glandarius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Columba oenas	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Columba palumbus	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1
Coracias garrulus	0	0	0	0	1	0	1	1	1	1	1	1	1	1	1	1	1	1
Corvus corone	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Corvus monedula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Coturnix coturnix	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
Cuculus canorus	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1
Egretta garzetta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Emberiza calandra	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
Emberiza cia	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
Emberiza cirlus	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1
Emberiza hortulana	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
Falco naumanni	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Falco subbuteo	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Falco tinnunculus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Galerida theklae	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1
Galerida cristata	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1
Hieraaetus fasciatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Himantopus himantopus	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Hippolais polyglotta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Hirundo daurica	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1
Hirundo rustica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Ixobrychus minutus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Jynx torquilla	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1
Lanius meridionalis	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Lanius senator	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Lullula arborea	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1
Luscinia megarhynchos	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1
Melanocorypha calandra	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
Merops apiaster	0	0	0	0	0	0	1	1	1	0	1	1	1	1	1	1	1	1
Milvus milvus	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Monticola solitarius	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1
Motacilla alba	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Neophron percnopterus	0	0	0	0	1	0	1	1	1	1	1	1	1	1	1	1	1	1
Nycticorax nycticorax	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Oenanthe hipsanica	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Oenanthe leucura	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1
Otus scops	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1
Parus major	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1

SPECIES PROTECTION STATUS BY CONSERVATION GOAL (CONTINUATION)

Species	0	5	10	15	20	25	30	35	40	45	50	55	60	65	70	75	80	83
Specie	0	5	10	15	20	25	30	35	40	45	50	55	60	65	70	75	80	83
Passer domesticus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Passer montanus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Petronia petronia	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Phasianus colchicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Pica pica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Picus viridis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Pterocles alchata	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
Saxicola torquatum	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1
Serinus serinus	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1
Streptopelia turtur	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Sturnus unicolor	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Sturnus vulgaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Sylvia borin	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Sylvia communis	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Sylvia conspicillata	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Sylvia undata	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Tetrax tetrax	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
Turdus viscivorus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Tyto alba	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Upupa epops	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1

2.H Soil and climate spatial estimates

Figure (A2) SOIL SPATIAL ESTIMATES



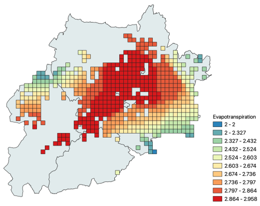
(a) (b) (c)

(d) (e) (f)

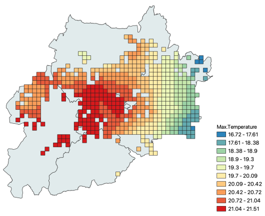
(g) (h) (i)

Notes: Soil variables- (a) Clay; (b) Total Carbonate Content (%); (c) Organic Nitrogen; (d) pH; (e) Gravimetric water content at Field Capacity of the 1st soil layer (%); (f) Gravimetric water content at Field Capacity of the 2nd soil layer (%); (g) Gravimetric water content at Wilting Point of the 1st soil layer (%); (h) Gravimetric water content at Wilting Point of the 2nd soil layer (%); (i) Thickness of the 2nd soil layer. Red colors represent higher values for each variable and blue colors lower ones. Values are displayed using Natural Breaks Optimization (Jenks) for better visualization of different variable characteristics.

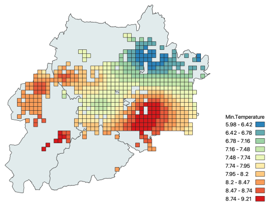
Figure (A3) CLIMATE SPATIAL ESTIMATES



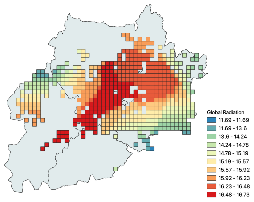
(a)



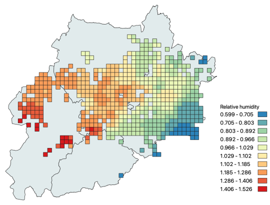
(b)



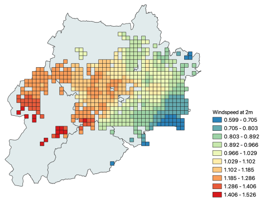
(c)



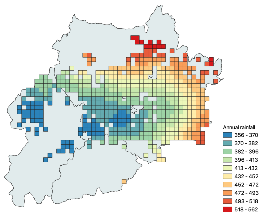
(d)



(e)



(f)



(g)

Mean climate variables- (a) Evapotranspiration (mm/day); (b) Max Temperature (^{o}C); (c) Min Temperature (^{o}C) ; (d) Global radiation (mJ/m^2); (e) ; (f) Vapor pressure (mb) ; (g) Windspeed at 2m (2m/s); (h) Annual precipitation (mm/year). Red colors represent higher values for each variable and blue colors lower ones. Values are displayed using Natural Breaks Optimization (Jenks) for better visualization of different variable characteristics.

2.I Habitat suitability index components

Table (A.4) COEFFICIENTS OF INTENSIFICATION FOR IRRIGATION AND FERTILIZATION

		Intensification				
		0 mm	150 mm	350 mm	650 mm	900 mm
Fertilizers (Kg)	Barley/Wheat	75	125	200	-	200
	Corn	50	75	150	300	300
COEF.Fert.	Barley/Wheat	1.1	1.3	1.6	-	1.6
	Corn	1	1.1	1.4	2	2
Irrigation (mm)		0	150	350	650	900
Irrigation coefficient		1	1.2	1.4	1.7	2

Table (A.5) HABITAT SUITABILITY INDEX: FOOD SUPPLY COMPONENTS

Practice		Dry cereal		Irrigated cereal		Corn	
		Spring	Summer	Spring	Summer	Spring	Summer
Agro-chemical inputs	Loss of crop plant material	1	1	1	1	1	1
	Loss of seeds	1	1	1	1	1	1
	Loss of crop invertebrates	1	1	1	1	1	1
Irrigation	Loss of crop plant material	0	0	1	0	1	1
	Loss of crop invertebrates	0	0	1	0	1	1
Plow	Loss of crop invertebrates	0	1	0	1	0	0
	Loss of crop plant material	0	1	0	1	0	0
	Loss of seeds	0	1	0	1	0	0
	Loss of crop vertebrates	0	1	0	1	0	0
Plant availability		0.33	0.20	0.07	0.07	0.04	0.04
Seed availability		0.33	0.20	0.12	0.07	0.08	0.08
Invertebrate availability		0.33	0.20	0.07	0.07	0.04	0.04
Vertebrate availability		1.00	0.33	1.00	0.12	1.00	1.00

	0 mm		150 mm		350 mm		650 mm		900 mm	
CEREALS	Spring	Summer	Spring	Summer	Spring	Summer	Spring	Summer	Spring	Summer
Plant availability	0.48	0.28	0.29	0.26	0.25	0.24	-	-	0.22	0.24
Seed availability	0.48	0.28	0.43	0.26	0.38	0.24	-	-	0.38	0.24
Invertebrate availability	0.48	0.28	0.29	0.26	0.25	0.24	-	-	0.22	0.24
Vertebrate availability	1.00	0.40	1.00	0.40	1.00	0.40	-	-	1.00	0.40
CORN										
Plant availability	0.33	0.33	0.31	0.31	0.26	0.26	0.21	0.21	0.20	0.20
Seed availability	0.50	0.50	0.48	0.48	0.42	0.42	0.33	0.33	0.33	0.33
Invertebrate availability	0.33	0.33	0.31	0.31	0.26	0.26	0.21	0.21	0.20	0.20
Vertebrate availability	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00

Notes: Source: Own elaboration based on Cardador et al. (2014).

2.J Crop management: technical specifications

Table (A.6) DATES, AMOUNTS AND TYPES OF TECHNIQUES APPLIED TO EACH MANAGEMENT TYPE FOR BARLEY AND WHEAT

Technique	Date (Julian)	Type	Amount				
Crop residues	311	11				3.47	
Tillage	311	5, 15					
Sowing	321	3,400,3					
					Amount (mm)		
		0	150	350	Trad.	650	
	311	5	0	50	50	50	Not justified
Fertilization	409	1	75	75	75	75	Not justified
	440	1	0	0	75	75	Not justified
			0	150	350	Trad.	650
	426		0.0	3.7	8.5		Not justified
	428		0.0	3.7	8.5		Not justified
	430		0.0	3.7	8.5		Not justified
	432		0.0	3.7	8.5		Not justified
	434		0.0	3.7	8.5		Not justified
	436		0.0	3.7	8.5		Not justified
	438		0.0	3.7	8.5		Not justified
	440		0.0	3.7	8.5		Not justified
	442		0.0	3.7	8.5		Not justified
	444		0.0	3.7	8.5		Not justified
	446		0.0	3.7	8.5		Not justified
	448		0.0	3.7	8.5		Not justified
	450		0.0	3.7	8.5		Not justified
	452		0.0	3.7	8.5		Not justified
	454		0.0	3.7	8.5		Not justified
	456		0.0	3.7	8.5		Not justified
	458		0.0	3.7	8.5		Not justified
	460		0.0	3.7	8.5		Not justified
	462		0.0	3.7	8.5		Not justified
	464		0.0	3.7	8.5		Not justified
	466		0.0	3.7	8.5		Not justified
	468		0.0	3.7	8.5		Not justified
	470		0.0	3.7	8.5		Not justified
Irrigation	472		0.0	3.7	8.5		Not justified
	474		0.0	3.7	8.5		Not justified
	476		0.0	3.7	8.5		Not justified
	478		0.0	3.7	8.5		Not justified
	480		0.0	3.7	8.5	129.0	Not justified
	482		0.0	3.7	8.5		Not justified
	484		0.0	3.7	8.5		Not justified
	486		0.0	3.7	8.5		Not justified
	488		0.0	3.7	8.5		Not justified
	490		0.0	3.7	8.5		Not justified
	492		0.0	3.7	8.5		Not justified
	494		0.0	3.7	8.5		Not justified
	496		0.0	3.7	8.5		Not justified
	498		0.0	3.7	8.5		Not justified
	500		0.0	3.7	8.5		Not justified
	502		0.0	3.7	8.5		Not justified
	504		0.0	3.7	8.5		Not justified
	506		0.0	3.7	8.5		Not justified
Harvest	547						

Notes: Dates in Julian days. Amounts in mm. Types according to STICS numeration. Source: Plaza-Bonilla et al. (2018)

Table (A.7) DATES, AMOUNTS AND TYPES OF TECHNIQUES APPLIED TO EACH MANAGEMENT TYPE FOR CORN

Technique	Date (Julian)	Type	Amount				
Crop residues	94	11	15.2				
Tillage	94	10, 25					
Sowing	102	5,8.4,7					
			Amount (mm)				
			0	150	350	Trad.	650
Fertilization	94	3	0	0	50	50	50
	154	3	50	75	100	125	125
	174	3	0	0	0	125	125
			0	150	350	Trad.	650
Irrigation	102		0.0	4.0	4.7	8.8	
	104		0.0	0.0	4.7	8.8	
	106		0.0	4.0	4.7	8.8	
	108		0.0	0.0	4.7	8.8	
	110		0.0	4.0	4.7	8.8	
	112		0.0	0.0	4.7	8.8	
	114		0.0	4.0	4.7	8.8	
	116		0.0	0.0	4.7	8.8	
	118		0.0	4.0	4.7	8.8	
	120		0.0	0.0	4.7	8.8	
	122		0.0	4.0	4.7	8.8	
	124		0.0	0.0	4.7	8.8	
	126		0.0	4.0	4.7	8.8	
	128		0.0	0.0	4.7	8.8	
	130		0.0	4.0	4.7	8.8	
	132		0.0	0.0	4.7	8.8	
	134		0.0	4.0	4.7	8.8	
	136		0.0	0.0	4.7	8.8	129
	138		0.0	4.0	4.7	8.8	
	140		0.0	0.0	4.7	8.8	
	142		0.0	4.0	4.7	8.8	
	144		0.0	0.0	4.7	8.8	
	146		0.0	4.0	4.7	8.8	
	148		0.0	0.0	4.7	8.8	
	150		0.0	4.0	4.7	8.8	
	152		0.0	0.0	4.7	8.8	129
	154		0.0	4.0	4.7	8.8	
	156		0.0	0.0	4.7	8.8	
	158		0.0	4.0	4.7	8.8	
	160		0.0	0.0	4.7	8.8	
	162		0.0	4.0	4.7	8.8	
	164		0.0	0.0	4.7	8.8	
	166		0.0	4.0	4.7	8.8	
	168		0.0	0.0	4.7	8.8	129
	170		0.0	4.0	4.7	8.8	
	172		0.0	0.0	4.7	8.8	
	174		0.0	4.0	4.7	8.8	
	176		0.0	0.0	4.7	8.8	
	178		0.0	4.0	4.7	8.8	
	180		0.0	0.0	4.7	8.8	
	182		0.0	4.0	4.7	8.8	
	184		0.0	0.0	4.7	8.8	129
	186		0.0	4.0	4.7	8.8	
	188		0.0	0.0	4.7	8.8	
	190		0.0	4.0	4.7	8.8	
	192		0.0	0.0	4.7	8.8	
	194		0.0	4.0	4.7	8.8	
	196		0.0	0.0	4.7	8.8	
	198		0.0	4.0	4.7	8.8	
	200		0.0	0.0	4.7	8.8	129
	202		0.0	4.0	4.7	8.8	
	204		0.0	0.0	4.7	8.8	
	206		0.0	4.0	4.7	8.8	
	208		0.0	0.0	4.7	8.8	
	210		0.0	4.0	4.7	8.8	
	212		0.0	0.0	4.7	8.8	
	214		0.0	4.0	4.7	8.8	
	216		0.0	0.0	4.7	8.8	129
	218		0.0	4.0	4.7	8.8	
	220		0.0	0.0	4.7	8.8	
	222		0.0	4.0	4.7	8.8	
	224		0.0	0.0	4.7	8.8	
	226		0.0	4.0	4.7	8.8	
	228		0.0	0.0	4.7	8.8	
	230		0.0	4.0	4.7	8.8	
	232		0.0	0.0	4.7	8.8	129
	234		0.0	4.0	4.7	8.8	
	236		0.0	0.0	4.7	8.8	
	238		0.0	4.0	4.7	8.8	
	240		0.0	0.0	4.7	8.8	
	242		0.0	4.0	4.7	8.8	
	244		0.0	0.0	4.7	8.8	
	246		0.0	4.0	4.7	8.8	
	248		0.0	0.0	4.7	8.8	
Harvest	284						

Notes: Dates in Julian days. Amounts in mm. Types according to STICS numeration. Source: Expertise

Table (A.8) DATES, AMOUNTS AND TYPES OF TECHNIQUES APPLIED TO EACH MANAGEMENT TYPE FOR VINEYARD

Technique	Date (Julian)	Type	Amount				
Crop residues	STICS	STICS	STICS				
Tillage	STICS	STICS					
Sowing	STICS	STICS					
			Amount (mm)				
			0	150	229	Trad.	650
Fertilization	90	3	0	No	30	Not justified	Not justified
			0	150	350	Trad.	650
Irrigation	122		0	Not justified	2.1	Not justified	Not justified
	126		0	Not justified	2.1	Not justified	Not justified
	130		0	Not justified	2.1	Not justified	Not justified
	134		0	Not justified	2.1	Not justified	Not justified
	138		0	Not justified	2.1	Not justified	Not justified
	142		0	Not justified	2.1	Not justified	Not justified
	146		0	Not justified	2.1	Not justified	Not justified
	150		0	Not justified	2.1	Not justified	Not justified
	154		0	Not justified	3.9	Not justified	Not justified
	156		0	Not justified	3.9	Not justified	Not justified
	158		0	Not justified	3.9	Not justified	Not justified
	160		0	Not justified	3.9	Not justified	Not justified
	162		0	Not justified	3.9	Not justified	Not justified
	164		0	Not justified	3.9	Not justified	Not justified
	166		0	Not justified	3.9	Not justified	Not justified
	168		0	Not justified	3.9	Not justified	Not justified
	170		0	Not justified	3.9	Not justified	Not justified
	172		0	Not justified	3.9	Not justified	Not justified
	174		0	Not justified	3.9	Not justified	Not justified
	176		0	Not justified	3.9	Not justified	Not justified
	178		0	Not justified	3.9	Not justified	Not justified
	180		0	Not justified	3.9	Not justified	Not justified
	182		0	Not justified	3.9	Not justified	Not justified
	184		0	Not justified	3.9	Not justified	Not justified
	186		0	Not justified	5.7	Not justified	Not justified
	188		0	Not justified	5.7	Not justified	Not justified
	190		0	Not justified	5.7	Not justified	Not justified
	192		0	Not justified	5.7	Not justified	Not justified
	194		0	Not justified	5.7	Not justified	Not justified
	196		0	Not justified	5.7	Not justified	Not justified
	198		0	Not justified	5.7	Not justified	Not justified
	200		0	Not justified	5.7	Not justified	Not justified
	202		0	Not justified	5.7	Not justified	Not justified
	204		0	Not justified	5.7	Not justified	Not justified
	206		0	Not justified	5.7	Not justified	Not justified
	208		0	Not justified	5.7	Not justified	Not justified
	210		0	Not justified	5.7	Not justified	Not justified
	212		0	Not justified	5.7	Not justified	Not justified
	214		0	Not justified	5.7	Not justified	Not justified
	216		0	Not justified	5.7	Not justified	Not justified
	218		0	Not justified	3.5	Not justified	Not justified
	220		0	Not justified	3.5	Not justified	Not justified
	222		154	Not justified	3.5	Not justified	Not justified
	224		0	Not justified	3.5	Not justified	Not justified
	226		0	Not justified	3.5	Not justified	Not justified
	228		0	Not justified	3.5	Not justified	Not justified
	230		0	Not justified	3.5	Not justified	Not justified
	232		0	Not justified	3.5	Not justified	Not justified
	234		0	Not justified	3.5	Not justified	Not justified
	236		0	Not justified	3.5	Not justified	Not justified
	238		0	Not justified	3.5	Not justified	Not justified
	240		0	Not justified	3.5	Not justified	Not justified
	242		0	Not justified	3.5	Not justified	Not justified
	244		0	Not justified	3.5	Not justified	Not justified
	246		0	Not justified	3.5	Not justified	Not justified
	250		0	Not justified	1.7	Not justified	Not justified
	254		0	Not justified	0.0	Not justified	Not justified
	258		0	Not justified	1.7	Not justified	Not justified
	262		0	Not justified	0.0	Not justified	Not justified
	266		0	Not justified	1.7	Not justified	Not justified
	270		0	Not justified	0.0	Not justified	Not justified
	274		0	Not justified	1.7	Not justified	Not justified
	248		0.0	0.0	4.7	8.8	
Harvest	284						

Notes: Dates in Julian days. Amounts in mm. Types according to STICS numeration. Boncompte et al. (2013). STICS (Inrae)

Chapter 3

Optimal conservation policy acceptance: The case of the Lleida plain

3.1 Introduction

In the previous chapter, we conducted a series of multi-objective optimization processes. In that regard, one of the most significant challenges for agricultural management is to optimize land-use decisions while ensuring sustainable provision of ecosystem services and economic output (Kaim et al., 2018). Thus, designing land-use plans that guarantee sustainable provision of ecosystem services and agricultural economic output is necessary. Moreover, it is essential to ensure that these land-use and conservation plans are accepted by landowners and other decision-makers. Strategies should not be developed without considering political and economic factors as they may generate more opposition and fewer chances of implementation (Polasky et al., 2007).

Policymakers should be aware of the trade-offs between environmental conservation and individual interests, particularly economic benefits, by monitoring the effects that optimal solutions at the landscape level may have on individual farmers. Many studies have analyzed and quantified these trade-offs (Barraquand Martinet (2011); Polasky et al. (2007); Polasky et al. (2008); Bateman et al. (2013)). Kaim et al. (2021) devised a land-use multi-objective optimization algorithm between biodiversity conservation and social welfare, pointing out the importance of identifying winners and losers of any optimal solution for future work. However, few studies have considered the relevance of the acceptance issue explicitly. One exception is Pitafi et al. (2009), who modeled welfare gains from efficient allocation of groundwater over time and space with respect to the status quo, focusing on the importance of opposition from losers, suggesting the possibility that they may be compensated by winners. However, to the best of our knowledge, this issue has not yet been analyzed from a spatially explicit agricultural policy perspective aimed at protecting biodiversity. We consider this to be crucial since conflicts between biodiversity conservation and agricultural activities have been found to be relevant, with several examples of opposition to policy implementation, such as in France in 1996, when the implementation of *Natura 2000* was suspended (Alphandery & Fortier, 2001), or in Germany, where strong local opposition has often emerged (for instance, in 1999, shortly after the designation of the Elbtalaue nature park, a regional group successfully managed to have it cancelled) (Stoll-Kleemann, 2001). In many countries, public demonstrations and organized gatherings opposing the designation of protected areas have become a prevalent occurrence (Pretty & Pimbert, 1995). In alternative instances, such as those observed in north-western European nations, the imposition of rigorous legal restrictions on the expansion of intensive agriculture has driven farmers from these areas to migrate to Eastern European countries in pursuit of land acquisition and the reorganization and intensification of agricultural operations. This chain of events has precipitated political, social, and environmental discord in Eastern European nations (Konecny, 2004), leaving the issue unresolved. Our goal is to show, in a specific spatial setting - the Lleida plain -, that biodiversity preservation often requires the existence of losers and to propose policy schemes likely to facilitate the implementation of measures that allow both biodiversity and economic sustainability of agricultural areas.

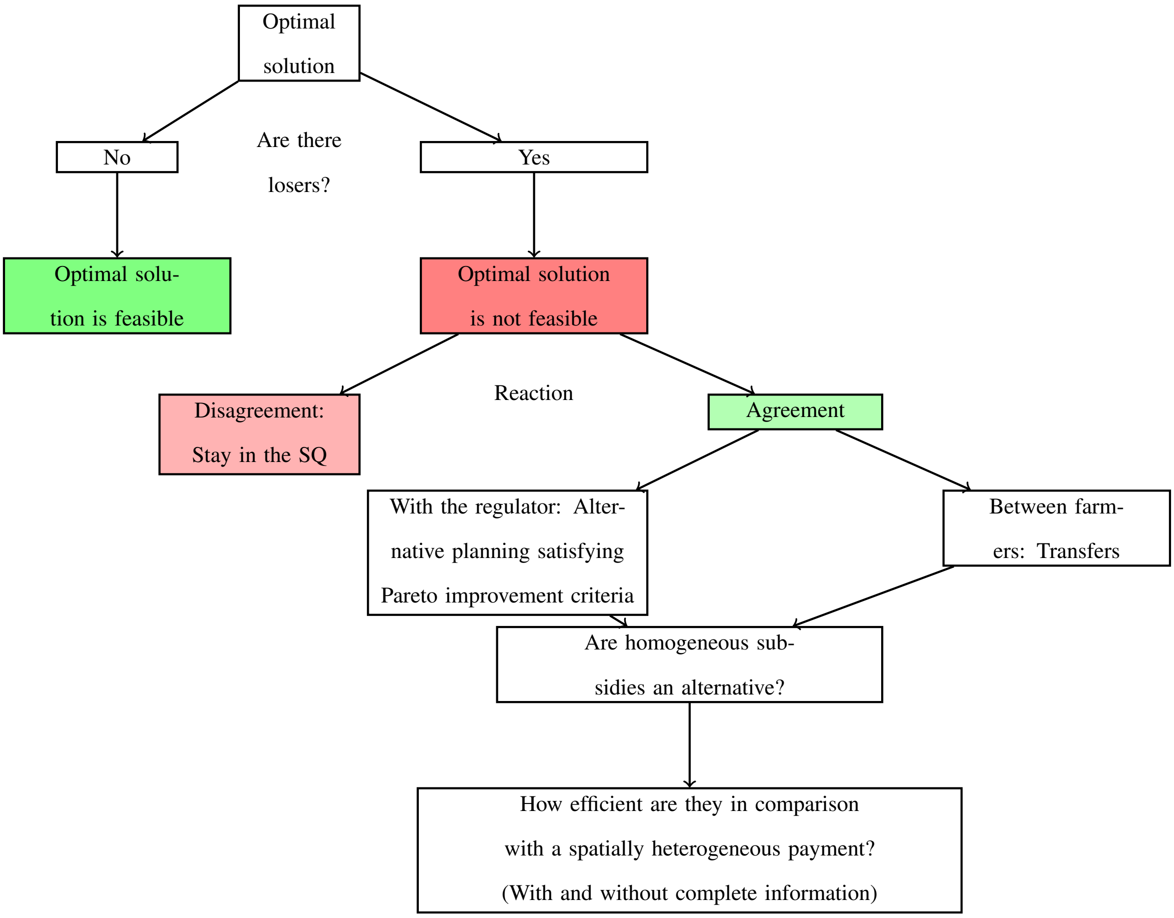
The approach of this study is based on the spatially explicit model of land use selection presented in Chapter 2, where we account for optimal multi-criteria analysis. The resolution of this issue involves maximizing the economic value of agriculture while adhering to specific constraints regarding bird habitat requirements, which is referred to as *Conservation goals* throughout this paper. To achieve this, we simulate crop yields using the STICS crop growth model (Brisson et al., 2008) and assess the impact of agricultural land use on birds through habitat suitability indexes. The outcome of the maximization problem by a regulating agency may imply winners and losers with respect to their status quo payoffs.

We consider several mechanisms capable of reducing the losers-winners ratio or directly satisfying the Pareto improvement criteria among farmers - situation where all farmers win at least the status quo values -. The paper firstly explores compensation mechanisms where the regulator proceeds “ex-post”. An ex-post mechanism serves as a corrective tool applied after the resolution of the unconstrained problem, aiming to satisfy the Pareto improvement constraint requirements. In particular, we consider a system of self-financed transfers where winners of the policy directly compensate losers according to a rule. The compensation rule can be proportional or egalitarian. It can be proportional, in the sense that winners pay losers proportionally to their obtained profits by ensuring their status quo profits. On the other hand, the implementation of the Nash Bargaining Solution is likely to ensure that each farmer receives their status quo profit plus an equal share of the surplus created. Additionally, the compensation can be complete, in the sense that winners and losers bargain over the entire surplus that will have been created. In our second mechanism we justify the feasibility of introducing an “ex-ante” explicit Pareto improvement constraint into the regulator’s maximization program which ensures that all farmers will be winners and allows us to measure the regulator’s economic cost to ensure that every farmer receives at least their status quo profit. An ex-ante mechanism is deemed to exist if the set of constraints is included in the optimization problem and may consequently restrict the optimal land-use pattern. In all cases, we show how these compensation schemes impact other variables of interest: low-intensification area, required budget, water use, and agricultural economic value.

However, the instruments employed by the Common Agricultural Policy (CAP) typically resemble a homogeneous subsidy system (where a constant subsidy per hectare is granted to every farmer that grows a specific crop) financed by an exogenous budget, representing an ex-ante approach. In this study, we compare the performance of the previously commented mechanisms with a stylized homogeneous subsidy system to rainfed winter cereal, evaluating its impact on biodiversity conservation, agricultural economic value, budget requirements, and water usage. We also assess the potential inefficiencies generated by this mechanism and explore how these inefficiencies may be reduced through the use of spatially explicit heterogeneous payments to losers - where payments are based on losses rather than land use. This analysis considers both scenarios where the regulator has complete information about individual losses, and where this assumption is relaxed. Under conditions of incomplete information, we assume that the regulator estimates losses and compensates farmers accordingly. In addition to these outcomes, we also examine the effects on social welfare using an alternative measure proposed by Sen (1973) that accounts for inequality levels in policy-induced gains among farmers.

Considering all this, this chapter is structured as follows: we begin by presenting the economic and the ecological models and define the maximization problem. We then explore whether there are any losers under optimal solutions to the maximization problem. Subsequently, we compare these outcomes with those resulting from the transfer system and the agreement with the regulator. We then contrast these results with those of the homogeneous payment, followed by an examination of the outcomes of the heterogeneous payment both with and without complete information. Additionally, we extend our analysis by comparing results while considering not only total economic changes but also the inequality of gains among farmers. The conceptual diagram of the chapter can be found in Figure 3.1.

Figure (3.1) KEY POINTS' OVERVIEW



Notes: SQ: status quo

3.2 Winners and losers

The conflict between bird conservation and agricultural development has been a persistent issue in many regions, including the one under study in this chapter. Balancing these competing interests requires careful consideration and analysis. In this context, the case presented in this chapter is particularly relevant, as it provides an opportunity to explore the trade-offs between preserving biodiversity and promoting economic growth through agriculture but emphasizing the potential negative impacts on farmers and developing solutions that take into account the needs of all stakeholders. This chapter aims to contribute to the development of sustainable approaches addressing this complex issue.

3.2.1 Benchmark maximization problem under biodiversity scenarios

As in the previous chapter, we aim to maximize the economic value (EV) of agriculture in the Lleida plain while taking into account the conservation requirements of a bird community. The problem is spatially explicit, as location affects crop yields due to varying climate and soil conditions. Beginning with a basic framework (benchmark), we modify it to incorporate alternative requirements.[1] For each framework, the optimization problem is solved under different conservation goals. A summary of the mathematical terms used throughout the paper is presented in subsection 3.A.

Consequently, the *Benchmark* optimization problem can be defined as [2]:

$$\begin{aligned} &\max_{x \in X} EV(x) \\ s.t. &\sum_{j=1}^{J} \sum_{r=1}^{R} x_{irj} = 1 \quad \forall \quad i \\ &x_{ijr} \in [0,1] \quad \forall \quad i,j,r \\ &\sum_{s=1}^{S} Z_s \geq Z^* \end{aligned} \tag{3.1}$$

where Z_s is a binary variable indicating whether the species s has obtained the conservation goal ($Z_s = 1$ if yes, 0 otherwise). Z^* is the number of species to be protected (a threshold). We include a total of 83 birds with presence in the region ($S = 83$), 11 of them being classi-

[1] Analogously to Chapter 2, we take as our main scenario the case where the *SPAs* are immobile. However, supplementary information relaxing this assumption can be found in subsection 3.B.

[2] This problem is the same as in the previous chapter. We enter it here to facilitate consultation.

fied as *Steppe* birds.

The set of optimal crop and management decisions can be defined as a vector $x^* = [x_1^*, x_2^*, \ldots, x_I^*]$ of dimension $1 \times I$ where $X : x^* \in X$ is the set of possible and feasible choices and x^* is the optimal solution which represents the set of land uses (x_{ijr}) selected at the landscape level which solve the maximization problem under certain constraints.

Note that the biodiversity constraint in Equation 3.1 corresponds to a triage approach, where bird conservation priorities are not considered - all species are equally important -, and consequently the less costly birds or combination of birds will be protected first - only the number of species is valued. Although it is possible to perform the optimization process for any arbitrary level of conservation goals, i.e., for any number of species above the threshold Z^*, we have chosen to focus our results only on three easily interpretable objectives. This is because presenting results for all possible levels of conservation goals would be difficult to interpret and tedious.[3] First, we consider an indiscriminate conservationist approach scenario where *all species* are conserved ($Z^* = 83$). Second, we include a *non-conservationist* approach ($Z^* = 0$) where there is not any policy oriented to bird conservation.[4] Finally, we include a conservationist policy only oriented to *steppe birds* ($Z^* = Steppe$). In this case, the constraint can be defined as a math equality expressing that those species catalogued as *steppe* must be conserved - independently from the fact that other species may also be conserved with this objective.[5]

We use a habitat suitability index to determine whether or not a species is conserved. We adopt the same methodology used in subsection 2.3.2 to describe the land use impact

[3]However, when necessary, we do construct Pareto Frontiers for all values of Z^* ranging from 0 to 83.

[4]This does not mean that no species is conserved, but that policies are not oriented to conservation and, as a result, any preserved species in the resulting scenario is merely circumstantial.

[5]Mathematically, steppe birds can be understood as a subgroup of species $S_{steppe} : S_{steppe} \subseteq S$. Therefore, the problem is formulated with a modified conservation constraint:

$$Z_s = 1 \qquad \forall \qquad s \in S_{steppe}$$

.

on biodiversity. In summary, for each of the 83 bird species our index calculates the habitat suitability of each of the 15 land uses listed above.[6] During the optimization process this index is calculated for each species and location decision after each optimization round, and an aggregate index is computed for each species s as the average of the indices of each location. For each species, if its aggregate suitability score is equal to or above the status quo score (i.e. better in comparison with the average value of the land uses nowadays), we consider that the species s is improving its preservation level ($Z_s^* = 1$). We repeat this process for each species, and our final biodiversity measure is the number of species above this threshold: $Z^* = \sum_{s=1}^{S} Z_s$. To simulate crop yield formation, and consequently the economic value, we use the crop growth simulation model *STICS* for each land use option in each cell.

A spatial configuration x^* is said to satisfy the *Pareto improvement criteria* (PI) if this new allocation results in an improvement over the status quo for each farmer i [7]:

$$EV_i(x_i^*) \geq EV_i(x_i^{SQ}) \quad \forall \quad i \tag{3.2}$$

When the optimal solution x^* does not satisfy the *Pareto improvement criteria* (i.e. not all farmers benefit from the policy), there are winners and losers as occurs in most collective optimization problems. The definitions of *winner* and *loser* in this study is established by the difference between the outcome of the optimization problem and the outcome of the status quo. For that, we can define the indirect utility function of a farmer i as $V_i = V_i(EV_i(x_i^*)) = EV_i(x_i^*)$, where $EV_i(x_i^*)$ is the economic value of the farmer associated with their corresponding optimal land use choice x_i^*. Since the farmer's solution is not about where to cultivate, but the crop and the management plan, the losers and winners are identified by the following rule: A farmer is a *winner* if:

[6]The variables considered to construct the habitat index account for the management practices of each land use and how they affect the resources demanded by each species regarding foraging and nesting.

[7]It is important to note that this Pareto improvement may be confused with the habitat suitability Pareto improvement constraint (as shown in Equation 3.1). However, unless explicitly stated otherwise, throughout this paper we will use the term 'Pareto improvement' to refer only to the farmers' perspective, while the biodiversity perspective will be differentiated through the categorization of conservation goals.

$$EV_i(x_i^*) \geq EV_i(x_i^{SQ}). \tag{3.3}$$

On the contrary, a farmer is a *loser* from the policy if:

$$EV_i(x_i^*) < EV_i(x_i^{SQ}). \tag{3.4}$$

where $EV_i(x_i^{SQ})$ accounts for the economic value of the status quo land use choice x^{SQ}. In our framework, the status quo value has been obtained using land use digital maps from *DUN (2019)*[8] and simulated crop yields - with their associated economic outcomes -. Consequently, we have information on crop selection and intensification that we used to approximate the EV for each cell - and its corresponding SQ values -.

Moreover, the optimal solution of the maximization problem under different specifications is analyzed considering different criteria (Table 3.1). Then, beyond considering the number of winners and economic value, we also considered the low intensification areas - that we define as the total area devoted to rainfed winter cereals (km^2) -, required budget to implement the policy and water usage (hm^3) in the optimal scenario.

[8]Declaración única agraria. DARP 2019.

Table (3.1) VARIABLES CONSIDERED

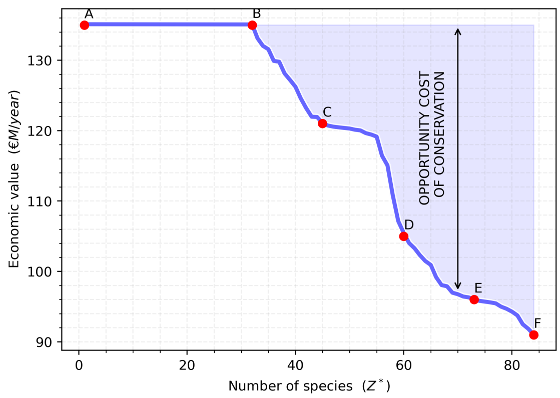
Criteria (measurement variable)	Description
(1) No. of winners	Represents the number of farmers (cells) aiming to adopt the policy. We assume that the farmers will support it if it is economically profitable for them with respect to their status quo - the economic profitability that they obtained before the policy was applied -. This variable can be interpreted as a policy acceptance proxy, since as it increases, less opposition is expected.
(2) Total economic value (€ M)	Indicates the sum of the agricultural economic value among farmers, deducting potential subsidies.
(3) Low-intensification areas (km^2)	Indicates the total area devoted to rainfed winter cereals.
(4) Budget/Subsidy (€ M)	Shows the money spent by the external agent (e.g. agency regulator) to compensate farmers for the opportunity costs of conservation.
(5) Water usage (hm^3)	Water scarcity is expected to be a major problem during the following decades. Consequently, we also analyze the impact that optimal economic land use planning constrained by conservation goals and subject to subsidies has on irrigation water needs. It is computed as the sum of the water in each cell.
(6) Total losses (€ M)	Amount of loss of economic profit among losers.
(7) No. of species conserved (Z^*)	Number of species which are at least as in the status quo.

3.2.2 Results: *winners' and losers' pay-offs under conservation goals*

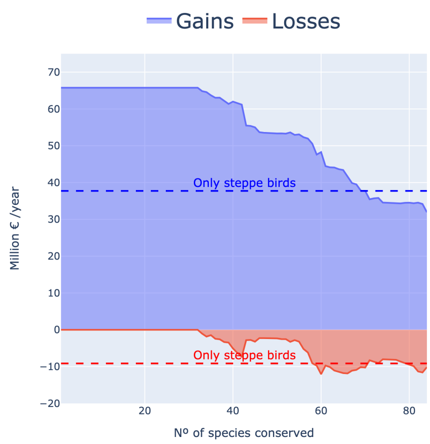
We solve the optimization problem described in Equation 3.1 for each conservation objective Z^*. Results are presented for all levels of Z^* (i.e., from 0 to 83 species) to provide a smoother representation of the winners and losers - although later, as mentioned, we will only use 3 different conservation goals -. The set of solutions is grouped to construct a Pareto Frontier (see Figure 3.2a), where each point corresponds to the maximum agricultural economic value (EV) that can be obtained under a specific Z^*.[9] This representation is decreasing with respect to Z^* since solutions associated with more demanding biodiversity goals are sub-

[9]For a more in-depth analysis of the land use transitions and spatially explicit changes see subsection 3.C.

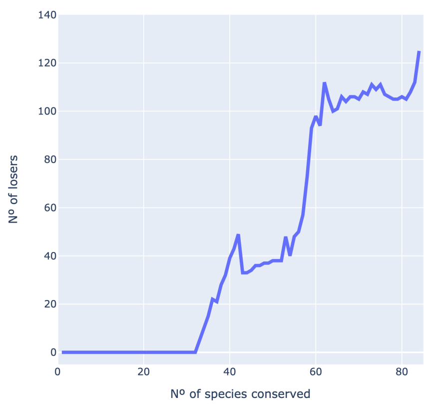
Figure (3.2) ECONOMIC VALUE CHANGE OF LOSERS AND WINNERS AND NUMBER OF WINNERS AFTER THE OPTIMIZATION PROCESS



(a) Pareto frontier



(b) Economic value (Winners-losers)



(c) No. of losers

Notes: Dashed lines represent values associated to the sole conservation of the 11 steppe species in the territory. Dashed line in red is associated with the losses corresponding to the optimal landscape when conserving steppe birds only. Analogously, blue line represents the wins.

ject to more stringent constraints, thereby reducing the set of possible solutions. It reveals that there is a trade-off between biodiversity conservation through habitat adaptability and agricultural economic outcome, which does not decrease smoothly since it depends on the combination of species' preferences for different habitats and land uses. For each point on the Pareto Frontier, it is possible to calculate the net gains with respect to the status quo payoffs. We compute total economic value in the status quo and find it to be € 69.3 M annually. For each point on the Pareto Frontier, the net gain is positive since the optimal solution must be at least as high as the status quo. Additionally, this comparison enables us to differentiate the net economic gain obtained by winners from the losses suffered by losers.

Nevertheless, positive economic value net gains with respect to the status quo do not necessarily imply that each farmer benefits from the new allocation. Some farmers may be adversely affected. The blue area in Figure 3.2b represents the economic gain of the winners, while the red area represents the economic loss of the losers under each specific level of biodiversity constraints denoted by Z^*. Graphically, it can be observed that the net gain is positive, as described above, since for each Z^* the distance from zero (along the vertical axis) to the blue line is greater than the distance to the red line. As Z^* increases, two simultaneous effects become apparent: first, the economic gain of the winners decreases - blue area in Figure 3.2b - since higher biodiversity constraints restrict the number of potential solutions, reducing the total potential for net improvement. Second, the loss of the losers tends to increase - red area in Figure 3.2b - because environmental constraints require some farmers to choose less profitable options.

As the set of biodiversity constraints becomes more demanding, maximizing EV at the landscape level requires the existence of losers. This is illustrated in Figure 3.2c, where, as the conservation objectives become more demanding along the horizontal axis, the number of losers increases and the amount of winners decreases. For example, when all species are required to be conserved ($Z^* = 83$), the number of winners is 477 and the average gain of the winners is € 342/ha[10]; however, the number of losers is 125 (26.2% of the total), where each loser losses on average € 453 /ha. As a consequence, the likelihood of facing implementation obstacles increases. Nevertheless, it is worth noting that there are no losers - and therefore no losses - for $Z^* \leq 31$ (point B). This indicates that there are 31 species whose conservation does not incur any cost, and therefore their interests are aligned with those of the farmers.

Now we focus solely on the conservation of steppe birds. To facilitate its graphical visualization, the EV obtained by preserving the 11 steppe species is represented by a dashed horizontal line in Figure 3.2a. It should be noted that its value is close to the economic value obtained when all species ($Z^* = 83$) are required to be preserved, suggesting that preserving steppe birds involves a high conservation cost. The gain of the winners and the loss of the losers are represented by dashed horizontal lines, in blue and red, respectively, in Figure 3.2b. As before, the gain of the winners is greater than the loss of the losers. In this case, the number of losers decreases to 99 (16.4%) as shown in Figure 3.2c. This translates into an average gain of € 384/ha of the winners and an average loss of € 517/ha for the losers. Thus, although preserving all bird species results in a higher number of losers compared to preserving only steppe birds, the loss per hectare is lower.

These findings support the notion that in our case study, policies that do not consider the existence of losers - despite being designed to be optimal - may have a high risk of opposition. It should be expected that practices that involve a loss of income lead to enforcement difficulties. Additionally, bird sustainability cannot be attributed to a single identifiable set of individuals but rather to the joint action of all the agents in the area, since it is the entire habitat that facilitates their sustainability. Therefore, no agent should lose on applying conservationist farming strategies, and therefore compensatory policies should be implemented. Next we present and analyze the performance of several mechanisms.

[10] We obtained this number by dividing the total economic value by the number of winners for each Z^*

3.3 Enforcing agents' participation

Our findings support the notion that to maximize social economic returns requires some farmers to be losers. That is, maximizing social economic returns under high conservation goals requires the existence of losses and losers, which are often the result of the land management changes required to meet conservation goals. These changes represent transitions from highly productive options to less profitable options. In particular, our optimal solution forces a number of farmers to see their profit reduced from their status quo. In our case study, we can identify agents who would be required to shift to less profitable regimes and without whom it would be unfeasible to guarantee the species sustainability but who, ironically, are forced to carry out agricultural practices that lead them to losses.

We hereby propose two mechanisms that facilitate the attainment of a solution devoid of any losers. We then analyze the performance of these mechanisms that can lead to an allocation of agricultural practices that represents a Pareto improvement over the status quo. In such a scenario, no agent shall possess economic grounds for non-participation, as all agents shall either maintain or augment their profits relative to the status quo. The first entails a self-financed transfer system between winners and losers. In the second mechanism the regulator proposes a specific allocation of farming practices such that it ensures, for each location i, that total economic profits are maximized and no agent is worse off than in the status quo - in other words, a solution without losers. Obviously, these two measures to attain high efficiency levels requires regulator complete information about wins and losses.

3.3.1 Agreement between farmers: compensation with self-financed transfers

The first mechanism to consider is based on a self-financed transfer system where winners compensate losers, and consequently requires the *Kaldor-Hicks* criterion (Hicks, 1939;

Kaldor, 1939) to be satisfied.[11] In section 3.2 we computed the solution of the optimization problem described in Equation 3.1. We have shown that our optimal allocation represents an increase in the total economic value EV respect to status quo. This implies that the total economic gains of the winners exceeds the loss incurred by the losers, and that the resulting winners can compensate the resulting losers. Therefore, such a solution could be enforced thorough a self-sustained transfer systems able to ensure that there are no losers among farmers.

To implement this compensation mechanism, we assume that each winner will pay a transfer (t_w) to the losers, and each loser will receive a transfer (τ_l) from the winners. Furthermore, the transfers paid by winners should not be too high to result in a loss with respect to the status quo and, on the other hand, the transfers received by the losers have to ensure that they will receive at least their status quo payoffs. Based on the definitions of winners and losers stated previously (in Equation 3.3 and Equation 3.4), we assume that each winner will pay at most a transfer t_w^{max} with $w = 1, \ldots, W$ to the losers such that:

$$t_w^{max} = EV_w(x_w^*) - EV_w(x_w^{SQ}) > 0, \tag{3.5}$$

Similarly, each loser will receive at least a transfer τ_l^{min} with $l = 1, \ldots, L$ such that:

$$\tau_l^{min} = EV_l(x_l^{SQ}) - EV_l(x_l^*) > 0. \tag{3.6}$$

where x_w^* represents the optimal allocation solution of a winner and x_w^{SQ} represents the status quo allocation of a winner. Analogously, x_l^* represents the optimal allocation solution of a loser and x_l^{SQ} represents the status quo allocation of a loser. Note that a policy will increase social welfare, if implemented, if and only if:

$$\sum_{w=1}^{W} (EV_w(x_w^*) - EV_w(x_w^{SQ})) > \sum_{l=1}^{L} (EV_l(x_l^{SQ}) - (EV_l x_l^*)) \tag{3.7}$$

[11] According to the Kaldor-Hicks criterion, an outcome is considered more efficient if those who benefit from a change could hypothetically compensate those who are made worse off by the change so that everyone is no worse off than before.

as occurs in our case and which is equivalent to:

$$\Phi = \sum_{w=1}^{W} t_w^{max} - \sum_{l=1}^{L} \tau_l^{min} > 0, \tag{3.8}$$

Thus, our optimal allocation will allow a self-sustained transfer system to be implemented that satisfies the Kaldor-Hicks criterion, as the resulting winners can compensate the resulting losers. Further, note that this is an ex-post mechanism; therefore, the optimization goal remains the same as our benchmark problem (Equation 3.1). The solution x^* is identical, and transfers are only required to ease the implementation of this optimal allocation. Further, as the allocation to be implemented is optimal, this self-sustained transfer system does not presents any inefficiency loss. The total payoff $EV(x^*)$ is the same as the total payoff of the optimal solution, but once the transfer mechanism has been implemented there are no losers.

Additionally, we can calculate the net payoffs for winners and losers that result from this transfer mechanism. We calculate this net payoff as the difference between the payoffs in case of agreement with the implementation of the mechanism EV^a, and the payoff in the case of disagreement - stay in the SQ - with the same implementation, EV^d.

In case of agreement, the payoffs of the winners and losers are, respectively:

$$EV_w^a = EV_w(x_w^*) - t_w, \quad EV_l^a = EV_l(x_l^*) + \tau_l. \tag{3.9}$$

where agents grant or receive a transfer. Meanwhile, in case of disagreement, both payoffs will coincide with their status quo allocation payoff:

$$EV_w^d = EV_w(x_w^{SQ}), \quad EV_l^d = EV_l(x_l^{SQ}). \tag{3.10}$$

Therefore, the net payoffs for winners of a transfer t_w would be:

$$EV_w^a - EV_w^d = EV_w(x_w^*) - EV_w(x_w^{SQ}) - t_w \tag{3.11}$$

where $t_w^{max} \geq t_w$. Therefore, $EV_w^a - EV_w^d \geq 0$, that is, the net payoff for winners of a transfer t_w would be positive (equal to or larger than zero). And the net payoff for losers would be:

$$EV_l^a - EV_l^d = EV_l(x_l^*) - EV_l(x_l^{SQ}) + \tau_l. \tag{3.12}$$

where $\tau_l^{min} \leq \tau_l$. Also $EV_l^a - EV_l^d \geq 0$, that is, the net payoff for losers of a transfer τ_l would be equal to or larger than zero. Therefore, there will be no losers. This approach enables us not only to compute who should pay and who should be compensated but also the amount of these compensations. [12]

3.3.2 The regulator solution

Here we show an alternative solution proposed by the regulator without using transfer mechanisms or compensation payments. It can be interpreted as a proposal from a regulator that takes into account the risk of opposition.

The satisfaction of the *Pareto* improvement criteria - which implies that every single agent i obtains at least status quo profits - can be forced by adding to the maximization problem (Equation 3.1) a new set of constraints:

$$EV_i(x_i^+) \geq EV_i(x_i^{SQ}) \quad \forall \quad i \tag{3.13}$$

where we define x^+ as the solution to the original problem adding this new set of constraints. Nevertheless, x^+ does not necessarily equal x^*, since the accomplishment of Equation 3.13

[12] To calculate individually explicit transfers paid by winners, we specify certain redistribution rules in section 3.6.

will limit the total set of options and decrease the maximum attainable economic value for certain biodiversity constraints. As such, we expect that ensuring that there are no losers in the optimal solution will be costly and we aim to quantify this cost.

As before, we compute the net payoff, as the difference between the payoffs received by farmers in case of agreement EV^a and the payoff received in the case of disagreement EV^d. A resulting configuration under Equation 3.2 would consist solely of winners, rendering the distinction between winner and losers irrelevant.

In case of agreement, the payoffs of the winner will be EV_w^a:

$$EV_w^a = EV_w(x_w^+), \tag{3.14}$$

while in case of disagreement they will be EV_w^d

$$EV_w^d = EV_w(x_w^{SQ}). \tag{3.15}$$

As there are no losers, the net payoffs are then:

$$EV_w^a - EV_w^d = EV_w(x_w^+) - EV_w(x_w^{SQ}) \geq 0 \tag{3.16}$$

3.3.3 Results

Our analysis reveals relatively minor differences between the two types of mechanism, (Table 3.2) for the three conservation goals considered. They arise because an agreement among farmers does not limit the options available to each individual farmer as long as each loser is compensated and no winner incurs losses with respect to the benchmark solution in subsection 3.2.2 -. In contrast, the regulator's alternative proposal restricts the set of solutions, as some management regimes are precluded in certain cells if they result in lower profitability. Consequently, the farmer's agreement mechanism results in larger EV than the regulator solution.

Table (3.2) OPTIMAL LANDSCAPE OUTCOMES AGREEMENT VS. STATUS QUO

Status quo (Disagreement):			
Economic value (€ M)	69.3		
Water usage (hm^3)	425.9		
Low-intensification area (km^2)	469		

	$Z^* = 0$	$Z^* = Steppe$	$Z^* = 83$
Regulator solution:			
Economic value (€ M)	135.1	95.9	89.6
Water usage (hm^3)	526.3	380.1	427.8
Low-intensification area (km^2)	484	556	492
Agreement between farmers:			
Economic value (€ M)	135.1	97.9	91.0
Water usage (hm^3)	526.3	376.0	427.7
Low-intensification area (km^2)	484	568	492

Notes: The status quo is the current situation that is not subject to any conservation goal Z^*. Then, in this case, there would be a unique value for each variable. We do not present the number of winners since, under these scenarios, the model is designed to provide solutions without losers.

Our findings suggest that circumventing the issue of losers through an alternative proposal devoid of transfers incurs a comparatively negligible cost in contrast to the benchmark solution - and consequently those of the transfer system -.[13] In the absence of biodiversity constraints ($Z^* = 0$), the solution proposed by a regulator has no bearing as, under this conservation objective, each farmer makes their individual profit-maximizing decision, rendering them all winners. Conversely, when conserving steppe birds or all species, the negative impact of complying with the regulator's solution with respect to the benchmark scenario (or under transfers) is -2% and -1.5%, respectively, equivalent to a decrease in eco-

[13] Refer to subsection 3.D for a more detailed analysis.

nomic value of € 2M and € 1.4M. This can be translated into a decrease of only € 14/ha and € 10/ha respectively. Then, the cost of satisfying the *Pareto* improvement constraint is higher when targeting only steppe species for preservation.

Nevertheless, the biggest differences are observed between agreement and disagreement scenarios. Our analysis shows that economic value increases substantially for all conservation goals - evidently, the net increase is more pronounced if the conservation goal is lower -. When $Z^* = 83$ economic value increases 29% and 31% if there is an agreement with the regulator and between farmers, respectively. [14] If the conservation goal $Z^* = Steppe$ is target, these differences increase, with 38% and 41%, respectively. Considering an agreement between farmers, this implies an increase relative to the status quo of € 568/ha if $Z^* = 0$, € 247/ha if $Z^* = Steppe$ and € 187/ha if $Z^* = 83$. [15] On the other hand, if the agreement is with the regulator, there would be an increase relative to the status quo of € 568/ha if $Z^* = 0$, € 229/ha if $Z^* = Steppe$ and € 175/ha if $Z^* = 83$.

It is noteworthy that this enhancement is not confined to economic value alone, but also encompasses an expansion of low-intensification areas and a reduction in the water required for irrigation - especially for $Z^* = Steppe$. In order to preserve steppe birds without incurring any losers, the low-intensification areas increases from 18% in the case of an agreement with the regulator to 21% if it is between farmers. Analogously, this translates into a 16% and 18% reduction of water for irrigation (46 hm^3 and 41 hm^3), respectively. To provide a clearer intuition, this represents approximately 5% and 14% of the total irrigation and domestic water consumption in Catalonia in 2018, respectively (INE 2018; IDESCAT, 2018). If $Z^* = 0$, total water usage increases by 23%. Slight increases are also observed if $Z^* = 83$ (+0.4% for both types of agreements). This is noteworthy because it shows that optimal solutions coupling lack of losers and species protection do not necessarily mean increasing

[14] They have been calculated simply by obtaining the relative change in economic value from the status quo to the agreement scenario. For instance, if the agreement is with the regulator: $\Delta\% = \frac{89.6-69.3}{69.3} \times 100 = 29\%$

[15] In this case, considering that all parties are winners, the total gain has been calculated by dividing it by the total number of hectares in the landscape. For instance, in the case of $Z^* = 0$ and an agreement among farmers: Δ €/ha $= \frac{(135.1-69.3)\times 10^6}{115675} = €568/ha$, where the denominator indicates the total number of hectares in the study area.

water usage. Furthermore, we demonstrate that protecting all species while simultaneously increasing economic value without the existence of losers results in a negligible increase in total water usage.

Related with the transfer mechanism, we can calculate the proportion of gains obtained by winners required to compensate the losers, (P_w), as:

$$P_w = \frac{\sum_{l=1}^{L}(EV_l(x_l^{SQ}) - (EV_l x_l^*))}{\sum_{w=1}^{W}(EV_w(x_w^*) - EV_w(x_w^{SQ}))} \tag{3.17}$$

Interestingly, we found that if the agreement between farmers is carried out when $Z^* = 83$, the compensation to the losers can be achieved with only $P_w = 31.8\%$ of the winners' gains, which confirms the feasibility of the transfer system and demonstrates the fulfillment of the *Kaldor-Hicks* optimal criterion. This percentage decreases to 24.3% if the goal is to preserve only steppe species. This proves that it is theoretically possible to achieve an optimal solution without losers by redistributing the gains under reasonable values. However, meeting this criterion results in changes in the distribution of economic profits. [16]

3.4 Sustainability through CAP-type payment schemes

3.4.1 Theoretical framework

The mechanisms discussed in the previous section address two problems, preserving biodiversity and obtaining Pareto-improving allocations, where there are no losers. However, those mechanisms are not widely implemented in agricultural policies. On the contrary, as is the case in the European CAP, the agricultural policy is based on payment schemes externally funded that are provided on a per hectare basis to land use choices characterized

[16] See subsection 3.E for the details.

by low intensified management. The new reform of the CAP (2023-30) continues to reaffirm its use of decoupled direct payment systems to promote agricultural practices aimed at protecting biodiversity and the environment (European Commission, n.d.). Next, we study the performance of a system of externally financed homogeneous subsidies that represent a fixed payment per hectare of land allocated to low intensified uses. These payments are often implemented to promote conservationist practices and to complement farm income. We call this a homogeneous subsidy since it is usually implemented as a payment per hectare which is the same for all farmers of a given area that follow the same conservationist practice. Furthermore, this payment is intended for low intensity and therefore low profitable land uses and acts as an income complement able to reduce the number of losers even if it were not designed to do so. It could be argued that it could reduce the number of losers as long as it is high enough or not mandatory. By testing various subsidy amounts, we aim to determine whether these objectives, birds conservation and Pareto-improving allocations, can be met through this approach. Additionally, we will compare the effectiveness of these subsidies to previously discussed mechanisms, such as a system of transfers between winners and losers and an agreement with a regulator. However, it is important to note that such subsidies may not be the most efficient means of achieving our goals. This is because they may result in over-payment to some farmers who would have been willing to accept lower compensation for their efforts to support biodiversity. As such, while subsidies can provide an incentive for farmers to adopt more sustainable practices, it is crucial to carefully consider their design and implementation in order to maximize their effectiveness. In the next sections we focus on this issue, introducing a set of heterogeneous subsidies.

We assume that the set of low-intensity choices encompassed the cultivation of barley and wheat without irrigation, and we represent them by x_{LI}.[17] Under a homogeneous payment, our economic value function $EV(x_i)$ is replaced with:

$$EV_i^{Subsidy}(x_i) = EV_i(x_i) + \upsilon_i \times a_i$$

[17]In subsection 3.F we test the results with a complementary subsidy to 150mm and 350mm barley and wheat.

where

$$\upsilon_i = \begin{cases} \bar{\upsilon} & if \quad x_i^* \in x^{LI} \\ 0 & \text{otherwise} \end{cases} \tag{3.18}$$

Note that $EV_i^{Subsidy}$ represents the economic value obtained in location i when a homogeneous subsidy scheme has been implemented. υ_i is an indicator function referencing whether or not location i is subsidized. This rule depends on the land use choice at location i. If a land use in location i is included in the set of low intensified options (say x_i^{LI}), as defined above, a lump-sum subsidy $\bar{\upsilon}$ (€/ha) will be granted to location i. Otherwise, it will be zero. a_i represents the number of hectares in location i. It can be repeated for different values of $\bar{\upsilon}$. We assume that the budget, BU, required to implement this policy is equal to the sum of the subsidies paid in each location - accounting for its arable land a_i - :$BU = \sum_{i=1}^{I} \upsilon_i \times a_i$. Consequently, the optimization problem can be defined as:

$$\begin{aligned} &\max_{x \in X} \sum_{i=1}^{I} EV_i^{Subsidy}(x_i) \\ s.t. &\sum_{j=1}^{J} \sum_{r=1}^{R} x_{irj} = 1 \quad \forall \quad i \\ &x_{ijr} \in [0,1] \quad \forall \quad i,j,r \end{aligned} \tag{3.19}$$

Note that we remove the conservation constraints. In this setting, both the number of species protected and the reduction in the number of losers are outcomes of the problem but not constraints. In the absence of subsidies, a farmer following this low-intensification practice will remain in the most profitable option x_i^*. On the contrary, the introduction of a subsidy υ_i can make that $EV_i(x_i^{LI}) + \bar{\upsilon} \times a_i > EV_i(x_i^*)$ and farmer i would choose option x_i^{LI} that is more profitable. Mathematically, a farmer would accept the arbitrary subsidy level $\bar{\upsilon}_i$

if:

$$EV_i(x_i^{LI}) + \bar{\upsilon} \times a_i \geq EV_i(x_i^*) \tag{3.20}$$

We define the efficient subsidy υ^* as the payment at which agent i will be indifferent between adopting a conservationist strategy - and receiving subsidy υ^* - or maintaining the most profitable strategy as:

$$EV_i(x_i^{LI}) + \upsilon_i^* \times a_i = EV_i(x_i^*)$$

Obviously, as the subsidy is homogeneous across i (farmers/locations), and each location presents a different potential crop yield, there is no single subsidy that can be efficient for all farmers i. For some farmers the subsidy υ_i may be greater than the minimum compensation required to follow a low-intensification option. To measure the performance of this instrument we need to measure the loss associated with overcompensation. The overcompensation of this homogeneous subsidy scheme will be measured for each farmer i using $OC_i(x_i^{LI})$, that we define as:

$$OC_i(x_i^{LI}) = max\{ \quad 0, \quad \min\{ \quad \bar{\upsilon} \times a_i, \quad \bar{\upsilon} \times a_i + EV_i(x_i^{LI}) - EV_i(x_i^*)\}\} \tag{3.21}$$

where $OC_i(x_i^{LI}) \in [0, \quad \bar{\upsilon} \times a_i]$. We can envision several possible outcomes: The first one occurs when the payoff associated with the low intensified option being subsidized $EV_i(x_i^{LI})$ is greater than the most profitable option $EV(x_i^*)$. In this case, no subsidy would be needed, farmer i would have implemented the optimal land use choice x_i^{LI} without any payment. In this case, the overcompensation is equal to the total subsidy paid $\bar{\upsilon} \times a_i$. The second one arises when $EV(x_i^*)$ exceeds $EV_i(x_i^{LI})$ and the difference between these two values is smaller, in absolute value, than the subsidy granted $\bar{\upsilon} \times a_i$. In this case, the subsidy will be able to incentivize the land use change, and farmer i will need to be compensated to implement the conservationist land use allocation x_i^{LI}. The overcompensation level will be equal to $(\bar{\upsilon} \times a_i - (EV_i(x_i^*) - EV_i(x_i^{LI})))$, a value between zero and the subsidy.

The third possible outcome appears when $EV(x_i^*) \geq EV_i(x_i^{LI}) + \bar{\upsilon} \times a_i$. In this case, the difference between these two values is greater than or equal to the subsidy. Then, the individually optimal strategy is not to accept the subsidy, as it returns a lower economic value. In such a case, the subsidy would not suffice to compensate - or exactly compensates - the loss of agent i when choosing conservationist strategies. In this case, there is zero overcompensation, that is $OC_i(x_i^{LI}) = 0$.

This measure can be utilized to compare and contrast with alternative policies, as elevated levels of inefficiency may exert a significant detrimental effect on the economy.

3.4.2 Results: the accomplishments of homogeneous subsidies

In our model, we exclusively implement per-hectare subsidies for rainfed crops, specifically barley and wheat (see Figure 3.3 for the results). We administer subsidies at varying levels, ranging from € 0/ha to € 3000/ha in increments of € 50. Although this range of payments may seem significantly broad in comparison to real-world payments, it has been deliberately chosen to represent the full spectrum of effects that such subsidies may have on the variables under consideration. It should be noted that, in this region, the current per-hectare value of CAP payments falls between € 191/ha and € 487/ha, depending on irrigation and crop type.[18] Furthermore, these payments are taken into account when optimizing solutions; thus, homogeneous subsidies should be understood as additional payments. Initially, it should be pointed out that homogeneous subsidies per hectare of a similar amount to the current CAP subsides fail to alter farmers' land use choices. This is to be expected, as CAP payments are generally designed to incentivize practices that are potentially less costly for farmers (e.g., crop rotation or the allocation of a small portion of land for environmentally

[18]Irrigated crops receive higher payments than rainfed ones, and permanent crops such as vineyards are also subject to specific payment schemes.

beneficial practices) than, for example, reducing irrigation as in our case. If the policy truly aims to effect large-scale transformations, higher compensations would be required.

Furthermore, we observe that the total economic value of crops produced in this area - calculated without including the rent provided by the subsidy - decreases as $\bar{\upsilon}$ increases (Panel 1, Figure 3.3). Essentially, an increase in per-hectare subsidy leads to a greater likelihood of farmers selecting less intensive, rainfed land uses, which are less profitable in the absence of subsidies. This decision is primarily made by farmers with low opportunity costs of not irrigating their land, who are generally those with a relatively small difference between the EV obtained from irrigated crops and that obtained from non-irrigated crops. Without subsidies for low intensification practices, the landscape will be dominated by highly intensified crops, such as irrigated corn, where those practices are viable. Consequently, $\bar{\upsilon}$ will need to be sufficiently high to compensate for the profit gap between this intensive crop and non-irrigated barley/wheat. This explains why low intensified areas (rainfed winter cereals) (panel 3, Figure 3.3) will require high subsidies to be increased. The same holds for other associated variables such as EV and water usage, which will only be affected if $\bar{\upsilon} \geq$ € 1000/ha (see Figure 3.3).

The number of conserved species is expected to increase with higher subsidies per hectare (Panel 5, Figure 3.3). As before, biodiversity goals will only begin to be achieved with subsidies over € 900/ha. Furthermore, we can see that out of the 83 species considered, the maximum number of birds that can be conserved simultaneously is 77, which is achieved with a subsidy of € 2350/ha (€ 225M of total budget). It should be noted that the graph exhibits a peak, indicating that increasing subsidies for these land uses do not necessarily result in the conservation of more species and also alter the composition of the species conserved. As such, it can be anticipated that lower subsidies will primarily benefit waterfowl or generalist species, while higher subsidies may disadvantage these species in favor of those more closely associated with the subsidized lands. Focusing on steppe birds (Panel 6, Figure 3.3), the number conserved increases monotonically, but only with a sub-

Figure (3.3) HOMOGENEOUS PAYMENT TO WINTER CEREALS IMPACT

Notes: Subsidies are paid to rainfed wheat and barley. X-axis format indicates: Payment to rainfed winter cereals (€/ha/year). Each dot corresponds to a specific solution of the problem with different payment levels.

sidy of € 2100/ha (€ 180M of total budget) would all 11 steppe species considered be conserved. Moreover, we find that for $\bar{\upsilon} \leq$ € 950/ha, only 2 steppe bird species are conserved. Consequently, we observe that homogeneous subsidies will mostly fail in their goals of conservation under reasonable budgets.

Table (3.3) HOMOGENEOUS SUBSIDY OVERCOMPENSATION

	Total overcompensation (€ M)	**Farmers overcompensated** (nº)
$\bar{\upsilon}$ =*100*	10.7	242
$\bar{\upsilon}$ =*200*	14.1	242
$\bar{\upsilon}$ =*500*	24.3	242
$\bar{\upsilon}$ =*1000*	47.5	242
$\bar{\upsilon}$ =*1500*	89.9	303
$\bar{\upsilon}$ =*2000*	164.6	414
$\bar{\upsilon}$ =*3000*	346.3	601

Notes: $\bar{\upsilon}$: *Subsidy to rainfed barley/wheat (€/ha).* Number of farmers corresponds to the number of cells - farmers -. Total overcompensation is the amount of money spent on subsidies that was not required by farmers to participate, since it exceeds the most profitable economic alternative payoffs.

As previously mentioned and in relation to the previous results, homogeneous payments are expected to be inefficient, implying that compensation payments do not equal their real opportunity costs. We quantify this inefficiency in monetary terms (Equation 3.21) and the number of overcompensated farmers under a set of subsidy levels (Table 3.3). For low $\bar{\upsilon}$, the policy will only overcompensate those who were already going to undertake those land uses - primarily because they are inside *SPAs*, and therefore no intensification is allowed -. Only with a high subsidy of approximately € 1000/ha do we see that they begin to affect other farmers' land use decisions. More farmers will accept to change their land use as more farmers are being overcompensated, and therefore total overcompensation will also increase. These overcompensations are indeed substantial, particularly for values where we can see that the policy is truly effective in terms of conservation, with values that can even exceed the agricultural economic value under the most profitable option.

As a result, subsidizing less profitable agricultural options may meet conservation goals, but we find that equally ambitious goals could be more easily achieved through agreements, and without inefficiencies. Moreover, these agreements do not discourage the production of economic value, while a homogeneous payment does.

3.5 Spatial heterogeneous payments for policy acceptance

Given the inefficacy of homogeneous subsidies, which need to overcompensate numerous farmers to achieve conservation objectives, we propose a spatially explicit heterogeneous subsidy. Under complete information, it would be possible to implement a mechanism that enhances efficiency and compensates losers. In this case, the regulator could identify the economic implications of land use change that farmer i must undertake to reach the socially optimal allocation $EV_i(x_i^*)$ from the status quo $(EV_i(x_i^{SQ}))$. This difference could be different for each farmer, this variation stemming from the unique opportunity cost of conservation for each farmer due to specific conditions such as soil or climate variables, and from individual status quo choices. However, complete information is not often a realistic assumption. Next, we assume that the regulator knows the economic value of each land use option for each farmer and we analyze what the performance of such an instrument would be. The assumption of perfect knowledge is then removed and, following Salle (2019), we analyze the performance of a heterogeneous subsidy under imperfect information.

3.5.1 Target compensation under complete information

Unlike a spatially homogeneous payment scheme, a spatially heterogeneous payment scheme allows for different payments for each farmer, even when making the same change in land use. The regulator may then strictly subsidize each farmer i, who will receive a per hectare subsidy $\bar{\upsilon}_i$. A heterogeneous subsidy for a farmer i can be defined as:

$$\bar{\upsilon}_i = max\{EV_i(x_i^{SQ}) - EV_i(x_i^*), 0\} \tag{3.22}$$

In such an allocation, farmer i will be indifferent between remaining in the status quo or adopting the socially optimal conservation strategy. The regulator can ascertain the precise amount of the compensation subsidy required ($\bar{\upsilon}_i$) as it has complete information on the biophysical characteristics of cell i.

Our heterogeneous payment is modeled as an ex-post approach, whereby optimization is performed first and the required total subsidy is subsequently computed, instead of incorporating additional constraints such as a budget constraint. Consequently, outputs of the rest of the variables are identical to the benchmark optimization problem (Equation 3.1). Further, the payment only compensates for potential losses, but does not penalize gains. Consequently, the total budget required will be equal to the sum of the economic loss of the losers. Then, a system of heterogeneous subsidies is expected to be much more efficient in terms of the budget required to avoid the existence of losers, since it matches exactly the value needed to satisfy the Pareto improvement constraint.

We calculate the solutions for the three conservation objectives. If $Z^* = 0$, the total required budget is zero since there are no losers. If $Z^* = 83$, the necessary annual budget would increase to € 10.4 M. This conservation cost would be comparable to the cost of focusing solely on steppe birds (€ 9.1 M). As a result, targeted conservation efforts would save € 1.3 M per year. These amounts represent a significant budget saving compared to a homogeneous payment for conservation. As said before, the budget required for a homogeneous payment to conserve steppe birds represents a subsidy of €100/ha (as can be seen in panel 6 Figure 3.3) with a total budget of approximately 180M; that is, the cost is 18 times higher. Further, recall that no homogeneous payment could simultaneously satisfy the preservation requirements for all 83 species. The maximum number of species that can be sustained under a homogeneous payment is 77 (92.7% of the total), and the cost would be 22 times higher than conserving all species under a heterogeneous payment.[19]

[19]Still, it is important to note that the opportunity cost of the agreement with the regulator has been estimated as € 2M and € 1.4M for protecting steppe and all species, respectively (see subsection 3.3.2).

Furthermore, we use the budget computed for heterogeneous payments to explore the possible outcomes if the same budget were employed for a homogeneous payment. The results can be seen in Table 3.4. For budgets of € 9.7M and € 10.4M, payments over € 287/ha and € 306/ha per hectare for rainfed winter cereals were infeasible since the demand for subsidies would exceed the payment capacity. Therefore, the conservation objectives are not met (only 31 species of birds are preserved). Note that those are the 31 species that would be preserved without enforcing any conservationist policy, and only 2 steppe birds. In fact, only 37% of all species and 18% of steppe species benefit from the policy. Most of these species are generalists and are not threatened. Furthermore, on the economic side, a budget that allows us to attain the proposed conservationist goals if a heterogeneous payment were enacted would have no impact in the case of a homogeneous payment. The *EV* of the third column in Table 3.4 remains at its maximum, meaning that none of the farmers would be willing to change their land uses due to the subsidy.

Table (3.4) HOMOGENEOUS PAYMENT OUTCOMES UNDER THE HETEROGENEOUS PAYMENT BUDGET

Budget (€ M)	$\bar{\upsilon}$ (€/ha)	EV (€ M)	Total species (n^o)	Steppe species (n^o)	LI areas (km^2)	Water (hm^3)	OC (€ M)
9.7	287	135.1	31	2	339	526	9.7
10.4	306	135.1	31	2	339	526	10.4

Notes: € 9.7M and € 10.4M are the total annual budgets for a heterogeneous subsidy system to sustain $Z^* = Steppe$ and $Z^* = 83$, respectively. Further, $\bar{\upsilon}$: represents the subsidy to rainfed barley/wheat. *OC*: Overcompensation. Total species: Number of species being conserved. Steppe species: Number of steppe birds being conserved. LI: Low intensified.

We observe that the difference between the two budgets is negligible, the only differ-

Therefore, it could be argued that compensating losers through heterogeneous payments may not be desirable, as the cost to the external agency would outweigh the foregone agricultural economic value (4.5 - 7.4 times for $Z^* = Steppe$ and $Z^* = 83$, respectively). Consequently, from the regulator's perspective, it is more cost-effective to propose an opposition-risk scenario without losers than to compensate them for their foregone benefits.

ences being due to the overcompensation of farmers who are already willing to participate (as indicated by the increase in *OC*).

Summarizing, perfect information is rarely achieved; often, most of the required information is hidden and unobservable. In such cases, regulators choose other types of mechanisms that are not efficient but require less information to be implemented, such as a system of homogeneous subsidies. However, we showed that the homogeneous subsidy, in our case study, is not only inefficient but also too costly to be considered feasible. Furthermore, we have compared the performance of the homogeneous subsidy scheme with a heterogeneous subsidy scheme under perfect information and showed that the later will be much more efficient than the homogeneous one. Next, we analyze the performance of a heterogeneous subsidy under incomplete information.

3.5.2 The role of incomplete information on economic value and target compensations

In our case study, we use the STICS simulation model to obtain crop yields - from which we can calculate their economic value. STICS performs a complex analysis of the interactions of a multitude of biophysical and management variables to determine, for each cell i, each of the outcomes possible under every combination of these variables. Furthermore, from this set of outcomes, we can solve our benchmark optimization problem which allows us to identify the optimal crop and management regime, x_i^*, for each cell i, that is, the characteristics and outcome for any cell are perfectly known. Also with STICS, the status quo outcome for any cell is perfectly known. We can use STICS to simulate perfect information.

However, even if there is no perfect information, the regulator may have some knowledge about farmer-land characteristics. This raises a question about the possibility that the implementation of a heterogeneous subsidy system with incomplete information will reduce the efficiency loss associated with a homogeneous subsidy. The lack of efficiency of these incomplete-information heterogeneous mechanisms seems to depend on the amount of information available to the regulator. The larger the information available to the regulator, the greater the possibilities of reducing efficiency loss. In this section, inspired by the work of Salle (2019), we propose comparing, first, the efficiency of an incomplete-information heterogeneous subsidy with the efficiency of a homogeneous subsidy and, second, the efficiency of our incomplete-information heterogeneous subsidy with the perfect information one (from subsection 3.5.1).

Under incomplete information, the precise value of the heterogeneous subsidy, $\bar{\upsilon}_i$, is unknown because the regulator does not have enough information to clearly identify the returns that farmer i is obtaining either from the status quo or the optimal allocation. The efficiency of an incomplete heterogeneous subsidy will depend on the ability of the regulator to obtain a good estimate $\hat{\upsilon}_i$ of the optimal subsidy $\bar{\upsilon}_i$ with the available limited information. Moreover, the optimal solution to the problem under complete information (defined in Equation 3.1) is not identical to the one with incomplete information. That is, under incomplete information, the regulator does not precisely know the payoffs of a given land use for each location. In fact, the regulator, under incomplete information, estimates payoff values that, with high probability, will differ from the real ones, leading to a potentially different optimal solution, that we name x^*_{II}.

To represent an incomplete setting we assume that the regulator does not have complete information about cell characteristics. To obtain this information, we assume that the regulator would carry out a set of surveys among farmers, obtaining values about predictors (crop and management regime) and explained variables (economic value). That is, from our STICS simulated data and to avoid sample biases, we obtain the estimates of the effect of land use on economic value individually from 10,000 samples of 100 plots selected randomly by running a Monte Carlo simulation. [20] We use a simple OLS regression with the

[20] In practice, many plots from different owners can be inside the same cell. Consequently, we allow the regulator to survey different plots - even with the same land use - inside each cell. Consequently, we allow replacement in each random sample.

interaction of intensification and crop selection as a predictor, and the economic value EV (€/ha) as the dependent variable:

$$EV_{ijr} = \beta_0 + \sum_{k=1}^{k=J\times R-1} \beta_k \quad x_{kijr} + \varepsilon_{ijr} \tag{3.23}$$

where management and crop independent variables are categorical, with values 0 and 1 depending on the selected option of farmer i - 1 is selected, 0 is not.[21] The parameters described as β_k are those describing the influence of a given land use option with respect to economic value. β_0 is the intercept. ε_{ijr} is the residual term, which is assumed to follow standard assumptions. As a result, we obtain the corresponding coefficients $\hat{\beta}_k$ and their distribution, where k represents the specific land use combination.[22]

Once the distribution of $\hat{\beta}_k$ from all samples is obtained (Figure 3.4), we used $\hat{\beta}_k$ to carry out our analysis.[23] This estimated $\hat{\beta}_k$ will allow us to predict the estimated economic value under incomplete information $\hat{EV}(x)$. In fact, we use equation:

$$\hat{EV}_{ijr} = \hat{\beta}_0 + \sum_{k=1}^{k=J\times R-1} \hat{\beta}_k x_{kijr} \tag{3.24}$$

to predict for each farmer both the status quo estimated economic value $\hat{EV}_i(x_i^{SQ})$ and the estimated socially optimal economic value under incomplete information $\hat{EV}_i(x^*_{i,II})$.[24] We assume that the status quo land uses are known by the regulator since they are publicly available through *DUN*. On the other hand, before being able to predict the economic values of the optimal land uses under incomplete information we need to identify those uses. To identify $x^*_{i,II}$, we run our benchmark optimization process considering the set of all possi-

[21] We use the option of rainfed wheat as a reference point, excluding it as a categorical value in the regression.

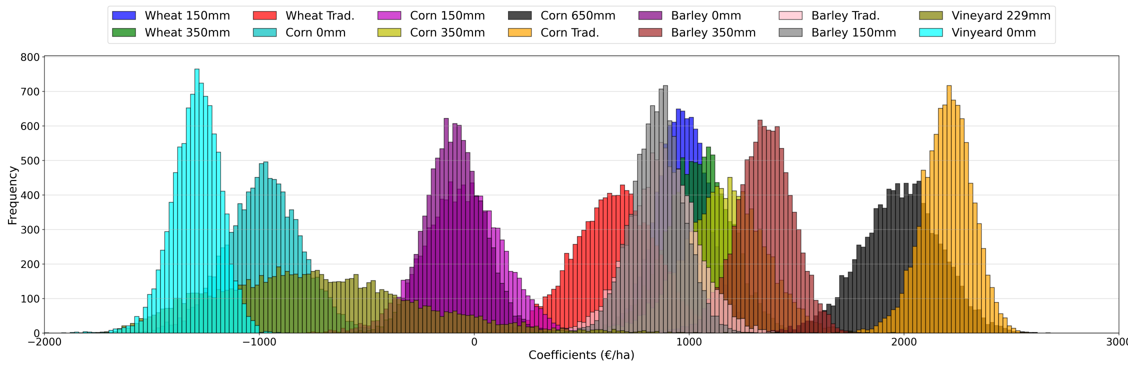
[22] k is the combination of management r and crop j.

[23] We could not use all the simulation samples since some of them did not have enough variability for the estimations. In total, we used data from 9,619 samples.

[24] To predict $\hat{EV}_i(x_i^{SQ})$, we use for each cell i the values in the status quo x_i^{SQ}. Similarly, for predicting the optimal economic value for each cell i we use the values $x^*_{i,II}$.

ble economic values predicted by $\hat{EV}(x)$ for two biodiversity objectives: $Z^* = Steppe$ and $Z^* = 83$ with seeded random samples. We exclude $Z^* = 0$ from this analysis since we assume that not imposing biodiversity constraints would imply that the regulator does not need to compensate anyone.

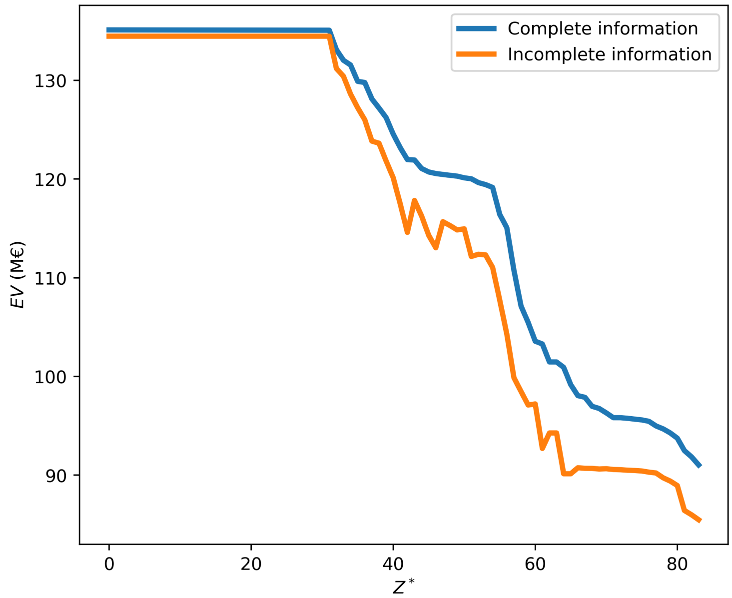
Figure (3.4) β_k DISTRIBUTIONS



Notes: W=Wheat; C=Corn; B=Barley; V=Vineyard; 0=Non-irrigated; 150=150mm; 350=350mm; 650=650mm; T=Traditional

Once we have identified the optimal land use solution under incomplete information $x^*_{i,II}$ we can compute the real economic value obtained through optimal land use solutions under incomplete information $EV(x^*_{i,II})$. This leads to a sub-optimal solution which can be represented as a *Pareto Frontier* (Figure 3.5) with lower total economic value in comparison to the solution under perfect information.

Figure (3.5) PARETO FRONTIER COMPLETE VS INCOMPLETE INFORMATION



Notes:

Consequently, we show that lack of information can lead to estimated optimal solutions with lower economic values, especially for higher conservation goals. Then, we could expect a loss due to lack of information from € 0.63M if $Z^* = 0$ to € 5.6 M if $Z^* = 83$. Regarding the payment schemes under incomplete information, the *Estimated compensation* for farmer i is:

$$\hat{\upsilon}_i = max\{\hat{EV}(x_i^{SQ}) - \hat{EV}(x_{i,II}^*), \quad 0\} \tag{3.25}$$

where Equation 3.25 indicates that compensation is only payed to farmers expected to be losers, while expected winners receive a zero lump-sum subsidy. The *target bias* for farmer i (θ_i) is, consequently:

$$\theta_i = |\bar{\upsilon}_i - \hat{\upsilon}_i| \tag{3.26}$$

which represents the difference between the unknown exact compensation needed (i.e. $\bar{\upsilon}_i = max\{EV_i(x_i^{SQ}) - EV_i(x_{i,CI}^*)\}$), and the estimated compensation $\hat{\upsilon}_i$, where we rewrite the optimal solution under perfect information for farmer i as $x_{i,CI}^*$ for a more intuitive comparison. The total target bias is the sum of the target bias of each farmer: $\theta = \sum_{i=1}^{I} \theta_i$. The average target bias is $\bar{\theta} = \frac{\theta}{I}$. Note that we can then calculate if a farmer i has been overcompensated - receives more than needed - or undercompensated - receives less than needed, and consequently is still a loser. Formally, a *winner* is a farmer who satisfies the following condition:

$$EV_i(x^*_{i,CI}) + \hat{\upsilon}_i \geq EV_i(x_i^{SQ})$$

with a total overcompensation $OC_i = EV_i(x^*_{i,CI}) + \hat{\upsilon}_i - EV_i(x_i^{SQ})$. A *loser* will be any farmer i such that:

$$EV_i(x^*_{i,CI}) + \hat{\upsilon}_i < EV_i(x_i^{SQ})$$

with a total undercompensation $UC_i = EV_i(x_i^{SQ}) - EV_i(x^*_{i,CI}) - \hat{\upsilon}_i$.

Some interesting variables to compare with the other mechanisms set out in the paper relate to the number of losers and the total level of inefficiency. It is straightforward to derive them from previous equation, where the total number of losers I^{UC} is the number of farmers being undercompensated. Mathematically, $I^{UC} = \sum_{i=1}^{I} \mathbb{1}_{\{UC_i>0\}}$. Conversely, the total number of overcompensated farmers is $I^{OC} = \sum_{i=1}^{I} \mathbb{1}_{\{OC_i>0\}}$.

Table (3.5) INCOMPLETE INFORMATION INEFFICIENCIES

	θ (€ M)	$\bar{\theta}$ (€/ha)	Budget (€ M)	Av. OC (€/ha)	Total OC (€ M)	I^{OC} (Nº)	Av. UC (€/ha)	Total UC (€ M)	I^{UC} (Nº)
$Z^* = Steppe$	18.8	78.0	14.9	48.9	11.8	271	29.1	7.0	85
$Z^* = 83$	16.7	69.6	16.5	48.0	11.5	305	21.6	5.2	70

Notes: $\bar{\theta}$: Average target bias. $OC = Overcompensation$, $UC = Undercompensation$. Av.=Average.

We present our results in Table 3.5. We find that the total number of losers - those being undercompensated - is considerable, regardless of the conservation goal, as a result of the target biases. These are the compensation levels that minimize the overall *target biases*, since they are expected to reduce the gap between estimated and real compensations, although this does not necessarily require everyone to be better off. As a result, when $Z^* = Steppe$, the number of losers is 85 (14.1% of the total), while if $Z^* = 83$, it is 70 (11.6% of the total). This indicates that a heterogeneous payment under incomplete information can fail in its goal of satisfying the Pareto improvement criteria, which did not occur under complete information. Incomplete information also results in many farmers being overcompensated (271-305 for $Z^* = Steppe$ and $Z^* = 83$, respectively) - also, this did not occur under complete information-. Moreover, it requires higher budgets to be carried out. If we computed that a heterogeneous payment under complete information required € 9.1M and € 10.4M for $Z^* = Steppe$ and $Z^* = 83$, respectively, it now requires € 14.9 - € 16.5 M. In addition, it is coupled with uncertainty since it depends on the chosen sample.

We compare its performance with respect to a homogeneous payment in terms of budget required to satisfy the conservation goal, as we did in subsection 3.5.1. We find that to satisfy $Z^* = Steppe$, the required budget would still be 12 times lower. If $Z^* = 83$, it would be 14 times lower. On the other hand, the inefficiencies also differ. While an estimated heterogeneous payment accounts for total inefficiencies of € 18.8 M - € 16.7 M ($Z^* = Steppe$ and $Z^* = 83$, respectively), in the case of a homogeneous payment it would be € 179 M - € 225 M. Therefore, it implies a much better use of budget - although lower than with complete information. However, under incomplete information the main risk arise as a result of losers, which are found to exist. However, its significant savings with respect to the homogeneous payment could motivate higher payments even than those estimated to increase the likelihood of reducing losers and opposition blocking.

Consequently, we demonstrate that in our case study, heterogeneous payments under incomplete information, where opportunity costs are estimated rather than known, can still be more effective than a homogeneous payment. Even though the regulator may not have complete information about the opportunity costs and economic value of each farmer and land use, they can still use available information to estimate these values and tailor the payments accordingly. While the estimates may not be perfect, they can still provide more accurate compensation to farmers than a homogeneous payment. By using heterogeneous payments under incomplete information, the regulator can better incentivize farmers to adopt the desired land use and attain more optimal results through a more efficient utilization of the budget. Although a heterogeneous subsidy may result in some losers, its greater efficiency could be used to make larger payments than initially estimated, thereby increasing the likelihood of participation.

3.6 Adding inequality to the policy design

Although the results shown throughout this chapter are compared exclusively from a utilitarian perspective, their outcomes can differ if we use a different welfare measure. Namely, we did not exploit the possibility that inequality may also play a role in acceptance. Moreover, this is one of the most relevant criticisms when we refer to optimal policies, where social welfare is not only about total outcomes - as utilitarian metrics suggest - but also about its distribution (Farrow, S. 1998). Additionally, it could be argued that even if two farmers receive the same gain through policy, it does not necessarily result in the same increase in welfare for both, as this depends on a multitude of factors such as pre-existing conditions or their utility function. For that, we use a new metric of social welfare which not only accounts for total agricultural economic value but also for its distribution. This distribution only accounts for the differences after the policy (i.e. $EV_i(x_i^*) - EV_i(x_i^{SQ})$ rather than $EV_i(x_i^*)$), since we want to analyze the policy performance rather than preexisting conditions.

It is commonly assumed in the literature that a social welfare function (SW) must satisfy four properties (Johansson, P., 1987). Firstly, it is assumed that *welfarism* holds, meaning that utility only depends on the utility levels of each farmer. Secondly, we assume *ceteris paribus*, which means that social welfare increases with an increase in each individual farmer's utility. Thirdly, the *intensity* property implies that social welfare depends on the degree of inequality in society. Finally, the *anonymity* principle indicates that it does not matter who enjoys higher or lower levels of utility. Under this definition of social welfare, we can observe that it necessarily increases when all farmers benefit from the policy and decreases when all are worse off. However, when there are both losers and winners, the result remains unclear. We choose the social welfare SW measure proposed by *Amartya Sen (1973)*:

$$SW = \Phi(1 - G) \tag{3.27}$$

where $\Phi = EV(x^*) - EV(x^{SQ})$ is the created surplus given by Equation 3.8 and G is the *Gini* coefficient showing how this surplus is distributed among farmers. [25] Because some land uses in some locations may imply economic losses - and the *Gini* coefficient is only valid for positive values - we use an adjusted *Gini* index designed by *Raffinetti et al. (2015)*.[26]

In the case of an agreement among farmers through a transfer system, the distribution of economic gains did not affect the measure of welfare in the previous sections. This is because the utility was only obtained as the sum of the economic gains or losses of each farmer. However, if we use a new welfare metric as in Equation 3.27, it is determined by the transfer rule employed - how much each winner transfers to the losers or how much each loser receives from the winners - which will alter the results. It is not important to link the payments between individual farmers, but to determine how much each farmer receives or pays. Therefore, we explicitly define these mechanisms considering two rules. Firstly, we

[25] Alternative social welfare measures could have also been employed. For instance, see subsection 3.I.

[26] This index ranges between 0 and 1, with higher values corresponding to more unequal distributions.

consider a strict compensation rule where winners of the policy directly compensate losers according to a proportional rule. The compensation is strict in the sense that winners compensate losers proportionally to their obtained profits by ensuring that losers will exactly obtain their status-quo profits. In the second case, we implement a Nash Bargaining Solution (NBS) where each farmer receives their status quo economic value and an equal share of the surplus created.

In the first case, for each winner w we defined their coefficient of proportionality λ_w as the ratio between the economic value of the gain obtained by winner w and the sum of all winners' gains:

$$\lambda_w = \frac{EV_w(x_w^*) - EV_w(x_w^{SQ})}{\sum_{w=1}^{W}(EV_w(x_w^*) - EV_w(x_w^{SQ}))}, \tag{3.28}$$

then the transfer paid by winner w, t_w, will be a fraction λ_w of the total amount of transfers needed to strictly compensate the losers, that is:

$$t_w = \lambda_w \sum_{l=1}^{L} \tau_l^{min} \tag{3.29}$$

Recall that $\tau_l^{min} = EV_l(x_l^{SQ}) - EV_l(x_l^*) > 0$, then the payoff of winner w after they have granted their corresponding compensation transfer is:

$$\widetilde{EV}_w^* = EV_w(x_w^*) - t_w = EV_w(x_w^*) - \lambda_w \sum_{l=1}^{L}(EV_l(x_l^{SQ}) - (EV_l x_l^*)) \tag{3.30}$$

while the payoffs of a loser l after they have received their compensation would be:

$$\widetilde{EV}_l^* = EV_l(x_l^{SQ}). \tag{3.31}$$

The compensation scheme can be complete in the sense that winners and losers can bargain over the entire surplus that will have been created.

In the second case we implement a Nash Bargaining Solution (NBS), which is likely to ensure that each farmer receives their status-quo profit plus an equal share of the surplus created. The NBS is defined as the product of the net payoffs of both winners and losers, where the net payoff of winners, $EV_w(x_w^*) - EV_w(x_w^{SQ})$, is equal to t_w^{max} (see Equation 3.5) and the net payoff of losers, $EV_l(x_l^{SQ}) - EV_l(x_l^*)$, is equal to τ_l^{min} (see Equation 3.6). Then the NBS is defined as follows:

$$NBS = \prod_{w=1}^{W} (t_w^{\max} - t_w) \prod_{l=1}^{L} \left(\tau_l - \tau_l^{\min} \right). \tag{3.32}$$

The maximization of the NBS with respect to the transfers paid by the winners and received by the losers yields the following outcomes (proof in subsection 3.G):

$$EV_w^* = EV_w(x_w^{SQ}) + \frac{1}{W+L}\Phi \quad EV_l^* = EV_l(x_l^{SQ}) + \frac{1}{W+L}\Phi \tag{3.33}$$

Since all farmers obtain more than their status-quo payoffs the Pareto improvement criteria are warranted. As we have assumed identical bargaining power for the farmers, the NBS yields an egalitarian rule.[27]

Nevertheless, we use this metric as an output of the model specifications considered throughout this paper, rather than maximizing with respect to it. This is because to carry out the maximization with Equation 3.32 as the objective function would imply a non-linear optimization with exponentially increasing numbers of possible solutions. Although some authors have proposed certain approaches to this problem (Acuna et al., 2020; Charkhgard et al., 2020) it is beyond the scope of this work. At this stage, the main intended objective is to observe how the previous problems behave with respect to this new metric of social welfare.

[27] In general, a multiplicity of rules can be considered.

Unlike EV, the values of SW in Table 3.6 lack a direct interpretation, and must be analyzed comparatively. Firstly, in case of disagreement $SW = 0$, which serves as a reference point.[28] With respect to the rest of the scenarios, we can firstly mention that mechanisms relying on high homogeneous payments tend to produce higher social welfare if payments are included for its computation. However, if we compute SW with homogeneous payments net of subsidies, the values obtained are the lowest, as inequality is high and the agricultural outcome is reduced. Moreover, when these subsidies reach a given amount (e.g. $\bar{\upsilon} \geq 1000$), for some conservation goals the resulting SW is even lower than in case of disagreement. On the other side, we observe that high SW values associated with homogeneous payments are usually at levels that we have tested to be incompatible with a realistic budget constraint.

From among the scenarios relying on agreements, we found that those where farmers cooperate between them should be preferred with respect to the agreement with the regulator, since their inequality levels are lower, and as seen in the previous section the EV is slightly higher. If agreement is between farmers, we found that using the NBS transfer rule is much higher since the Gini coefficient is reduced to 0 - gains equally split between transfers -. However, it would be possible to argue against this solution since winners from the policy may not agree to win the same as every other farmer even if their potential economic gains are larger. However, regardless of the transfer rule, we found that the computed SW value of the transfers between farmers is higher than in the benchmark case, which does not happen when the agreement is with the regulator. This is because the agreement with the regulator is costly in comparison with the benchmark case while the agreement between farmers is not. Moreover, if the agreement is with the regulator, inequality is almost identical to the benchmark case, while if the agreement is between farmers it is reduced, since the gain differences between losers and winners are smaller through the transfer rule. This means that not only are there no losers, but also the gains of the winners are not too excessive in

[28] This directly derives from Equation 3.27. In this case, the net payoff change would be zero, and then the SW function. However, note that the Gini coefficient accounting for inequality can take any value between zero and one.

Table (3.6) SOCIAL WELFARE ANALYSIS

		Z^*	
Status quo (Disagreement)	0.0 (0.45)		
Scenarios	0	Steppe	83
Benchmark	31.4 (0.52)	7.2 (0.75)	5.5 (0.75)
Agreement with regulator	31.4 (0.52)	6.7 (0.75)	5.1 (0.75)
Agreement between farmers (Proportional rule)	31.4 (0.52)	8.4 (0.71)	6.5 (0.70)
Agreement between farmers (NBS)	65.28 (0.00)	28.6 (0.00)	21.8 (0.00)
Homogeneous payment (With subsidies)			
$\tilde{\upsilon} = 250$	43.8 (0.41)		
$\tilde{\upsilon} = 500$	53.9 (0.35)		
$\tilde{\upsilon} = 1000$	68.8 (0.32)		
$\tilde{\upsilon} = 2000$	119.6 (0.27)		
$\tilde{\upsilon} = 3000$	210.2 (0.22)		
Homogeneous payment (Net of subsidies)			
$\tilde{\upsilon} = 250$	31.4 (0.52)		
$\tilde{\upsilon} = 500$	31.2 (0.52)		
$\tilde{\upsilon} = 1000$	10.2 (0.69)		
$\tilde{\upsilon} = 2000$	-9.7 (0.66)		
$\tilde{\upsilon} = 3000$	-6.4 (0.68)		
Heterogeneous (Complete information)	31.4 (0.52)	11.0 (0.70)	9.5 (0.70)
Heterogeneous (Incomplete information)	30.6 (0.53)	6.1 (0.75)	3.2 (0.77)

Notes: Numbers in parentheses show the *Gini* associated inequality index (higher means more unequal). In the case of homogeneous payments, first number is the annual subsidy to rainfed winter cereals (in €/ha).

comparison with those who benefit less from the policy.

Heterogeneous subsidies under complete information appear to yield higher social welfare results than the benchmark solution and agreements, except for those based on the Nash Bargaining solution. In the case of incomplete information, this would result in lower levels of social welfare than in the case of complete information, particularly for more demanding conservation objectives. This is due to a lower real economic value and also to slightly greater inequality.

3.7 Conclusions & discussion

In terms of spatial optimization policies, it is crucial to ensure that land-use and conservation plans are accepted by landowners and other decision-makers. Strategies must take into account political and economic factors to avoid generating opposition and reducing the likelihood of implementation, as has been observed in several instances where conservation plans have been slowed down or halted due to opposition (Alphandery & Fortier, 2001; Pretty & Pimbert, 1995; Stoll-Kleeman, 2001).

The motivating objective of this chapter was to investigate the acceptance of a policy aimed at conserving a bird community while maximizing agricultural economic value in a real agricultural region (the Lleida plain, Spain). We use the number of negatively affected farmers as a proxy for the difficulty of acceptance. Additionally, we aim to examine the effectiveness of different policy strategies aimed at promoting bird conservation by improving the ratio of winners to losers in the agricultural area. We compare these results with those associated with a homogeneous subsidy for rainfed cereals, which eliminates the problem of losers by definition, to determine if it can satisfy the same conservation goals as each of the proposed mechanisms. Subsequently, we assess whether a spatially heterogeneous payment can perform better, even without complete information about the required compensations.

Our results suggest that if no measures are taken, optimal solutions may result in scenarios where there are losers, particularly for high conservation objectives. This implies that an optimal policy may encounter opposition in the absence of incentives, regulation, or coordinated planning. An agreement with the regulator proposing a solution without losers would only reduce agricultural economic value by less than 2% relative to the benchmark case, depending on the conservation objective. We find that an agreement between farmers is preferred since the policy can be implemented without losers and without reducing agricultural economic value. Regardless of the type of agreement, reaching them should be preferred to the status quo due to the increased economic value without harming species conservation. Also, it should be preferred to the benchmark solution since it avoids the problems of policy losers. Our results indicate that compensating losers can be achieved with only 31.8% of the winners' gains if all species must be preserved, or 24.3% if only steppe species are prioritized, while meeting the Kaldor-Hicks efficiency criteria.

On the other hand, we found that a homogeneous payment is highly inefficient in satisfying conservation goals, where only an exceptionally high subsidy and required budget would achieve this, and it would be unattainable if the objective is to protect all species. Furthermore, it disincentivizes agricultural economic value production. A heterogeneous payment is more efficient and promotes agricultural economic value, requiring 18-22 times lower budgets if the objective is to protect steppe birds or the entire community, respectively. Thus, heterogeneous payments may be an option to satisfy the Pareto improvement constraint, as the necessary budget is low. By relaxing the regulator's complete information assumption, we can see that, even though the existence of losers is present, the budget of a heterogeneous payment is significantly reduced compared to a homogeneous one.

When considering inequality as a welfare asset in relation to the benefits provided by the optimal policy, we find that homogeneous subsidies yield higher social welfare values. However, subsidy amounts that are compatible with a Pareto improvement would necessitate an immense budget or may even be unfeasible if the conservation goal is to preserve all species. Among scenarios that require agreements, the transfer system reports higher social welfare values. However, this depends significantly on the redistribution rule, with the NBS rule expected to provide higher social welfare than the proportional rule. The heterogeneous payments also seem to report higher social welfare values than the benchmark solution with losers, although on relaxing the assumption of regulator complete information these improvements disappear. Regardless of the scenario, any agreement is preferable to disagreement.

Therefore, this study encompasses various approaches to addressing the issue of implementing conservation policies that avoid rejection by farmers, so that they can be carried out and thus be effective. Many of the proposed measures are not applicable on their own but demonstrate the improvements that would be possible if they could be implemented, and therefore should motivate their practical application. Homogeneous payments, such as the CAP subsidies, appear to be frequently highly inefficient and to be the result, in many cases, of their easy implementation. If we were able to make better use of the budget, or even reach agreements where no party is harmed, we could see significant improvements not only in the environment but also in competitiveness and economic value. This same approach that we have applied to an agricultural environment could be applied to any other policy that potentially involves the existence of losers.

However, some insights can be derived from the assumptions made in this work. We assume that agents maximize profits and consider the cost of any conservation measure on their income. Furthermore, we assume that farmers do not derive any utility from biodiversity conservation. However, there is a wide range of evidence indicating that tradition, cultural habits, altruism, and other characteristics influence human behavior and could shape the relationship between humans and the environment in a more complex and generous manner than the classical homo economicus hypothesis (Ostrom, 1990; Fehr and Schmidt, 1999; Fehr and Gintis, 2007). A natural question that arises is how our results would change if we assumed that farmers' values were somehow affected by biodiversity loss.

We recognize that deviating from the classical behavioral assumption of profit maximization could influence and challenge some of our results. It is difficult to determine how a change in agents' behavioral assumptions would affect the results of our model. A priori, we anticipated that the introduction of more environmentally friendly attitudes would facilitate the sustainability of bird species and potentially allow the achievement and implementation of no-lose solutions, provided that adequate distribution of environmentally friendly agents is present in the area.

On the other hand, one of the inspirations for this work was the complaints from farmers following the imposition of irrigation limitations in the irrigable area of the Segarra – Garrigues Channel (a subarea of our study area). These complaints can be considered as evidence that profit maximization was an appropriate behavioral hypothesis to represent a large proportion of the farmers in the study area. Additionally, it has been recognized that the influence of behavioral factors on agents' decisions depends on various factors, such as moral values, the social and environmental context and opportunity cost (e.g., Schluter et al., 2017). In this specific area, the economic differences between the returns obtained from irrigated land and rain-fed land are substantial, ranging from 2 to 7 times on average higher - according to our results - depending on the crop. This significant difference in returns further underscores the economic motivations of the farmers in the area.

Consequently, farmers' behavior should be considered when designing policy instruments but there are many challenges for incorporating alternative farmer behavioral hypotheses in mathematical models. Schluter et al. (2017) argue that there are many theories on human decision making that only cover certain aspects of decision making, have a low degree of formalization and do not specify any causal mechanisms in decision making, which makes it difficult to include them in these models.

On the other side, the implications of land use configurations have been simplified and may not necessarily reflect sustainability thresholds representing the real implications on birds. Additionally, aspects such as spatial connectivity or migration capacity of certain species were not included. Furthermore, our land use choices are based on a predefined set of management types, so non-computed solutions with better outcomes resulting from non-considered practices should be considered for further analysis. Finally, it would also be recommended to consider the implementation of more comprehensive payment schemes that accurately reflect the intricacies of the current CAP policy, thereby providing a more precise representation of the policy's implications for conservation, economic development, and acceptance.

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Appendices

3.A Summary of main mathematical terms

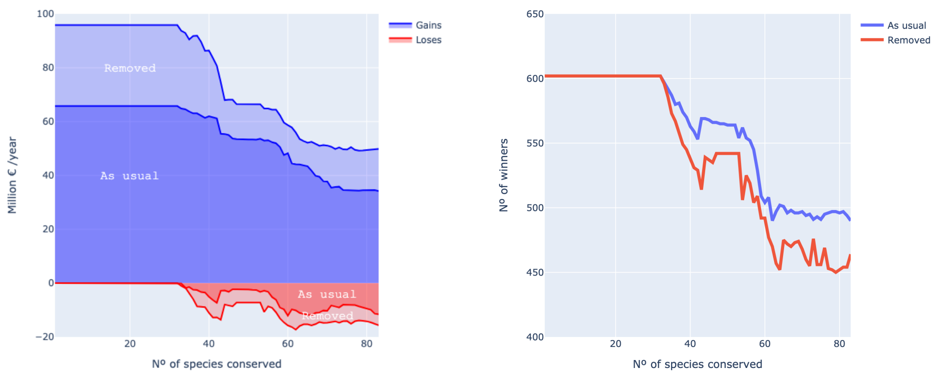
Table (B.1) SUMMARY OF MATHEMATICAL TERMS

Term	Description	Term	Description
EV_{ijr}	Economic value (€ M/year) associated with crop j and management r if they are applied in location i.	Z_s	Indicator function: 1 if species s is better adapted to the landscape configuration than in the status quo.
x_i^*	Optimal land use choice corresponding to location i.	λ	Parameter measuring the proportion of the losses that a given winner must pay based on a proportional rule.
x_i^{SQ}	Land use choice in the status quo.	υ_i	Indicator function: 1 if a certain given practice is carried out; 0 otherwise.
$EV_i(x_i^*)$	Economic value of farmer i associated with their optimal land use choice.	$\bar{\upsilon}$	Homogeneous lump-sum subsidy to given land use choices.
$EV_i(x_i^{SQ})$	Economic value of farmer i associated with their status quo land use choice.	$\bar{\upsilon}_i$	Heterogeneous payment to farmer i.
Z^*	Conservation goal: Number of species required to have a landscape habitat suitability level as in the status quo.	t_i, τ_i	In a scenario where a Pareto improvement is solved through an agreement between farmers, it indicates the transfer that a winner pays to the losers (t_i) or that a loser receives from the winners (τ_i).
$x_{i,II}^*$	Optimal solution under incomplete information	θ	Target bias between real and estimated compensation requirements.
x_i^+	Optimal solution if we impose the Pareto improvement criteria.	OC_i	Overcompensation of farmer i.
UC_i	Undercompensation of farmer i.	$x_{i,CI}^*$	Optimal solution under complete information.

3.B Analysis modifying conservation areas

We depart from the previous assumption that *SPAs* , i.e. zones with irrigation constraints, are immobile. We consider two different possibilities: 1) *SPAs* are removed. 2) *SPAs* can be spatially allocated in other territories, while holding their current extension. In both cases, we only consider modifications with respect to regulatory constraints, rather than infrastructure limitations. Thus, certain areas without access to irrigation will still have limited land use options. Moreover, even after removing these constraints, we still account for different conservation goals (Z^*). Increasing the set of options available in many locations may change the proportion of winners and losers, as well as their associated wins and losses, which could affect policy acceptance.

Figure (B.1) PARETO FRONTIER GAINS AND LOSSES WITH REMOVED INTENSIFICATION CONSTRAINTS



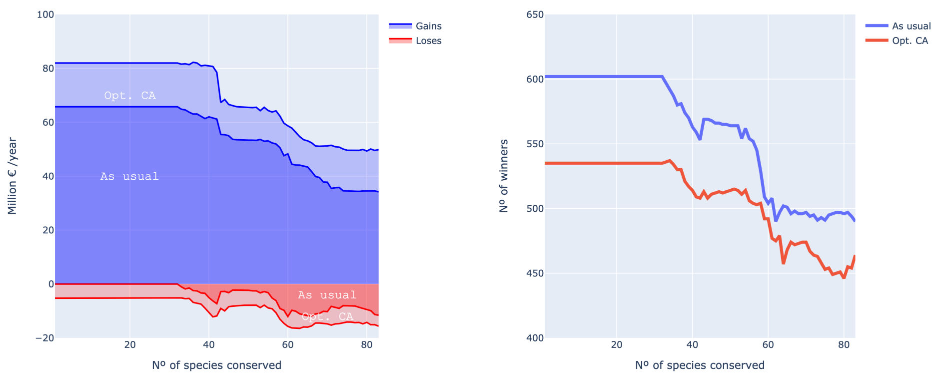
(a) Gains/losses (b) No. winners

Notes: The losses and gains are represented as the change from the status quo values (which have been normalized to zero).

The obtained solutions reveal an interesting result. By removing *SPAs* and allowing more intensified land uses in those places, under high conservation goals, the losses are higher than if the *SPAs* remains unaltered (Figure B.1). On the other hand, removing *SPAs* results in a higher gain for the winners, which overcompensates the increased loss of the losers. This implies that relaxing irrigation constraints may not reduce the losses of the losers, but it will increase the gains of some locations at the expense of others. Consequently, relaxing some constraints potentially increases net gains, but their potential Pareto improvements may be associated with higher opposition risks.

We also consider the possibility that *SPAs* are not eliminated but reallocated spatially (Figure B.2). The results show a similar trend, although with some differences. Reallocating optimal areas involves reducing the gains of the winners, especially when conservation goals are low. For example, when $Z^* \leq 31$, the gains are € 81.9 M contrasting with € 95.8 M if we remove the *SPAs* . The main difference between both assumptions about *SPAs* is that, if we allow them to be optimally allocated to maximize economic value under conservation goals, losers exist even when conservation goals do not exist or are low (i.e. when $Z^* \leq 31$). This occurs because *SPAs* must be placed at given locations that were not constrained, at least not as severely, and some farmers become losers, even though conservation goals are null or low. Obviously, carrying out this reorganization necessarily requires imposing *SPAs* at some locations to be compensated by relaxing these restrictions in previously conserved areas. This can be observed in Figure B.2a, where the area associated with the gains of the winners (blue region) increases more than the loss of the losers (red region). Moreover, not only the loss of the losers increases, but also the number of losers (Figure B.2b) in comparison with the original optimal solution.

Figure (B.2) PARETO FRONTIER GAINS AND LOSSES WITH OPTIMAL *SPAs* ALLOCATION



(a) Gains/losses (b) No. winners

Notes: The losses and gains are represented as the change from the status quo values (which have been normalized to zero).

3.C Analysis of the land use transitions and spatial implications with respect to losers

We investigate the land use transitions that lead to the presence of losers. To accomplish this, we generate a land use transition matrix that calculates the proportion of farmers who experience losses for each potential land use change (see Table B.2). Since both the status quo and optimal solutions usually involve a variety of land uses in each cell, we consider, to simplify the analysis, the predominant land use as the state value - that is why some values in the diagonal are lower than one, since remaining in the same state does not necessarily imply the exact land use composition -. This approach is justified by the assumption that the predominant land uses are representative of general trends, enabling us to identify the specific transitions that result in losses. The most frequent transitions are the shifts from corn 650mm to 350mm irrigated wheat, and from traditionally irrigated corn to traditionally irrigated wheat. Additionally, changes associated with transitions from traditionally irrigated corn and irrigated barley to non-irrigated barley are also associated with losers. Finally, changes to vineyards generally result in losses, especially to non-irrigated vineyards. On the other hand, those changes that generate winners are those shifting to irrigated corn or to non- or low-irrigated wheat from barley. Small differences can be discerned comparing the results obtained on Panel A of Table B.2, where only steppe species are preserved and those on Panel B, where all birds are preserved. For instance, it appears more suitable under targeted conservation policies to transition from 350mm barley to 150mm barley, despite resulting in greater losses. This shift is replaced with changes from 350mm barley to 650mm corn in case of conserving all species.

We further observe that, when conservation goals are binding and affecting total economic value, there is a clear trend toward using less irrigated barley and irrigated wheat, since the former commonly provides higher yields under low water inputs than wheat, but lower yields under more intensified regimes. In practice, crop yield differences are low, so farmers may be relatively indifferent between both options. Corn is also expected to be reduced, although it is still employed in more productive areas. These findings may prove useful in formulating policies that provide support to farmers who may occasionally experience losses.

Table (B.2) LOSERS PROPORTION BY LAND USE TRANSITION (MATRIX FORM)

PANEL A ($Z^* = Steppe$):

	B0	B150	B350	BT	C0	C150	C350	C650	CT	W0	W150	W350	WT	V0	V Irr.
B0	1	-	-	-	-	-	-	-	-	1	-	-	-	-	-
B150	-	1	-	-	-	-	-	-	-	-	1	-	-	-	-
B350	-	0.19	0.86	-	-	-	-	-	-	-	-	0.94	-	-	0.75
BT	0	-	-	-	-	-	-	-	1	-	-	-	0.35	-	-
C0	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-
C150	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
C350	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
C650	-	0	-	-	-	-	-	-	-	-	-	0.18	-	-	-
CT	0	-	-	-	-	-	-	-	0.99	-	-	-	0	-	-
W0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W150	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W350	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
WT	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
V0	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
V Irr.	-	-	-	-	-	-	-	-	1	-	-	1	1	-	-

PANEL B ($Z^* = 83$):

	B0	B150	B350	BT	C0	C150	C350	C650	CT	W0	W150	W350	WT	V0	V Irr.
B0	0.99	-	-	-	-	-	-	-	-	1	-	-	-	0	-
B150	-	1	-	-	-	-	-	-	-	-	1	-	-	-	-
B350	-	-	1	-	-	-	-	1	-	-	-	0.94	-	-	0.53
BT	0	-	-	-	-	-	-	-	1	-	-	-	0.26	-	-
C0	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-
C150	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
C350	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
C650	-	-	-	-	-	-	-	-	-	-	-	0.18	-	-	0
CT	0	-	-	-	-	-	-	-	1	-	-	-	0	-	-
W0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W150	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W350	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
WT	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
V0	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
V Irr.	-	-	-	-	-	-	-	-	1	-	-	1	1	-	-

Notes: Values represent the proportion (from 0 to 1) of farmers who have changed from a given state to another and have become winners (or remain equal). Rows represent original state while columns show ending state. Lower values mean higher proportion of losers for a given transition. (-) No transitions between these two states have been observed. To determine the states the predominant land use in a given cell has been selected. B=Barley, C=Corn; W=Wheat; V=Vineyards. Numbers indicate irrigation regime in *mm*. T=Traditional irrigation. Irr. = Irrigated vineyard.

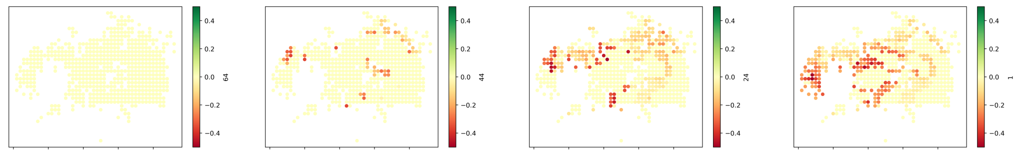
In Figure B.3, we present the spatial distribution of (EV) changes resulting from two different optimization approaches. Firstly, we show the spatial distribution of EV changes between the optimal allocation for each Z^* and the non-conservationist approach ($Z^* = 0$) (Figure B.3 (a-d)). As expected, when Z^* is sufficiently low, there are no trade-offs between conservation and economic value in any location, as demonstrated in Figure 3.2a. However, as conservation goals become more stringent, relative losses emerge in certain locations (represented in red in Figure B.3 (a-d)). It should be noted that the only restrictions that vary between these locations are biodiversity restrictions, while physical restrictions (such as the lack of an irrigation system) or legal restrictions (such as legally protected areas) remain constant. Therefore, cells lacking an irrigation system, such as many areas in Segarra county, are minimally affected by stricter conservation goals, since the only land use change possible is crop selection but not management regime, resulting in minimal crop yield impacts.

The first locations to experience economic losses as biodiversity restrictions become more stringent (e.g. $Z^* = 40$) mostly coincide with cells where full irrigation (650mm and traditional) is allowed. These cells switch from crops with high economic returns to other crops that are also economically valuable but more compatible with bird conservation. As biodiversity restrictions become more demanding ($Z^* = 60$), the number of cells that must switch crops and/or management regimes increases. In contrast, areas with a larger opportunity cost of irrigation, which tend to coincide with areas that currently have a higher irrigation water allocation, do not experience negative impacts as the number of biodiversity constraints increases, and are expected to be the last to abandon irrigated crops.

Second, we present the spatial distribution of the differences in EV between the optimal and the SQ allocations (see Figure B.3 (e-h)). In this case, when $Z^* \leq 31$, there are no losers since farmers can select the individual profit-maximizing option without any loss due to biodiversity preservation restrictions. There is no trade-off between conservation goals and EV. The EV increases or remains unaltered (green/yellow, respectively) as the optimal solution often represents an improvement over the SQ in every cell i, $EV_i \leq EV_i^{SQ}$. Following the same reasoning as before, as biodiversity constraints become more demanding ($Z^* = 40$)

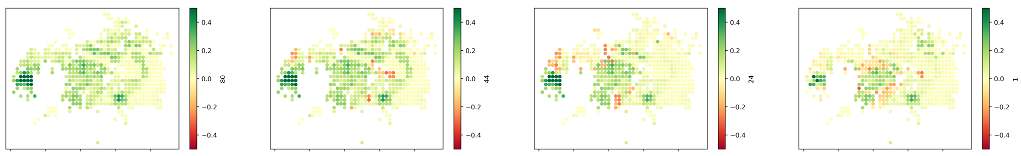
Figure (B.3) SPATIAL IDENTIFICATION OF ECONOMIC GAINS AND LOSES

With respect to $Z^* = 0$



(a) $Z^* \leq 31$ (b) $Z^* = 40$ (c) $Z^* = 60$ (d) $Z^* = 83$

With respect to the *SQ*:



(e) $Z^* \leq 31$ (f) $Z^* = 40$ (g) $Z^* = 60$ (h) $Z^* = 83$

Notes: Values are in €M€

losses begin to appear, with some locations experiencing losses in comparison with the *SQ* outcome (in red). In these places the loss is necessary at the expense of a gain at other locations. However, in Figure B.3 (e-h) it can easily be seen that the number of winners (cells in green) is larger that the number of losers (cells in red). This result is independent of the strictness of the conservationist goal. Comparing these results with that obtained in the former paragraph we observe, looking at the first and second row of Figure B.3, that the patterns of change present similarities. Most of the cells that become losers as restrictions become more strict coincide with cells that were first forced to switch crops as the number of preservation constraints increased, showing that the first locations that have to face losses are those with lower opportunity costs.

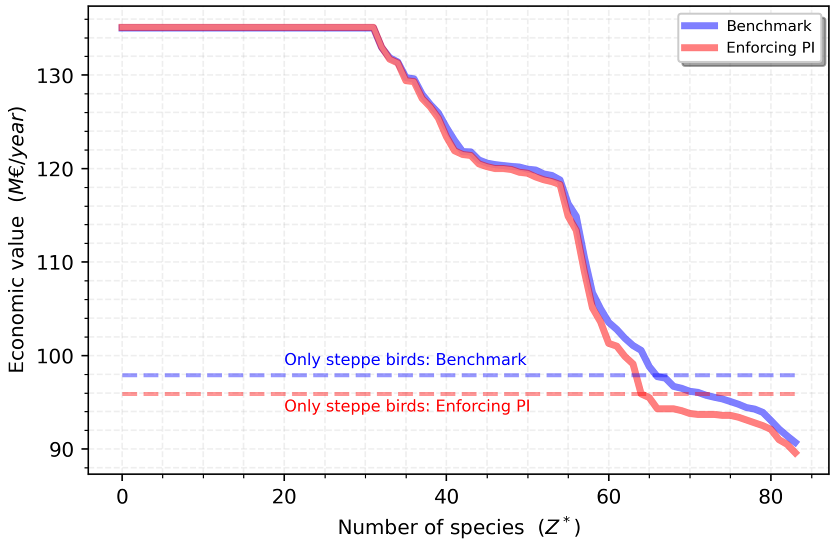
Independently from the approach used, the spatial distribution of irrigation constraints (Figure 2.4) significantly affects the results (see subsection 3.H for map representation overlapping irrigation constraint and gains/losses). More specifically, those cells with very restrictive irrigation constraints are not affected by coordinated spatial planning (for example, eastern parts of *Segarra* or some cells within the *Segarra-Garrigues* irrigation system). The irrigation regulation precludes planting crops that require abundant water for any set of biodiversity conservation constraints and consequently these areas do not benefit from easing these conservation constraints. Also, since they are already obtaining low profitability, the corresponding optimal solution pattern will not affect them negatively either.

However, those areas without restrictive irrigation limitations may benefit more when conservation goals are less ambitious, but are also those with potentially higher losses when conservation goals are high, as occurs in the central parts of the plain. This happens because in the center of Lleida plain irrigation is highly required - since it is characterized by higher temperatures and lower precipitation levels -, and therefore the opportunity cost of shifting to lower irrigation is high. Consequently, the optimal solution is to continue irrigating that territory, while towards the periphery, it is preferred to abandon more profitable options so as to satisfy the biodiversity constraint. Eastern parts are not affected by the policy since there is no infrastructure to irrigate.

3.D Pareto Frontier analysis comparing the benchmark case and the regulator solution

A more detailed comparison of the Pareto Frontier, with and without satisfying the Pareto improvement constraint, is presented in Figure B.4. The Pareto Frontier represents all the possible trade-offs between economic value and conservation goals (Z^{*}) for the benchmark case and for the case where a Pareto improvement is imposed.

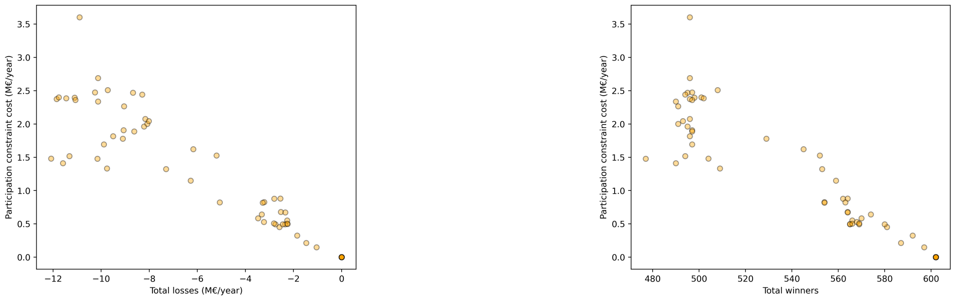
Figure (B.4) PARETO FRONTIER IMPOSING PARETO OPTIMALITY BETWEEN FARMERS



The opportunity cost of enforcing the Pareto improvement criteria (PI) increases as the conservation goals become more demanding, up to a certain point, after which these costs begin to decrease as the conservation goals become more stringent. The reason for this is that, for conservation goals beyond a certain threshold (e.g., $Z^* > 31$), the difference in economic value between imposing the *PI* or not widens because individual and social interests do not coincide. Therefore, optimal solutions require the existence of losers, and solutions satisfying the *PI* deviate from the optimal solution, resulting in lower objective function values. However, when conservation goals are sufficiently high (e.g., $Z^* \geq 65$), the economic value differences between both scenarios become narrower again. This is because the difference between satisfying or not satisfying the *PI* depends not only on the economic value of the scenario where the *PI* is imposed but also on the number of losers and losses associated with each Z^*. From Figure 3.2b, we observe that, when $Z^* \geq 65$, the gain of the winners generally decreases while the loss of the losers also decreases. Thus, scenarios with higher conservation goals may have a higher win-loss ratio than others with lower conservation goals. This is possible because we do not minimize total losses in the optimization problem, but rather maximize net gains (i.e., gains minus losses). Generally, those conservation goals Z^* that involve a higher economic value gap between satisfying the

PI or not are those with higher losses/losers (see Figure B.5), although, even with similar values, the cost of satisfying the *PI* can differ, especially when losses are higher.

Figure (B.5) COST OF SATISFYING PARETO IMPROVEMENTS DEPENDING ON TOTAL LOSSES AND LOSERS



(a) (b)

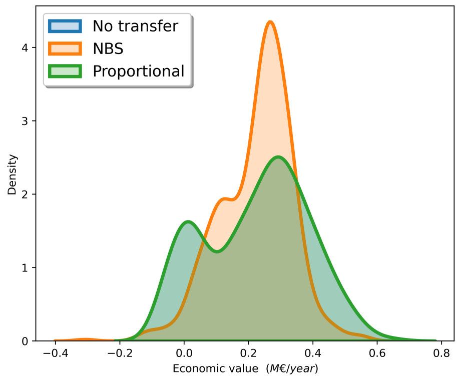
Notes: Each dot represents a solution (Z^*) of the Pareto frontier. The Pareto improvement constraint cost is the difference between the maximum *EV* obtained satisfying Pareto improvements or not under the same conservation goal Z^*.

3.E Economic value change distributions after transfer systems

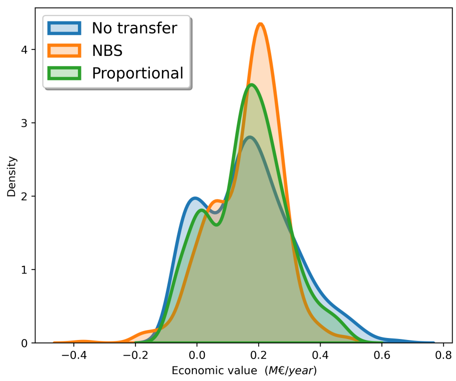
In Figure B.6, we explicitly illustrate how income is distributed among farmers under different conservation goals and transfer mechanisms. Specifically, we compare three scenarios: 1) without transfers, 2) the egalitarian rule given by the Nash Bargaining Solution, and 3) the proportional rule under the strict compensation scheme. Note that the three panels of Figure B.6 exhibit bimodal distributions, which result from the distribution of the geophysical characteristics of our case study. In the scenario where $Z^* = 0$, the solution without transfers is the same as that of the proportional redistribution rule because, under the proportional rule, redistribution occurs only if there are losers. Since there are no losers in the no-conservation scenario, no redistribution process occurs. In contrast, the compensation transfers are carried out independently of the existence of losers, resulting in a different distribution of agricultural income that shifts the farmers' payoffs towards more central values, reducing both those with higher incomes and those with lower incomes.

When the conservation objectives are more demanding, such as conserving steppe species or all species, the distribution of economic payoff values shifts to the left for all transfer mechanisms compared to the situation where $Z^* = 0$. This result implies that, on average, farmers have lower incomes, and there is a higher proportion of farmers who are losers. As observed, both the proportional and the egalitarian rules help to displace farmers' income to values closer to the average. In particular, the egalitarian transfer rule seems to be more effective in reducing inequality between farmers Additionally, note that even with a proportional rule, which was designed to satisfy the Pareto improvement constraint, some farmers have negative payoffs. This occurs because some farmers were already obtaining negative outcomes in the status quo.

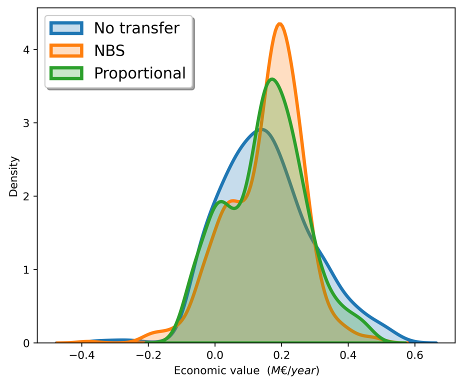
Figure (B.6) FARMERS' ECONOMIC VALUE DISTRIBUTION USING PROPORTIONAL AND EGALITARIAN TRANSFERS



(a) $Z^* = 0$



(b) $Z^* = Steppe$

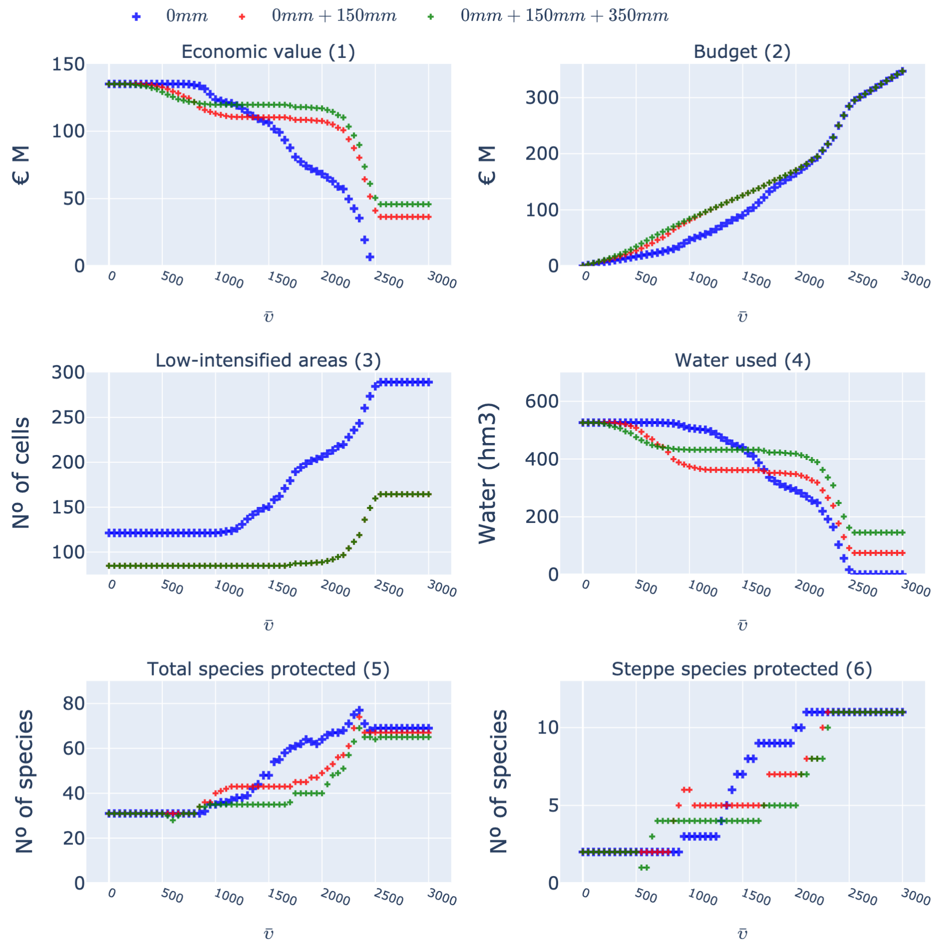


(c) $Z^* = 83$

3.F Homogeneous payments with additional compensation to barley/wheat from 150mm and 350mm management regimes

We also investigated the impact of moreover subsidizing irrigated lands with less than 350*mm* with barley and wheat crops (Figure B.7). We consider two different cases: subsidizing rainfed and 150*mm* winter cereals, and subsidizing rainfed, 150*mm* and 350*mm* winter cereals. We assume that the payment is equal for all the land use choices being subsidized (e.g., rainfed wheat will receive a compensation equal to 350*mm* wheat).

Figure (B.7) HOMOGENEOUS PAYMENT TO WINTER CEREALS IMPACT (RAINFED, 150MM AND 350MM)



Notes: The x-axis represents the subsidy to rainfed barley and wheat. All the solutions are computed considering a subsidy to barley and wheat 150mm of € 500/ha.

The results indicate trends similar to the previous case where subsidies were granted exclusively to rainfed winter cereals. However, we find that subsidizing low-irrigated barley and wheat may influence farmers' land use decisions with lower payments per hectare. For instance, with a payment of €500/ha, conservation goals will increase in contrast to the €1000/ha required if only rainfed lands were subsidized. Nonetheless, this does not imply that the total budget is lower (Figure B.7). Additionally, low-intensification areas classified as rainfed would be lower since we would incentivize other land use choices.

3.G Proof Nash Bargaining solution

Consider the maximization problem of the NBS with respect to the transfers t_w and τ_l:

$$\max_{t_w,\tau_l} NBS = \prod_{w=1}^{W} (t_w^{\max} - t_w) \prod_{l=1}^{L} \left(\tau_l - \tau_l^{\min}\right).$$

Since the system is self-financed, this means that for one farmer, say the last loser L, we have

$$\tau_L = \sum_{w=1}^{W} t_w - \sum_{l=1}^{L-1} \tau_l. \tag{B.1}$$

The derivative of the NBS with respect to one winner gives

$$\frac{\partial NBS}{\partial t_w} = 0 \Leftrightarrow t_w = t_w^{\max} - \tau_L + \tau_L^{\min}, \tag{B.2}$$

while the derivative of the NBS with respect to one loser gives

$$\frac{\partial NBS}{\partial \tau_l} = 0 \Leftrightarrow \tau_l = \tau_l^{\min} + \tau_L - \tau_L^{\min}. \tag{B.3}$$

Taking the sums of (Equation B.2) over all the winners and (Equation B.3) over all the losers gives

$$\begin{aligned} \sum_{w=1}^{W} t_w &= \sum_{w=1}^{W} t_w^{\max} - W\left(\tau_L - \tau_L^{\min}\right), \\ \sum_{l=1}^{L-1} \tau_l &= \sum_{l=1}^{L-1} \tau_l^{\min} + (L-1)\left(\tau_L - \tau_L^{\min}\right) \end{aligned}$$

Substitute in (Equation B.1) yields

$$\tau_L = \tau_L^{\min} + \frac{1}{W+L}\left(\sum_{w=1}^{W} t_w^{\max} - \sum_{l=1}^{L} \tau_l^{\min}\right).$$

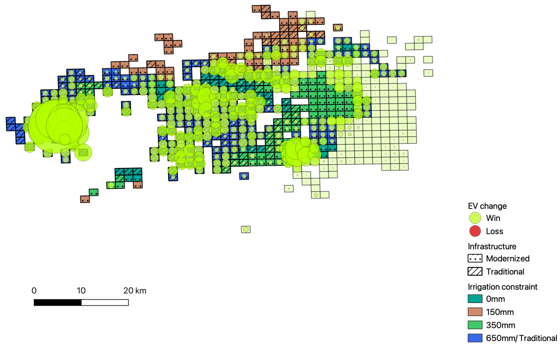
After substitution in the transfers given by (Equation B.2) and (Equation B.3) we obtained

$$\begin{aligned} t_w &= t_w^{\max} + \frac{1}{W+L}\Phi, \\ \tau_l &= \tau_l^{\min} + \frac{1}{W+L}\Phi. \end{aligned}$$

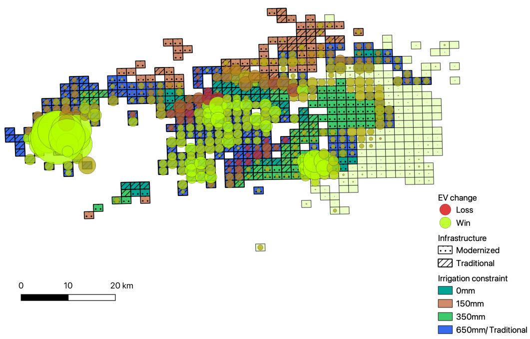
3.H Association between *SPAs* and gain/loss after the optimization process for different biodiversity goals

These maps overlap the gain/losses of each farmer (cell) with the current *SPAs* for different conservation goals (Z^*). The main interest of these results is to display how *SPAs* shape the potential gains/losses of the farmers.

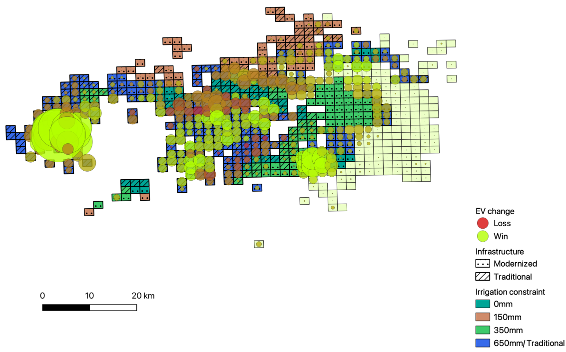
Figure (B.8) SPAs AND GAINS/LOSSES AFTER THE OPTIMIZATION PROCESS



(a) $Z^* = 0$



(b) $Z^* = Steppe$



(c) $Z^* = 83$

Notes: Circle size corresponds with the amount of gain/loss in a representative cell. The color of the circle indicates if it is a gain (green) or a loss (red).

3.I Alternative social welfare measure

Alternative *Social Welfare* measure:

Based on *Nash Social Criteria* (Nash, J. (1950); Kaneko, M. & Nakamura, L. (1979)):

$$SW = \prod_{i=1}^{I} (EV_i(x^*) - EV_i(x^{SQ}))^{\alpha_i}$$

In case of implementing a transfer system, it should be:

$$SW = [\prod_{w=1}^{W} (EV_w(x_w^*) - t_w - EV_w(x_w^{SQ})) \prod_{l=1}^{L} (EV_l(x_l^*) + \tau_l - EV_l(x_l^{SQ}))]^{\alpha_i}$$

where α_i is a parameter indicating the weight of each farmer in the social welfare function.

Chapter 4

Climate change impact on crop yields and irrigation demands: An application to the Lleida plain

4.1 Introduction

For millennia, humans have been altering natural systems. Among these changes, the modification of the global carbon cycle and its associated climate effects represent the most complex transformation with significant impacts on the economy and energy balance. The accumulation of greenhouse gases (GHGs) in the atmosphere hinders the re-radiation of sunlight reaching Earth, resulting in an increase in energy at the Earth's surface (Hsiang & Kopp, 2018). This effect, driven by economic expansion and population growth, is the primary cause of global warming (Kouhestani et al. 2016). Since the late 19th century, the global mean surface temperature has increased by 1.0 C° , with an acceleration after 1980. Projections indicate that the global mean surface temperature will rise by 0.9-2.3 C° by 2080-2100 compared to preindustrial levels under low-emission scenarios and by 3.2-

5.4 C° under high-emission scenarios (Collins et al., 2013). Precipitation patterns are more uncertain but are expected to increase overall (Hsiang & Kopp, 2018), with significant regional disparities. For instance, the likelihood of drought has increased in areas such as the Mediterranean (Hartmann et al. 2013) and generally in arid regions (Collins et al., 2013; Estrela et al., 2012). This results in spatially and temporally varying impacts on agricultural systems, with some locations benefiting and others being harmed (Kukal et al., 2018), affecting yields differently (Rashford et al. 2016; Ray et al., 2019). Consequently, the opportunity costs of biodiversity conservation may also change differently, with implications for opportunity cost patterns of conservationist landscape (Gerling, C. et al., 2021). Therefore, not only agricultural systems will be affected but also the conservation of biodiversity, especially that which depends on the characteristics of the habitat and its variations.

If climate changes are not extreme, the negative impacts of rising temperatures on crop yields may be offset by increased CO_2 concentrations (Smith et al., 2013). For instance, it has been found that climate change has mainly positive effects in the UK (Fezzi et al., 2010), where water is abundant. However, while increased atmospheric CO_2 concentrations have been shown to have a positive impact on crop growth yields through the fertilization effect on photosynthesis, this effect varies between crops. Differences in photosynthesis pathways mean that wheat generally benefits more than corn (Antle & Stöckle, 2020; Paul et al., 2020). Some studies have found that the effect of CO_2 on wheat yields by mid-century will be between +8 and 26% (Asseng et al. 2013). In the case of corn, irrigation is expected to reduce climate change impacts in the mid century but may be insufficient in the long term (Paul et al., 2020). Climate change may affect water use, particularly irrigation, the need for which is positively associated with rising temperatures in the case study area (IPCC, 2007). Thus, climate change presents a challenge for agriculture in the medium and long term, with spatially heterogeneous impacts and direct implications for agrosystem processes such as crop production (Vaghefi et al., 2014), biodiversity conservation and water quantity and quality (Estrela et al., 2012).

This study has three objectives. Firstly, to develop a methodology for estimating the economic impact of climate change on a real agricultural landscape scenario, specifically the Lleida plain in Spain, where agricultural profits have been historically constrained by aridity and irregular precipitation patterns (Reguant, 2017). Secondly, to ascertain the effects on the conservation of biodiversity of the changes in the Lleida plain agricultural landscape resulting from climate change and also to know the consequences of biodiversity conservation on the economic value of agricultural production in the Lleida plain in climate change scenarios. And, last but not the least, to forecast the implications of water scarcity on agricultural economic value in future scenarios using a spatially explicit land-use model with a regulator coordinating to maximize economic value under total water capacity constraints.

In the literature, three approaches have been used to measure the impact of climate change on agriculture (Schlenker et al., 2006): Ricardian analysis (Mendelsohn et al., 1994), Computable General Equilibrium models (Nordhaus & Yang, 1996), and Agronomic models. Due to the challenges of assessing the impact of climate change on crop yields at a regional scale, considering complex interactions between climate, plant, soil, and management values for future conditions, we used an agronomic model. These dynamic systems relate inputs to predict outputs such as grain yield or other biological or physical variables. These inputs include all atmospheric climate conditions, and represent all non-climate biophysical factors such as soils, water, government policies, and regulations (Antle & Stöckle, 2020). Crop growth simulation models offer several advantages over other approaches, such as econometric models. They provide reliable estimates of future outcomes, account for non-linear effects and interactions, and are not reliant on sufficient variation in climate variables to estimate parameters. They can identify potential thresholds not observed in historical data and are efficient tools for research and development, predicting crop phenology and yield, optimizing resource utilization, forecasting pests and diseases, mitigating climate change, and supporting decision-making (Antle Stöckle, 2020; Divya et al., 2021).

We determine the potential crop yields for each land use and location using STICS crop growth simulation model under climate projections from regional downscaled climate models. We consider two different emission scenarios corresponding to the Representative Concentration Pathway (RCP): 4.5 and 8.5. We obtain the potential economic value associating the results of the crop simulation with economic data. STICS has been used to forecast climate change impact on crops in many works (e.g.: Butterworth et al., 2010; Fraga et al., 2016; Tribouillois et al., 2018). Most of them can be included in the agronomic field research, and therefore we expand this literature by considering four different economic approaches regarding land-use choices: continuing with the same land uses as now (Business-as-usual), maximizing economic profit land-use choice criteria (Coping strategy), a regulator's approach maximizing aggregate economic value under available water constraints, and finally a regulator's approach maximizing aggregate economic value under combined constraints on water availability and biodiversity conservation. We perform these analyses for three periods: 2020-2040 (Current), 2040-2070 (Medium term), and 2070-2100 (Long term).

The structure of this study is divided into three main parts. First, we set out our theoretical framework where the spatial model, the different scenarios proposed and the biodiversity theoretical components are presented. Second, we detail our applied framework, where we briefly show the case study and the data used. Finally, we present and discuss our results.

4.2 Theoretical framework

4.2.1 Spatial model

Following the model applied in Chapter 2 and Chapter 3, we now develop a model in which representative farmers i make land-use decisions in each period t. As before, we partition the landscape into I farmers (or locations/cells) where each $i = (1,2,...,I)$ must decide on two elements: crop j and management r at period t. Now , variable $x_{tijr} \in [0,1]$ indicates the proportion of available land devoted to crop j and MR r planted in cell i at t. If it is not planted, $x_{tijr} = 0$. Therefore, we have a set of decision variables that indicate which land uses are selected in each location for each period. Consequently, the set of crops and MG selected at period t can be defined as a vector $x_t = [x_{t1jr}, x_{t2jr}, ...x_{tIjr}]$ of dimension $1 \times I$. The set of feasible choices is $X : x \in X$. [1] The economic value (EV_{tijr}) for each location i is represented as the *Net Cash Income*:

$$EV_{tijr}(x_{tijr}) = a_i \times (Q_{tijr}(x_{tijr}) \times P_j - C_{jr} - W_{ijr}) \tag{4.1}$$

where Q_{tijr} - which is a function of the land-use choice x_{tijr} - is the potential crop yield at time t (tons/ha) for each cell i for any given combination of crop j and MG r - note that each cell presents different soil and climate characteristics and therefore location variability responds to the differences in these characteristics. $EV_{tijr}(x_{tijr})$ also depends on a_i, the number of hectares corresponding to location i. P_j(€/ton) is the market price for a given crop j, which we keep fixed using average prices from 2010 to 2020 to avoid price variability and make them comparable; C_{jr} is the cost (in €/ha excluding water cost) of crop j under management regime r and W_{ijr} is the water cost (€/ha). Water costs are location-specific since we account for heterogeneity in channel irrigation costs. The aggregate agricultural economic value for each period is therefore $EV_t(x_t) = \sum_{i=1}^{I} \sum_{j=1}^{J} \sum_{r=1}^{R} EV_{tijr} x_{tijr}$.

4.2.2 Economically driven land-use change impact on biodiversity

We also utilized our climate projections to estimate the economically driven land-use changes that may occur and the implication of these changes on biodiversity. We employed the same methodology as delineated in Chapter 2 to analyze the implications of land use on biodiversity. Thus, for each location i, there will be a matrix B_{jrs} of unique parameters representing the suitability between species and habitats. As in the preceding chapters, we consider a community of 83 bird species with varying preferences for land uses. Of these, 11 are classified as steppe birds, with special protection interest. This index incorporates both dietary

[1] As in previous chapters, we consider legal and infrastructure irrigation constraints depending on the location.

and nesting requirements into the habitat preference index designed by Estrada et al. (2004).

Farmers' land-use choices, under each RCP and time period, may result in different land uses and landscapes, which will have implications for most species. Consequently, biodiversity conservation will not be imposed in the model but rather will be a consequence of economically driven decisions. [2] For each cell, RCP, time period, and behavioral assumption, an x_{ijr} is chosen. The habitat in each cell, that is, crop j and management regime r chosen by agent i, would be affected by the corresponding RCP and period considered. We compute a species s adaptation to the area's current conditions by obtaining our biodiversity index presented in Equation 2.9, that is $B_s = \Sigma_{i=1}^{I} \sum_{j=1}^{J} \sum_{r=1}^{R} B_{jrs} x_{ijr}$. If for a given species s, given an RCP scenario and a time period t, the corresponding biodiversity index is higher than in the status quo (the current landscape configuration obtained from *DUN*) biodiversity index, we consider that a given species is being conserved and $Z_s = 1$. Otherwise, it will be considered that s is not preserved ($Z^* = 0$). The total number of species conserved is $Z^* = \sum_{s=1}^{S} \in [0, 83]$.

4.2.3 Scenarios

In our analysis, we contemplate four distinct scenarios, as outlined in Table 4.1. The first assumption posits that farmers will maintain their current land-use practices, referred to as the *Business as usual* scenario. Under this assumption, farmers do not alter their selection of crops or management techniques in response to changing climatic conditions. That is, farmers' choice x_{tijr} will be equal to the status quo:

$$x_{tijr} = x_{ijr}^{SQ} \quad \forall \quad i, j, r, t$$

where x_{ijr}^{SQ} represents the status quo land-use choice of farmer i. As such, land-use choices are not selected to maximize economic value for each period. Note that, for a given location

[2]This is the general rule, although at the end of the chapter (fourth scenario) we create a scenario where conservation constraints are imposed in the model.

i, for two different periods (e.g. t_1 and t_2), the associated EV_{tijr} does not necessarily coincide even if the same land-use choice is selected because the climate conditions can differ. Then, for two arbitrary time periods t_1, t_2, it could be the case that $EV_{t_1ijr}(x_{ijr}^{SQ}) \neq EV_{t_2ijr}(x_{ijr}^{SQ})$.

Table (4.1) MODEL ASSUMPTIONS CONSIDERED

Scenario	Description	Type
1. Business-as-usual	Farmers do not react to new climatic conditions and carry out the status quo choices.	Individually taken
2. Coping strategy	Farmers react to new climatic conditions by selecting the most profitable option in each period.	Individually taken
3. Optimal solution under water availability constraints	A regulator selects land uses at each location to maximize aggregate economic value considering different water availability levels.	Regulator
4. Optimal solution under water availability and biodiversity constraints	A regulator selects land uses at each location to maximize aggregate economic value considering different water availability levels and habitat suitability constraints .	Regulator

However, it is highly likely that farmers will adjust their decisions in response to new climatic conditions. For this reason, our second scenario assumes that farmers will select land uses that maximize their individual economic returns in each period. We refer to this scenario as the *Coping strategy*. In this case, the land-use choice in location i at period t is:

$$x_{tijr} = x_{tijr}^{'} \qquad if \qquad \nexists \quad x_{tijr}^{''} \quad s.t. \quad EV(x_{tijr}^{''}) \geq EV(x_{tijr}^{'})$$

which implies that there is no other choice $x_{tijr}^{''}$ at period t and location i with higher economic value than the selected option x'_{tijr}. The net gain obtained by coping with respect to remaining in the status quo is defined as:

$$\Delta EV^{COPING} = EV_i(x'_{tijr}) - EV_i(x^{SQ}_{tijr}) \tag{4.2}$$

We propose a third scenario based on a centralized solution in which the regulator maximizes the aggregate economic value - the sum of the economic return of each farmer - subject to a constraint on water availability (Equation 4.3). Water is expected to be a determinant factor of agricultural production in this area (Noto et al., 2023). The intensification level at which crops are produced is determined by the amount of water that can be employed (mm) at each location (L_{ijr}). That is, we consider that this is particularly noteworthy in light of the potential for future reductions in water availability and therefore we introduce water availability as a constraint in our model. The total amount of water used is calculated as the sum of the water used in each cell i $\sum_{i=1}^{I}\sum_{j=1}^{J}\sum_{r=1}^{R} L_{ijr}x_{ijr}a_i$. The restriction is completed by adding a maximum amount of water allowed in the region L^*, which we assume to be time-invariant. Then, it differs from the farmers' decision assumptions in the previous chapters, where farmers did not have water constraints.

$$\begin{aligned} &\max_{x_t \in X} EV_t(x_t) \\ s.t. &\sum_{j=1}^{J}\sum_{r=1}^{R} x_{tijr} = 1 \quad \forall \quad i \\ &\sum_{i=1}^{I}\sum_{j=1}^{J}\sum_{r=1}^{R} L_{tijr}x_{tijr}a_i \leq L^* \\ &x_{tijr} \in [0,1] \quad \forall \quad t,i,j,r \end{aligned} \tag{4.3}$$

The first constraint stipulates that the sum of land-use choices for a given location must equal one, indicating that there is no land abandonment. The solution of this problem is defined as $x^*_t = [x^*_{t1jr}, x^*_{t2jr}, \ldots, x^*_{tIjr}]$, where each term of this vector is the optimal solution at cell i.

Finally, in the fourth scenario, we incorporate to Equation 4.3 a new constraint which incorporates biodiversity goals as a constraint:

$$\sum_{s=1}^{S} Z_s \geq Z^* \tag{4.4}$$

which indicates that the number of species not being worse than now should be greater than Z^*. Our interest is to combine the water availability constraint with the biodiversity conservation one to construct a matrix of combinations, where for each biodiversity goal we will calculate the outcomes for a set of water availability levels.

4.3 Case study, data and methods

4.3.1 Case study and crop yield simulation

The case of study corresponds to the one already described in Chapter 2 and Chapter 3. It includes the counties of *Segrià, Urgell, Pla d'Urgell, Segarra, Garrigues* and *Noguera* in the Province of Lleida (Catalonia, Spain). Recall that the area has been stylized in 2×2 km squared cells, which represent the minimal decision units.

We estimate the potential yield of the crops selected in the study area for each cell and period under different management regimes. Since no data on potential crop yields were available at this resolution scale, we used a crop growth simulation model. Due to its precision we used, as in previous chapters, *STICS* that is able to simulate the behavior of the interactions between climate, soil, management and crop systems.

Table (4.2) STICS INPUT DATA

Categories	Source	Input names	Nº of sample points
Soil	Institut Cartogràfic i Geològic de Catalunya	pH, calcium carbonate content, water content at field capacity and wilting point, Clay, Organic carbon, rooting depth, Organic Nitrogen	216
Climate	CORDEX (IPSLP-SMHI-RCA4)	Precipitations, max. air temperature, min. air temperature, vapor pressure, global radiation, windspeed at 2m, evotranspiration, CO_2.	602
MG	Expertise	Supply of organic residues, tillage, sowing, fertilization, irrigation, harvest.	-
Plant	STICS default	Barley, wheat, corn, vineyard.	-

Notes: The STICS crop growth simulation model requires four categories of input data to simulate crop yields: soil, climate, management, and plant information. Each category comprises various variables, as shown in the third column. The final column represents the number of spatially located sources of information, including trial pits and weather stations.

We used the data on soils, management regimes and plant characteristics that were used in previous chapters. Consequently, we used 3 crops: barley, wheat and corn, under 5 management regimes: rainfed, 150mm, 350mm, 650mm and traditional irrigation, under different combinations representing realistic options (see Table 4.3). However, and unlike Chapter 2 and Chapter 3, we excluded from this analysis vineyards due to their difficulty to be simulated under future climate conditions. Moreover, spatial restrictions on irrigation were considered.

Table (4.3) COMBINATIONS OF CROPS AND IRRIGATION SCHEDULES

MG/Crop	**Wheat**	**Corn**	**Barley**
Rainfed	*Yes*	*Yes*	*Yes*
Surface	*Yes*	*Yes*	*Yes*
150mm	*Yes*	*Yes*	*Yes*
350mm	*Yes*	*Yes*	*Yes*
650mm	*No*	*Yes*	*No*

Notes: Yes=considered; *No*=Not considered.

On the contrary, in this chapter we are using a climate values database different from that used in previous chapters. The data used were retrieved from the CORDEX database, which is part of the Copernicus European Union Earth observation program. This creates some differences between simulated crop yields in this chapter and previous chapters, where we used data from Meteocat.[3] We obtained daily climate forecasts for key climatic variables, including maximum temperature (C°), minimum temperature (C°), solar radiation (Mj/m^2), relative humidity (mbar), wind speed (m/s), and precipitation (M/day). We used the global climate model IPSL-CM5A-MR and the regional climate model SMHI-RCA4. The latter model downscales data from the global model to a finer resolution to better represent smaller areas, with a resolution of $0.11^{\circ} \times 0.11^{\circ}$ for the European domain. To obtain values for each cell and day for the period 2020-2100, we computed, for each cell and day, the average value of each variable, correcting biases using the Delta change method (Hawkings et al., 2013), by comparing historical simulations from the model with observed data for the period 1991-2005 [4]. These models have been found to be suitable for this region since they represented realistic current conditions.[5]

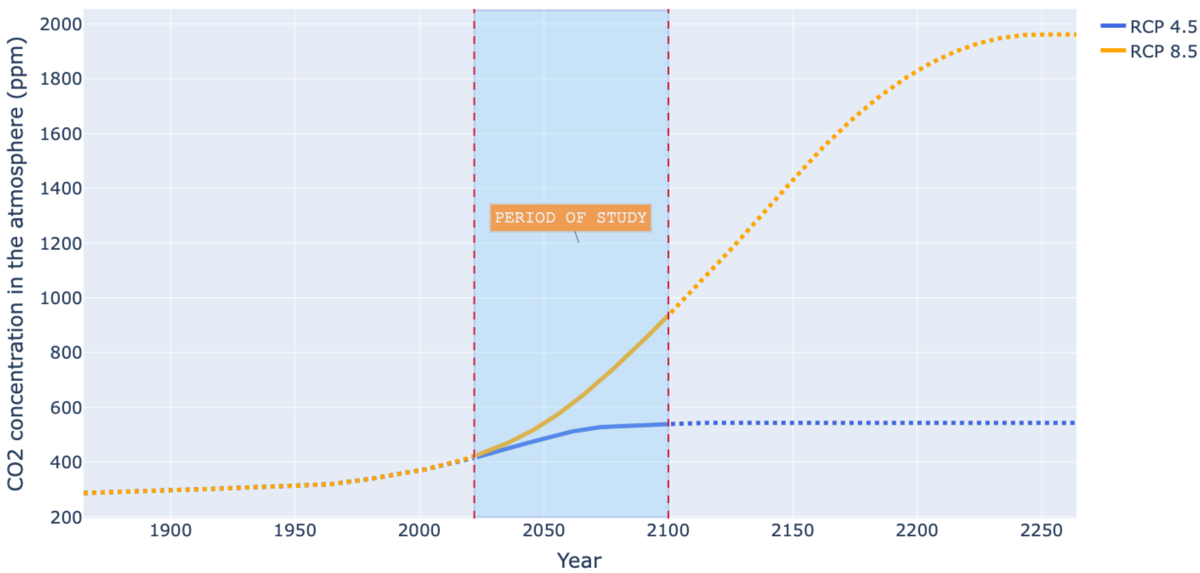
[3]We use this different data set because Meteocat did not provide enough forecasting climate values for the variables required by STICS.

[4]Global and Regional Climate models contain systematic errors in their output, which can lead to unrealistic results when applied to impact models or climate impact assessments. To address these biases, various bias correction methods have been developed. The simplest approach is the Delta change method, which involves adding or multiplying future forecast values by a constant, representing the differences between model values applied to a past period and the corresponding observed values (Copernicus Climate Change Service, s. f.).

[5]Some methodological notes: Evapotranspiration was calculated from STICS using Penman equations. The ensemble member is r1i1p1. The annual average differences between observed and simulated values were added to/subtracted from the forecast values. We used Climate Data Operator (CDO)

Given that non-mitigation pathways may have potentially severe consequences for the global economy and food security (CNA Corporation 2007), we distinguished between two different Representative Concentration Pathways (RCPs): RCP 4.5 and RCP 8.5 W/m^2. These RCPs represent different time series of aerosol emissions worldwide (Moss et al. 2010) and exhibit different trends in radiative forcing until 2100.[6]

Figure (4.1) EVOLUTION OF CO_2 CONCENTRATION UNDER DIFFERENT RCP SCENARIOS FOR THE PERIOD 1850-2250



Notes: Graph generated utilizing data obtained from the RCP database (v2.0), accessible at https://tntcat.iiasa.ac.at/RcpDb/dsd?Action=htmlpagepage=welcome. The blue region delineates the time frame encompassed by our study.

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to rotate coordinates to regular ones (latitude, longitude), and Zonal Statistics for Multiband Rasters (https://github.com/dymaxionlabs/qgis-zonal-statistics-multiband, QGIS 3.22.7).

[6]Radiative forcing, intuitively speaking, is the difference between incoming energy absorbed by the Earth and that radiated back into space. If it is positive, this means that more energy enters than goes back. For a more precise definition, which is related to the change in surface temperature, see (Shine et al. 1990).

Table (4.4) SPECIFICATION FOR DIFFERENT PATHWAYS OF CO_2 EMISSIONS FOR THE 21TH CENTURY

Scenario	**RF**	**RF trend**	**CO_2 in 2100**
RCP 4.5	*4.5 W/M*^{2}	Stable in 2100	538 ppm
RCP 8.5	*8.5 W/M*^{2}	Growing	936 ppm

Notes: Ppm= Parts per million. RF= Radiative forcing.

As shown in Figure 4.1 and Table 4.4, both scenarios are significantly different. Measured in W/m^2 with respect to preindustrial levels (1750), scenario RCP 8.5 involves almost 1.9 times more relative radiative forcing than scenario RCP 4.5 W/m^2.[7] Moreover, the levels of radiative forcing in RCP 4.5 scenario would be stabilized by 2100, while in the second scenario they would still be growing in 2100. That is, RCP 4.5 and RCP 8.5 can be interpreted as a stabilization and an expansionist scenario, respectively. Other scenarios have been developed (e.g. RCP 2.6, which would represent accounting for emission abatement policies limiting climate change to 2°). However, the climate data required were only available for the two mentioned scenarios.

The simulated trends for CO_2 concentrations worldwide are presented for previous periods (i.e. before 2020), in Figure 4.1, for the period of study (i.e. from 2020 to 2100) and subsequent periods (i.e. after 2100). As shown, the increase in CO_2 concentrations has been exponential during the last two decades. However, the paths diverge from 2050, reaching a difference of 397 ppm of CO_2 in 2100. While CO_2 concentrations stabilize in 2070 in the case of RCP 4.5 [8], they would stabilize, approximately in 2230, for the RCP 8.5. Consequently, while differences in concentrations between the two scenarios at the end of our period of study account for 398 ppm (+ 74%), in 2250 they are of 1420 ppm (+361%). Then the impact of mitigation policies could be potentially higher than the results shown in this

[7] Where W/m^2 stands for Watt per square meter.

[8] Small increases ($\leq$ 1 ppm per year) are observed, but they do not represent significant changes.

study.

4.4 Results

We present our results in three steps. First, we show the results of the climate projections in our case study region for the period 2020-2100. Second, we take into account the results of these climate projections and simulate the crop yields in our study region for three periods (2020-40, 2040-70 and 2070-2100) and two RCP scenarios (4.5 and 8.5). Finally, we show the results considering four different assumptions about farmers' behavior.

4.4.1 Climate forecasting

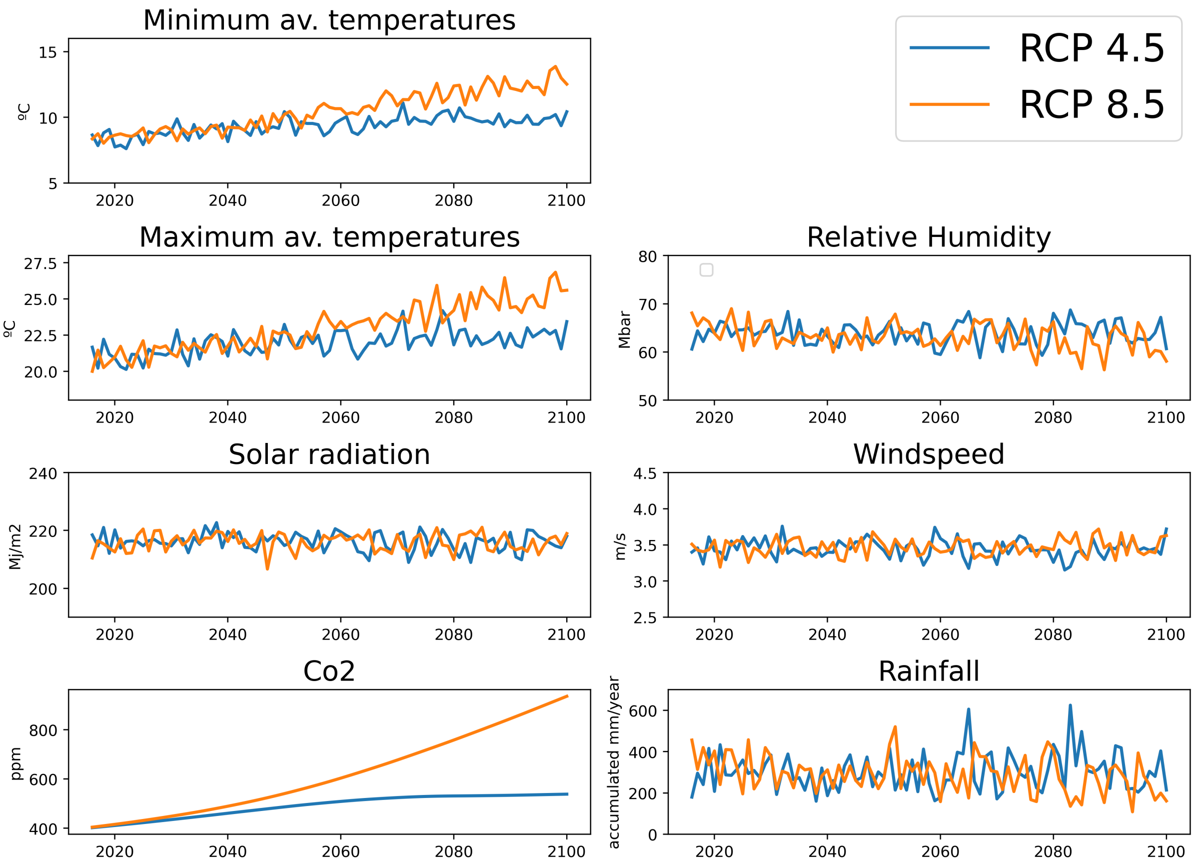
The climate projections for the area of study show significant changes beyond CO_2 levels (Figure 4.2). Both minimum and maximum temperatures are expected to rise during the coming decades. For the period 2020-2060 [9], minimum temperatures will increase moderately (+ 2.1 C° and + 2.9 C° for RCP 4.5 and RCP 8.5, respectively). For the period 2020-2100, the minimum temperature increases will be even more significant (+ 2.95 C° and + 4.8 C° for RCP 4.5 and RCP 8.5, respectively). Maximum temperatures are also expected to rise, for period 2020-2060 (+ 1.91 C° and + 2.5 C° for RCP 4.5 and 8.5, respectively). Also they would increase for period 2020-2100 (+ 2.5 C° and + 4.7 C° for RCP 4.5 and 8.5, respectively).[10] Precipitation levels are expected to decrease in the long term in the case of the RCP 8.5, while they will remain stable in the RCP 4.5. Windspeed is also anticipated to rise, especially for the scenario of high emissions, with a difference of +2m/s until 2100. The rest of the variables remain stable for the period 2020-2100. In Figure 4.3 we represent average climate values' evolution for the three periods considered (*2020-2040*, *2040-2070* and *2070-2100*) to differentiate more clearly the climatic conditions in the medium term and

[9] This period does not coincide with any of the 3 periods considered throughout the paper (present, medium term and long term). It is just established from data observations representing more identifiable trends.

[10] Increases have been computed using year 2020 temperature values of RCP 4.5 as a reference.

long term.[11] Temperatures are expected to rise in both scenarios, although the increase is more noticeable in the RCP 8.5. Difference in average maximum temperatures between both scenarios in the medium term is $+0.79C^{\circ}$ while in the long term it is $+2.28C^{\circ}$. Analogously, difference in average minimum temperatures rises from $+0.91C^{\circ}$ to $+2.24C^{\circ}$. Precipitation levels differences are not significant in the medium term ($+10mm$), but in the long term, in the high-emission scenario they decrease by $45mm$. Interestingly, precipitation levels are expected to rise in the long term in the low-emission scenario.

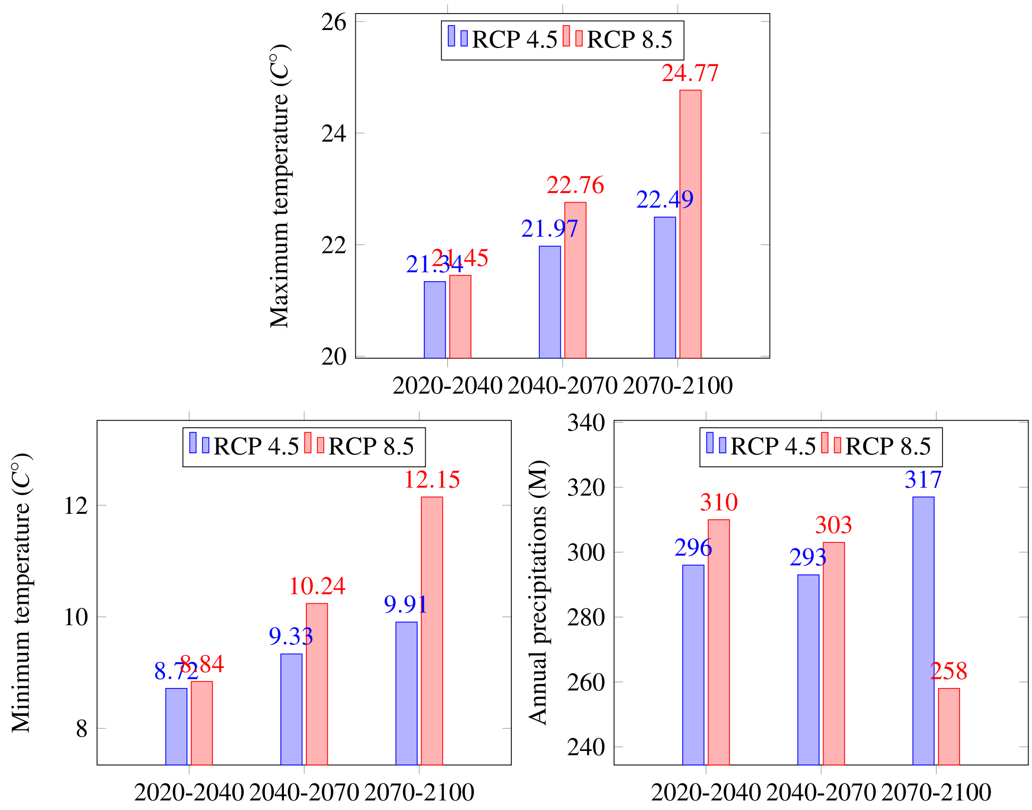
Figure (4.2) EVOLUTION OF CLIMATE VARIABLES UNDER DIFFERENT RCP SCENARIOS FOR THE PERIOD 2020-2100



Notes: Evolution of main climatic variables for the period 2020-2100 in the area of study (Lleida plain). For each variable, values represent the annual average for all the cells and days.

[11] We show the results for max. temperature, min. temperature and precipitations, which are key determinants of crop productivity. The rest of the variables are not shown to maintain simplicity.

Figure (4.3) AVERAGE PRECIPITATIONS AND TEMPERATURE EVOLUTION UNDER DIFFERENT RCP SCENARIOS



Given this, the impact of anthropogenic greenhouse gas emissions on climate will be evident in the medium term, but particularly in the long term in the Lleida plain. Considering the importance of the agricultural sector in this region, we examine its implications in the following section.

4.4.2 Climate change impact on crop yields

The impact of climate change on crop yields is projected to be significant in the coming decades. For each RCP and time period considered, daily climate values were calculated for each cell and year to obtain average values for each period. Crop yields were then simulated for each location and land use for each of our six scenarios (two RCPs for each of the three time periods). Average values were obtained for each land use, period, and emission scenario. Focusing on RCP 4.5 for the period 2040-2070, it is observed that - as expected - crop yields increase with irrigation, particularly with modern irrigation, for all crops. 650mm corn is found to have the highest yield, with average values exceeding 14 tons/ha. Winter cereals such as wheat and barley have lower yields but these last also increase with irrigation.

Table (4.5) AVERAGE CROP YIELD FORECASTING (Annual ton/ha)

Land use	RCP 4.5			RCP 8.5		
	2020-40	*2040-70*	*2070-100*	*2020-40*	*2040-70*	*2070-100*
Barley:						
Rainfed	1.01	0.95	1.99	1.24	1.55	1.38
150mm	4.69	4.19	4.53	4.67	4.95	3.51
350mm	7.08	6.6	6.42	7.31	6.95	5.13
Traditional	2.21	1.94	3.33	2.42	2.67	2.07
Wheat:						
Rainfed	1.00	0.73	1.62	1.09	1.18	1.04
150mm	3.13	2.78	3.80	3.36	3.34	2.82
350mm	6.27	5.64	6.54	6.37	6.07	5.58
Traditional	3.07	2.74	4.05	3.43	3.32	2.75
Corn:						
Rainfed	0.01	0.05	0.27	0.00	0.07	0.06
150mm	0.19	0.45	1.64	0.24	0.53	0.20
350mm	4.08	4.88	6.39	4.24	3.95	1.84
650mm	14.21	13.99	14.12	14.35	12.65	7.25
Traditional	14.40	14.35	14.53	15.04	13.24	7.72

Notes: Each value is the crop yield average between cells for a given scenario representing the period and *RCP*. Unit: ton/ha .

Surprisingly, crop yields for non-irrigated crops are projected to slightly increase in the long term. This may be due to climate projections that anticipate increases in CO_2 levels and precipitation, despite rising temperatures. The increase in yield is generally higher for barley and wheat than for corn. Additionally, traditional irrigation appears to be of more benefit than modern irrigation. This suggests that traditional irrigation methods, which involve fewer application dates but higher water consumption, may benefit more from higher precipitation levels in the long term [12] than modern irrigated crops, which have more evenly distributed irrigation dates and amounts throughout the year and thus have their water needs met and yields closer to their potential.

With respect to the RCP 8.5 scenario, medium-term crop yields are projected to be higher than under the RCP 4.5 scenario. This can be attributed to higher CO_2 levels, relatively small differences in temperature, and slightly higher precipitation levels. However, in the long term, crop yields are expected to decline significantly for all land-use types. These results suggest that the effects of climate change on crop yields in the Lleida plain will be more pronounced in the long term than in the medium term. Comparing the crops, highly intensified corn is significantly more affected than barley or wheat. Despite higher productivity in terms of production per hectare (e.g., long-term values for traditionally irrigated corn with 650mm yield average values of approximately 7 tons/ha, while wheat or barley yields 5 tons/ha for 350mm), the increased costs associated with corn production may result in a reduction in its cultivation due to lower profitability. Moreover, these adverse effects are projected to occur only under the RCP 8.5 scenario, underscoring the potential significance of mitigation efforts in preserving the economic value of agriculture.

4.4.3 Scenario analysis

Initially, we present the results in relation to the first two scenarios regarding the behavior of farmers. (see Table 4.6).[13] In the *Business-as-Usual* scenario, where farmers do not adapt, current land-use data were obtained from the 2019 *Declaración Única Agraria (DUN)*. We also keep the current SPA restrictions in place and data on crop types. In the case of irriga-

[12] See Figure 4.3 to check this.

[13] It should be noted that the values presented here differ from those in Chapter 2, even under current conditions. This discrepancy can be attributed to two factors. First, as this chapter does not take vineyards into account, the total area is reduced due to the exclusion of land currently used for vineyard cultivation, resulting in a 10.1% decrease. Second, the climate data used in this analysis differ from those obtained through Meteocat and yield lower crop productivity. While the difference in crop yields between the two datasets is not substantial, it may have a significant impact on the economic viability of certain land uses that are close to their profitability threshold, causing them to shift from profitable to non-profitable outcomes. This is particularly relevant for winter cerealwinter cereal cultivation.

tion, as in previous chapters, and due to the lack of precise data, we assumed that farmers choose the maximum irrigation level that is allowed at each location. In the medium term, the differences in economic value between the 4.5 and 8.5 RCP scenarios are small, with total economic values of 21.7 and € 27.6 M, respectively. However, in the long term, the forecast economic impact is clearly negative in the RCP 8.5 scenario. While the total economic value increases to € 36.2 M under the low-emission scenario, it decreases to € -5.8 M under the high-emission scenario. In this last case, economic value is negative under the business-as-usual 8.5 RCP scenario, which would imply a dramatic impact on regional agriculture as costs would exceed income in many cases. We could expect an average loss of € -317/ha under these conditions compared with the period 2020-2040.

Table (4.6) FORECAST AGGREGATE ECONOMIC VALUE, WATER USE AND BIODIVERSITY

	RCP 4.5			RCP 8.5		
Scenario	*2020-40*	*2040-70*	*2070-100*	*2020-2040*	*2040-70*	*2070-100*
Economic Value ***No coping***:						
€ M	29.7	21.7	36.2	33.3	27.6	-5.8
Δ €/ha	-	-71	58	32	-18	-317
Economic Value ***Coping***:						
€ M	71.7	60.9	72.3	76.2	57.7	22.2
Δ €/ha	374	278	380	415	249	-66.4
Total irrigation ***Coping***:						
hm^3	554	576	546	548	479	475
Biodiversity ***Coping***:						
Total (Z^*)	27	27	26	27	12	18
Steppe (Z^*)	1	1	1	1	0	2

Notes: Each value is the crop yield average between cells for a synthetic year representing the period and *RCP*. Units are in annual € M . Increments Δ are computed with respect to the *SQ* choices (using RCP 4.5 for period 2020-2040 as reference point). Biodiversity values represent number of species which are better off or equal to in the status quo.

In the scenario where farmers adapt to the new climate context, the expectations for both

RCPs in the medium term are similar, with only a small difference of € 3.2 M between € 60.9 M and € 57.7 M. However, while these differences may not be significant in the medium term, they are expected to become more pronounced in the long term. In the case of RCP 8.5, the economic value is projected to decrease to € 22.2 M. Coping strategies can mitigate these losses, with the total economic value of € 22.2 M representing a significant increase compared to the losses associated with a no-coping scenario. Nevertheless, this value is still lower than current levels (€ 29.7 M). On the other hand, irrigation water usage is projected to be lower in the RCP 8.5 scenario than in the RCP 4.5 scenario. This result arises from the fact that crop yield gaps between high-irrigated and low-irrigated crops decrease under climatic stress factors in the long term [14] and there may be situations where it is not worth intensifying irrigation.

Conservation of biodiversity under future climate conditions reveals significant differences. Regarding this issue, our analysis focuses on the coping strategy scenario since no coping will not have implications for biodiversity - considering that the bird biodiversity indices are the same as the current ones. On the other hand, no coping evidently implies that the water usage remains constant (we compute it to be 426 hm^3). If they cope, landscapes are driven solely by economic interests and biodiversity constraints have not been introduced in the maximization problem. As a result, only a portion of species will be preserved. Currently, without introducing biodiversity constraints, 31 species are preserved. Under RCP 4.5, biodiversity conservation levels remain constant with 27 species currently and 26 in the long term. However, under RCP 8.5, the number of protected species decreases from 27 to 12 in the medium term and recovers slightly to 18 in the long term. Focusing on steppe birds, farmers' economic decisions do not significantly contribute to their conservation, with only 1 out of 11 steppe bird species being conserved under RCP 4.5 for all periods and between 0 and 2 species conserved under RCP 8.5. In the long term under RCP 8.5, an additional steppe bird species is conserved as a direct consequence of crop yield results. In the RCP 8.5 scenario in the long-term, the reduction of the crop yield gap between highly irrigated

[14] See Table 4.6.

crops and low irrigated crops like corn and winter cereals could paradoxically increase the conservation of steppe species as more of the second crop is farmed. Winter cereals such as wheat and barley are less harmful for many bird species than corn and as the hectares harvested of these increase more species are preserved. Nevertheless, most steppe species would still not be able to overcome their status quo threshold under this climatic scenario.

Associated with this, the data presented in Table 4.7 reveal several trends in land use evolution under both RCP 4.5 and RCP 8.5 scenarios - considering that farmers cope with the different climate conditions -. For some land uses, such as rainfed barley, the surface of land remains constant from 2020 to 2100 under both scenarios. In fact, the surface allocated to barley at all levels of irrigation remains fairly stable under the RCP 4.5 scenario. However, for other crops, such as wheat 350mm, the surface decreases to zero under the RCP 4.5 scenario. In the case of traditionally irrigated barley there is an increase under the RCP 8.5 scenario. Conversely, under the RCP 8.5 scenario the land-use extension for highly irrigated corn (both 650mm or traditional) seems to be severely reduced, being replaced with irrigated wheat and barley.

Table (4.7) LANDSCAPE LAND USE OUTCOMES (km^2)

	RCP 4.5			*RCP 8.5*		
Land use	**2020-40**	**2040-70**	**2070-100**	**2020-40**	**2040-70**	**2070-100**
Barley: Rainfed	472	456	472	472	472	472
Barley: 150mm	52	72	24	20	24	0
Barley: 350mm	440	564	480	116	1024	172
Barley: Traditional	0	0	0	0	4	608
Corn: Rainfed	0	0	0	0	0	0
Corn: 150mm	0	0	0	0	0	0
Corn: 350mm	0	0	0	0	0	0
Corn: 650mm	568	544	696	500	40	0
Corn: Traditional	736	736	736	736	732	16
Wheat: Rainfed	0	16	0	0	0	0
Wheat: 150mm	0	0	0	0	0	0
Wheat: 350mm	140	20	0	564	112	1028
Wheat: Traditional	0	0	0	0	0	112

Notes: Values are in km^2. These values correspond to those of the coping strategy scenario.

We now consider the third scenario proposed, where a regulator optimizes land uses to maximize overall economic value under different levels of irrigation availability at the landscape level (Table 4.8). In other words, we examine the maximum economic value which can be obtained for the whole area analyzed if there is a maximum amount of water permitted for irrigation - which is spread optimally between locations -. Consequently, the first places to be irrigated would be those with higher benefits from irrigation in comparison with low or no irrigation (rainfed). We consider 7 water availability levels from 0 hm^3 to 600hm^3. Higher amounts were not considered because they did not have implications for economic value since they were not binding.

Table (4.8) OPTIMAL LAND-USE SCENARIO ECONOMIC VALUE UNDER DIFFERENT AGGREGATED IRRIGATION CONSTRAINTS (€ M)

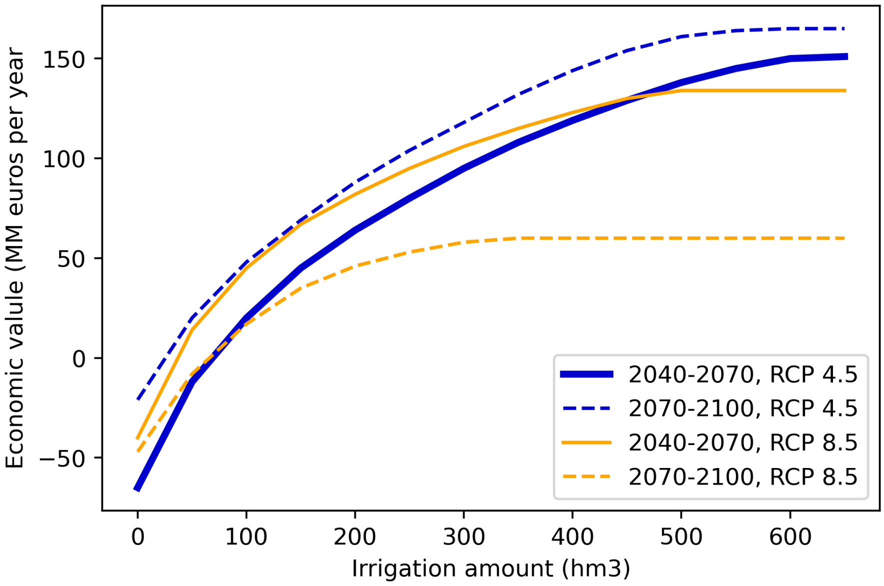
	RCP 4.5			**RCP 8.5**		
Irr. Constr.	*2020-40*	*2040-70*	*2070-100*	*2020-40*	*2040-70*	*2070-100*
$= 0 \ hm^3$	-63	-65	-21	-53	-40	-47
$\leq 100 \ hm^3$	31	20	48	36	45	17
$\leq 200 \ hm^3$	77	64	88	82	82	46
$\leq 300 \ hm^3$	111	95	118	117	106	58
$\leq 400 \ hm^3$	139	119	144	148	123	60
$\leq 500 \ hm^3$	161	138	161	172	134	60
$\leq 600 \ hm^3$	173	150	165	184	134	60

Notes: Each value is the crop yield average between cells for a synthetic year representing the period and *RCP*. Units are in annual € M . The increments are computed comparing for each *RCP* the values between the long term and medium term.

Although intuitive, the first result is that total economic value increases with larger amounts of water availability. However, this growth differs between periods and RCP. Under no irrigation constraints, productiveness coincides with those presented in Table 4.6 under the *Coping* strategy, since the optimal solution arises from the set of choices that maximize economic value at each location.[15] At the other extreme, when no irrigation is allowed (i.e., when the x-axis is zero), the most affected scenarios under high water scarcity are those of the period 2020-2040. This is due to the fact that, in these cases, crop yield differences between irrigating or not irrigating are higher. Following the same reasoning, it is RCP 8.5 for the period 2070-2100 where, in general, water scarcity has lower implications. This suggests that, under sever water scarcity, scenarios will yield similar outcomes while they will differ greatly if water is available.

[15]This can be seen in Table 4.6. Still, we presented them as two separate solutions since this was considered more intuitive in order to compare between *Coping strategy* and *No coping* strategy.

Figure (4.4) TOTAL ECONOMIC VALUE UNDER DIFFERENT AGGREGATE IRRIGATION CONSTRAINTS



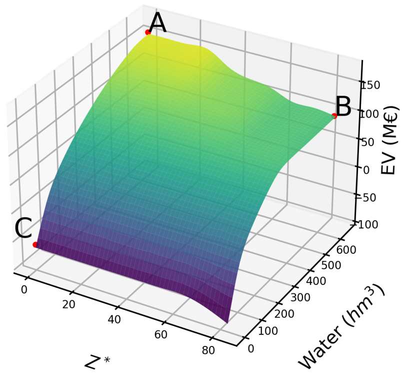
Notes:

As permitted irrigation increases, irrigation yield impact becomes inefficient for RCP 8.5 in the long term - marginal increases in irrigation on economic value decrease - since highly intensified crop yields - especially corn - present much lower returns in comparison with other scenarios (Figure 4.4). Since the costs of these crops are higher, the potential yield increase from adding these land uses does not benefit or increase economic return. However, it is interesting to note that, in the medium term, the functions representing economic return under different irrigation constraints cross each other. This occurs because, under the RCP 8.5 scenario, rainfed winter cereals have higher productivity than under the RCP 4.5, but it is lower in the case of highly irrigated crops (mainly *650mm* and *traditional*). Consequently, when no irrigation is allowed, the economic value curve corresponding to RCP 8.5 scenario is higher, but as more irrigation is allowed the curve corresponding to RCP 4.5 scenario grows faster and exceeds the former.

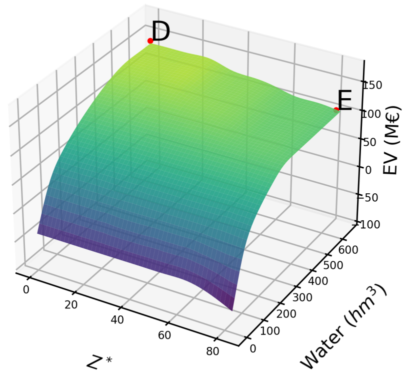
We have observed that different scenarios regarding emission pathways and time periods can have implications for economic value, water usage, and biodiversity conservation. We question what optimal solutions could be obtained if we sought to balance these three variables to meet the required objectives. To this end, as our fourth scenario, we solve the problem with three objectives using economic value as the objective function and water availability for irrigation and number of species conservation as constraints. These two objectives have been introduced into the model as a combination of constraints. That is, for each biodiversity objective Z^*, we have solved the problem for a subset of required water levels similar to that introduced in Equation 4.3. Note that this increases the number of solutions exponentially.

Graphically, these results can be represented on a three-dimensional map as a mesh of results where all combinations that maximize economic value under each combination of water and biodiversity constraints are obtained. This can be understood as a three-dimensional Pareto frontier (Figure 4.5, and Table 4.9 for some data values). Recall that each species s in cell i is affected by crop j and management regime r and that these are affected by the RCP and period considered. We observe that water usage and economic value are positively related - as shown throughout this paper -, while conservation goals are negatively related. Without water usage constraint - and expectedly - biodiversity constraints decrease maximum economic value independently of the scenario considered. Under RCP 4.5 we obtain the highest achievable economic values if no conservation goals are pursued. If all species must be conserved, the most productive scenario is found to be RCP 4.5 in the long term.

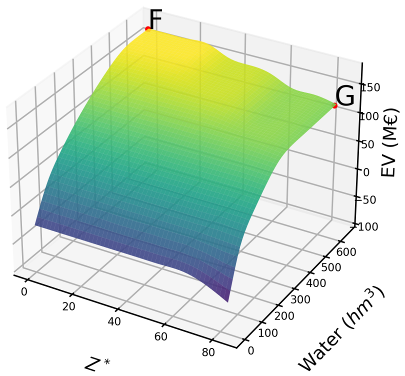
Figure (4.5) TRADE-OFFS BIODIVERSITY, ECONOMIC VALUE AND WATER USAGE



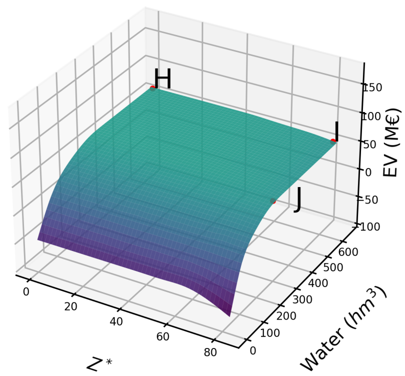
(a) Medium term RCP 4.5



(b) Medium term RCP 8.5



(c) Long term RCP 4.5



(d) Long term RCP 8.5

Table (4.9) HIGHLIGHTED POINTS ON THE 3D PARETO FRONTIER

Scenario	**Letter**	Z^*	**W** (hm^3)	**EV (€ M)**
RCP 4.5 medium term	A	0	650	151.4
	B	83	650	92.9
	C	0	0	-65.6
RCP 8.5 medium term	D	0	650	134.8
	E	83	650	103.1
RCP 4.5 long term	F	0	650	165.3
	G	83	650	118.5
RCP 8.5 long term	H	0	650	60.3
	I	83	650	51.1
	J	83	288	51.1

Notes: Letters correspond to those solutions indicated in Figure 4.5. Z^* is the number of species required to be conserved.

Moreover, we can see that water usage generates greater trade-offs than biodiversity conservation goals, which implies that biodiversity conservation regardless of the conservation goal Z^* can be achieved with a much lower opportunity cost than could be achieved under low water availability. For example, for the case of RCP 4.5 in the medium term, from point A to B, which represents the change in economic value obtained without water limitations from $Z^* = 0$ to $Z^* = 83$, the economic value difference is € 58.6 M. However, from point A to C, which represents the difference in economic value obtained without biodiversity preservation limitations $Z^* = 0$ from $L^* = 650$ to $L^* = 0mm$, is € 217.0 M [16]. This implies that the trade-off for water is 3.7 times higher than the trade-off for biodiversity conservation. Similar results can be obtained from the rest of the scenarios.

The trade-off with respect to biodiversity seems much less pronounced in the case of RCP 8.5 in the long term (point H to I) than in the case of RCP 4.5 (point F to G), which is a result of the smaller differences between the most productive irrigated herbaceous crops (corn) and barley or wheat. With total water availability, the difference between conserving all species or not is € 9.2 M.

[16] From A to B computations we just performed the following calculation: 151.4-92.9. From point A to C the calculation was 151.4-(-65.6).

Moreover, the trade-offs between conservation goals and economic value are much more pronounced if water availability is higher, since under lower water availability the options are reduced to rainfed lands, with smaller differences between them and consequently lower trade-offs. Also, note that there are no trade-offs between water usage and economic value up to a certain water amount value with higher conservation goals (e.g. point J to I, where increasing water from 288 hm^3 to 650 hm^3 does not increase economic value). However, when there are no conservation goals water will generate trade-offs with economic value even under higher water availability levels.

Consequently, high emission scenarios will presumably reduce trade-offs in the long term between conservation goals and economic values but this does not imply that it will be good for conservation, since we show that low-emission scenarios can achieve high conservation goals yielding higher economic value than under high-emission scenarios (compare point G with I).

4.5 Conclusions & discussion

In this study, we quantify the impact of climate change on main agricultural crops for a real landscape - the *Lleida* plain, a semiarid region in the North-East of the Iberian Peninsula -. For that, we firstly simulate crop yields under future climate conditions, using the crop growth simulator model *STICS* combined with climate projections from downscaled global climate models until 2100, soil features, management regimes and crops. The climate projections for our case study indicate general trends for the next decade, which will be exacerbated under the RCP 8.5. In general, temperatures are expected to rise, precipitations will decrease slightly in the long term for the RCP 8.5, and CO_2 levels will grow exponentially.

The results suggest that the climate change impact in the *Lleida* plain may be negative on agriculture production if the path of emissions continues as at present (RCP 8.5), which is associated with higher temperatures and lower precipitations. However, these changes will be especially noticeable in the long term (2070-2100), while in the medium term (2040-2070) they would remain stable - or even improving -. In the long term, the conjoint effects of temperatures, precipitations and CO_2 will reduce crop yield productivity for corn, wheat and barley for the management regimes analyzed in the RCP 8.5 emission scenario, while under the RCP 4.5, crop yields are even expected to increase. The total economic value loss in the long term will mainly be the consequence of high-irrigated corn productivities, which reduce the crop yield gap between low-rate irrigated winter cereals. This will also provoke a decrease in water demand for irrigation. Independently of the emission scenario, to continue with current land-use choices results in a significant agricultural production loss, while coping would reduce the impact loss. For example, in the long term under the RCP 8.5, non adaptation measures would imply a negative economic balance in the region. Consequently, these changes are also expected to be accompanied by landscape transformations with a higher presence of winter cereals, or other crops which have not been considered in this work, and a progressive abandonment of corn. Another possible option is that more irrigation will be needed to satisfy current conditions. However, higher intensification levels were beyond the scope of this work, since we only simulated limited predefined irrigation schedules to enhance computation feasibility. Moreover, we should expect that water availability will be reduced, so these results are plausible. Another expected consequence is that plant material and management practices will be adapted to this climate scenario.

We built upon these findings by conducting a comparative analysis of the effects of each RCP and term period on a community of 83 bird species, utilizing habitat suitability indices. Our results indicate that while trade-offs between biodiversity and agricultural economic value are lower in a high-emission scenario, economic values are higher in a low-emission scenario for any conservation objective.

If we attempt to maximize economic value under a combined set of water availability and conservation goals, we find that the highest achievable economic values are obtained under RCP 4.5 without conservation goals, while the most productive scenario with all species conserved is RCP 4.5 in the long term. On the other hand, water usage generates greater trade-offs than biodiversity conservation goals, implying that biodiversity conservation can be achieved with a much lower opportunity cost than under low water availability. Also, we find that trade-offs between conservation goals and economic value are more pronounced with higher water disposal.

These results add to the climate change impact analysis literature, and provide some insights into negative implications of a continuationist emissions path. They intend to build a bridge between different disciplines and take advantage of new methods which have not usually been adopted in the economic literature. Although general trends seem to be clear, absolute quantifications should be taken with caution. Moreover, we have considered a limited set of land uses which are carried out at present but, in the future, they could be replaced with other options not included in the model, or technological advances could overcome some of the problems that we pose throughout this work. Additionally, we considered only 3 time periods for simplicity and to avoid excessive computation workload. We could expand this work for simulations year by year to obtain more precise trends. Still, we expect it to give some useful insights into the implications of climate change in an agricultural region, with quantified results.

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Chapter 5

General conclusions and final remarks

This thesis explores the trade-offs between agricultural economic value and biodiversity through the use of a spatially explicit model framework in a real landscape: the Lleida plain. It builds on existing literature on the economics of agriculture and on the provision of ecosystem resources. It also draws on studies of spatial models and optimization. Additionally, it employs tools and methodologies from the fields of agronomy, biology, and meteorology, giving this work an interdisciplinary character that helps to achieve objectives that may be useful in a realistic scenario. However, this thesis has distinguished itself from pre-existing literature in several ways. Firstly, it focuses exclusively on agricultural environments, delving into land uses within this category rather than expanding to more general land uses. To this end, the use of agronomic models, whose precision and versatility have allowed us to simulate crop yields under very diverse conditions, as well as the adaptation of biological models to these specific land uses, has been crucial. On the other hand, it addresses the issue of policy acceptance when there are losers, which is a topic that has been scarcely explored in these type of models with real data. It also explores the implication for the economic value of the agricultural performance and biodiversity of climate change. The use of these techniques and the results obtained with them could be useful for making decisions aimed at satisfying conservation objectives without compromising economic development.

The first essay establishes the methodological basis for constructing the required data and performing my analysis. In this stage, I quantified the trade-offs between agricultural development in monetary terms and the requirements of a bird community through habitat suitability indices, using spatial optimization processes. The results obtained in this first study have allowed us to realistically model and simulate the yields of 4 predominant crops under various management practices associated with agricultural intensification for an entire region, where productivity was conditioned by location, which is characterized by specific aspects of climate and soil. On the other hand, I have been able to assign the preference for each of these land uses for each species in a community of 83 birds - with special emphasis on 11 species considered steppe species. By introducing all these values into a multi-objective spatial maximization model solved through a computational algorithm, I have been able to construct a Pareto frontier defining the highest possible economic value if we want to conserve a specific number of species. This model also took into account current conservation areas, where irrigation is limited, as well as areas without irrigation infrastructure. My results indicate that although meeting biodiversity objectives may have an opportunity cost, optimal landscape-level land uses can propose significant economic benefits without compromising bird sustainability. I also expanded these results by removing the assumption that conservation areas must remain fixed, in order to observe their optimal location with respect to the current one, again observing significant differences that suggest alternative solutions to the current ones if we want to meet our objectives. However, all these results are based on a series of assumptions that facilitate their resolution and interpretation. In my study, I assume that farmers are homogeneous in terms of their ability to generate economic value, and are based on the assumption that the agronomic model used simulates ideal situations where the existence of biotic stresses (weeds, diseases and pests) or some extreme climatic events is ruled out. Also, the habitat suitability indices employed are a necessary simplification of the bird requirements to carry out such a demanding optimization problem, although it represents an advance in economic models to accurately reflect those factors that may benefit or harm each species. Last, it is a static model, where some factors cited in the biodiversity ecology literature have not been considered, such as connectivity, migration capacity, extinction thresholds, maximum capacity or competition between species.

In the second study, I have used much of the work and data generated from the first one to be able to take the study towards a new goal. I observed that optimal solutions that maximize agricultural economic value under demanding biodiversity constraints require the existence of a significant proportion of losers. I wanted to delve deeper into the existence of losers based on the solutions obtained, as a relevant factor that could block their implementation. I expanded these results, proposing alternative solutions or policy mechanisms and comparing their performance. In particular, my results suggest that an agreement between farmers involving the transfer of winners to losers in such a way that all farmers obtain a greater economic benefit than in the status quo is possible. On the other hand, an optimal solution in which there are no losers, maintaining biodiversity objectives for all species and without transfers, is also possible, albeit at a cost. Next, I compare the effectiveness of these agreements with the effectiveness of a homogeneous subsidy for various less intensive land uses. I was able to observe that homogeneous type subsidies are very inefficient and would entail unaffordable budgets if biodiversity objectives obtained under agreements are to be achieved through this type of subsidies. If policy is to be carried out through subsidies, I suggest that an individualized subsidy for each farmer may be a much more efficient option than a homogeneous one, as it adjusts much more to the actual costs of each of them. However, as this would require information from the regulator that is not normally available, I assume that the regulator only has partial information and estimates the losses and gains of each farmer based on surveys. Although this incomplete information may result in the existence of losers, I observe that it obtains much more efficient biodiversity results than a homogeneous subsidy.

Finally, in the last chapter, I departed from the current situations and considered the implications that climate change may have on economic value and biodiversity during the following decades. I use my model to predict the impact of climate change on the economic value and biodiversity levels until 2100 using climate projection data, and perform the analysis under different assumptions regarding farmer crop and management regime choices and under two different RCP scenarios. In addition, I propose an economic value maximization scenario for cases where available water is not sufficient to meet crop needs. I observe that under an RCP 8.5 emissions scenario the productivity of crops, especially corn, can be severely affected. In contrast, a scenario with fewer carbon emissions (RCP 4.5) could curb this trend by maintaining productivity in the long term. However, my model did not consider another series of factors such as the introduction of land uses not currently existent in the area, the adoption of more resilient crops or the use of more intensive water uses. However, given the expected water shortage, I consider that this last option may not be valid in the future. I expect to take this into account in future work.

This research has posed a challenge from several points of view. Some of them involve the database that I had to build, since there were no existing databases to carry out the analysis. On the other hand, the interdisciplinary nature has involved managing information with experts from each branch. The computational aspect regarding geostatistical analysis, coding and data management has also involved a significant workload. However, I am aware that these results can be expanded and improved from several points of view that involve greater realism in terms of having implications for the design of conservation policies that combine several objectives that compete with each other. Nevertheless, I believe that they can provide interesting results and aspects that contribute to scientific knowledge.