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Dissertation

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'Field and Farm Size Optimization of Agricultural Ecosystems: Decision Making at the Local Government Level in Intensively Used Agricultural Lands in Germany'

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Gießen, den 15.11.2017 Amanda Eigner

Eidesstattliche Erklärung

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Abstract

In recent decades, species diversity has been greatly reduced within agro-ecosystems, although political efforts to protect biodiversity exist (Batàry et al., 2015; Pe'er et al., 2014). The aim of this study is to show the spatial effects of long-term farming decisions regarding field and farm sizes in order to improve political impact analyses on biodiversity protection. The concept of socio-ecological resilience has a systemic and dynamic view that considers several interrelated scales and domains. Changes in one domain-scale combination might trigger others and lead to an irreversible regime shift throughout the whole socio-economic system (Kinzig et al., 2006) that is often less desired by society (Matthews and Selman, 2006). By applying this concept, our economic model creates relationships with other domains in order to get a comprehensive view of the socio-ecological system of the case study. In a sequence of linear mixed integer and non-linear programming models it covers spatial and temporal dimensions of structural change processes by considering economies of size. Farmers of three different farm types decide on their livestock production, the cultivation of crops and flowering strips, as well as their field and farm sizes. Using our innovative approach, the cropping and field size results were spatially translated via GIS. Based on the resulting maps, three different spatial biodiversity indicators were calculated and analyzed. The model was applied to a small municipality within intensively used agricultural lands in Hesse and tested three alternate political incentive schemes. Two of them were inspired by the new CAP reform 2020 and the results showed that for our case study, both were ineffective with respect to habitat quality. Moreover, the new CAP reform even accelerated structural change processes and led to a more homogeneous landscape with larger fields and fewer farms. In the third policy design, alleviation of structural change and the establishment of many more semi-natural habitats through incentive-based payment schemes led to higher levels of biotic diversity. Dairy farmers quit the agricultural sector in all simulated futures. This had impacts on the labour market and might influence population dynamics in rural areas. The interplay of spatial scales may shift dynamic agro-ecosystems into other ones, where they might be locked in and less resilient against perturbations (Walker et al., 2004). Therefore, political impact analyses need to consider economic long-term decisions at the farm scale and their spatial meaning at the field scale. Both scales interact and have impacts on the landscape scale.

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List of Abbreviations

ABM	Agent-based model
AES	Agri-environmental schemes
ANP	Agri-nature premium
CAP	Common agricultural policy
CBD	Convention on Biological Diversity
DC	Direct costs
DM	Dry matter
DP	Direct payments
EFA	Ecological focus area
EoS	Economies of size
ES	Ecosystem services
GAMS	General algebraic modeling system
GIS	Geographical information system
HALM	Hessisches Programm für Agrarumwelt- und Landschaftspflegemaßnahmen
HLUG	Hessisches Landesamt für Umwelt und Geologie
KW	Kilowatts
MC	Machine costs
MW	Megawatts
MWh	Megawatts per hour
NABU	Naturschutzbund
SC	Service costs

SES	Socio-ecological system
SP	Sustainability premium
WTO	World Trade Organization

1 Chapter 1. Introduction

1.1. Loss of species diversity in agricultural landscapes

‘Where have all the flowers gone?’ is the beginning of an American pop song written by Peter Seeger and later translated into German known under the title ‘Weißt du wo die Blumen sind?’. The song is against war in which, after flowers, girls, men, soldiers, and finally the graveyards also disappear. Although the topic of this work is somewhat different from war, the song text reveals some inspiring notions. It indicates that at the beginning of a horrible development, aesthetic values, having a deep meaning for individual sensations, vanish first and have to make way for overriding aims. The view of cultural landscapes are often shaped by aesthetic, romantic, and historical ideas and therefore inherit high cultural values. However, when taking a walk through rural areas where agricultural production dominates land use, these ideas and expectations are often not met. Instead of enjoying the sound of busy insects, singing birds, or the beauty of plentiful butterflies and sweet-smelling flowers while walking along naturally shaped ditches surrounded by alley trees and idyllic lush pastures and cornfields, agrarian deserts with monoculture and only a few natural landscape elements that provide shelter and food for a diverse fauna open up. What we often actually smell is diesel from big machinery, pesticides, and fertilizer, giving space only for the intended and economically valuable crops.

Over the last decades, one third of the breeding bird species have disappeared from German landscapes, and farmland birds are affected most severely (Wahl et al., 2015). Besides birds, populations of other species also continue to decrease. As an example, Sorg et al. (2013) found out that the biomass of fluctuating insects declined by more than 75% between 1989 and 2013 in a nature conservation area near Krefeld, which is close to the border of the Netherlands. Insects not only deliver essential ecosystem services such as pollination or pest control but also serve as fodder for birds and other predators. As another example, in central Germany, the regional species pool of vascular plants declined by 23% since 1950 (Meyer et al., 2013). Mammals are also affected by high extinction rates. The population of hares has declined by 50% within the last ten years (Wildtierschutz Deutschland e.V., 2015). The list of species extinction is long and cannot be presented here. There is a general trend to biotic homogenization, which means that specialists are gradually displaced by generalists with yet unknown impacts on future ecological and evolutionary processes (Julliard et al., 2004; Olden et al., 2004).

First countermeasures for decreasing environmental quality and its associated biodiversity loss in Europe were already implemented on behalf of the MacSharry reform of the common agricultural policy (CAP) in 1992 in the shape of so called agri-environmental schemes (AES). At the same time, CAP reforms promoted farm productivity and a higher efficiency through farm modernization (Burrell, 2009). This endeavor is grounded in increasing competition pressures originating from liberalization processes that are advanced by the World Trade Organization (WTO). Whilst political countermeasures against environmental degradation failed (Kleijn et al., 2006), farm productivity goals were reached by utilizing economies of size (EoS). Larger fields lead to lower costs and thus increase productivity (Johnston and Mellor, 1961). Hötker and Leuschner (2014) hold agricultural field consolidation accountable for many habitat type changes, which led to an intended homogeneity and thus simplification of landscapes. The main reasons for biodiversity losses are agricultural intensification (Hendrickx et al., 2007; Bengtsson et al., 2005; Tscharntke et al., 2005; Fuller et al., 2005), the loss of extensively used grassland, fallow ground, other habitat types (Settele et al., 2010; Benton et al., 2003; Sanderson et al., 2009; Batàry et al., 2010; Cornulier et al., 2010; Wahl et al., 2015), and the decline of landscape heterogeneity (Hendrickx et al., 2007; Sanderson et al., 2009; Batàry et al., 2010; Vandermeer, 2011; Tscharntke et al., 2011; Doxa et al., 2012).

The new 'CAP towards 2020' reform seems to put a high priority on environmental issues (European Commission, 2013). However, in the public debate the new reform is discussed controversially. There is a lot of concern that the new CAP reform follows a green-washing strategy (Matthews, 2013), which would imply that the European Commission is sustainable with respect to their mission to be steeled for market competition without being serious about environmentally destructive economies. In any case, improving ecosystem services and species diversity has not yet been effectively addressed by the CAP reforms (Batàry et al., 2015; Pe'er et al., 2014). Against warnings of the German scientific advisory board for biodiversity and genetic resources to reduce landscape elements such as hedges, natural habitats, field strips, or fallow land to less than 7% of the agricultural area (Gerowitt et al., 2012), the new 'CAP towards 2020' reform planned only 5% of these semi-natural habitats, which are implemented as so called 'ecological focus areas' (EFAs). On these EFAs, certain crop types and cropping management practices are even allowed[1]. The German advisory board further emphasized the importance of diversifying agricultural production (*ibid.*). Especially with respect to climatic changes, diverse or mosaic landscapes that offer habitat requirements for species are urgently needed. In earlier times, temperature changed in comparable dimensions, and species could react easily either through dispersion or through microclimatic variations. Nowadays habitat fragmentation hampers these strategies and climatic change could in fact be fatal for biodiversity (Pearce-Higgins et al., 2015).

Following the new crop diversity strategy of the 'CAP towards 2020' reform, in most regions not many farmers are supposed to change their crop rotation schemes (European Commission, 2016). For example, in the Wetterau county in central Germany, which is an agriculturally intensified area, the main crops in 2010 were wheat (45%), rapeseed (13%), and barley (7%) (Hessisches Statistisches Landesamt, 2012j). The most intensively producing municipality within the Wetterau county cultivates 51% wheat, 16% sugar beets and 11% rapeseed as main crops. Even there farmers easily fulfill the new crop diversification plans of the EC council without being forced to change their usual crop rotation schemes. Similar criticism arose for EFAs. A new study of Lakner and Bosse (2016) showed that the implementation of EFAs has high deadweight effects.

Academics have been developed sustainability indicators with the aim to improve natural resource management and policy (Woodhouse et al., 2000). However, by means of effectiveness, resource management ought to change from a rather relativistic to an absolute view including absolute limits. The gap between what is known to be done in order to improve the allocation of natural resources and natural life-supporting systems, and what has actually been done rises, and normative aspects of humans' impacts on the ecological system should not be excluded from the scientific agenda. Small political steps in the right direction are positive but won't improve the situation and won't stop the system from collapsing (Fischer et al., 2007).

First political attempts to set absolute limits on critical biodiversity thresholds for Germany were made in 2010 by the Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety (BMBU). The BMBU (2010) summarized all catastrophic phenomena regarding biodiversity losses on the basis of several indicators describing the status of flora and fauna. They found that eleven of thirteen biodiversity indicators are far beyond what is required to reach the desired biodiversity goals of the CBD.

[1]Annex X of the supplementing Regulation (EU) No. 1307/2013 of the European Parliament

1.2. Research objectives

The aim of our study is to analyze spatial impacts of political incentive schemes. Using GIS, we seek to show local landscape dynamics induced by farmers' decision making and the impacts on biodiversity. Within the ecological impact assessment, we focus on pollinators such as bees who deliver important ecosystem services. The spatial representation of farmers' decision making is characterized by the following:

- field and farm sizes.
- cropping changes.
- participation in environmental measurements in the mode of flowering strips.

Our case study is a municipality within an intensified agricultural area in Hesse, Germany. Particularly in intensified areas, political measurements to enhance biodiversity in agro-ecosystems are not effectively implemented (Batàry et al., 2015). Temporal planning plays a crucial role in political targeting since long-term processes may undermine biodiversity goals (Beunen et al., 2013). Therefore, we take into account structural change as a slow but concurrent social and economic process. It will be modeled by considering incentives to use economies of size.

This leads us to the hypothesis that the new CAP 2020 reform does not match with the goal of enhancing biodiversity due to structural change within the study site. Our research questions are further formulated as follows:

1. To what extent is the socio-ecological system of our case study influenced by the new CAP 2020 reform?
2. How efficient is the CAP 2020 reform with respect to the intended European biodiversity targets within the study site?
3. Is a restructure of governmental payments in order to improve biodiversity protection within the research area superior?

The research questions are further divided into several secondary questions, which we will address before we set up our conceptual and analytical framework. They are summarized in the following:

1. Why is it necessary to sustain biotic diversity within agricultural farmlands?

2. Which processes within socio-ecological systems need to be considered for governmental impact assessments on biotic diversity?

3. How can farmers' decisions technically be linked with the field and landscape level?

1.3. Study structure

Chapter two starts with theoretical concepts about the connectivity between biodiversity and human well-being. It elaborates on resilience thinking since we regard agricultural systems as being part of social and cultural systems. Since we emphasize biodiversity protection, resilience thinking within socio-ecological systems forms the conceptual and analytical framework of our study. The following part of the chapter covers our methodological background. It summarizes philosophies of landscape-oriented agricultural modeling approaches that strive to assess governmental impacts on human-environmental systems and leads to a synthesis upon which our applied modeling approach is based. In the last part of the chapter we summarize the common agricultural policy (CAP) reform towards 2020 in short.

Chapter three is dedicated to our research area. It outlines the socio-ecological system of our case study with respect to the conceptual framework of the previous chapter. In doing so, it depicts the ecological, socio-cultural, and economic domain of the case study. Afterwards, an overview of the agricultural production system is given. It builds the data foundation for our mathematical model.

In chapter four our mathematical model is finally described in detail. The first section starts with explanations regarding the model content as well as the programming language used. The mathematical model formulations are presented in the second section. Our three alternate political incentive schemes are delineated therein. They are aligned towards the CAP 2020 reform and other more radical policy suggestions. The third section covers the description of our chosen biodiversity indicators that we will apply to measure changes in biodiversity in order to assess the political impacts of our three different policy designs.

Chapter five gives an overview of the model results. Results are evaluated with respect to our analytical framework and are set in relation to findings of other authors. After that, simulated spatial land use patterns are visualized and presented. They form the sandbox for our spatially explicit biodiversity indicator calculations, which are analyzed at the end of the chapter.

In chapter six, we classify our findings in a broader context and contrast them with respect to our conceptual framework. Further clues on changes in biotic diversity due to political control within the study site are given here. Thereafter we outline methodological issues within the given framework conditions. At the end of this chapter, research needs and model proposals are elaborated.

The last chapter summarizes our work and gives final conclusions.

Chapter 2. State of the art and model frameworks

This chapter is concerned with theoretical foundations of human-environmental systems and practical sides of their modeling applications. In the first section, theoretical concepts about biodiversity and the connection with human well-being are depicted. Thereafter, resilience theory within socio-ecological systems is considered. Based on the socio-ecological resilience theory, the conceptual framework of our model has been developed. It leads us to our analytical framework against which we are going to evaluate our model results (in chapter five). The fourth section outlines the methodological background of landscape-oriented agricultural modeling approaches. After a literature review that summarizes several modeling techniques, we develop our modeling approach. It is a synthesis of the presented model approaches. The fifth section gives a short overview of the new common agricultural policy (CAP) reform 2020. It delivers an orientation for the development of three alternate policy schemes, which we will apply to our model.

2.1. Theoretical background

2.1.1. Biodiversity and human well-being

Academics involved in human-environmental disciplines often capture the impact of natural resources and processes on human well-being through the concept of ecosystem services (ES) established by the *Millennium Ecosystem Assessment* founded in 2001. It considers nature as an important and essential provider of basic living conditions from which humans benefit. The intention of the ES concept was to operationalize essential natural processes for humans' natural resource use due to critical resource depletion and degradation processes occurring around the globe. This ought to shift attention onto the preservation of natural resources and ecosystems (Millenium Ecosystem Assessment, 2005). However, some conflicts with respect to definitions and the usefulness of the ES concept exist (De Groot et al., 2010). Mainly ecologists are critical of the connotation that nature provides its services *for* humans. They would rather talk about ecosystem functions that maintain the ecosystem and all living species therein. This directs the focus onto the natural system as a whole, which operates due to biotic diversity (Spangenberg and Settele, 2010).

Conceptual and methodological uncertainties about how to address natural systems and how to connect ES with biodiversity remain (Kremen, 2005). This is mainly caused by big knowledge gaps in underlying biological processes and due to interdisciplinary communication issues between social and natural scientists when it comes to the valuation of ES (Bockstael et al., 1995; Noss, 2007; Jackson et al., 2007; Meinard and Grill, 2011). Nevertheless, there are agreements about the supporting function of biotic diversity on food production and other ES, such as nutrient cycling, pollination, pest control, and cultural recreation (Millenium Ecosystem Assessment, 2005).

Biodiversity as such is not easy to grasp (Duelli, 1997). The multidimensional notion of biodiversity is reflected in a huge variety of indicators measuring biodiversity. One hundred highly diverse indicators are nominated for the evaluation of the 2020 goals of the Convention on Biological Diversity (CBD) (Pereira et al., 2013). By increasing one of these indicators, another might decrease (Mouysset et al., 2012). Most attention and protection is given to charismatic species. The decision about which biodiversity indicator to take depends on certain basic framework conditions of the study under investigation and is additionally influenced by intrinsic values (Metrick and Weitzman, 1998)[1]. In order to overcome normative biases, other ecological concepts with broader views have been developed. They have the notion of not knowing everything about the functioning of ecosystems and the living species therein. The concept of ecological memory, for example, considers the health of whole ecosystems and assumes that their functioning maintains the provision of services upon which humans depend (Bengtsson et al., 2003). The health of ecosystems strongly depends on regulatory processes, and the ability to self-organize is the highest predictor for the delivery of ES (Müller, 2005; Kay, 1993).

Connected to ecosystem health are other concepts such as ecological integrity or resilience. Ecological integrity as a general principle strives to prevent unspecific ecological risks by protecting ecosystem patterns and processes (Barkmann et al., 2001). Ecological patterns and processes are strongly interlinked and mutually adapted, which makes the system more efficient in generating ES (Müller, 2005; MacCann, 2000; Ives and Carpender, 2007; Keesing et al., 2010; Tilman et al., 2002). The older an ecosystem is, the more disparate biotic and abiotic patterns emerge. The more numerous and diverse these patterns are, the more complex their interactions are and the higher

[1]This is then called the *Noah's Ark problem*.

the degree of information (Forman and Godron, 1986) is. Therefore, '[...] the appropriate level of biodiversity is a necessary condition for the sustainability of any managed system' (Perrings, 1998, p. 514). This goes hand in hand with the insurance hypothesis claiming that high biodiversity levels keep ecosystems functioning (Naeem and Li, 1997).

2.1.2. Resilience theory

The resilience theory evolved in the early eighties and was significantly shaped by C.S. Holling. Holling (1973) challenged the usefulness of traditional equilibrium-oriented and quantitative analyses of ecological systems. He suggested focusing on the behavior of ecological systems and their elements. According to that, the total number of species or the change in numbers should not be the focus, but rather what kind of conditions are needed for the probability of species survival. Holling [*ibid.*] considered resilience and stability as two unique characteristics of ecological systems as a result of evolutionary strategies and defines them as follows: 'Resilience determines the persistence of relationships within a system and is a measure of the ability of these systems to absorb changes of state variables, driving variables, and parameters, and still persist' and: 'Stability [...] is the ability of a system to return to an equilibrium state after a temporary disturbance' (Holling, 1973, p. 17).

Ecological systems can have high resilience, but a low stability and vice versa. The higher the heterogeneity in space and time, the higher the resilience. Management approaches adapted to the resilience theory need a shift in perspective since they do not predict the future in numbers of species left. They rather assess the capacity to absorb and accommodate future events (Holling, 1973). Ecological models need to be open and consider heterogeneity in time and space where unexpected events occur. Studies confirm that the lower the diversity of a system is, the lower the capacity to recover and the lower the ecological resilience is (Bengtsson et al., 2003).

Gunderson and Holling (2001) supplemented the resilience theory with the concept of adaptive cycles. They claim that an ecological system passes through a typical two-dimensional cycle visualized as a recumbent eight. The two dimensions are stored capital and connectedness. Depending on the starting point and direction, four phases follow upon each other: (a) an exploitation phase, in which stored capital and connectedness are low but increasing, (b) a conservation phase, where stored capital and connectedness are highest, (c) a release phase, where stored capital decreases to the utmost and connectedness drops, and (d) a reorganization phase, where the stored capital again accumulates, having a low connectedness. As such, the resilience theory is free of normative aspects. It tries to quantify the state of an ecosystem and whether the ecosystem remains in that state, or how far it is from switching into another one (Brand and Jax, 2007).

In social, economical, and environmental science, resilience theory is often mixed up with the normative concept of sustainability and is further used in a broader perspective in order to to analyze socio-ecological systems at multiple scales (Cumming, 2011; Folke, 2006). Due to its use in several scientific disciplines, a clear distinction between ecological resilience and social-ecological resilience has to be made. Ecological resilience is a well and precisely defined concept for applications in ecological science. On the contrary, socio-ecological resilience, which is used in interdisciplinary studies, does not claim to be quantifiable and has a rather vague conceptual framework including normative connotations (Brand and Jax, 2007).

2.1.3. Socio-ecological resilience

Matthews and Selman (2006) used resilience thinking to combine human and natural aspects by focusing on the landscape as a multifunctional and holistic entity that underlies intrinsic dynamic and cyclic changes as described in Gunderson and Holling (2001). Landscapes are created through human practices and particularly through agricultural land management. Therefore, landscape changes are driven by natural pressures, social and economic needs, unconscious actions (e.g. demographic trends or human-induced climatic changes), and human perceptions of how landscapes should look. As a result, today's cultural landscapes are characterized by agricultural land intensification or rural abandonment, both leading to landscape homogenization and perceived cultural value losses within the society. According to that, Matthews and Selman (2006) differentiated between virtuous and vicious cycles, meaning desired and undesired basins of attractions. They promoted modeling approaches that test changes in cultural landscapes before they are altered. These modeling approaches should take into account economic, social, and natural components interacting with each other. The authors suggested modeling policy impacts and how far they stabilize socio-ecological systems or whether they lead them into other (virtuous) basins of attraction.

The concept of socio-ecological resilience combines resilience theory with socio-ecological systems. It has the capacity to shift the manner of thinking about economic-ecology dynamics from an equilibrium paradigm to a broader view of system interactions at multiple scales that operate around instead of at an equilibrium (Anderies et al., 2006). The concept of socio-ecological resilience is in its infancy but bears lessons from management practices since it reveals important parts of system dynamics. Socio-ecological systems (SES) are open systems that permanently adapt to material, energy, and information flows. They are determined by human and natural components (Berkes et al., 2003).

Walker et al. (2004) borrowed the adaptive cycle concept of resilience theory and transferred it to SES in order to contribute to sustainability research. They assumed that an SES passes through dynamic development phases regarding stored capital and connectedness, i.e. without changes in the framework conditions, and moves within its so-called *basin of attraction*. Precariousness as one of the components making up a basin of attraction is defined as the distance of the current state of the system from a threshold limiting the basin. The probability to cross a critical threshold is determined by the resistance of a system, which can be understood as the depth component of the basin. The level of the system's (social) ability to control the components of a basin without leading it to another one is defined as adaptability. Transformability is the capacity of a basin to transform into a totally new and desired system. Within a so-called stability landscape, several basins that are separated by their thresholds can exist. Scales above or below a considered variable influence the dynamic behavior of SES, which is referred to as panarchy.

SESes need clear definitions of their state variables with regard to location and level as well as their connections. In order to measure changes in the resilience of a socio-ecological system, a reference system is needed (Cumming, 2011). However, numbers for the components of a stability landscape are often not quantifiable or even understood completely, and the location as well as thresholds of a current state (regime) are not known. Thresholds are hard to define since they underlie altering processes of slowly changing variables at higher levels of ecological self-organization, on stochastic events outside of the system, and on dynamics within and between the sub-systems (Scheffer and Carpenter, 2003; Carpenter, 2003). Interrelations across ecological, social, and economical domains may lead to changes in or passings of thresholds (Walker and Meyers, 2004). Every kind of connectivity changes within ecosystems affect the socio-ecological resilience of a system. A full description of an SES and its current location within a basin of attraction according to its components in order to measure the socio-ecological resilience is challenging and has not yet been reached (Cumming, 2011). More detailed information is needed in order to understand forces and feedbacks that might trigger regime shifts. Models that take into account network structures at multiple scales and domains are required in order to gain more insights into the dynamics of SES (Anderies et al., 2006).

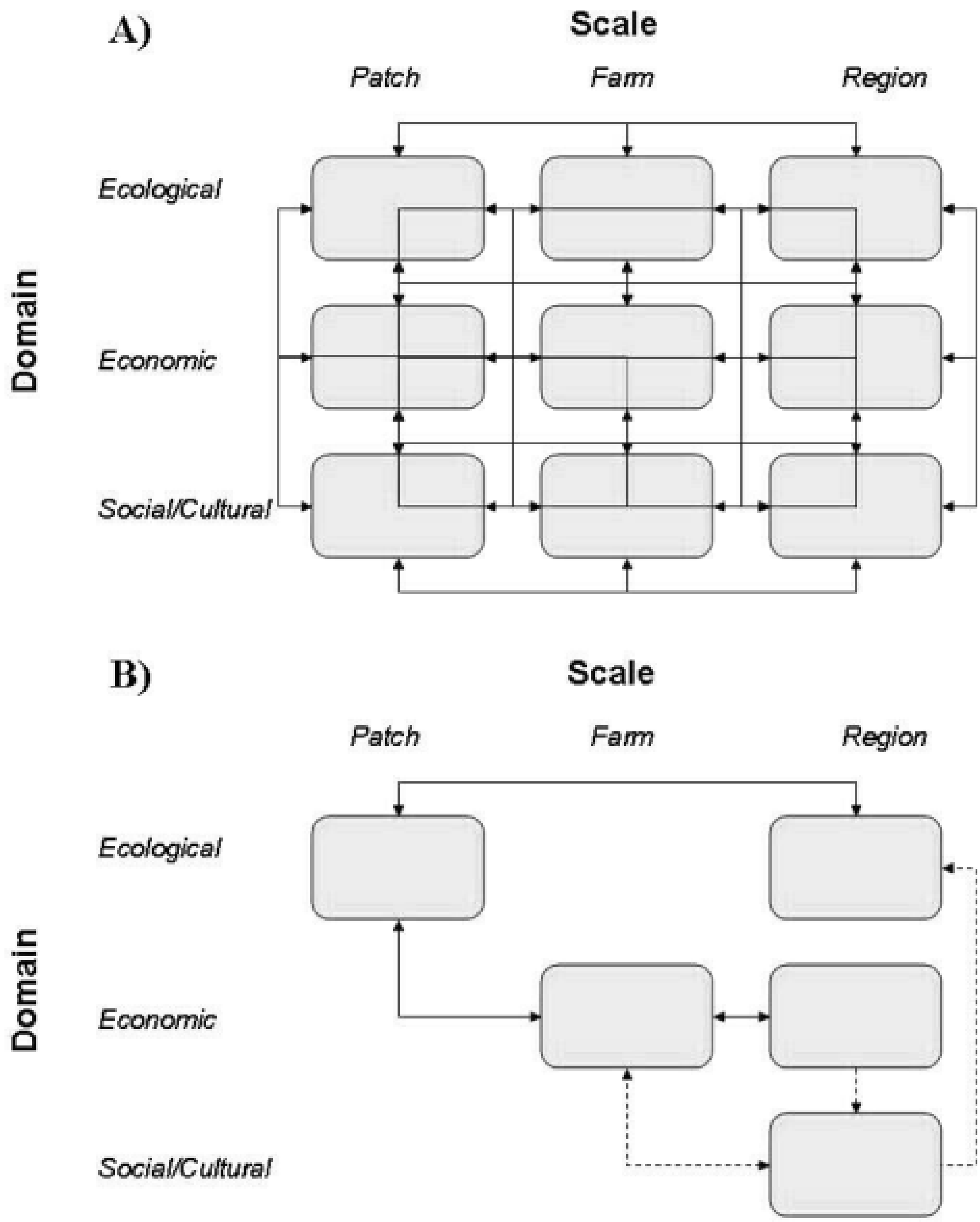

Figure 2.1.: A: Domain-scale combinations and their interaction possibilities (arrows). B: Critical domain-scale interactions of four case studies (solid lines indicate cascading effects in all case studies, whereas the dotted lines indicate cascading effects in two case studies) (Kinzig et al., 2006).

2.2. Conceptual framework

The systemic view regarding the delivery of ES is one strategic level of implementing a political impact assessment within our study. It requires a holistic understanding of human-environmental systems. The other strategic level of conducting our political impact assessment refers to the systematic and technical procedure in which we focus on the spatial translation of farmers' short and long-term decisions. This procedure will be described in more detail in section 2.4.2 after a comprehensive literature review of practical modeling applications in section 2.4.1. In the following, conceptual framework suggestions of socio-ecological systems are presented. These conceptual frameworks deliver practical guidance to define thresholds and system components of our case study.

With respect to the research questions, we test whether the new CAP 2020 reform is capable of leading the system of our case study into a desired basin of attraction. As described above, a switch from one to another basin of attraction is determined by the resilience of an SES. Therefore, a conceptual framework that allows one to operationalize resilience thinking is required. What has been 'desired' is aligned to the German national indicator report of the BMBU (2015). According to that report, only 66% of the biodiversity target within agricultural land was reached in 2008. The political aim was to reach 100% by the year 2015. Newest results showed that in 2011, the biodiversity indicator level had even decreased to 56% (BMBU, 2015). Since then the BMBU has not released an updated version of the German national indicator report. We take these numbers as a benchmark. Assuming that the target is reached at the one hundred percent mark, biodiversity indicators need to increase at rather high rates in order to reach the biodiversity targets.

However, which variables drive the system into a desired state? Ecological and social systems are intertwined, and the conditions of resilient systems need to be examined. The socio-ecological resilience concept is targeted to deliver a holistic view of human-environmental systems (Fischer et al., 2007; Perrings, 1998). With the concept of socio-ecological resilience, several domains and scales of a system are addressed (Walker et al., 2004). They are delineated in the following.

Kinzig et al. (2006) used the socio-ecological resilience theory to develop a conceptual framework that focuses on regime shifts and cascading effects. They considered interrelated scales with respect to their ecological, economic, and sociocultural domains. Within a domain, influences on the dynamics of SES at three scales can occur:

- small scale (patches).
- the medium scale (farms, managed entities).
- the large scale (regions).

All possible interactions between scales and domains are depicted in figure 2.1 (A). Within one domain-scale combination (gray box), changes can occur without leading to a regime shift. However, if a critical threshold is reached, domain-scale regime shifts can induce a regime shift in the whole system, which results in a new basin of attraction.

The authors characterized the current state of a system with a few main variables that were defined by simplistic thresholds. Thresholds were, for example, a certain commodity system, the level of farm debt, the size of the rural population, or the area of soil salinity. For practical reasons, the systems under investigation was defined by a manageable number of state variables. These variables determined the current location of the SES within a basin of attraction. If a threshold was reached, a regime shift followed. The authors defined a regime shift as '[...] any drastic change in the properties of a system resulting from smaller perturbations or smooth changes in independent controlling variables [...]' (Kinzig et al., 2006, pp.2). In analyzing several case studies, they found that at most five (of nine) domain-scale combinations occurred in which regime shifts took place (see figure 2.1 B). This amount of regime shifts always led to a new regime of the whole SES and was less desired than the regime before and irreversible (hysteresis effect). The authors further found out that a single domain-scale regime shift can lead to a cascading effect, resulting in a new basin of attraction. Interrelated ecological regime shifts took place at the 'patch' scale, whereas economic regime shifts happened at the medium ('farm') scale. Regime shifts at the regional scale, however, occurred in all three domains. Large-scale changes within the socio-cultural domain have been much faster than large-scale changes within the ecological system. Therefore, feedbacks from social systems onto ecological systems might appear to be time delayed. On the other hand, small-scale changes within the socio-cultural system have often been slow. In order to understand the behavior of SES, short as well as long-term perspectives of socio-ecological systems need to be taken into account (Kinzig et al., 2006).

According to these findings we focus on the critical domain-scale combinations of figure 2.1 (B). The following scales are addressed:

- the field scale by recognizing of field sizes, which determine field margins.

- the farm scale by taking into account farmers' decisions on cropping patterns, farm and field sizes, as well as the participation in environmental measurements in form of flowering strips.
- the landscape scale by considering the interaction between several farms and farm types as well as by assessing impacts on biotic diversity.

In order to capture long-term effects on biodiversity, structural change processes as slowly changing variables within the economic domain are also considered. The following domains are addressed:

- the ecological domain via the calculation of spatial biodiversity indicators.
- the economic domain by simulating farm optimizations.
- the socio-cultural domain by taking into account farm type developments and labour possibilities within the agricultural sector.

Equipped with these aspects and the basic conceptual framework of socio-ecological resilience, our study shows possible future cascading effects of policy changes leading to regime shifts before they actually happen. A regime shift within a domain and scale is defined as a change in the respective variables so that a critical threshold is reached. These thresholds are dynamic and are affected by the state of other variables below or above the considered variable (Walker et al., 2004). Due to knowledge and information gaps, science has lacked adequate models to identify and define critical thresholds until now. Arbitrary threshold assumptions are made based on available data, and focus is put on the regime shifts that follow (Kinzig et al., 2006; Cumming, 2011). In the next section, we will define our critical thresholds.

2.3. Analytical framework

The economic argument at the farm scale is a slowly changing process of structural change induced by the exploitation of economies of size (EoS). EoS are defined as an average (fixed) cost reduction per unit of production due to an increase in farm size (Duffy, 2009). Larger farms have lower per unit costs than smaller farms (Chavas and Barham, 2007). In order to increase labour productivity, technical progress is needed. Therefore, EoS are often accompanied by capital-intensive investments and a lower complexity in production steps. This also includes spatial attributes such as field structures (Chavas, 2008). EoS have successfully increased land

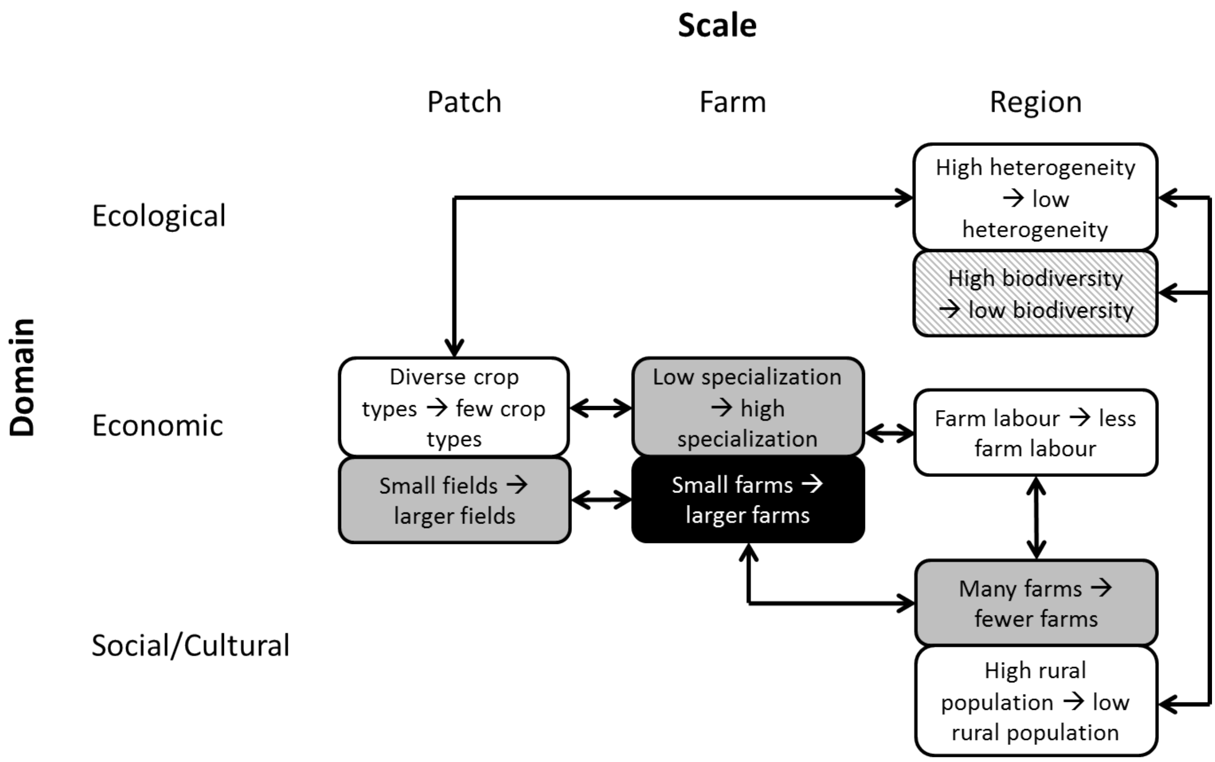

Figure 2.2.: Potential regime shifts at different domains and scales as well as interactions among them. Boxes represent potential regime shifts, while arrows show interrelations among them. The black box indicates the starting point of our analysis. Gray boxes indicate variables with defined critical thresholds related to our case study. The figure is adapted from Kinzig et al. (2006).

and labour productivity through land consolidation and farm specialization (Johnston and Mellor, 1961; Chavas, 2008). Expensive machinery is only profitable if scale enlargements are feasible (De Roest et al., 2017). This is the starting point of our analysis.

Figure 2.2 summarizes the potential regime shifts of our case study and their interactions. These are simplistically formulated potential shifts in a certain direction (e.g. from ‘high’ to ‘low’). With this figure we want to show interrelations among the observed domain-scale combinations and link them with biodiversity, which is our target value. We identified seven domain-scale combinations that may directly or indirectly affect biodiversity. For three of these combinations, we defined critical threshold levels. They are indicated as gray boxes and are subsequently defined as driving variables for which we will evaluate their trajectories.

Kinzig et al. (2006) have deducted critical thresholds from historical observations. In one of the case studies[2] analyzed, they found that a critical threshold within the socio-cultural domain was reached when the number of farming enterprises declined from 23,000 to 8,000 between the late 1960s and 2003. In parallel, from the beginning of the 1970s, consolidation and amalgamation of farms took place, which led to increasing land and water degradations (Walker and Salt, 2006, chapter 2). At around the same time in Germany, between 1975 and 1990, the indicator for biotic diversity on farmland of the national report fell from the one hundred percent mark by around 20%. It arrived at a state far below the target value from where it could not recover up to now BMBU (2015). Between 1979 and 1991, the amount of farms in Germany reduced by around 30% (Statistisches Bundesamt, 2013). On the basis of these historical values and the case study mentioned above, we define the critical threshold with respect to the amount of farms. The critical threshold is reached if one-third of the farms of our case study close down.

As noted before, farm growth is the starting point of our analysis (see the black box of figure 2.2). On the one hand, if farms grow bigger (due to EoS), field sizes increase at the 'patch' scale. The critical threshold at the 'patch' scale is reached if average field sizes increase more than double. Unfortunately, neither in general nor at the local level was information about historical changes in field sizes available. However, farm size increased by 83% between 1979 and 1991 (Statistisches Bundesamt, 2013). Brady et al. (2012) and others assumed a correlation between farm and field size. By including an additional buffer (of 17%) this value seems to be legitimated.

The critical threshold at the farm scale is linked to the degree of specialization. In Hesse, the specialization of farms is already high (Berger, 2012). If farms become more specialized while mixed farm types disappear, fewer crop types might be cultivated at the 'patch' scale. As specialization processes weaken the economic resilience of farms due to greater risk exposures (De Roest et al., 2017), each loss of farm type crosses the critical threshold at the farm scale.

Regionally, both effects at the economic 'patch' scale may lead to lower landscape heterogeneity and therefore reduce biodiversity within the ecological domain. A less heterogeneous landscape as well as low biodiversity may also lead to a reduction in rural population since the landscape is less attractive. It no longer represent cultural values (Manos et al., 2013). If farms close down due to farm size changes, at the regional scale, the amount of rural population might decrease (Knickel et al., 2017). By using EoS and simplifying farm structures with respect to their types, less farm labour might be required since fewer farms remain as potential employers. This, too, may lead to a reduction of the rural population.

[2] 'The Western Australian Wheatbelt'

With this systemic view of the three different domains and scales, we try to get deeper insights into background processes that occur within the landscape of our study site. In considering interrelated regime shifts, we will delineate possible scenarios in the case study that depict different political incentive schemes.

2.4. Methodological background

The theoretical background covers the conceptual and analytical framework of our study. It combines a systematic approach with several domains and scales and is the first strategic level of conducting a comprehensive political impact assessment. The second strategic level for the political impact assessment focuses on the temporal and spatial scale. Therein the methodological base for our case study is developed, which will be outlined in the following sections.

2.4.1. Landscape-oriented agricultural modeling approaches

Interdisciplinary modeling approaches associated with agricultural production are widely used in human-environmental research (Kissinger and William, 2010). A huge amount of applications exist and to review all of them would be far beyond the scope of our work. Reviews can be found in Rossing et al. (2007), where they focused on integrative modeling approaches, in Barthel et al. (2012) with the focus on climate change and water resources, in Kragt et al. (2016) with the main emphasis on bio-economic modeling, or in Kanter et al. (2016) who summarized all kinds of trade-off analyses within the agro-ecological context regarding their tools, parameters, and methods.

Bio-economic farm models stress economic farming decisions and the effects on ecosystems, which enable them to assess policy changes and technological developments (Jones et al., 2016). There is a vast variety with respect to the spatial scale (from the global to the local and even the field scale), the farming decision model type (e.g. linear programming or agent-based models), and the temporal scale (different types of stochastic and dynamic models) (Janssen and van Ittersum, 2007).

The following sections are dedicated to applied economic process-based agricultural modeling approaches, which put the landscape at the core of their research. In order to narrow the literature review, we focus on economic models being part of some selected interdisciplinary modeling approaches mainly conducted in Germany. We outline the differences between the chosen approaches with respect to their agricultural system modeling and particular attention is given to the spatial scale.

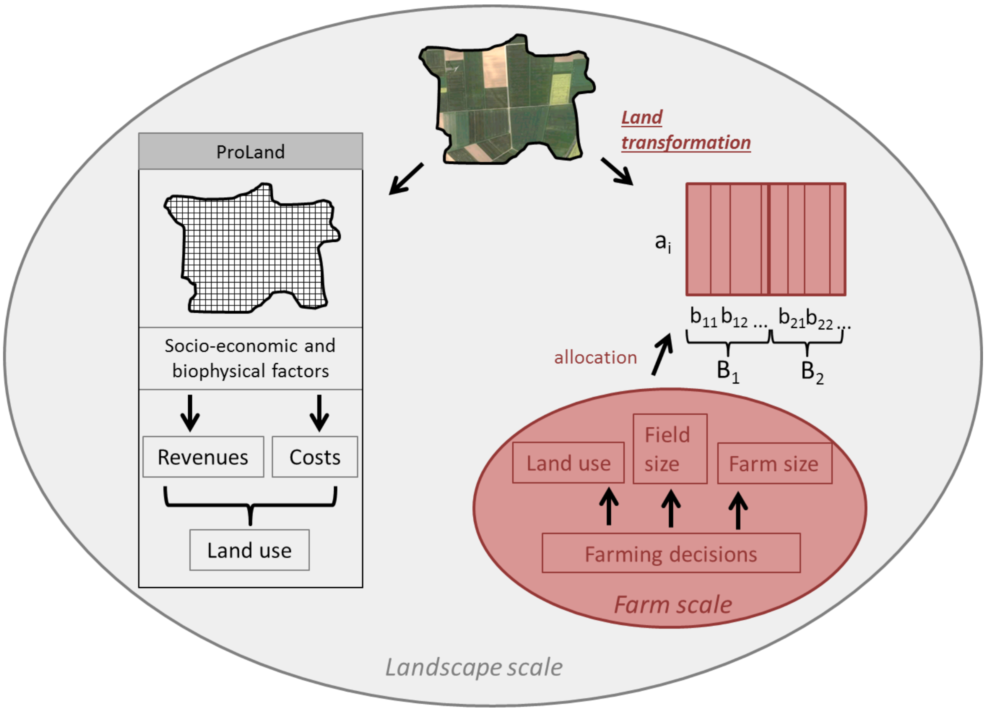

Figure 2.3.: Our methodological framework in comparison to the ProLand model.

2.4.1.1. ProLand model

As part of a project on the development of land use concepts for peripheral regions[3], the Justus-Liebig University of Giessen employed more than 30 researchers from thirteen institutes working either in the Faculty of Biology and Chemistry or the Faculty of Agricultural Sciences, Nutritional Sciences, and Environmental Management. From 1997 to 2005, the Insititute of Agricultural Business Operations, one of the latter faculties, was commissioned to develop a spatially explicit land use model in order to predict land use patterns with respect to technological and political developments: the ProLand model (Möller et al., 1998). The Aar watershed in central Germany was taken as the research area, representing a marginal region that faces land abandonment and demographic decline. Several scenarios about the effects of policy interventions or changes in socio-economic, technical, or natural conditions were applied (Weinmann et al., 2006; Schroers, 2005; Möller et al., 2000). For further calculations, the resulting land use maps were transferred to abiotic and biotic models (Weber et al., 2001; Fohrer et al., 2002). Waldhardt et al. (2010) applied the ProLand model to the Wetterau region in Hesse. They used ProLand in order to calculate normative scenarios based on the results of ecological models.

[3]Sonderforschungsbereich 299: Landnutzungskonzepte für periphere Regionen.

ProLand produces agricultural and silvicultural land use patterns based on bio-physical and socio-economic characteristics. With respect to graphical implementation, ProLand is a GIS-based[4], comparative-static, and deterministic economic simulation model that is suitable for raster or vector elements as decision units. Depending on the size of the research area, it works with 25 m by 25 m grid cells (Weinmann et al., 2006). For each grid cell, it chooses the optimal land use by maximizing the land rent. This procedure, however, causes friction since the land rent theory does not represent the actual value of the land (Czyzewski and Matuszczak, 2016). There is evidence that agricultural subsidies, for example, increase land rents and reward land owners instead of farmers (Patton et al., 2008; Ciaian and Kancs, 2012). Intrinsic land values such as recreational and visual values partly determine land rents (Wasson et al., 2013; Delbeq et al., 2014). Taking the land rent as a single decision criterion for land use does not seem appropriate. By working at the regional scale on maximizing land rent per pixel, the modeling design abstracts from the agricultural farm level. The farm level, though, is the crucial scale where decisions are made and is affected by individual resource availabilities and socio-economic factors (Kenny, 2017)[5]. In ProLand, interrelations between farms and microeconomic coordination processes within farming homesteads that drive land management are not considered. The authors argue that this is necessary for stressing spatial aspects and for improving interfaces with natural science models (Schroers, 2005). However, we challenge this argument and have developed a model that integrates economical and spatial land use decisions by taking into account the field, farm, and regional scale.

Figure 2.3 shows a comparison between our framework and the methodological framework of ProLand. In ProLand, simulations take place at the field level, where revenue and cost calculations on the base of socio-economic and biophysical factors lead to a certain land use type that maximizes the land rent within each grid cell (Weinmann, 2002; Schroers, 2005). In Fohrer et al. (2002), different field sizes were taken as input parameters for modeling production costs with the ProLand model. Farming decisions on field sizes were not simulated, though. On the contrary, our approach transfers spatial landscape characteristics into spatial decision units[6] and operates at the field, farm, and landscape scale. The spatial decision units have certain land uses (cropping patterns), field sizes defined by $a_i \times b_i$), and farm sizes ($a_i \times B_j$). a_i of Fig. 2.3 defines the length of a field and b_i the field width. Based on mathematical optimization routines, we use an iterative process to calculate field and farm sizes separately but not independently from each other. First of all, the model calculates the optimal land use and field sizes under the given (predefined)

[4]Geographical information system
[5]Farm models will be described in the next sections.
[6]For all calculations, ArcGIS from ESRI group is used.

farm sizes. After that, our model optimizes the farm sizes under the given (preprocessed) land use decisions. How this will be conducted and which model outputs will be transferred to the next iteration will be shown in section 2.4.2 after a detailed description of the farm scale. Therefore, it is necessary to look into the literature on farm models, based on which we will develop our model.

2.4.1.2. Farm models

Farm models can be used in interdisciplinary modeling approaches that investigate human-environmental interrelations in taking into account farmers' decisions (Jones et al., 2016). In order to simulate farmers' decision making at the farm level, mathematical programming models are often used (Wossink et al., 1992). Simple linear programming models maximize a linear function by calculating the optimal combination of a set of variables subject to linear inequality and equality resource constraints (Dantzig, 1963).

According to Hazell and Norton (1986, pp. 10-11) a simple linear programming farm model that maximizes the total gross margin Z can be written as follows:

$$Z = \sum_{j=1}^{n} c_j x_j = max! \tag{2.1}$$

such that

$$b_i \geq \sum_{j=1}^{n} a_{ij} x_j \tag{2.2}$$

and

$$x_j \geq 0, \tag{2.3}$$

where

- x_j is the level of land use activity for n numbers of activities ($j = 1, 2, \ldots, n$).
- c_j is the gross margin of one unit of the jth activity.
- a_{ij} is the demand of the ith resource required to produce one unit of the jth activity.
- b_i is the resource endowment for m numbers of resources ($i = 1, 2, \ldots, m$).

Depending on the model complexity, several variables with more than one index can be optimized in order to maximize the target value. The elements of i and j cover the components of any optimization problem. In GAMS[7], the modeling software we used, they are summarized as 'sets'

[7]General Algebraic Modeling Software

(Rosenthal, 2017).

One example of a spatially simple representation of land use is the 'Landschaftsmodell Kraichgau'. It describes the effects of land management on the profitability of farms, soil erosion, nitrate depletion, and habitat quality according to different political scenarios. It was developed by Dabbert et al. (1999). The approach includes socio-economic as well as biotic and abiotic aspects, which were modeled by corresponding disciplines. As ProLand, it operated with a geographical information system (GIS), which served as interface between the interdisciplinary modules of the model complex. In order to unify data exchange, the spatial reference units of all models were 50 m by 50 m grid cells. The modeling concept was strongly influenced by landscape planning views, emphasizing the visualization of results and scenarios at the spatial scale (Herrmann, 2000). The economic module as part of the whole model complex was based on a comparative static linear programming approach. It aggregated farms of several municipalities and calculated the optimal land use decisions of the grouped (average) farmers. The land use results were allocated based on biophysical or stochastic models (Dabbert et al., 1999). Aurbacher and Dabbert (2011) enhanced the allocation of (economic) land use results with a Markov chain approach in combination with a minimum cross entropy approach. In order to cover soil erosion variability, conventional farming practices were extended by more environmentally friendly soil treatment practices, and nitrate balances were computed based on land use results. As it is conventional in such optimization models, maximization of total income of all aggregated farms was taken as an objective function. Large areas were homogenized, which led to a high spatial imprecision regarding farm management decisions and practices. Landscape and field structures were not considered at all. However, within the same research area but for two smaller case studies, Aurbacher (2010) developed a linear optimization model at the farm scale that took into account the field scale in order to calculate production costs of different field shapes.

Kenny (2017, pp. 5) noted, that '[...] some element of spatially explicit mapping is especially important for any farm model seeking to integrate considerations of natural capital.' Since spatial scales between socio-economic decisions and the provision of environmental services and biodiversity do not match in these kinds of modeling approaches (Mottet et al., 2005), environmental effects such as carbon sequestration or groundwater strains were investigated (see for example Barthel et al. (2012)). These ES do not occur within the same location where the actual agricultural management takes place.

Several attempts to overcome these spatial mismatches were made with continuously improving results. One example is a spatial application of a linear programming model that was developed by Kächele (1999). He showed that it is possible to connect economic land use results with ecological models on a very accurate spatial scale if enough data is available. On behalf of a project, the author developed a decision support system for stakeholders within the context of severe conflicts between land users and conservationists. The research area was the German national park 'Lower Odra Valley', where, along with total reservation areas without any human interventions, grassland production mainly takes place. The study included 23 farms that cultivated 334 agricultural fields. A survey of all farms was conducted in order to gather information about the location of each farmer's fields and other production factors such as holding capacities, capital, and contracts. The developed MODAM model (multiple objective decision support tool for agroecosystem management) simulated the optimal amount and allocation of nature preservation areas by considering their spatial allocation. The author applied multiple goal programming, and economic income maximization was constrained by ecological preservation goals. The ecological goals were a certain amount of land and connected area for conservation purposes. However, the land belonged to farmers. In order to find the best solution, income losses of farmers were minimized through spatial considerations of conservation land that farmers were forced to enact. Each field was represented by a polygon of the underlying GIS-database, and land use scenarios were visualized (see also Kächele and Dabbert (2002)).

The background of the MODAM model was very specific. For example, typical agri-environmental schemes (AES) such as political conservation incentives were not implemented. Schönhart et al. (2011) developed a spatially explicit farm model including AES and applied it to a small (550 ha) research area in Austria. They used linear programming techniques in order to assess the cost effectiveness of AES within their case study (in total 430 fields and 20 farms). For each farm, land use activities were optimized with a mixed integer linear programming model chosen from a pre-processed selection of crop rotations[8] that caused certain effects on soil and climate. The authors focused on landscape development and appearance, which is why fields were regarded as landscape elements in their actual shape and location. By accounting for different land use types such as crops, orchard meadows, and grasslands, and different intensification levels of these land use types, they applied a Shannon-Wiener index in order to measure landscape heterogeneity. AES were limited to management practices that reduce nitrogen loads, soil loss, and soil organic carbon emissions to protect water and soil, and mitigate climate change. Happe et al. (2004) applied another method. They used agent-based models to allocate land use results. In doing so, they added information of land ownership and transportation costs in order to distribute land use results

[8]These crop rotations were based on historical observations.

to pixels. The different approaches can be divided into top-down and bottom-up approaches. While the latter approach requires high levels of information and data acquisition, the former ones are often used to allocate land use results to larger areas. Only a few studies considered field size changes, although farmed landscapes have experienced severe changes due to increasing agricultural production units (Hötker and Leuschner, 2014).

The examples above showed that it is possible to conduct spatially explicit political impact assessments based on farm models. However, these farm models did not consider interrelations between farmers, and land use activities were the only driver of landscape changes. Our approach considers structural change processes, and changes in farm and field sizes represent additional aspects of landscape developments.

2.4.1.3. Agent-based models

Matthews and Selman (2006) suggest using agent-based models (ABMs) in order to to understand underlying dynamic processes of socio-ecological systems. In ABMs the landscape can be seen as a system that evolves out of the interaction of its users and components. It has no equilibrium state that the system adapts or tends to (Kay et al., 1999). Furthermore, by linking the dynamics of farm structures, ABMs cover important parts of structural change and landscape processes (Schouten et al., 2013). Their inherit conceptual framework is based on spatial farm interconnections (Verburg and Veldkamp, 2005). Parker et al. (2003) gave an overview of agent-based land use models, and An (2012) summarized ABMs linked with coupled human and natural systems.

ABMs enable users to depict farms as heterogeneous entities with respect to different variables such as capital, asset structure, contracts, management capacities, individual adoption costs, or even to bounded rationality issues (Ligtenberg et al., 2001). Interrelations between farmers are explicitly taken into account and directly influence the farmers' behaviors (Verburg and Veldkamp, 2005). The advantages of these bottom-up farm behavior models are that - in theory - many current and complex conditions regardless of the region can be integrated (An, 2012). When a large number of individual farmers are chosen, a huge amount of data is necessary to run the simulations and utilize these potentials (Murray-Rust et al., 2014). Another strength of applying ABMs in agricultural economics is to show how differently heterogeneous farmers behave based on similar external changes such as prices or political incentives. Therefore, the dynamics of structural change can be delineated more adequately (Uthes et al., 2011).

An important cornerstone for ABM applications in landscape-related agro-ecological science was laid by Balmann and his working group at the Leibniz Institute of Agricultural Development. Balmann (1997) investigated the dynamics and complexity of structural change process within the agricultural sector by working on path dependencies. In Balmann (1999), he showed how sunk costs affect structural change and the implementation of economies of size in regions where small family farms dominated large scale farms. These first modeling efforts served as a basis for the development of the AgriPoliS model, which is used to show structural change processes. It takes into account farm size changes, farm exits, and land use changes (Happe et al., 2004). In AgriPoliS, farmers (agents) decide independently from each other on their production choices, investments, and land rentals. They are connected through product and factor markets. Farming decisions are modeled using linear programming tools such as multi-integer programming with the objective of maximizing household income. AgriPoliS models the land rent market with auction mechanisms through which it ties together all agents[9].

Regarding space, AgriPoliS works with a stylistic landscape of grid cells with equally sized plots, which can have different possible land use types (e.g. grassland, arable land, forest, or roads). In addition to the land use structure, ownership and land quality attributes can be included. Transportation costs are determined by the distance between the location of the farm and the managed plot. In Happe et al. (2004), the farmsteads were randomly distributed, and transportation costs were calculated with an Euclidean distance function.

AgriPoliS was applied to several (developed) regions, and research questions related to policy switches affecting farms in different manners creating varying adaptation strategies among farmers (Balmann et al., 2001; Happe et al., 2006; Schnicke et al., 2007; Happe et al., 2008; Uthes et al., 2011). Therein the focus was the complexity of agents (farmers) and their long-run production adjustments. Brady et al. (2012) extended the AgriPoliS model in order to conduct a spatially explicit nature conservation study. They focused on how policy affects farmers' decisions on land use differently depending on farm types and spatial heterogeneity. Furthermore, they elaborated different effects on biodiversity and ecosystem services. The research area comprised two low-intensive regions in Sweden. In the study, the field level was addressed through plots as basic elements of an artificial landscape grid. The plots were kept constant during the simulations and only ownership changed. Several plots of the same type (e.g. arable land) formed a land block, which was later fixed and addressed in order to allocate cropping results. An algorithm minimized costs by allocating each farmer's crop production to fields, i.e. for minimizing production costs. Therefore, the initially created farm blocks formed the spatial field size constraint of the respective farmers. After the standard procedure of AgriPoliS, farm sizes changed subject

[9] For more information about land markets in ABMs, see Kellermann et al. (2008).

to agents' decisions, which led to field size changes. The authors assumed a positive correlation between farm and field sizes. Together with a Shannon-Wiener index measuring the diversity of land use types, field sizes formed a landscape indicator. The model was applied to three different political scenarios: coupled production (Agenda 2000), decoupled production (CAP reform 2005), and a reduction of DP by 30%. Results showed significant impacts on the landscape mosaic, biodiversity, and ecosystem services that differ between the regions. In a similar application, Happe et al. (2011) investigated abiotic environmental effects. Both studies showed the importance of considering complex interactions between farmers and the dynamics of structural change in order to promote biodiversity and ES. Nonetheless, linear landscape elements as part of agricultural fields were not considered. However, due to the use of grid cells, important linear landscape elements such as field margins or strips along field edges are missing.

Linear landscape elements can serve as niches for certain species that contribute to the delivery of ecosystem services within agricultural landscapes (Zhang et al., 2007; Davies and Pullin, 2007; Hinsley and Bellamy, 2000). Therefore, spatial maps including such landscape elements are required for biodiversity assessments at the regional scale (Van der Zanden et al., 2013; Meyer et al., 2012). A lot of literature about the impact of the intensive margin of agricultural production on biodiversity and ES are present but only few studies about the 'extensive' margin exist (Rizzo et al., 2013). Extensive margins of agricultural lands are supposed to comprise natural elements and field sizes (Antle and McGuckin, 1993). Linear artificial landscape elements such as flowering strips are often the only landscape structures in highly intensified agricultural areas (Bengtsson et al., 2003). Although they provide habitat for insects and facilitate pollination (Geert et al., 2010) we rarely see them. In order to account for potential insect and pollinator habitats, our model includes the cultivation of flowering strips, implemented AES. According to HALM guidelines[10], flowering strips need to have at least a width of 5 m, a minimum area of 0.1 ha, and a maximum area of 1 ha. Modeling such field characteristics requires a greater spatial awareness of the field scale. In the following section, examples of optimization models that take into account spatial field and landscape characteristics are presented.

2.4.1.4. Field and landscape pattern optimization models

Following suggestions of Hanf (1994), models with the focus on landscape patterns should transform spatial units into decision units. Non-dimensional economic models are not sufficient

[10] Hessian program for agri-environmental and landscape measures ('Hessisches Programm für Agrarumwelt- und Landschaftspflegemaßnahmen'

for modeling landscape interaction.

From this perspective, Wossink et al. (1998) searched for optimal wildlife conservation at lowest costs. Their modeling approach included decision making at the farm level and the spatial extent and location of nature conservation measures within fields (sprayed vs. unsprayed field margins). They followed a landscape-centered spatial approach. The ecological benefits of different disposals of unsprayed field margins were calculated and compared to the costs. Through the assessment of an ecological network, a wildlife cost function at the landscape level was delineated. The authors used a simplified and stylized landscape digitized with GIS, and for the baseline scenario they randomly distributed crops and field margins. Normative scenarios were developed in order to ensure species dispersion. By taking into account the economic preferences of farmers, land use patterns for minimal costs were calculated. They found that selective control of field margins through farmers' preferences leads to win-win situations.

A French research team further developed a model appreciating this basic approach. They investigated agricultural landscape changes and the impacts on biodiversity. With their approach, both landscape appearance (e.g. landscape mosaic and networks) and dynamics on farm structure (induced by policies) were addressed. The configuration and composition of certain landscape elements (depending on the habitat of focal species) was used to assess population dynamics (Gaucherel et al., 2010). The farm-based mathematical optimization model was linked to three spatial scales: the field, farm, and landscape scale (Havlík et al., 2008).

The modeling approach was applied to intensively used farmland in France, where the abundance of the Little Bustard (*Tetrax tetrax*) has decreased. The model first calculated the optimal reserve for the Little Bustard and used the simulated landscape appearance as a normative scenario in order to test how it can be achieved with lowest costs. Moreover, the authors developed positive scenarios with political incentives for farmers and compared the outcomes in terms of conservation achievement and governmental payments. The underlying economic model was a mixed integer linear programming model, which optimized the sum of all farms' gross margins within a stylized landscape. Abiotic factors of each field (e.g. soil or slope) influenced farmers' land use decisions. Each farm had a similar size with similar amounts of fields lying adjacent to each other. The authors divided the farms into two different farm types: crop and mixed dairy farms. The decision variables consisted of crop rotations. The political framework referred to the CAP reform of 2003 with 10% obligatory set-aside area and the voluntary participation of farmers in AES (permanent and temporary grassland and alfalfa production). The AES are possible habitat elements for the Little Bustard. At the regional scale, the spatial extent of land use and habitat measures determined the habitat quality and the conservation success of the Little Bustard. The spatial analysis was conducted using Ripley K and L functions, which count for a combination of densities and distances of habitat patches. In another work, they focused on optimal policy strategies in the form of contracts (Bamière et al., 2011) and auction schemes (Bamière et al., 2013) in order to reach desired landscape structures.

In another alternative approach, Cong et al. (2016) investigated spatial aspects of farming with respect to the provision of ecosystem services (ES). They assumed that farmers conserve certain habitats on their fields purposely for the need of crop pollination, which increased the yield up to a certain threshold. Habitat configuration and composition served as indicators for crop pollination. Individual farmers were represented within a landscape of grid cells where each farmer had the same size of land and amount of fields. Farmers had perfectly consolidated fields and the same production conditions. The field was the smallest decision unit and was represented by a grid cell. In order to model crop pollination, the optimization model included a feedback loop that linked yields with pollination. The higher the distance between the field and the habitat was, the less 'yield' was positively influenced. Since the parameters for yield effects originating from the presence of pollinators are difficult to measure from an ecological point of view, yield efficiency was changed in a sensitivity analysis. The authors calculated the optimal allocation of habitat measures[11] for individual farmers at the farm scale and compared the results to those at the landscape scale, where they optimized the total profit of all farms.

Results showed that landscape management enhances the efficiency of ES management. Habitat dispersion was much higher in the landscape management scenario than in the farm scale management scenario (individual farm optimization). At the farm scale, farmers aggregated habitat measures in the very middle of their farmland so that they profit the most from ES provision. If modeled at the landscape scale, habitat measures became more dispersed over the whole landscape. Note that the quality of the ES provider (scale and mobility) effected the habitat configuration.

With their analysis, Cong et al. (2016) pointed out that synergies between habitat conservation and agricultural production are too weakly considered in policy and landscape management. Having the intention to promote biodiversity protection is not enough. The search for win-win situations should be examined. The authors showed that habitat dispersion supports the provision of some intermediate ecosystem services such as pollination or pest control but effects of other ES were limited. Up to now, interactions between farmers as well as aspects of structural change were not considered in the models. Practical applications are scarce and not yet well adapted to real landscape parameters.

[11] Farmers had an obligation to dedicate 5 % of their land to 'habitat areas'.

2.4.2. Applied modeling approach

In the following sections, our modeling approach is described in more detail. It is a synthesis of several modeling approaches described above.

2.4.2.1. FOLAS model: a synthesis

The literature review of human-environmental modeling approaches revealed important system components that need to be captured. Commonalities in the models described above (except for the ProLand model) are the simulation of farmers' land use decisions; they address the farm scale. Our approach is also based on farm optimization in mode of mathematical programming models. By implication, promoting biodiversity on agriculturally used land needs to consider farmers as decision-making units and spatially explicit biological processes in order to delineate a nature production function at the landscape scale (Nuppenau and Helmer, 2007). In order to account for farmers' spatial decisions, optimization routines must be applied in an interactive way. This investigation will show how it can be done.

The modeling approach is landscape oriented and particularly takes into account the spatial distribution of farmers' decisions. In order to account for variations in the behavior of individual farmers as decision units, it includes some agent-based modeling aspects since it divides between farmers. Farms are treated differently with respect to their production capacities (farm sizes, financial capital, and livestock husbandry capacity) and production modes. Three different farm types are modeled:

- arable farms.
- pig-fattening combined with arable farming.
- dairy production combined with arable farming.

Field sizes play a central role in landscape complexity since they effect the landscape mosaic, patterns, and processes (Antle and McGuckin, 1993). Depending on the species involved, the field scale is crucial for biodiversity conservation (Gaucherel et al., 2010). Therefore, a closer look at field sizes is also relevant. Farmers apparently decide on sizes and fields. These kind of decisions are much slower than choosing appropriate cropping patterns for the next year. Our approach covers long-term decisions by considering structural change. Most field and landscape pattern optimization models do not yet explicitly consider structural change processes (e.g. Havlík et al. (2008); Cong et al. (2016)), since this requires modeling farm interrelations. Our approach captures farm interrelation through the exchange of land (consolidation) and land leases between neighbouring farmers. Therefore, farm size enlargements and farm closings can be modeled with our approach. As a limitation, interactions between different social groups or institutions and social networks are not captured with our model, albeit they play an important role in socio-ecological systems (Adger, 2000). In order to spatially reflect structural change, our model stylizes the actual landscape through landscape simplification measures (see fig. 2.3). The simplified landscape and the modeling results are visualized via GIS. In doing so, our model follows a certain execution logic, which is outlined in the next subsection in more detail.

In view of the political impact assessment, new political measurements can be tested with respect to their capability to provide a habitat for rich biodiversity. Habitat loss and isolation mainly threaten farmland biodiversity (Benton et al., 2003; Watling and Donnelly, 2006). Although they are hardly represented in agricultural areas (Bengtsson et al., 2003), semi-natural habitats promote species abundance (Hendrickx et al., 2007) and are of great importance in order to maintain ES such as pollination (Kremen, 2005; Klein et al., 2007). The dispersal distance between habitat patches depends on the land use matrix (Roland et al., 2000). For bees and smaller insects, the travel radius is much smaller than for birds, for example. According to Jauker et al. (2009), a radius of around 250 m can be used as buffer zone for the abundance of bees and hoverflies within agricultural lands. For our study, we chose three different spatial biodiversity indicators and applied them accordingly. The biodiversity indicators encompass (1) the total area of semi-natural habitats defined as grassland and flowering strips within a buffer zone, (2) the Simpson's diversity index in order to account for cropping diversity, and (3) the number of patches/fields within a buffer zone targeting the landscape structure.

With these aspects, the field, farm, and landscape scales are taken into account. In addressing the three domains mentioned in section 2.2, potential regime shifts are delineated based on the analytical framework (see figure 2.2). This allows us to develop a comprehensive picture of the case study and deduct possible futures. Our applied modeling approach captures the following:

- the farm scale using optimization models.
- spatial aspects of landscape pattern models.
- farm interrelations.
- structural change processes.
- far-reaching spatial decisions on field sizes.
- farm sizes.

- flowering strips as an example of linear semi-natural landscape elements.
- spatially explicit biodiversity indicators.

In the following sections, our model is shortened to 'FOLAS' (**F**arm **O**ptimization at **LA**ndscape **S**cale).

2.4.2.2. Spatial and temporal model assumptions

In FOLAS, short and long-term decisions are modeled. Short term decisions refer to cropping decisions each farmer makes at the beginning of a cropping year. A core issue for our modeling approach is to consider structural change within the agricultural sector. This determines the time scale of our model. For the present study a comparative static approach is chosen, where several discrete points in time are compared to each other. FOLAS captures a long time period in which a farm might close down, reduce, or expand its production in terms of farm size. With respect to long-term decisions, investments play a key role in mathematical programming approaches. However, in this work, investment activities are not modeled since this would outgrow the temporal and financial scale of our study. Based on the new farm sizes, new field size decisions are made. Changes in field sizes are modeled by explicitly taking into account EoS as a driver of structural change. Structural change is typified by farm and field sizes, cropping patterns, and the variety of farm types. For modeling EoS, we made some assumptions regarding the equipment of farmers' machine power. Within the time span of a model run, each farmer was given the same engine power of 120 KW. Therefore, the observed mechanical equipment of farmers served as productivity boundaries. The level of machine power determined the labour requirement per field size. It showed how strong existing machinery power can be exploited by increasing field sizes. In fig 2.4 the average labour requirements (depending on the field size) with engine power of 120 KW is depicted. With this machine power set as baseline, EoS can mainly be exploited between farm sizes of 1 - 10 ha.

The FOLAS model run consists of four model iterations and use information from the previous iterations. Each one is linked to the next one via the transfer of output data, which leads to a stepwise solution. The four iterations contain short and long-term decisions of farmers and underlie certain spatial and temporal assumptions, which are delineated in the following paragraphs.

- **1st Iteration**: land use decisions are modeled assuming a dispersed location of all farmers' fields. Therefore statistical information about farm size and types, as well as field sizes and

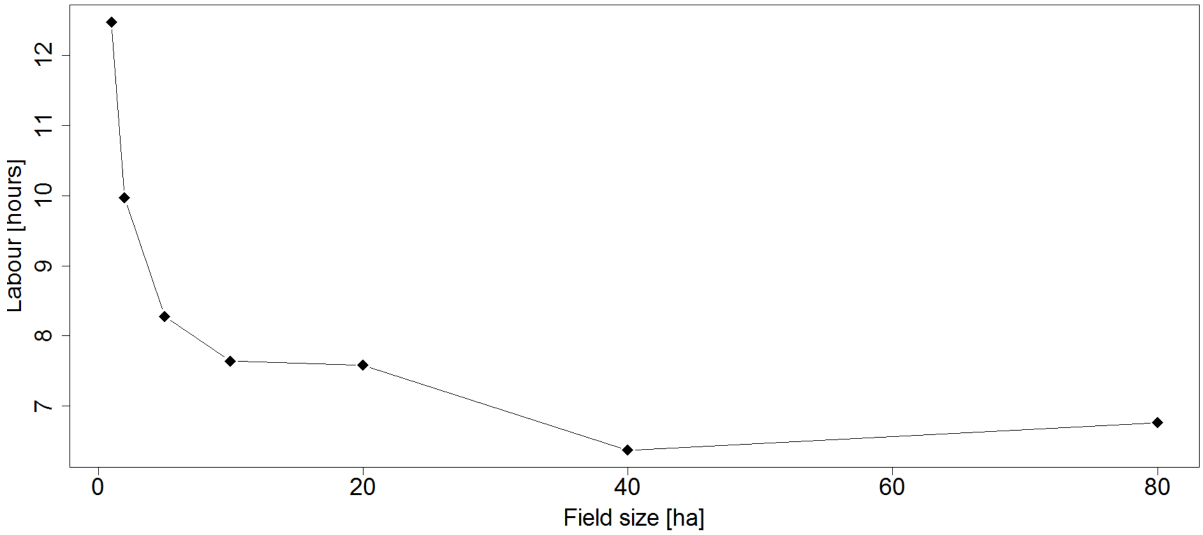

Figure 2.4.: Average labour requirements of all crops in the model depending on field size. Data refer to an engine power of 120 KW (received from Ktbl on request).

livestock keeping of the case study for the calibration year (2011), were used as orientation. It was sufficient to set up the basic model parameter structure. FOLAS most efficiently allocates the total amount of grassland (due to statistical information) to the farms by maximizing the total gross margin at the municipality level. This is done since we assume that farm structures with their grassland proportions evolved due to path dependency in an optimal manner. Results on grassland proportions per farm are transferred to the next iteration.

- **2nd Iteration**: In order to use economies of size and economics of transport, farmers require fields located next to each other so that field enlargements are possible. According to these efforts we assume that farmers exchange their fields until they reach full consolidation. This might take several years. However, the grassland proportions from the 1st iteration stay the same for each farmer; albeit the location of the grassland can change. Cropping results, information about the participation in AES, and livestock keeping are passed on to the next iteration.

- **3rd Iteration**: After consolidation, farmers are capable of renting out lands to their neighbouring farms in order to further exploit EoS by increasing farm and field sizes. FOLAS optimizes the amount and size of the farms with respect to labour savings by creating bigger fields. Therefore, land use proportions of the former iteration are fixed, and EoS are modeled explicitly. In order to model EoS for each crop type, a proximity function of labour per field size based on Ktbl data was calculated. As suggested by Nuppenau and Helmer (2007), approximations in order to receive computable were applied. The resulting functions per arable crop were included in the model to calculate labour requirements for each crop and field. Detailed mathematical model formulations follow in section 4.2.3.1.

- **4th Iteration**: Farming activities including crops, livestock keeping, and AES are modeled again to check how farmers adapt to the new land capacities calculated in the previous model step. This model run is necessary since it checks whether farmers use their new lands or rent them out again. In this case the former iteration would not lead to an optimal farm size.

FOLAS maximizes the total gross margin at the municipality level and assumes Kaldor-Hicks compensations. This enables the model to distribute certain cropping patterns such as grassland or sugar beets to single farms in an optimal manner. Since single farm data is often missing and restrictions at the farm scale make the model less flexible at the landscape level, a strict landscape scope is pursued.

When land exchange mechanisms are included in the modeling process (2nd iteration) and farm interactions take place (3rd iteration), maximization at the municipality level implies that those farmers who are able to increase the total gross margin of all farms (the most), will survive. From a short-term perspective, this would mean that farmers give up farming only because they do not contribute to the total gross margin of the farming community, which is rather unrealistic. But considering a much longer time span (with the current hassles of farmers facing global markets with increasing competitive pressures and the lack of farm successors) this assumption has a more solid basis. In order to show long-term labour flows between agricultural and other sectors as well as due to important socio-cultural regime shifts represented by population developments, off-farm work is modeled from a macroeconomic perspective[12]. It applies to each iteration.

2.4.2.3. Simplification of spatial outlay and the modeling approach

In our study, farmers are the decision-making units. We translate farming decisions into spatial units, though. The landscape serves as a spatial interface between the economic and ecological domain (Bockstael et al., 1995), not only with respect to the land use activities of farmers, but also with respect to overarching farming decisions of field and farm sizes (Nuppenau and Helmer,

[12]Off-farm labour is not modeled in the sense of household economics since the contribution of off-farm work to total farm utility is not the focus of the study.

2007). Besides cropping and livestock patterns, they also decide on field and farm sizes. As already indicated in figure 2.3, the landscape is stylized as coherent rectangles. These have a certain length and width and represent the fields and farms, respectively. Figure 2.5 gives an overview of the consecutive model iterations and their graphical representations. The starting point is an image of a real landscape at the left edge of the figure. In preparation for land transformation into rectangles, the fields within the study area were aggregated to bigger spatial units. Each spatial unit represents one agricultural farm (farm B_1 and B_2 in figure 2.5).

This procedure is based on general trends within mainstream agriculture, where full consolidation by exchanging land is seen as a golden standard for crop production[13]. In doing so, spatial conditions such as roads, streams, and housing areas were taken into account, insofar as they do not belong to the agricultural area and aggregation over these spatial units is unfavorable. Due to this procedure, the farm size depends on spatial aggregation possibilities, and each farm differs with respect to its size. Within a farm, aggregation over fields is assumed to be possible and also sought by the farmer due to utilization of EoS. In the next step, farm types were allocated randomly to the farm outlet. Since from public statistics no information about land ownership or farmstead locations was available, it had to be done this way[14]. Based on the spatial farm consolidation, achieved farm sizes were taken as the baseline for the modeling procedure in the 1st iteration (see B_j^* of figure 2.5, where j is the number of farms ($j = 1, 2, \dots, n$)).

In order to model land rentals, artificially aggregated farm units were simplified to rectangles. These rectangles have the same size as the previous aggregated spatial farm units. In the 3rd iteration, neighbouring farms enlarge their cropping area and use EoS since the neighbouring fields have shapes similar to their own fields. These farms are assumed to be willing to rent land from the other farm and are modeled as described in section 4.2.3.4. In order to display farm size changes, the rectangles of farms are located next to each other and have the same edge, which is defined as the field length a_i of the respective farms. The farms are assumed to arrange all of their fields next to each other and therefore only one field length per farm exists. In this way, farms are stringed together, and farm size changes can occur through parallel shifts of the farm edge of two neighbouring farms (change in B_j). The field length stays the same while the sum of all field widths (b_i) may change for one farm after the 3rd iteration. Based on the new farm sizes, new field sizes and farming activities are calculated within the last iteration. The resulting map of the aggregation procedure as well as the stylized rectangles of the case study can be found in Appendix A.

[13] Research into the development of software enabling agricultural floor exchange exists and gets even remunerated (Agrarzeitung, 2013).

[14] In projects this might be different.

The first iteration is calibrated to the current agricultural landscape. It assumes scattered fields that are allocated randomly throughout the farmland. This is simulated by restricting the upper limit of the field size variable to an observed average of our case study. Since land amalgamation is not yet realized, EoS through field merging cannot be exploited, and fields are smaller. The output of the farm optimization model for each field b_j comprises the cultivated crop ('crop'), field size ('fdsize'), and flowering strip ('fs'). These variables refer to the field scale. For example, the first variable indicates which crop is cultivated on that specific field. $B_{j,'gl'}$ is the grassland proportion of farm B_j. It is the central variable of the first iteration and will be fixed for all following iterations. The underlying assumption is that regardless of field consolidation processes, the proportion of grassland per farm should not be subject to change. However, the location of grassland after consolidation (the 2nd iteration) may change. In the 2nd iteration, consolidation of farmland is assumed. Farmers are able to exploit EoS and fields get bigger. Based on the grassland proportions of the previous model run and without restrictions on the field size variable, new model output is produced: $b_{j,'crop'}$, $b_{j,'fdsize'}$, $b_{j,'fs'}$ and $b_{j,'gl'}$. The latter variable for grassland ('gl') now refers to the field and indicates whether it is managed as grassland or not. Within the 3rd iteration, the farm size B_j is simulated. Therefore, the model output of the 2nd iteration is taken as input. The new farm sizes (together with the grassland proportions of the farms $B_{j,'gl'}$) serve as the basis for the newly simulated field scale variables in iteration four ($b_{j,'crop'}$, $b_{j,'fdsize'}$, $b_{j,'fs'}$ and $b_{j,'gl'}$). Results show the land use patterns after structural change processes.

During the iterations, the model provides information on farm and field size changes as well as on land use and flowering strips for each farm. Results are collected and semi-automatically transferred to GIS. The field size results need to be translated into field widths with a respective coordinate for GIS presentation. Basically two python scripts were developed: one to generate rectangles representing farms and one to generate fields and their cropping patterns. In order to generate the rectangles, information about length and width of each farm was used with coordinates for the central points. The fields were generated by first calculating the coordinates for the points on both sides of each farm width, and second we created polygons out of those points as well as concurrently transferred the correct cropping patterns. The results for the 1st, 2nd, and 4th iterations were mapped. They contain farm and field sizes as well as land use information on crops and flowering strips.

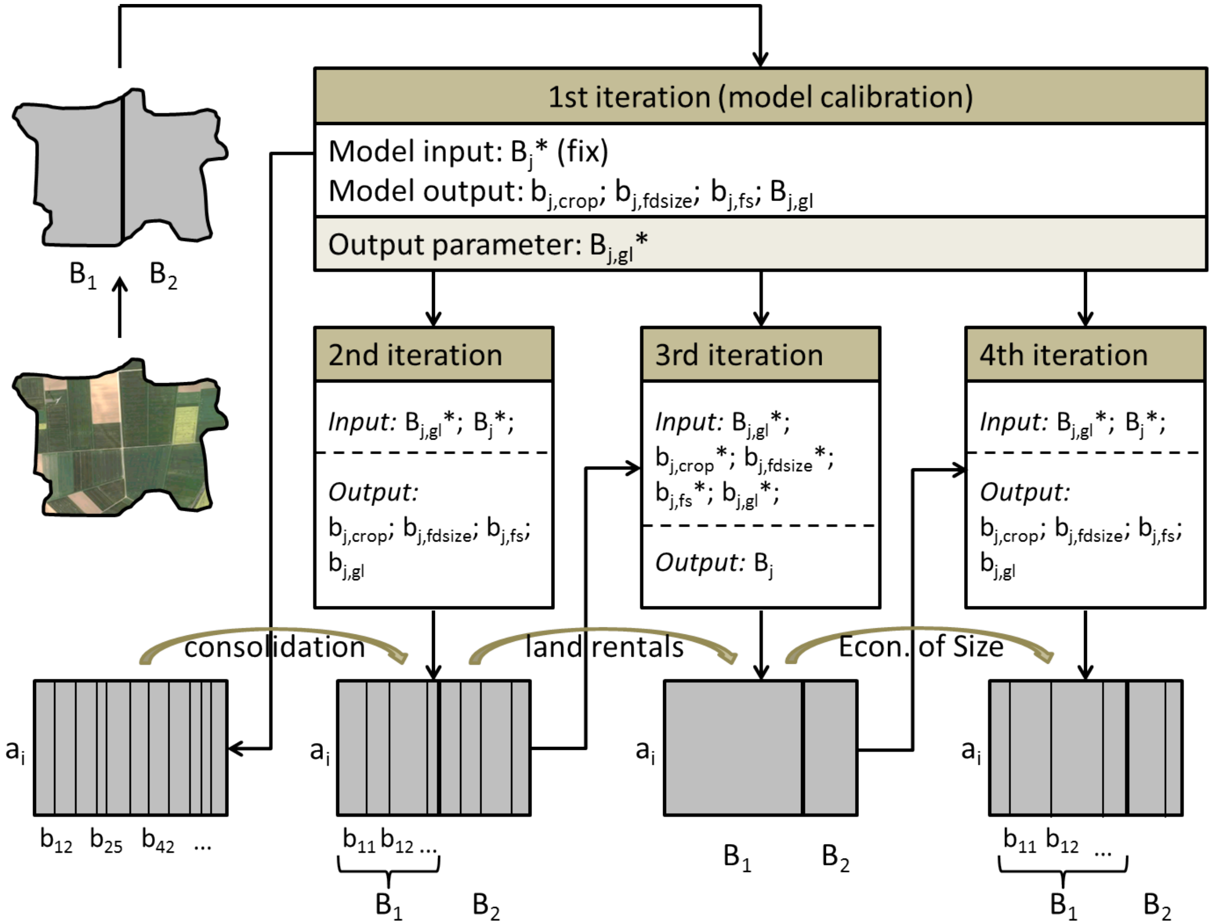

Figure 2.5.: Overview of the consecutive model iterations and graphical representation of the model results. Variables that are fixed for the next iteration serve as input parameters. They are marked with an *. See text for variable definitions.

2.5. CAP 2020 in short

In farming decisions, political incentives play a crucial role (Ewert et al., 2005, 2011; Mandryk et al., 2012). Since our model is calibrated to the year 2011, the past CAP reform (decoupled payments and agri-environmental schemes) was set as the baseline. As we intended to test the achievements of the 'CAP towards 2020' reform with respect to the biodiversity targets of the EC, policy interventions were introduced by applying parts of the new CAP reform. Additionally, due to critical responses to the new measurements (Lakner and Bosse, 2016, e.g.), an alternative political incentive scheme was tested. It covers a fully restructured governmental payment scheme. The different policies applied in the model will be described in section 4.2 before the results of the policy choices are presented. In the following, the new CAP reform is described roughly. It gives an overview of the main characteristics of the current incentive structures of the European Commission. Detailed information can be found in the utilized references.

In 2013, the Council of the EC agreed upon the new CAP Reform 2014-2020. Therein, they seek to support a 'sustainable' agricultural production with respect to economic (food security, globalization, price volatilities, etc.), ecological (e.g. biodiversity, soil and water quality, and climate change), and territorial (e.g. demographic changes and urban migration) aspects (European Commission, 2013). Its financial framework in order to fulfill theses promises is around 400 billion EUR for 2014-2020, paid by European citizens. This budget is separated into direct payments (DP) of the first pillar and rural development schemes, which form the second pillar. Around one quarter of the money is dedicated to the second pillar and also contains the budget for agri-environmental schemes (AES).

One of the main novelties of the new CAP 2014-2020 reform is the so-called mandatory greening component of the first pillar. 30% of the DP are coupled with environmentally friendly measures covering the following:

- a crop diversification strategy.
- the preservation of permanent grassland.
- 5% ecological focus areas (EFAs).

The crop diversification strategy provides that farms bigger than 10 ha need to grow at least two different crop types. Farms larger than 30 ha need to grow at least three crop types, provided that not more than 95% of the arable land is covered by the two main crops. The main crop is restricted to cover not more than 75% of the arable land. For the EFAs, a farmer has to dedicate

5% of agricultural area in order to get the mandatory DP. Therefore, a bunch of measures from fallow land to landscape elements (single trees, hedges, etc.) were determined by the EC council. The measurements are weighted according to their natural capital. Weighting factors[15] are used in order to calculate the actual area.

Another novelty is the opportunity to use DP in order to support small and medium-sized farms. The German implementation of the new CAP reform makes use of this redistributive payment and pays 50 EUR/ha for the first 30 ha, and 30 EUR/ha for the following 16 ha farmland. Farmers below the age of 40 receive an additional payment[16].

The second pillar of the CAP supports the rural areas. Besides the provision of AES, it includes investment funding as well as development projects for municipalities. They ought to increase the attractiveness of rural areas for tourism and the population living therein.

[15] Fixed in Annex X of the supplementing Regulation (EU) No. 1307/2013 of the European Parliament.
[16] 44 Euros per ha for five years and 90 ha.

Chapter 3.
Research site

The first section of this chapter is dedicated to the description of the socio-ecological system of our study site. We adapt the socio-ecological resilience approach to our rural study region in order to meet our conceptual framework (see figure 2.2 of the previous chapter). Since we are investigating the effects of agricultural production at all three domains (with the respective scales), the study site is screened according to the ecological, socio-cultural, and economic domains. Afterwards, particular interest is given to the agricultural production system of our case study. This second section is further used in order to refer to the chosen values for the model parameters of the baseline model and of the sensitivity anaylses.

3.1. Socio-ecological system of the case study

3.1.1. Location and land cover: the ecological domain

Hesse consists of three administrative districts: Darmstadt, Kassel, and Giessen. Wetterau county belongs to the Darmstadt district. The study site is a small municipality named Wöllstadt and is located in the middle of the most intensively cropped agricultural area of Wetterau county in central Hesse, Germany. It has an area of 15.38 km^2.

Fig. 3.1 shows the Wetterau region. As can be seen, in the eastern part of the Wetterau region as well as in a small strip of the westernmost Wetterau area, the landscape is covered with a mix of forest, agricultural, and residential areas. In the middle, there is an elongated strip from the north to the south where predominantly agricultural production takes place. The study site is outlined in red and lies in the midst of this highly agriculturally used area.

Wetterau county is well known for its intensive agricultural production, with an agricultural area of 53%, followed by 29% forest cover in 2010. Infrastructure and housing area contribute around 7% each to the land cover of Wetterau county, the rest being water bodies (1.2%), recreational area (0.8%), and other areas (Hessisches Statistisches Landesamt, 2012b). Due to highly fertile loess soils, which evolved in the Tertiary Period and reach a depth of up to twenty meters, good climatic conditions, and plenty of water bodies, Wetterau has been used for agricultural production for more than two thousand years (Wetterauische Gesellschaft, 2012; Waldhardt et al., 2010). Consequently, Wetterau counts for one of the oldest cultural landscapes in Europe.

According to Bundesamt für Naturschutz (2012), smaller conservation areas totaling 1.5% are scattered within Wetterau. Moreover, bird protection areas encompass 11.6% of the total Wetterau area, whereas 4.4% are dedicated to FFH[1] areas. FFH areas belong to the European protected Natura2000 network of indigenous habitats[2]. Therein, agricultural production is subject to stricter nature conservation requirements.

Compared to Wetterau county, Wöllstadt has an even higher area of agricultural production (81%). In Wöllstadt, only 1.1% is covered with forest. There is no land that belongs to the Natura2000 network. Housing and infrastructure together cover almost 15% of the area in Wöllstadt, the rest being water bodies (1.5%), recreation area (1%), and other areas (Hessisches Statistisches Landesamt, 2012a).

The geo-physical conditions and historical land use characteristics have prepared the ground for the intensively used agricultural lands of our case study. The ecological domain is confronted with a low land cover heterogeneity. Less then 5% of the landscape is used for something other than agricultural purposes or housing. There is no conservation area at all. Only a few semi-natural landscape elements exist, which deliver wildlife habitats. In our research area, a poor basis for the abundance of species diversity as well as for the delivery of ES is prevalent.

3.1.2. Population and demographic change: the socio-cultural domain

In 2011, Hesse hosted a population of around six million people, of which two thirds were in the Darmstadt district. Within this district, the Wetterau county is the second largest county with respect to the inhabitants with almost three hundred thousand people (Hessisches Statistisches Landesamt, 2012b). As we search for possible futures of our case study, we need to look into the dynamics of demographic change within and around the research area. The population structure in Hesse has developed in different directions. While the population in rural areas dramatically decreased, it increased in cities or regions adjacent to urban centers. The reasons for this development in Hesse were the insufficient supply of care facilities, price increases of mobility

[1] Flora-fauna habitat
[2] FFH guideline 92/43/EWG and Bird Directive 79/409/EWG

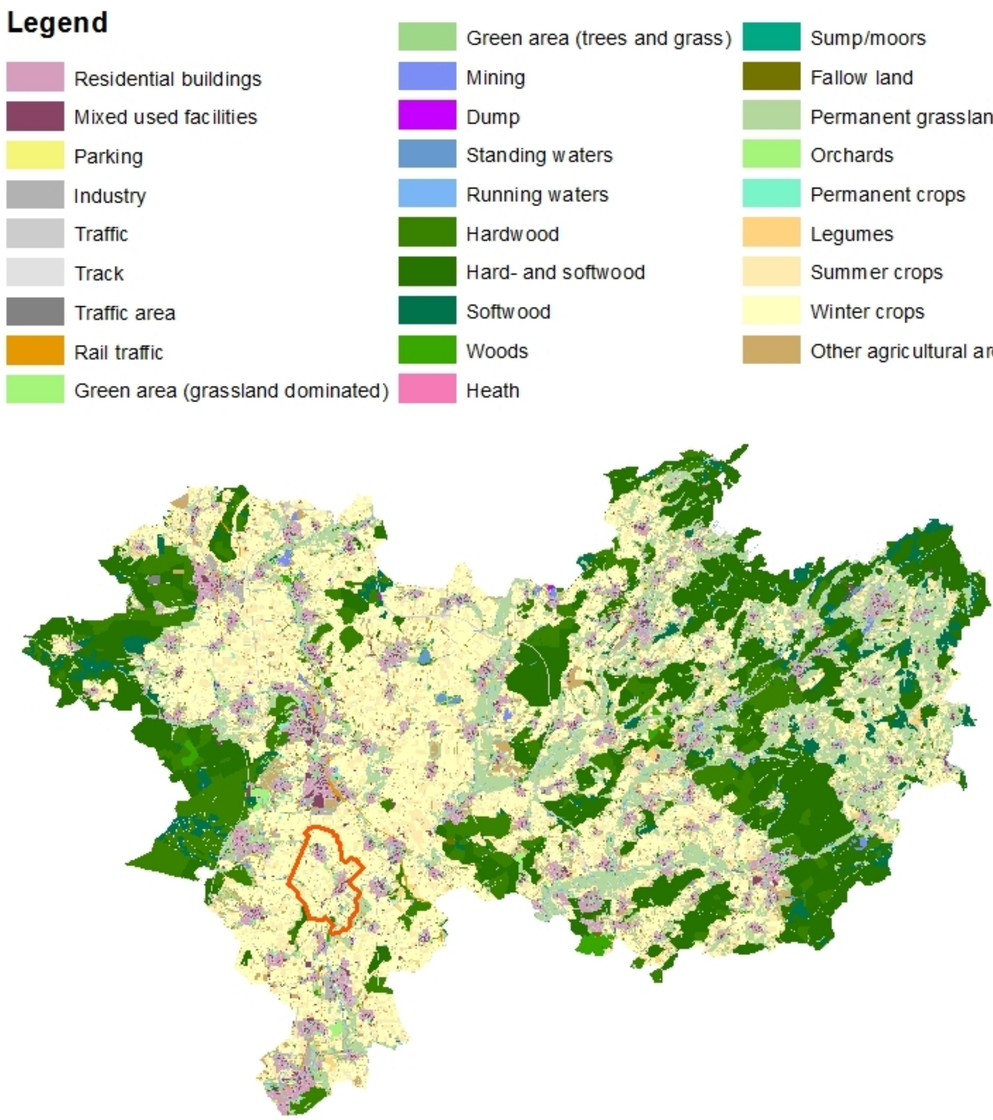

Figure 3.1.: Land use in Wetterau county in 2011. Source: own map based on Schlagkatatster, ALKIS, and ATKIS data provided by the 'Hessischen Landesamt für Umwelt und Geologie' (HLUG).

services, a lack of medical and social supply structures, the absence of economic potential, and an increasing importance of urban lifestyles (Mager, 2011).

Wetterau county attracts more people and exhibited a population increase of 19% between 1987 and 2011 due to its closeness to Frankfurt a. M. (Hessisches Statistisches Landesamt, 2012b). According to HMWVL (2011), the positive demographic change in Wetterau county continues, and the population will further increase by 5% by 2030.

Wöllstadt is a small municipality with only 6,153 inhabitants (Hessisches Statistisches Landesamt, 2012a). With respect to demographic change processes, negative population trends are registered. Between 2008 and 2011, population decreased by 1.8% (Statistische Ämter des Bundes und der Länder, 2013a). Afterwards, the number of inhabitants decreased by 0.5%. According to newest estimations, population will further decrease by 7.1% until 2030 (Bertelsmann Stiftung, 2014).

Within the socio-cultural domain of our case study, we are faced with a declining population. Population decline is an issue that is also addressed in the new CAP reform. Through investments and development plans, the EC tries to strengthen rural areas in order to support the livelihoods of the rural population (European Commission, 2013). With regard to our conceptual framework, we try to show whether this negative population trend is boosted, mitigated, stopped, or affected by the new CAP reform at all. As socio-ecological systems are linked together through several domain-scale combinations and cascading effects, modeling the farm scale also may give insights into the socio-cultural domain at the regional scale (see figure 2.2).

3.1.3. Economy and agriculture: the economic domain

At the level of the federal state of Hesse, the agricultural sector plays only a minor role, contributing 0.4% to the total gross value in 2012. Financial services contributed the most to the total gross value with 35%, followed by public services (21%), production industry (20%), transport and trade sector (20%), and the construction industry with around 4%. However, at the level of Wetterau county, the agricultural sector contributed 6.2% to the total gross value and played a major role within Wetterau county. The construction industry followed with 4.3%, public services with 3.5%, financial services with 3.1%, production industry with 3.0%, and transportation and trade with 2.9% (Hessisches Statistisches Landesamt, 2014a).

Although agriculture is an important part of the economy within our case study, the average agricultural net wage is at a low level compared to other sectors. In 2010, the total average gross wage of Wetterau county was 29,725 Euros per year (Hessisches Statistisches Landesamt, 2014b). After deductions, the average net wage was around 18,900 Euros, which resulted in an average net wage of 1,575 Euros per month or 9.8 Euros per hour. This hourly net wage is taken as off-farm wages for the model. The gross wage for the agricultural, silvicultural, and fishery sector is much lower with 16,350 Euro per year in 2010 (Hessisches Statistisches Landesamt, 2014b).

In order to reduce risk and for other lifestyle reasons, income resources are diversified, and many farms are managed as part-time farms (Hansson et al., 2013). Hesse is one of the German states with the highest agricultural area under part-time farming (35% in 2010) (Hessisches Statistisches Landesamt, 2013). The average size of part-time farms is 23.6 ha, which differs highly throughout the regions (Mawick et al., 2011). In Wetterau county, 63% of the farms are part-time farms (Hessisches Statistisches Landesamt, 2012a). Under these conditions in Wetterau county, 72% of all farms have an uncertain farm succession, and the average age of most farmers in Hesse lies between 45 and 64 (Hessisches Statistisches Landesamt, 2012j). Due to the 'Hessische Landgüterordnung' from 1947, after succession, there is no legal obligation to split land property into partial plots. Instead, farmland can be passed on to only one heir, and compensation for the coheirs is regulated by law. Fields keep their sizes without getting smaller throughout time. Nevertheless, most farms with a size of 50 ha or smaller do not have a secure farm succession in Hesse (Mawick et al., 2011).

In Wöllstadt, agricultural production with respect to area is even higher in relative terms[3]. Within the economic domain of our case study, farmers face an insecure farm succession and predominantly part-time farming (40%) (Hessisches Statistisches Landesamt, 2012a). With the prevalent agricultural practices, farming is in most cases not the only income source since more attractive ones (with higher wages) exist. In the coming years, structural change can be expected, which might lead to a regime shift within the economic domain at the farm scale. This domain-scale regime shift may trigger other domain-scale combinations, and a new regime of the socio-ecological system might evolve. In the following section, a closer look at the agricultural production system is taken. It also serves as a basis for our model calibration.

3.2. Agricultural production system of the case study

Parameters presented in this section refer to fixed values and constitute model data in order to calculate endogenous model variables, which we are going to present in the next chapter. Since the model was calibrated according to spatial land use data from the 'Hessische Landesamt für Umwelt

[3] See land use distributions from Hessisches Statistisches Landesamt (2012a) and Hessisches Statistisches Landesamt (2012b) above.

und Geologie' (HLUG) from 2011, most statistical data was collected from the agricultural census in 2010. If possible, statistics were collected at the level of the study site. However, due to data protection regulations, in some cases only information at the county or district level was available. All statistical values used as parameters in the model are summarized in Appendix B.1 unless specified differently at the spot. With the aim of presenting a comprehensive picture of the study site, some statistical values were compared to different spatial scales including the federal state, district, county, and municipality levels.

3.2.1. Farm types and sizes

In our model we include different farm types. Therefore, a closer look at the farm types within the study region is needed.

The officially established classification system organizes agricultural farms according to their common economic orientation: (1) arable farms producing, for instance, cereals, pulses, oil fruits, root crops, and vegetables among other crops; (2) horticultural farms; (3) permanent crop farms (e.g. wine); (4) grazing livestock farms (e.g cattle, goats, or sheep); (5) processing farms (e.g. pig-fattening or poultry); and three different mixed farm types: (6) mixed plant production, (7) mixed livestock production, and (8) plant plus livestock-producing farms (Statistisches Bundesamt, 2012).

In table 3.1, the proportions of farm types and their changes over time for Wetterau county are charted. Therein, arable farms had the highest relevance and covered more than 50% of the agricultural area in 2010, followed by plant and livestock-producing farms (23.1%) and grazing livestock farms (22.2%). Horticultural and permanent farms as well as mixed plant producing farms cover only a small area within the Wetterau county. With respect to the percentage of farms, these three farm types became less important over time. In 2010, 1% of the farm types were processing farms covering 1% of the total area. For comparison, in 2003, the same proportion of farms covered only 0.2% of the agricultural area. From 2003, the production area of processing farms increased fivefold. Mixed livestock production farms have also increased their production area from 0.7 to 1.2%, although the farm proportion sunk from 1.4 to 1.3%. These trends indicate an increase in farm sizes.

There are differences within the classification thresholds and counting units, which makes comparisons over time difficult. From the beginning of the '70s, the German classification criterion was the operating income[4]. If 50% of the operating income was earned through horticulture, the farm belonged to type two. However, in 2003, the European classification system was applied,

[4]Before, the classification was based on the size of the agricultural area under cultivation.

Table 3.1.: Proportions of farm types in Wetterau county regarding their economic orientation for 2003, 2007, and 2010 (%). Source: Statistische Ämter des Bundes und der Länder (2013b); Hessisches Statistisches Landesamt (2012j,i)

Farm type	Share in Wetterau county [%]					
	2003		2007		2010	
	Farms	Area	Farms	Area	Farms	Area
Arable farms	48.0	55.9	48.2	55.9	47.8	50.2
Horticulture	5.6	0.6	4.7	0.6	3.8	0.4*
Permanent farms	4.4	0.3	2.0	0.3	2.4	0.3
Grazing livestock	18.8	19.0	22.8	19.7	22.0	22.2
Processing farms	1.0	0.2	1.2	0.4	1.0	1.0
Mixed plant production	4.5	3.2	4.8	4.0	3.1	1.6
Mixed livestock production	1.4	0.7	1.6	1.2	1.3	1.2
Plant and livestock production	16.4	20.0	14.7	19.0	18.5	23.1*

*Imprecise values due to incomplete data in Hessisches Statistisches Landesamt (2012j).

and the threshold for the classification criterion rose to 75% (Blumöhr et al., 2006; Statistisches Bundesamt, 2012; BMELV, 2012). Since 2010, the standard output[5] has been used. Due to that, a separate consideration of livestock farming will be conducted later in this section.

The average farm sizes in Germany increased from around 15 ha in 1979 to 56 ha in 2010, and the amount of farms more than halved during this time (see fig. 3.2) (Statistisches Bundesamt, 2013). An even stronger trend can be found in Hesse. From almost 91,000 farms in 1971 only 18,000 in 2010 remained (Mawick et al., 2011). Concentration and specialization processes within the agricultural sector in Hesse have been observed, and mainly single production systems, such as pig fattening or purely arable farming prevail (Berger, 2012).

Within Wetterau county, 988 agricultural farms cultivate an area of more than 525 km^2 agricultural lands. Of these, half of all farms have a farm size between 20 and 100 ha. Only 15% cultivate more than 100 ha agricultural area, and the remaining 35% less than 20 ha (Hessisches Statistisches Landesamt, 2012a).

In Wöllstadt, 25 farms cultivate 1,432 ha agricultural area. 20 farms are arable, four produce plants and livestock, and one is a horticultural farm. The proportion of farms in Wöllstadt that cultivate more than 50 ha is 44%, which is even higher than the average for Wetterau county (37%). 80% of the farms in Wöllstadt have an area between 20 and 100 ha. Two farms cultivate total areas larger than 200 ha (Hessisches Statistisches Landesamt, 2012a), which indicates the agriculturally intensified production situation of the study site. There is only one farm in Wöllstadt that has an area less than five ha (Hessisches Statistisches Landesamt, 2012i), which might be the

[5]Gross value of agricultural products using farm-gate prices without subtracting variable costs.

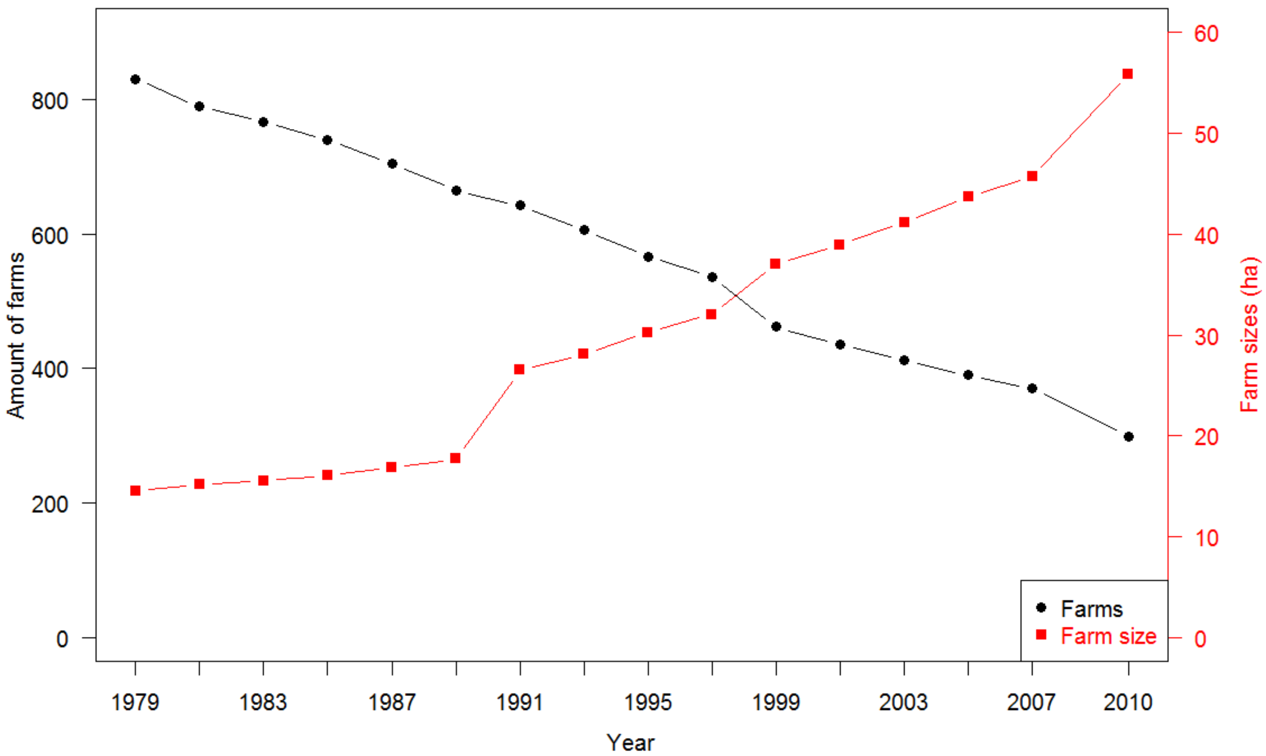

Figure 3.2.: Average farm size development in Germany (1979-2010). Source: Statistisches Bundesamt (2013).

horticultural farm[6]. This farm will be excluded from the modeling process.

3.2.2. Crop production and prices

In table 3.2 the percentage of cultivated crops in Wöllstadt are compared to the percentage crop area within Wetterau county.

For both regional levels, wheat is the most important crop. However in Wöllstadt, wheat production is particularly pronounced with 52% of the total agricultural area. Sugar beet production also plays an outstanding role in Wöllstadt with more than 16% compared to 5.5% land cover in Wetterau county. Rapeseed production in Wöllstadt and Wetterau county have similar proportions with around 11%. In Wöllstadt, the area of permanent grassland is much lower (5%) than in Wetterau county (24%). The proportion of potato production in Wöllstadt is more than twice as much as in Wetterau county. Due to one horse farm, oat production is relatively high in Wöllstadt[7]. However, silo maize production is much lower in Wöllstadt compared to the county level. Particularly apparent is the absence of forest land in the study site, which is already low in the entire Wetterau county. The emphasis on wheat, sugar beet, and potato production shows excellent soil conditions and again mirrors the intensive agricultural land use in Wöllstadt.

Crop prices are highly volatile throughout the year as well as annually. In figure 3.3, time series of grain prices between 2000/01 and 2014/15 for the Darmstadt district are depicted based on data from Ktbl (2016)[8]. As visible in the graph, grain prices are highly correlated. Density functions of the grain prices show a skewed distribution to the left (see Appendix D), leading to a lower median than mean value (see table 3.3). Farmers within our research area face volatile prices. The mean values for grain prices lie between 12.8 Euros/dt (triticale) and 14.6 Euros/dt (maize) with standard deviations of around 4.5 Euros/dt. For model calibration, relatively high crop prices, having the same level as the mean values plus one standard deviation are taken[9]. In order to check for model robustness, a sensitivity analysis is conducted with the median values for grain crops.

For model calibration, average crop prices of the Darmstadt district are taken from Ktbl (2016). For some crops such as grass silage and hay coming from permanent grassland, silo maize for biogas plants, corn cob mixes, and corn maize, data from the same Ktbl database was not available, and other sources needed to be found. In Appendix E, all land use activities included in the model, as well as their prices, year, regional level and data sources, are presented in a tabular form.

[6]Horticultural farms have an average size of only 6.8 ha (Hessisches Statistisches Landesamt, 2012f).
[7]Expert interview with the responsible farmer.
[8]Prices on lower spatial level were not available.
[9]This is adapted to crop prices of the calibration year 2011.

Table 3.2.: Land use in Wöllstadt and the Wetterau county in ha and % (2011). Source: GIS data (Schlagkatasterdaten) received from HLUG

Crop	Wöllstadt [ha] (%)	Wetterau county [ha] (%)
Winter wheat	627 (51.7)	19,593 (36.1)
Sugar beets	199 (16.4)	2,985 (5.5)
Rapeseed	129 (10.6)	5,804 (10.7)
Winter barley	95 (7.9)	2,990 (5.5)
Permanent grassland	60 (5.0)	13,132 (24.1)
Potatoes	28 (2.3)	606 (1.1)
Oats	16 (1.3)	378 (0.7)
Rye	16 (1.3)	401 (0.7)
Fallow	9 (0.7)	338 (0.6)
Silo maize	8 (0.6)	2,484 (4.6)
Grasses (silage)	6 (0.5)	525 (1.0)
Corn maize	5 (0.4)	1,055 (1.9)
Triticale	3 (0.2)	357 (0.7)
Winter oats	3 (0.2)	3 (0.0)
Vegetables	2 (0.1)	130 (0.2)
Strawberries	2 (0.1)	67 (0.1)
Mixed grains	2 (0.1)	7 (0.0)
Forest land	0 (0)	791 (1.5)
Miscellaneous	4 (0.3)	2,697 (5.0)

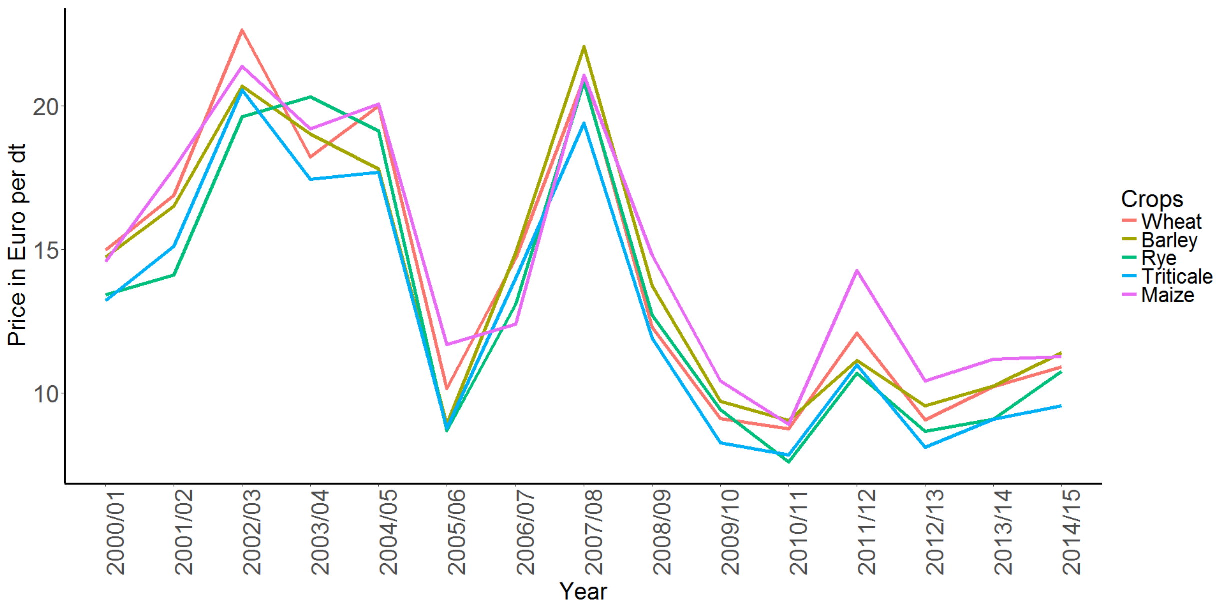

Figure 3.3.: Time series for grain prices: Darmstadt district (2000/01 - 2014/15). Source: Ktbl (2016).

Table 3.3.: Descriptive statistics for grain price time series of the Darmstadt district (2000/01 - 2014/15)

Crop	Min [Euros/dt]	Mean [Euros/dt]	Max [Euros/dt]	Sd [Euros/dt]	Median [Euros/dt]
Wheat	8.76	14.07	22.67	4.69	12.29
Barley	8.90	13.96	22.07	4.43	13.74
Rye	7.59	13.21	20.85	4.65	12.72
Triticale	7.84	12.80	20.57	4.38	11.90
Maize	8.90	14.63	21.39	4.25	14.28

3.2.3. Livestock keeping

Numbers for Hesse show an increase in the average amount of livestock per farm (Mawick et al., 2011). Between 1970 and 2010, the amount of cattle farms decreased by almost 94%, while the number of cattle decreased only by around 58%. Within the same time period, milk yield increased by more than 50%. This shows an intensification of milk and cattle production. Similar trends can be found for pig-fattening and pig-breeding farms. While in 1983 only 3% of all pigs were kept in farms with more than 100 stable places, in 2010, 64% were kept in such farms. Actually, in 2010, 42% of all pigs were kept in farms with more than 200 stable places. However, with respect to the total number of pigs in Hesse, between 1999 and 2010, there is a reduction of breeding sows (- 30%), while pig-fattening increased by 103% during this time (Mawick et al., 2011).

In Wetterau county, around 60% of the agricultural farms keep livestock. Among these farms, 60% still own cattle, and 40% own pigs. The average amount of livestock per farm is around 60 for cattle and 98 for pigs.

In the case study, four out of 25 farms keep 53 cattle, and five farms keep 401 pigs (Hessisches Statistisches Landesamt, 2012a). In Wöllstadt livestock keeping is not yet very intensified compared to the state average. Compared to the county level, cattle keeping is underrepresented, while pig fattening is emphasized. The type of livestock keeping (e.g. dairy, breeding, fattening) in Wöllstadt is not apparent from statistics. Therefore the most common mode of livestock keeping in Hesse was chosen for the model. For cattle keeping, this is dairy farming, and for pig farming, the most common mode is pig fattening (Mawick et al., 2011). In the model, four plant and dairy producing farms, each having 25 stable places and five plant and pig-fattening farms having 30 stable places[10] are modeled. According to husbandry methods for dairy, loose-housing stables with solid manure and for pig fattening, slatted floor, are assumed in the model.

[10] 2.8 pigs per stable place and year.

3.2.4. Soil treatments and yield levels

In Hesse, almost 90% of the farms cultivate their soils with conventional soil tillage, which is ploughing. With respect to the agricultural area, 57% is ploughed and 37% cultivated with conserving soil treatments (e.g. harrowing or with cultivators). The other area is cultivated with direct sowing. In Darmstadt district, 88% of the farms apply conventional soil tillage, which covers 55% of the agricultural area. 45% of the area is cultivated using conserving soil treatments (Hessisches Statistisches Landesamt, 2012h). In the model it is assumed that farmers plough their fields conventionally.

Wetterau county has one of the highest yield units with 61[11] compared to other counties of Hesse (Hessisches Statistisches Landesamt, 2012b). The average yield for the main crop types in Hesse (winter wheat, rapeseed, and sugar beets) more than doubled between the fifties and 2010. Further, the average silo maize yield doubled between 1950 and 1980. Since 1980, silo maize yield has fluctuated around the same level (Mawick et al., 2011).

Yield data for the calibration year are summarized in Appendix G. For the model, average crop yields are gathered at the county level from Hessisches Statistisches Landesamt (2012d). Since not all yields for the crops included in the model are covered with this reference, some crop yield levels are gathered at the district level from the Ktbl database (Ktbl, 2016).

3.2.5. Field structure

The field structure of Wöllstadt has a relatively uniform appearance. In 1945, on average, arable fields were smaller than in 1970 (compare fig. 3.4 and fig. 3.5). As can be seen, urban areas increased at the expense of agricultural areas. This trend has continued into the present year as apparent in fig. 3.6. On the orthophotos, the typical rectangular shape of fields is visible. Rectangular field shapes allow for simplified and cost-reduced cultivation practices.

For the spatially explicit modeling, GIS data from the HLUG about field and landscape structure in Wöllstadt is analyzed (see fig. 3.7). Since one of the central variables is field size, it needs to be compared to the current and actual field sizes. GIS data only show small field units with regard to the ownership and do not contain information about user rights. The field sizes from GIS did not fit to the actual cultivation units. They have an average size of 1.6 ha. In order to asses the size of

[11] Range: 0 - 100.

Figure 3.4.: The municipality Wöllstadt with its field structures in 1945. Source: Orthophotos received from HLUG.

Figure 3.5.: The municipality Wöllstadt with its field structures in 1970. Source: Orthophotos received from HLUG.

Figure 3.6.: The municipality Wöllstadt with its field structures in 2017. Source: GoogleMaps (2017-01-19).

the current cultivation units, GIS data is edited by aggregating polygons[12]. Thereby tracks smaller than management roads, which are usually 3 m wide, were deleted, and the fields became bigger (see fig. 3.7, right). After this data adjustment, the average field size was 4.2 ha and reflected the current field structure more accurately.

[12]Up to 2.9 m gap between fields.

Figure 3.7.: Agricultural field structure in Wöllstadt (2011). *Left:* Original land parcels; *right*: Land parcels after editing (see text). Source: GIS data received from HLUG

3.2.6. Farm labour

In 2010, 67% of the agricultural labour force in Wetterau county was provided by family labourers (Hessisches Statistisches Landesamt, 2012j). The labour force in Wöllstadt consists of 57.5% family, 35% foreign, and 7.5% seasonal labourers. The total labour force in Wöllstadt was around 40 labour units [13] in 2010. Of these, 23 labour units were delivered by family labourers (Hessisches Statistisches Landesamt, 2012i).

For the model, these family labour units were allocated evenly to each farm. It is assumed that only 50% of the family labour capacity is purely spent for crop and livestock-correlated production steps. The other 50% is needed for office work such as paperwork and farm communication. The total family labour capacity for field and stable work was 997 hours of a typical farm per year. Foreign labour hours can be bought for a gross wage expenditure of 15.5 Euros per hour[14]. Seasonal labourers could be hired for the minimum wage of 7.40 Euros per hour in 2011[15]. These values are taken as respective parameters for the model.

Labour requirements per ha depend on several parameters such as land use type, field size, distance between farm and field, and the degree of mechanization. For the model, the level of engine power is 120 KW (see section 2.4.2.2), and the distance between farm and field is assumed to be one ha due to field consolidation assumed by the modeling approach. The Ktbl database (Ktbl, 2013a) offers data on labour requirements for several land use activities for two, five, and ten ha. Depending on the modeling step, the model uses labour input coefficients as summarized in Appendix H. One exception is the input requirement for fallow land. Since fallow land is small and typically not used to exploit EoS, the labour coefficient refers to the smallest receivable data on field size (1 ha) and does not change.

3.2.7. Revenue and variable costs of production

Within the Darmstadt district, the standard output of family farms in 2010 was on average 34,997 Euros, whereas commercial farms generated 163,977 Euros[16] (Hessisches Statistisches Landesamt, 2012e). The profit of farm types, however, differs and changes annually. For example, in the financial year 2010/11, commercial arable farms participating in the Hessian regional statistics gained the highest profit with 76,886 Euros, followed by fodder farming with 70,781 Euros,

[13] One labour unit corresponds to 40 average labour hours per week.
[14] This corresponds to a trained labourer of wage group three according to the collective agreement for Hesse (LLH, 2015).
[15] According to the collective agreement of the agricultural sector in Hesse in 2011.
[16] Profit for fiscal purposes.

Table 3.4.: Financial model capacity parameters: liquid monetary amounts per farm size

Farm size [ha]	Liquid money [Euros/year]
< 50	6,500
50 - 150	9,750
> 150	13,000

and processing and mixed farms both with around 60,000 Euros. Compared to that, in 2011/12, processing farms obtained the highest profit (73,919 Euros), whereas arable and mixed farms gained the lowest profit with around 60,000 Euros. The reasons were increasing costs for farming inputs (e.g. fodder, fertilizer, seedlings, and fuel), which outweighed increasing revenues (LLH, 2013).

A key figure of the economic farm performance is capital formation, yet it fluctuates strongly from year to year. From 2011 to 2012, it sunk from around 18,000 Euros to 10,000 Euros for commercial farms (LLH, 2013). In the model, we assume that liquid money depends on the farm size (see table 3.4). Farm sizes lower than 50 ha have at their disposal 6,500 Euros per financial year, and farms between 50 and 150 ha have 50% more. Farms bigger than 150 ha own 13,000 Euro equity capital. Each agricultural activity requests available financial resources differently, which can be obtained from the Ktbl database. Variable costs of each activity also influence the gross margin and need to be considered in the model. Financial coefficients and variable costs for each activity, used in the model, are summarized in Appendix K.

3.2.8. Governmental payments and measurement participation

In Wöllstadt, governmental payments consist of DP and AES payments. As outlined above, there were no other payments, e.g. for Natura2000 areas. Similarly, there were no payments for compensatory allowances in the village since the study site benefits from excellent production conditions. The level of DP for the case study was calculated on the basis of a cluster analysis[17] conducted in the working group of the JAGUAR project[18]. All municipalities of the Vogelsberg and Wetterau county in 2011 were classified into intensive and extensive agricultural production areas. For the intensive agricultural production area (to which Wöllstadt belongs), the average level of DP was calculated using single municipality data from 2010[19] (Hessisches Statistisches

[17] For clustering, some important parameters have been urban, artificially vegetated, forest, abandoned, wood, grassland, orchard, or arable area.

[18] Sustainable futures for cultural landscapes of **Ja**pan and **G**ermany - biodiversity and ecosystem services as **u**nifying concepts for the management of **a**gricultural **r**egions

[19] Due to payment graduations, DP were not consistent throughout all farms.

Landesamt, 2012i). This average value of 275.6 Euros per ha is used in the model.

On the web page of the 'Bundesanstalt für Landwirtschaft und Ernährung' (BLE, 2014), information about the amount and types of payments for the current EC household year can be viewed for two years[20]. Wöllstadt received 1,027 Euros for AES in 2014 (the available year most closest to the calibration year). This value is chosen for the AES budget parameter in the model. AES in the model consist of annual flowering strips, which are rewarded with 750 Euros per ha according to the HIAP guidelines of the Hessian Ministry for the Environment, Climate Protection, Agriculture, and Consumer Protection[21] (HMUKLV, 2015b). Translated into flowering strips, only around 1.5 ha of semi-natural habitat was provided by AES. Taking this value as a baseline in the model, the variable production costs are assumed to be 160 Euros per ha since the difference equals the gross margin of winter wheat, which is often taken as a reference for the calculation of opportunity costs[22]. Labour and capital coefficients for flowering strips used in the model are 2.78 Euros per ha and 10.41 Euros per ha, respectively. They are calculated based on Ktbl data on seeding and milling, assuming 120 KW engine power, field size of one ha, distance to the farmstead of one km, and a working width of 2.5 m (Ktbl, 2013a).

For modeling the new CAP reform, farmers can also consider hedges since they count to the EFAs. For their cost calculation, it is assumed that hedges have lifespans of 20 years. Only variable maintenance costs are taken into account since often hedges already exist; alternatively the planting and establishing of hedges could be funded by programs or farmers. Then they can apply for grant funds (Meyerhoff, 2011). Variable costs in this case are calculated using an official cost catalog of the Bavarian National Office for the Environment (Beiersdorf, 2012). Labour time is not considered since the model assumes perfect outsourcing of hedge maintenance. Variable costs consist of (a) trimming (1.5 times per year in the first four years), (b) cutting parts of the hedge back to the trunk 5 times within 20 years, (c) disposing wood and shrubs (also five times within 20 years), and (d) the gross margin (opportunity costs) of wheat production. The total variable costs for hedges per ha and year totals 5,407 Euros.

[20] Due to Regulation EU EURATOM No. 966/2012, Regulation No. 1306/2013, and Regulation No. 908/2014 the amount of European agricultural payments have to be public.

[21] Entered into force on the 21st of September 2015.

[22] According to the CAP, AES should only compensate farmers for their losses and not put incentives to gain from implementing AES.

3.2.9. Energetic biomass production

From the early eighties to 2010, the amount of biogas plants in Hesse reached a level of 124. These plants supply around 48 MW and utilize an area of 10,400 ha, which covers 1.2% of the agricultural area. Almost 90% of that area is covered with silo maize and the rest with other grains. Compared to that, the area of rapeseed for oil and diesel production is much higher with 47,000 ha in 2010 (Mawick et al., 2011).

In 2013, the total capacity of biogas plants in Wetterau county was around 73,000 MWh per year (Dölling and Voß, 2014). Within a range of 15 km around Wöllstadt, the biogas capacity was around 16,000 MWh per year. According to LLH (2012), 62% of the biogas production in 2011 was supplied by silo maize. Depending on the silo maize yield for biogas, which is assumed to be 500 dt per ha, in the case study, the area for silo maize production utilized in biogas plants, is currently (2016) around 100 ha per municipality within the intensively used agricultural areas of the cluster analysis mentioned in section 3.2.8. This is around 1% of the area in Wöllstadt and is used as an upper limit for the model according to the maximum biogas capacity.

3.2.10. Land tenure

Within Hesse and the Darmstadt district, around 64% to 68% of the agricultural area is rented. In Wetterau county, 64% is rented; this is in Hesse no exception (Hessisches Statistisches Landesamt, 2012j). In 2010, the arable land rent in Darmstadt district was 201 Euros per ha, however, for newly rented land, the price was higher; about 255 Euros per ha arable land (Hessisches Statistisches Landesamt, 2012g). The average arable land rent for Wetterau county in total in 2010 was 214 Euros per ha, which is used as parameter for the model.

Since land rent is an important parameter in agricultural decision making, especially when looking at longer time periods, a second sensitivity analysis was conducted using different land prices. Compared to crop prices, land prices did not fluctuate annually around a certain price level but showed an increasing trend line for the last years. Within the same year, average rent prices differ from region to region. Therefore, average rent prices for arable land in all counties of Hesse for the year 2010 were taken from Hessisches Statistisches Landesamt (2012c) and compared. According to descriptive statistics [23], Wetterau county almost exactly mirrors the median value of all observed counties. For the sensitivity analysis one standard deviation lower and higher than the median value were used as parameters.

[23]Mean: 225.6; median: 216; min: 151; max: 342; Sd: 59.

4 Chapter 4.
FOLAS: Farm optimization at landscape scale

In this chapter, technical and mathematical dimensions of the model are described in detail. At first, the model content is presented in a programming language. In doing so, model components are summarized and classified into sets and variables. In the second section after an overview of the modeling procedure, mathematical model formulations are outlined. It follows a classification into two different modeling techniques, which are integral parts of the model procedure. At the end of the section, alternate political incentive schemes will be presented. They are simulated in order to compare effects of the new CAP reform on some chosen biodiversity indicators within our study region. These indicators will be described in the last part of the chapter.

4.1. Model content in GAMS language

In order to set up our model, General Algebraic Modeling System (GAMS) software was used. GAMS is an elaborated modeling system, solving mathematical programming problems through optimization. Before elaborating on the mathematical model formulations, which are used to explain the case study at hand, basic system components of the case study are introduced in the mode of sets and endogenous variables. The values of variables are unknown and will be simulated by the model. On the contrary, parameters are fixed values (data) that are used by the model in order to calculate variables (Rosenthal, 2017). They were presented in the previous chapter.

4.1.1. Sets

GAMS uses sets in order to integrate system components into the modeling process. Each set contains elements, capturing certain dimensions. In order to get a broad picture of our model world, all FOLAS sets are summarized in table 4.1 with a short description of their elements.

In total we have 24 farms g in the case study. They are classified into arable farms $a(g)$, plant and pig-fattening farms $m(g)$, and plant and dairy producing farms $d(g)$. Every farm type is randomly distributed to the farmland at the beginning of the modeling process and does not change throughout the model iterations. However, some farms and farm types may close down during farm size optimization and no longer exist after the model run. Potentially they are in the programming, but not chosen.

The set act covers all cropping and livestock farming activities, whereas the two sets $lu(act)$ and $li(act)$ are subsets of act and include only the cropping activities or only the livestock keeping activities, respectively. For the model, all crops in the list in table 3.2 (down to triticale, excluding oats) are included. The production method of permanent grassland is differentiated between silage and hay production. Silo maize production is further divided into maize silage as a fodder crop and silo maize for biogas plants. Additionally, corn cob mix and summer peas, as alternative land use activities, are included in the model. Thus almost the whole cropped area of Wöllstadt is covered by the model activities (97.6%).

For livestock-keeping activities, information on feeding practices is required. Fodder crops are summarized in the set $fc(lu)$, which is a subset of $lu(act)$. The set for feed nutrients $feed$ includes energy, proteins, RNB (ruminal nitrogen balance), and dry matter.

Farming inputs contain land, labour, capital, pig, and dairy stable places. They are separated into farming inputs i including all of them and the subset $lu(i)$ for farming inputs of only the cropping activities.

Each farm has a set (f) of 36 potential fields. The number of fields per farm is based on GIS information about the original field number of the study area divided by the amount of farms in the case study (24 farms). However, fields do not need to be fully utilized by farmers. The optimal field size, which is one of the model variables, determines the amount of fields since each farm has a predetermined amount of agricultural land within the linear programming part of the model (according to the spatial simplification explained in section 2.4.2.2). An overview of predetermined farm sizes used by the model can be found in Appendix C.

4.1.2. Variables

Variables used in the model are summarized in table 4.2, with their descriptions and variable types as they are implemented in GAMS. Prior to model execution, variable values are unknown.

Table 4.1.: Model sets: declarations and descriptions.

Declaration	Description	Elements
act	Cropping and livestock activities	Winter wheat, winter barley, triticale, rye, silo maize, silo maize for biogas, corn cob mix, corn maize, potato, sugar beet, oilseed rape, permanent grassland (silage), permanent grassland (hay), grasses (silage), summer peas, fallow, pig-fattening, dairy
$lu(act)$	Cropping activities	Winter wheat, winter barley, triticale, rye, silo maize, silo maize for biogas, corn cob mix, corn maize, potato, sugar beet, oilseed rape, permanent grassland (silage), permanent grassland (hay), grasses (silage), summer peas, fallow
$li(act)$	Livestock activities	pig-fattening, dairy
$fc(lu)$	Fodder crops	Winter wheat, winter barley, triticale, rye, silo maize, corn cob mix, corn maize, permanent grassland (silage), permanent grassland (hay), grasses (silage), summer peas
$feed$	Feed nutrients	Energy, proteins, ruminal nitrogen balance (RNB), dry matter
i	Farming inputs	Land, labour, capital, pig stable places, dairy stable places
$lu(i)$	Farming inputs for cropping activities	Land, labour, capital
g	Farms	Farm 1 - farm 24
$a(g)$	Arable farms	Farm 1, farm 2, farm 4, farm 5, farm 6, farm 7, farm 8, farm 12, farm 14, farm 15, farm 16, farm 18, farm 20, farm 23, farm 24
$m(g)$	Plant and pig-fattening farms	Farm 3, farm 10, farm 11, farm 13, farm 21
$d(g)$	Plant and dairy-producing farms	Farm 9, farm 17, farm 19, farm 22
f	Fields	Field 1 - field 36

FOLAS maximizes the total gross margin (TGM) of all farms in the municipality (TGM_mun) by summing up the TGM of each farm, field, and land use type. For example, $x_{g,f,lu}$ is one of the central variables, calculating the levels of cropping activities in ha per farm and field. It is a 'Specially Ordered Sets Type 1' (SOS1) variable. This means, that only one element of the last set can have positive values (with the rest being zero) (McCarl et al., 2012, chapter XXI). In that case, each field f per farm g of the variable x can have only one cropping activity lu. This is necessary because now a land use decision is linked with one field of a certain size. With this special type of variable, where mutual exclusivity of elements within a set is forced, the model changes to a mixed integer programming model, requiring another solver in GAMS: the SCIP (solving constraint integer programs) solver[1].

Livestock keeping activities $y_{g,li}$ and flowering strips $aes_{g,f}$ are both positive variables and provide information about activity levels regarding their sets: $y_{g,li}$ counts the level of occupied livestock stable places per farm (for cattle or pigs) and $aes_{g,f}$ the level of flowering strips per field and farm.

In GAMS, it is possible and often advisable to assign bounds or starting values to variables. This is done for the upper bound of $aes_{g,f}$ being one. Therefore, flowering strips cannot be bigger than one ha[2]. However, in order to model flowering strips more accurately, their width is taken into account as well. Therefore, another variable is included: $aes\_width_{g,f}$. It is a semi-continuous variable. If it is non-zero, it takes on a given level above a minimum or below a maximum. The minimum level for $aes\_width_{g,f}$ is five, meaning that flowering strips are at least five meters wide.

For livestock-feeding practices, another positive variable is implemented: $feedtrans_{g,fc}$. It calculates the total amount of fodder crops that is transferred from one's own crop production in order to feed livestock. This variable is calculated for each farm and depends on the livestock type and number.

Off-farm work (off_lab_g) covers the amount of family labour hours spent outside the agricultural farm. On the contrary, farmers can hire additional labour hours ($hire\_lab_g$). Both variables are positive variables and provide insights into the labour allocation patterns of agricultural farms.

The possibility of renting out land is covered by the variable $rentout\_land_{g,f}$. It shows whether a farmer gives up his or her land under the current (short-run) production conditions. Under the given production conditions, this variable should be zero since in the model calibration farming practices are assumed to be stable (in a short-term perspective). Renting out land is linked with farm size changes. However, farm size changes are modeled only in the 3rd iteration where

[1] For more information about the SCIP solver, see Maher et al. (2017).
[2] According to HMUKLV (2015b), flowering strips need to be at least five meters wide and smaller than one ha.

Table 4.2.: Model variables: declarations, types, and descriptions.

Declaration	Type	Description
$x_{g,f,lu}$	Specially Ordered Sets Type 1 (SOS1)	Land use activities per farm and field [ha]
$y_{g,li}$	Positive variable	Livestock activities per farm [amount of stable places]
$aes_{g,f}$	Positive variable	AES in form of flowering strips per farm and field [ha]
$aes\_width_{g,f}$	Semi-continuous variable	Width of flowering strips per farm and field [m]
$feedtrans_{g,fc}$	Positive variable	Fodder crops fed to livestock per farm [dt]
off_lab_g	Positive variable	Off-farm labour per farm [hours]
$hire\_lab_g$	Positive variable	Hired labour per farm [hours]
$rentout\_land_{g,f}$	Positive variable	Land rented out per farm and field [ha]
$fieldsize_{g,f}$	Semi-continuous variable	Field size per farm and field [ha]

farmers interact with each other. In all other iterations, this variable serves as a test of whether the modeled farming structure is stable under the current short-term conditions.

The field size variable $fieldsize_{g,f}$ also counts to the semi-continuous variables. It provides the possibility to allocate upper and lower bounds if the value is non-zero. In comparison to standard positive variables, semi-continuous variables need not have a value if forcing an upper or lower bound on it (Rosenthal, 2017). As noticed above, this is required since a farmer does not need 36 potential fields. $Fieldsize_{g,f}$ calculates the size of each field and thereby the amount of fields per farm.

4.2. Mathematical model formulations

Within this section, the main equations that constitute our model are written down and described comprehensively. As previously introduced in section 2.4.2.2, FOLAS follows a certain execution logic comprising four iterations. The iteration procedure is outlined and charted in the following subsection.

4.2.1. Iteration procedure

FOLAS acts at farm, field, and municipality level. In doing so, the logic of farm consolidation and farm size changes as part of long-term structural change processes is addressed. Within the model procedure, field-level information serves as a model input, which constraints model output at the farm level. Farm-level output is then used as another model input affecting the next iteration and so on. Figure 4.1 gives an overview of the iteration procedure at field, farm, and municipality levels.

The first iteration ought to simulate the current farming conditions. The amount of farms and field size are adapted to the real conditions of the case study. Since the actual fields are relatively small, the field size variable has an upper bound of 3.5 ha with respect to field size information mentioned in section 3.2.5. In order to account for the smaller field sizes, labour coefficients are adapted accordingly. Labour requirements (per ha) are based on an average field size of 2 ha (see Appendix H).

Within the 1st iteration, farms do not interact with each other. This delivers the calibration results of our case study. The objective of each farm is to maximize the total gross margin (TGM) considering their land, labour, stable, and capital constraints. In maximizing the sum of each farm's TGM, the aim of the 1st iteration is to allocate grassland proportions to the available farms in an optimal manner. Therefore, a grassland constraint is implemented at the municipality level. There are other farming aspects implemented at the municipality level. These are sugar beet and potato contracts, the maximum area of silo maize for biogas production, and the available budget for flowering strips. The model decides on their allocation. Often, such legal conditions are allocated 'a priori' to the farms either due to survey information (Kächele and Dabbert, 2002; Aurbacher, 2010; Kantelhardt, 2003) or averages (Dabbert et al., 1999). We assume that farms are different with respect to these aspects. Since no data is available, the applied mode of allocation seems plausible as due to path dependencies and farm developments, certain farm structures evolve (Balmann, 1999).

It has to be noted that the first iteration also simulates crops, livestock, and all other variables summarized in table 4.2 above. The land use and field size variables can be visualized via GIS. As we want to show how iterations are linked with each other, only the grassland variable is relevant as an iteration result at the moment.

In the 2nd iteration, land consolidation is assumed. If farmers reach full amalgamation of their land, they are able to increase field sizes. Under these new conditions, the field size bounds are relaxed, and each farmer just needs to have at least five fields[3] (due to crop rotation considerations). Furthermore, per ha labour coefficients are now based on labour requirements for fields, which have an average size of 5 ha (see Appendix H). Each farmer still maximizes the TGM through an optimal combination of crops and livestock production regarding their land, labour, stable, and capital endowments. The model structure stays the same except for the aforementioned

[3]Farm size divided by five.

changes in labour coefficients and field bounds. The amount of grassland per farm from the first iteration is transferred to the second iteration and fixed for the corresponding farm. These grassland constraints are now implemented at the farm level instead of the municipality level. The aim of the 2nd iteration is to model the optimal production of crops, livestock, and flowering strips per farm under the given grassland obligations. These iteration results are transferred to the next iteration.

The 3rd iteration has a fundamentally different model structure. It captures farm interrelations through land rentals within neighbourhoods. The aim is to simulate farm sizes based on the land use (cropping) results of the previous iteration. Therefore, cropping results are fixed, and changes in farm size are simulated by including a new variable. How this will be done is explained in more detail when we focus on the mathematical model formulations in section 4.2.3.1.

The aim of the 4th iteration is to simulate new farming decisions (crops, livestock, and flowering strips) based on the new farm sizes of the previous iteration. This iteration is similar to the 1st and 2nd iteration. It simulates final field sizes per farm. The upper bound on the field size variable is the same as in the 2nd iteration. However since farm sizes changed, field sizes might increase. Therefore, per ha labour requirements are based on labour coefficients that hypothesize fields of 10 ha (see Appendix H). Farms with grassland (from the 1st iteration) still have the fixed grassland proportions. If grassland farms quit the sector (in the 3rd iteration), their grassland is rented out to the neighbouring farm that increased its size at the expense of the abandoned farm.

The 1st, 2nd, and 4th iteration have similar model structures and operate as mixed integer linear programming models using the same mathematical formulations. Differences are the parameters mentioned above. The 3rd iteration, however, operates as a non-linear programming model due to an additional variable for the farm size simulation. In the following sections, these two model types are handled separately. A comprehensive list of variables and parameters is applied to both model types; variables are written in lower-case whereas parameters are written in upper-case.

4.2.2. Linear programming iterations

4.2.2.1. Objective function

Each farm increases its total gross margin (TGM) by producing an optimal combination of crops (including flowering strips) and livestock. Basically, our objective function is similar to linear programming models and takes into account revenues and variable costs of production (see also Henseler et al. (2009), Audsley et al. (2006), or Van Wenum et al. (2004)). Farmers can hire

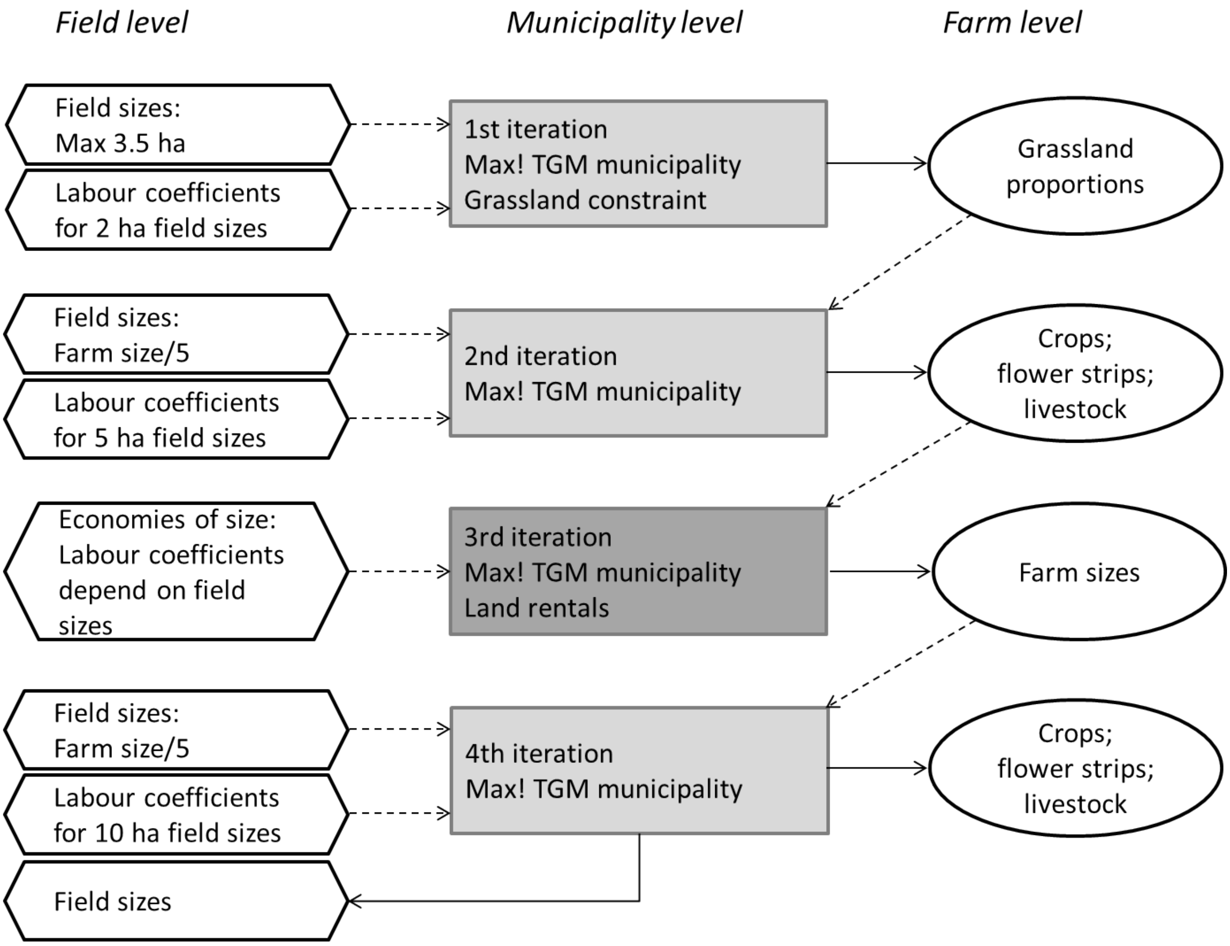

Figure 4.1.: Overview of the iteration procedures to simulate structural change processes at the field, farm, and municipality levels. Dotted arrows indicate information transfer and solid arrows indicate iteration results.

labour, which creates costs (see for example Júdez and Miguel (2002) or Lauber (2006)); however, it might pay off. On the other hand, off-farm wage increases the TGM if after farming a family labour force is left in order to work outside the farm. Moreover, each farmer has to pay land rents. They depend on the farm size[4] and are fixed cost factors. A farmer can also decide to rent out his or her own land (see also Lauber (2006) or Kantelhardt (2003)). He or she would do that, if the TGM of farming activities were negative or if the farm size reached a threshold where resources no longer sufficed to cultivate the whole land. However, the latter is not possible since the farm sizes stay the same within the 1st iteration. Renting out land would reduce land rent paid by farmers. The renting activity is implemented in order to prove whether the resources are allocated according to the needs of the farm sizes. If not, the farm would reduce farming under the given production conditions and the released land would be abandoned.

The objective function of this iteration is the sum of each farm's TGM (TGM_{mun}) and is written as follows:

$$
\begin{aligned}
TGM\_mun = \sum_{g,f,lu}(P_{lu} \cdot YIELD_{lu} + PREM_{lu} - VC_{lu})x_{g,f,lu} \\
+ \sum_{g,li}(P_{li} - VC_{li})y_{g,li} \\
+ \sum_{g,f}(P\_AES - VC\_AES + PREM\_AES)aes_{g,f} \\
- \sum_{g} hire\_lab_g \cdot LP \\
+ \sum_{g} off\_lab_g \cdot WOFF \\
- \sum_{g} RENTED\_LAND_g \cdot LRENT_g \\
+ \sum_{g,f} rentout\_land_{g,f} \cdot LRENT_g
\end{aligned}
\tag{4.1}
$$

where

- P_{lu} is the price vector for all cropping activities lu [EUR/ha].
- $YIELD_{lu}$ is the yield vector for all cropping activities lu [dt/ha].

[4]The amount of owned land vs rented land is taken from statistics (see section 3.2.10).

- $PREM_{lu}$ is the direct payments (DP) vector for all cropping activities lu [EUR/ha].
- VC_{lu} is the variable costs vector for all cropping activities lu [EUR/ha].
- $x_{g,f,lu}$ is the area of crop type produced per field f and farm g [ha].
- P_{li} is the price vector for livestock-keeping activities li [EUR/stable place and year].
- VC_{li} is the variable costs vector for livestock-keeping activities li [EUR/stable place and year].
- $y_{g,li}$ is the level of occupied livestock stable places per farm g.
- P_AES is the price for flowering strips [EUR/ha].
- VC_AES are the variable costs for flowering strips [EUR/ha].
- $PREM\_AES$ are the DP for flowering strips [EUR/ha].
- $aes_{g,f}$ is the size of each flowering strip per field f and farm g [ha].
- $hire\_lab_g$ is the level of hired labour per farm g [hours].
- LP is the labour price for hired farm workers [EUR/hour].
- off_lab_g is the level of off-farm labour per farm g [hours].
- $WOFF$ is the off-farm wage for work outside the farm [EUR/hour].
- $RENTED\_LAND_g$ is the amount of rented land for each farm g [ha].
- $rentout\_land_{g,f}$ is the level of land rented out per farm g [ha].
- $LRENT_g$ is the land rent for each farm g [EUR/ha].

4.2.2.2. Resource constraints

Farm resources consist of land, labour, capital, and stable place capacities. Each farm has a different amount of production that determines capital and stable place capacities. Family labour capacities, however, are the same for all farms.

For the land resources, two conditions need to be considered. First, only farm land is available for cropping activities and participation in AES in the form of flowering strips. The actual farm land can be reduced by the amount of land rented out (see also Lauber (2006) and Kantelhardt (2003)):

$$\begin{aligned} &\sum_{f,lu} LANDREQ_{lu} \cdot x_{g,f,lu} \\ &+ \sum_{f} LANDREQ\_AES \cdot aes_{g,f} \\ &\leq GWNNSIZE_g - \sum_{f} rentout\_land_{g,f} \end{aligned} \tag{4.2}$$

where

- $LANDREQ_{lu}$ is the amount of land required per ha cropping activity lu[5].
- $x_{g,f,lu}$ is the area of crop type produced per field f and farm g [ha].
- $LANDREQ\_AES$ is the amount of land required per ha flowering strip ($=1$).
- $aes_{g,f}$ is the size of flowering strip per field f and farm g [ha].
- $GWNNSIZE_g$ is the farm size for each farm g.
- $rentout\_land_{g,f}$ is the level of land rented out per farm g [ha].

Second, only owned land can be rented out:

$$\sum_{f} rentout\_land_{g,f} \leq OWN\_LAND_g \tag{4.3}$$

where

- OWN_LAND_g is the amount of owned land in ha for each farm g.

Farm labour resources are allocated to the cropping, livestock, and AES activities per farm. Hired labour increases and off-farm labour decreases the available labour stock (see also Júdez and Miguel (2002) or Lauber (2006)):

[5] Which is 1 for all cropping activities.

$$\sum_{f,lu} INPTREQ_{'labour',lu} \cdot x_{g,f,lu}$$
$$+\sum_{li} INPTREQ_{'labour',li} \cdot y_{g,li}$$
$$+\sum_{f} INPTREQ\_L\_AES \cdot aes_{g,f}$$
$$-hire\_lab_g + off\_lab_g \leq INPTCAP_{'labour',g} \tag{4.4}$$

where

- $INPTREQ_{'labour',lu}$ is the amount of labour required per ha cropping activity lu.
- $INPTREQ_{'labour',li}$ is the amount of labour required per year and livestock stable place li.
- $y_{g,li}$ is the level of occupied livestock stable places per farm g.
- $INPTREQ\_L\_AES$ is the amount of labour required per ha flowering strip.
- $hire\_lab_g$ is the level of hired labour per farm g [hours].
- off_lab_g is the level of off-farm labour per farm g [hours].
- $INPTCAP_{'labour',g}$ is the labour capacity available for each farm g.

However, the maximal amount of off-farm labour hours cannot exceed the family labour hours available per farm:

$$off\_lab_g \leq LABCAPFAM_g \tag{4.5}$$

where

- $LABCAPFAM_g$ is the amount of family labour hours available per farm g.

Another farm resource is the amount of financial capital, which is modeled similarly to the labour resources but with some small deviations:

$$\begin{aligned}&\sum_{f,lu} INPTREQ_{'capital',lu} \cdot x_{g,f,lu}\\&+\sum_{li} INPTREQ_{'capital',li} \cdot y_{g,li}\\&+\sum_{f} INPTREQ\_I\_AES \cdot aes_{g,f}\\&\leq INPTCAP_{'capital',g}\end{aligned} \tag{4.6}$$

where

- $INPTREQ_{'capital',lu}$ is the amount of capital required per ha cropping activity lu.
- $INPTREQ_{'capital',li}$ is the amount of capital required per year and livestock stable place li.
- $INPTREQ\_I\_AES$ is the amount of capital required per ha flowering strip.
- $INPTCAP_{'capital',g}$ is the capital capacity available for each farm g.

The constraint for the stable place capacity for pigs is specified as follows:

$$\sum_{li} INPTREQ_{'pigstableplace',li} \cdot y_{g,li} \leq INPTCAP_{'pigstableplace',g} \tag{4.7}$$

where

- $INPTREQ_{'pigstableplace',li}$ is the amount of pig stable places required per livestock activity li.
- $INPTCAP_{'pigstableplace',g}$ is the amount of pig stable places available within each farm g.

A similar restriction is implemented for dairy farming.

4.2.2.3. Biophysical and legal constraints

Due to biophysical constraints, farmers have some natural production limits that need to be captured by the model as well. This is typically done in programming models, and the way of implementing does not deviate from many other models (see for example Kantelhardt (2003) or Kächele (1999)). Potatoes, for example, cannot be planted on the same field for two or three years in a row without suffering yield reductions. It is recommended to wait at least four years until the next seeding. This constraint for potatoes is translated as follows:

$$\sum_{f} x_{g,f,'potato'} \leq GWNNSIZE_g \cdot 0.25 \tag{4.8}$$

where

- $x_{g,f,'potato'}$ is the area of potato produced per field f and farm g [ha].
- $GWNNSIZE_g$ is the size of each farm g [ha].

The same applies to sugar beets and rapeseed. Maize fields need at least one crop rotation after three planting periods. The same applies to a combination of halm crops. This is mathematically translated into less than 75% maize or halm crops per season. Wheat is especially sensitive if planted two years in a row. The model assumes that the yield of stubble wheat[6] decreases by 10%. Therefore, another wheat activity (stubble wheat) is included, and the (first) wheat activity is restricted to 50%.

Biophysical restrictions all apply to each farm, whereas legal constraints are implemented on the municipality level. Potatoes and sugar beets are traded under contracts, and thus the observed production level is assumed. For potatoes, this is 2.3% which translates into:

$$\sum_{g,f} x_{g,f,'potato'} \leq \sum_{g} GWNNSIZE_g \cdot 0.023 \tag{4.9}$$

The observed level of produced sugar beets in the calibration year is 16.4%. As described in 3.2.9, silo maize production for biogas plants covers around 1% of the study area. Participation in AES is restricted to the available budget within Wöllstadt as mentioned in 3.2.8. Legal obligations restrict the conversion of permanent grassland into arable fields. The amount of grassland, which is 5%, is restricted to at least the observed level. Farmers are allowed to convert arable land into grassland.

[6]The second wheat cultivation at the same location.

4.2.2.4. Livestock keeping

Livestock-keeping farms are endowed with a certain amount of stable places (see section 3.2.3). In the model, the stable place capacity of pig-fattening farms needs to be reached with at least 95%:

$$y_{m,'pig-fattening'} \geq INPTCAP_{'pigstableplace',m} \cdot 0.95 \tag{4.10}$$

where

- $y_{m,'pig-fattening'}$ is the level of occupied pig stable places per plant and pig-fattening farm m.
- $INPTCAP_{'pigstableplace',m}$ is the amount of pig stable places available per plant and pig-fattening farm m.

For dairy, at least 50% of the stable place capacity needs to be occupied. The maximum level of livestock keeping is the full occupation of the stable place capacity of each farm. Since silo maize (fodder) is only fed to dairy and not to pigs and since the model assumes that silo maize silage is fully used as fodder within the dairy farm, there is another boundary condition implemented:

$$\sum_{f} x_{d,f,'silomaize'} \cdot YIELD_{'silomaize'} \leq feedtrans_{d,'silomaize'} \tag{4.11}$$

where

- $x_{d,f,'silomaize'}$ is the area of silo maize production [ha] per plant and dairy-producing farm d.
- $YIELD_{'silomaize'}$ is the yield for silo maize [dt/ha].
- $feedtrans_{d,'silomaize'}$ is the amount of silo maize [dt] fed to dairy per plant and pig-fattening farm d.

Additionally, silo maize production of the other farm types is forced to zero. The very same applies to grass production, which is only fed to dairy as grass silage.

Livestock feeding practices are further modeled through several other equations. Our model formulations are aligned to Schönhart et al. (2011) and constitute the following equations.

In general, there is always a feed balance to make sure that the transferred fodder crops deliver the required nutrient contents for the produced amount of livestock. For pigs, the required nutrients are energy and protein, and for dairy, the dry matter (DM) content. According to GfE (2006), energy requirements of pigs are around 10.4 GJ per year and stable place. The protein requirements are 78.5 kg per year and pig stable place, which are partly delivered by the grain ratio in the fodder. The average nutrition contents of the fodder crops in the model are taken from LfL (2017) and are summarized in Appendix F. For cows, one important parameter is the feed amount in dry matter, which is 60.6 dt per year and stable place[7]. The following example shows the feed balance for dairy.

$$\sum_{fc} feedtrans_{d,fc} \cdot NUTCAPCROPS_{fc,'drymatter'} \geq NUTREQ_{'dairy','drymatter'} \cdot y_{d,'dairy'} \tag{4.12}$$

where

- $feedtrans_{d,fc}$ is the amount of fodder crops fc [dt] fed to dairy per plant and dairy-producing farm d.
- $NUTCAPCROPS_{fc,'drymatter'}$ is the DM content of fodder crops fc [kg DM/kg fresh matter].
- $NUTREQ_{'dairy','drymatter'}$ is the DM requirement per dairy stable place [dt/year].
- $y_{d,'dairy'}$ is the level of occupied dairy stable places per dairy-producing farm d.

A production balance makes sure, that the amount of fodder crops is produced by the same farm and thus available for feeding purposes. The following example shows the production balance for plant and pig-fattening farms and is similar to the livestock feed balances of Schönhart et al. (2011). A similar equation is implemented in the model for dairy-producing farms.

$$-\sum_{f} x_{m,f,fc} \cdot YIELD_{fc} + feedtrans_{m,fc} \leq 0 \tag{4.13}$$

where

[7] This is a recommendation of the GfE (2001) for dairy with 650 live weight giving 20 kg milk per day.

- $x_{m,f,fc}$ is the area of fodder crops fc produced per field f and per pig-fattening farm m [ha].
- $YIELD_{fc}$ is the yield of fodder crops fc [dt/ha].
- $feedtrans_{m,fc}$ is the amount of fodder crops fc fed to pigs per pig-fattening farm m [dt].

In order to reach certain proportions of fodder crops within the fodder ratio, further equations are needed. According to GfE (2006), 10.6% of the energy requirement needs to be delivered by wheat but must be smaller than 50%. In the following exemplary equation, the energy requirement per occupied pig stable place needs to be delivered by at least 26.7% barley.

$$\begin{aligned} &feedtrans_{m,'barley'} \cdot NUTCAPCROPS_{'barley','energy'} \\ &\qquad \geq NUTREQ_{'pig-fattening','energy'} \cdot y_{m,'pig-fattening'} \cdot 0.267 \end{aligned} \tag{4.14}$$

where

- $feedtrans_{m,'barley'}$ is the amount of barley fed to pigs per plant and pig-fattening farm m [dt].
- $NUTCAPCROPS_{'barley','energy'}$ is the energy content of barley [GJ/dt fresh matter].
- $NUTREQ_{'pig-fattening','energy'}$ is the energy requirement per pig stable place [GJ/year].
- $y_{m,'pig-fattening'}$ is the level of occupied pig stable places per plant and pig-fattening farm m.

The model assumes that the dry matter requirement is completely delivered by the farm's own crop production. The total permanent grassland yields (silage or hay) need to be used first as fodder. For dairy, 50-60% of the total DM requirement needs to be delivered by silage coming from grasslands, maize, or grasses from arable fields. However, the maximum amount of the latter is 10% of the required DM since we assume a maize dominated fodder ratio due to statistical information on land use. The maximum amount of grain in the dairy fodder needs to be lower than 6% with respect to the energy requirements (Meyer, 2005). Additionally, the RNB needs to be around zero (LfL, 2017, pp. 13), which is implemented in the model as being higher than -5 and lower than 5:

$$\sum_{fc} feedtrans_{d,fc} \cdot NUTCAPCROPS_{fc,'RNB'} \geq -5 \tag{4.15}$$

and

$$\sum_{fc} feedtrans_{d,fc} \cdot NUTCAPCROPS_{fc,'RNB'} \leq 5 \tag{4.16}$$

where

- $feedtrans_{d,fc}$ is the amount of fodder crops fed to dairy per plant and dairy-producing farm d [dt].
- $NUTCAPCROPS_{fc,'RNB'}$ is the RNB contribution per fodder crop.

With the following equation, the model makes sure that the total amount of hay and grass coming from permanent grassland is used as fodder:

$$\sum_{f} x_{d,f,'grasssilage'} \cdot YIELD_{'grasssilage'} + x_{d,f,'hay'} \cdot YIELD_{'hay'} \geq feedtrans_{d,'grasssilage'} + feedtrans_{d,'hay'} \tag{4.17}$$

where

- $x_{d,f,'grasssilage'}$ is the area of grass silage produced per field f by plant and dairy-producing farm d [ha].
- $YIELD_{'grasssilage'}$ is the yield for grass silage coming from permanent grassland [dt/ha].
- $x_{d,f,'hay'}$ is the area of hay produced per field f by plant and dairy-producing farm d [ha].
- $YIELD_{'hay'}$ is the yield for hay coming from permanent grassland [dt/ha].
- $feedtrans_{d,'grasssilage'}$ is the amount of grass silage fed to dairy per plant and dairy-producing farm d [dt].
- $feedtrans_{d,'hay'}$ is the amount of hay fed to dairy per plant and dairy-producing farm d [dt].

4.2.2.5. Grassland constraints

As described in the previous section, we have four iterations. The 1st iteration allocates the amount of grassland to the agricultural farms of the municipality. The assumption behind is that due to path dependencies and competitive pressure, permanent grassland is distributed in an optimal manner over time.

Within the 2nd iteration, the model does not allow for changes in the grassland proportions. The grassland area per farm from the 1st iteration is transferred to the 2nd iteration and allocated to the respective farms. This means that farmers can change the location of their grassland but not the total area under cultivation. In doing so, a new set for farms having grassland is introduced: $glfarms$. This ranges from one to fourteen[8], but similar to the fields, each element does not need to have a value. A maximum of fourteen farms can have grassland. Afterward, for each grassland farm simulated by the 1st iteration, a constraint is included in the 2nd model iteration, which fixes the amount of grassland. This constraint includes the proportion of permanent grassland, which is allocated to the new set and transferred as minimal amounts to the respective farm. The mathematical formulation for one of four farms[9] having grassland is:

$$\sum_f x_{'farm9',f,'grasssilage'} + x_{'farm9',f,'hay'} \geq GRASSLAND_{'glfarm1'} \tag{4.18}$$

where

- $x_{'farm9',f,'grasssilage'}$ is the area of grass silage produced per field f by farm nine [ha].
- $x_{'farm9',f,'hay'}$ is the area of hay produced per field f by farm nine [ha].
- $GRASSLAND_{'glfarm1'}$ is the area of permanent grassland for grassland farm one $glfarm1$ calculated by the 1st iteration [ha].

This kind of constraint is adapted to all other grassland farms. For those farms it is possible to cultivate more permanent grassland; however, they are not allowed to cultivate less.

After rearranging farm land within the farm size optimization (3rd iteration), some farmers quit farming and hand over the entire land to the neighbouring farm. In the case of a grassland farm, the grassland proportion is then also transferred to the neighbouring farm. This is modeled in the 4th iteration. For the example above, the grassland farm one ($'glfarm1'$), which is farm nine ($'farm9'$) quits farming and hands over its land to farm ten ($'farm10'$). Therefore, the grassland proportion is also transferred:

$$\sum_f x_{'farm10',f,'grasssilage'} + x_{'farm10',f,'hay'} \geq GRASSLAND_{'glfarm1'} \tag{4.19}$$

[8] Arbitrary number; in several model runs, this number of farms having grassland was never reached.
[9] Within the 1st iteration, the model generates four farms with grassland (see Appendix M).

where

- $x_{'farm10',f,'grasssilage'}$ is the area of grass silage produced per field f by farm ten [ha].
- $x_{'farm10',f,'hay'}$ is the area of hay produced per field f by farm ten [ha].
- $GRASSLAND_{'glfarm1'}$ is the area of permanent grassland for grassland farm one $glfarm1$ calculated by the 1st iteration [ha].

4.2.2.6. Equality constraints

As mentioned in section 4.2, $x_{g,f,lu}$ is a SOS1 variable and therefore needs to fulfill a certain condition. In this case, the SOS1 application makes sure that each field can have only one land use type. For the variable, it is necessary to define the amount to which $x_{g,f,lu}$ adds up. The possibility of planting flowering strips along fields f per farm g needs to be considered as well:

$$\sum_{lu} x_{g,f,lu} + aes_{g,f} = fieldsize_{g,f} \tag{4.20}$$

where

- $x_{g,f,lu}$ is the area of crop type produced per field f and farm g [ha].
- $aes_{g,f}$ is the size of flowering strip per field f and farm g [ha].
- $fieldsize_{g,f}$ is the size of each field f cultivated by farm g [ha].

Furthermore, the sum of a farm's field sizes equals the total size of the farm:

$$\sum_{f} fieldsize_{g,f} = GWNNSIZE_g \tag{4.21}$$

where

- $GWNNSIZE_g$ is the size of each farm g [ha].

With a lower bound on $aes\_width_{g,f}$, the model forces flowering strips to be at least five meters wide. The width of flowering strips is defined as follows:

$$aes\_width_{g,f} = aes_{g,f} \cdot 10,000 \div FIELDLENGTH_g \tag{4.22}$$

where

- $aes\_width_{g,f}$ is the width of flowering strips per field f and farm g [m].
- $FIELDLENGTH_g$ is the length of each farm g [m][10].

4.2.3. Non-linear programming iteration

Non-linear programming models are similar to linear ones in that they are composed of an objective function and several resource constraints. However, non-linear programming models have at least one nonlinear function. The optimization problem at hand has some nonlinear constraints related to farm labour and due to the consideration of economies of size (EoS). In order to solve the non-linear model with GAMS, the CONOPT solver was used. Before elaborating on the mathematical model formulations, an overview of sets and variables is given.

The sets of the model are the same as in the linear model above as summarized in table 4.1, although not all of them are used. Sets comprising livestock feeding practices are omitted since the amount of livestock and the feeding patterns are calculated in the linear model part and kept as a constant parameter during the nonlinear model. The same applies to flowering strip activities as well as to the proportions of cropping activities.

In the non-linear programming part of the model run, the total gross margin TGM_{mun} of all farms is calculated and maximized as an objective function. This is a typical exercise of regional farm models (see, for example Balmann (1997); Dabbert et al. (1999)). The difference to our linear programming model is that farms are assumed to be spatially interrelated and able to lease out their land resources. With this model specification, the long-term perspective and the issue of farm succession is indicated. In the longer run, farmers having a successor perform better than others and have a higher probability of keeping the agricultural production going (Van Passel et al., 2007).

The land exchanging patterns, though, are strongly linked to spatial issues. We assume that only neighbouring farms lease land to each other[11]. Otherwise they couldn't exploit EoS to a full extent. By including a land exchanging module, the model can optimize each farm's size under the condition to rent in and out. The non-linear iteration also includes variables for off-farm work (off_lab_g) and hiring labour ($hire\_lab_g$) for each farm as in the linear model. Farm sizes are optimized regarding labour requirements for each crop ($labreq_{g,f}$), which depend on the field size.

[10]The length of each farm is assumed to be a fixed parameter as documented in Appendix C and is deducted from spatial simplification (see section 2.4.2.3).

[11]According to Strohm (1998), neighbouring relations positively influence land leases.

The assumption behind this is that field and farm sizes are linearly linked to each other. Increases in farm sizes lead to increases in field sizes. This is a common assumption, also made by Brady et al. (2012).

4.2.3.1. Economies of size

In order to explicitly account for EoS, labour requirements need to be considered in a more specific manner. We show how this could be done.

Labour requirements, as well as variable machine costs for a farmer depend on the farm's field sizes. For each cropping activity[12], Ktbl data on permanent and seasonal labour in hours per ha and variable machine costs in Euros per ha were used. They apply to several field sizes and crop types. This database contains comprehensive information on labour requirement and machine costs for one, five, ten, and twenty ha. Based on these data, proximity functions for variable machine costs and labour requirements per ha were calculated for each crop. The relationships have been linear with R^2-values between 0.97 and 0.99. The slope values of the linear functions were taken as machine cost/labour coefficients, which were later included in the objective function. All machine cost/labour coefficients are listed in Appendix I.1.

Afterward, proximity functions for the amount of labour hours per field size were calculated for each crop. These logarithmic functions have R^2-values between 0.86 and 0.92[13]. Hereafter and as suggested by Nuppenau and Helmer (2007), an approximation was applied to each logarithmic function in order to receive broken rational functions around a certain field size level. According to statistical and spatial information we used an average of 3 ha for our case study. The resulting functions per arable crop were included in the model to calculate labour requirements for each crop and field. In order to allow for changes in farm sizes (and corresponding to that in field sizes), a farm size growth factor was included as a variable ($gwnn\_size_g$). The following example is for winter wheat[14]:

$$labreq\_wheat_{g,f} \geq 5.67154367 + 5.499 \div Xfix_{g,f,'wheat'} \cdot gwnn\_size_g \tag{4.23}$$

where

[12] Flowering strips are assumed to be independent from the farm size. As flowering strips are commonly planted along the field length, economies of size are unlikely to be considered by the farmer here.
[13] Logarithmic functions are listed in Appendix I.2.
[14] All other equations for the respective crop are listed in Appendix J.1.

- $labreq\_wheat_{g,f}$ is the labour requirement for wheat per farm and field [hours/ha].
- $Xfix_{g,f,'wheat'}$ is the area of wheat production for each farm and field [ha].
- $gwnn\_size_g$ is the growth factor per farm g.

Since for several fields the area of wheat production $Xfix_{g,f,'wheat'}$ is zero and division by zero is mathematically impossible, a condition for wheat production being greater than zero was included. This was made for every other equation calculating the labour requirement for a certain crop.

$Gwnn\_size_g$ values that are smaller than one indicate a reduction in the farm size. Values greater than one increase the size of the respective farm. In the objective function, this variable is multiplied by the fixed land use proportions $Xfix_{g,f,lu}$. Gross margins change due to increases or decreases in farm sizes. If farm sizes increase, EoS are exploited since less labour input is needed for bigger fields.

4.2.3.2. Objective function

The objective function maximizes the TGM at the municipality level TGM_mun, assuming land transferability among farms in order to account for the long-term perspective of the model. For TGM calculation, the costs are subtracted from the revenues.

As outlined in equation 4.24, the revenue consists of income from cropping activities ($Xfix_{g,f,lu}$) and livestock activities ($Yfix_{g,li}$), as well as from participation in AES ($AESfix_{g,f}$). Those are now fixed parameters adopted from the former model iteration in which they were treated as variables. Fixed land use proportions $Xfix_{g,f,lu}$ are multiplied by $gwnn\_size_g$ in order to address the monetary revenues of exploiting EoS. $Gwnn\_size_g$ values greater than one increase the TGM, and values lower than one decrease it. Off-farm income off_lab_g also promotes TGMs. As in the linear programing model, for each land use activity, a governmental payment is included. For alternate political incentive schemes, it is divided into two payment components as will be explained in the next section

On the other hand, variable costs of cropping activities, hiring labour, and renting land reduce the TGM. In contrast to the objective function of the linear programming model(s), variable costs for cropping activities are classified into direct (DC), service (SC), and machine costs (MC).

The reason for this division is that machine costs are assumed to correlate negatively with field sizes. Field sizes change due to farm size changes, whereas the other cost components are linearly positively correlated with the farm size. This means that direct and service costs increase (decrease) in the same shape as an increase (decrease) in farm size. Marginal machine costs, however, get smaller if fields increase due to reduced labour requirements. Therefore, machine cost coefficients as a function of labour requirements were calculated (see previous subsection) and multiplied by the cropping activity, the farm size factor, and the labour requirement $labreq_{g,f}$ of the respective cropping activity. The labour requirement itself depends on the field size and was calculated separately (see also subsection above).

The objective function below needed to be re-formulated and disaggregated due to the labour requirement variables, which must be calculated separately for each crop type. The dots [...] mark all similar terms for each cropping activity.

$$
\begin{aligned}
TGM\_mun = \sum_{g,f}&(P_{lu} \cdot YIELD_{lu} + PREM_{lu} - DC_{lu} - SC_{lu}) Xfix_{g,f,lu} \cdot gwnn\_size_g \\
&-[\sum_{g,f} labreq\_wheat_{g,f} \cdot Xfix_{g,f,'wheat'} \cdot gwnn\_size_g \cdot MC_{'wheat'}] - [\ldots] \\
&- \sum_{g,f} labreq\_potato\_seas_{g,f} \cdot Xfix_{g,f,'potato'} \cdot gwnn\_size_g \cdot SLP \\
&+ \sum_{g,f} (P\_AES - VC\_AES + PREM\_AES) AESfix_{g,f} \\
&+ \sum_{g,li} (P_{li} - VC_{li}) Yfix_{g,li} \\
&- \sum_{g} hire\_lab_g \cdot LP \\
&+ \sum_{g} off\_lab_g \cdot WOFF \\
&- \sum_{g} (GWNNSIZE_g \cdot gwnn\_size_g - OWN\_LAND_g) \cdot LRENT_g
\end{aligned}
\tag{4.24}
$$

where

- P_{lu} is the price vector for all cropping activities lu [EUR/ha].
- $YIELD_{lu}$ is the yield vector for all cropping activities lu [dt/ha].
- $PREM_{lu}$ are the DP for the area of all cropping activities lu [EUR/ha].
- DC_{lu} is the direct costs vector for all cropping activities lu [EUR/ha].

- SC_{lu} is the service costs vector for all cropping activities lu [EUR/ha].
- $Xfix_{g,f,lu}$ is the area of crop production for each crop type, field, and farm [ha].
- $gwnn\_size_g$ is the growth factor per farm g.
- $labreq\_wheat_{g,f}$ is the labour requirement for wheat per farm and field [hours/ha].
- $MC_{'wheat'}$ are the machine costs for wheat [EUR/ha].
- $labreq\_potato\_seas_{g,f}$ is the seasonal labour requirement for potatoes per farm and field [hours/ha].
- $Xfix_{g,f,'potato'}$ is the area of potato production for each field and farm [ha].
- SLP is the seasonal labour price for hired farm workers [EUR/hour].
- P_AES is the price for flowering strips [EUR/ha].
- VC_AES are the variable costs for flowering strips [EUR/ha].
- $PREM\_AES$ are the direct payments for flowering strips [EUR/ha].
- $AESfix_{g,f}$ is the area of flowering strips for each field and farm [ha].
- P_{li} is the price for livestock-keeping activities li [EUR/stable place and year].
- VC_{li} are the variable costs for livestock-keeping activities li [EUR/stable place and year].
- $Yfix_{g,li}$ is the level of livestock-keeping activity per farm [stable places].
- $hire\_lab_g$ is the amount of hired labour per farm [hours].
- off_lab_g is the amount of off-farm labour per farm [hours].
- LP is the labour price for hired farm workers [EUR/hour].
- $WOFF$ is the off-farm wage for work outside the farm [EUR/hour].
- $GWNNSIZE_g$ is the farm size per farm [ha].
- OWN_LAND_g is the amount of owned land per farm g [ha].
- $LRENT_g$ is the land rent per farm g [EUR/ha].

4.2.3.3. Resource constraints

The resources for farms in this model iteration consist of land and labour. Capital requirements shrink due to EoS. They play no important role as resource restrictions[15]. However, after farm

[15]Since we do not model investment decisions.

enlargement, it might be that capital no longer suffices. Therefore, in the followed model iteration, the capital capacities are adapted to new farm sizes according to table 3.4. This 4th and last model iteration is again a linear model calculating the final farming activities. At the same time, this validation step allows us to prove whether the new farm land is actually used by farmers. Husbandry capacities (as other resource constraints) were not included in the nonlinear programming iteration since livestock activities are assumed to stay the same and be independent from the farm size. In order to increase stable place capacities of livestock farms, investments are required. However, as mentioned above, investments are not the focus of this study, so they are not explicitly modeled.

In contrast to capital and husbandry capacities, land constraints are obviously needed. The sum of all cropping and AES activities of the former model step are restricted to the minimum amount of land available on the municipality level forcing the whole agricultural land to be used[16]. Cropping activities are multiplied with the farm growth factor $gwnn\_size_g$:

$$\begin{aligned} &\sum_{g,f,lu} LANDREQ_{lu} \cdot Xfix_{g,f,lu} \cdot gwnn\_size_g \\ &+ \sum_{g,f} LANDREQ\_AES \cdot AESfix_{g,f} \geq \sum_{g} GWNNSIZE_g \end{aligned} \tag{4.25}$$

where

- $LANDREQ_{lu}$ is the amount of land required per ha cropping activity lu[17].
- $Xfix_{g,f,lu}$ is the area of crop production for each field and farm [ha].
- $gwnn\_size_g$ is the growth factor per farm g.
- $LANDREQ\_AES$ is the amount of land required per ha flowering strip ($= 1$).
- $AESfix_{g,f}$ is the area of flowering strip for each field and farm [ha].
- $GWNNSIZE_g$ is the farm size for each farm g.

Moreover, labour constraints are required when modeling EoS. The calculated labour requirement for each crop type is multiplied by the fixed cropping activity level and the farm growth factor at the farm level. Labour requirements for livestock and AES activities need to be considered, as well as hired and off-farm labour. Therefore, a labour market is implemented as in the linear

[16]In the next iteration, upper limits on land are used in order to check if the whole farm land is used, rented out, or left as fallow.
[17]Which is 1 for all cropping activities.

modeling iterations based on proposals of Júdez and Miguel (2002). The equation is implemented as follows, where the [...] mark all similar terms for each cropping activity.

$$\sum_f [labreq\_wheat_{g,f} \cdot Xfix_{g,f,'wheat'} \cdot gwnn\_size_g] + [\ldots] + \sum_f AESfix_{g,f} \cdot INPUTREQ\_L\_AES + \sum_{li} INPUTREQ_{'labour',li} \cdot Yfix_{g,li} - hire\_lab_g + off\_lab_g \leq LABCAPFAM_g \tag{4.26}$$

where

- $labreq\_wheat_{g,f}$ is the amount of labour required per ha wheat production [hours/ha].
- $Xfix_{g,f,'wheat'}$ is the area of wheat production for each field and farm [ha].
- $gwnn\_size_g$ is the growth factor per farm g.
- $AESfix_{g,f}$ is the area of flowering strip for each field and farm [ha].
- $INPUTREQ\_L\_AES$ is the labour requirement per ha flowering strip [hours].
- $INPUTREQ_{'labour',li}$ is the labour requirement per occupied livestock stable place li [hours/stable place].
- $Yfix_{g,li}$ is the amount of occupied stable place per livestock-keeping activity li.
- $hire\_lab_g$ is the level of hired labour per farm g [hours].
- off_lab_g is the level of off-farm labour per farm g [hours].
- $LABCAPFAM_g$ is the labour capacity per farm g [hours].

Labour requirements per farm need to be less than or equal to the available amount of family labour force plus the amount of hired labour. The amount of off-farm work cannot exceed the available amount of family labour, which is modeled with the following equation:

$$off\_lab_g \leq LABCAPFAM_g \tag{4.27}$$

where

- off_lab_g is the level of off-farm labour per farm g [hours].

4.2.3.4. Land rentals and neighboring effects

Modeling land rentals is strongly related to spatial dimensions of the model. Depending on the possibility of cultivating fields located next to their own fields, farmers can enlarge the cultivation area at the expense of the neighbouring farm in order to gain from EoS. This enlargement takes place at the farm level, and ignores the current field structures of a farm. Some exceptions prevail; only two neighbouring farms are capable of leasing or renting land to each other, although they might have two farm neighbours. There are two farms modeled that have two neighbours with the possibility to lease land. This exception stems from the spatial locations of the farms. Each farm needs at least one neighbour (with the same field length). When looking at the rectangular stylized farms in Appendix A, this condition only holds true if farm 12 and 17 have two farm neighbours each. The following equation shows an example of a typical pairwise land change constraint:

$$\begin{aligned} &GWNNSIZE_{'farm1'} \cdot gwnn\_size_{'farm1'} - GWNNSIZE_{'farm1'} \\ &\quad \leq -(GWNNSIZE_{'farm2'} \cdot gwnn\_size_{'farm2'} - GWNNSIZE_{'farm2'}) \end{aligned} \tag{4.28}$$

where

- $GWNNSIZE_{'farm1'}$ is the farm size of farm 1 [ha].
- $gwnn\_size_{'farm1'}$ is the growth factor of farm 1.
- $GWNNSIZE_{'farm2'}$ is the farm size of farm 2 [ha].
- $gwnn\_size_{'farm2'}$ is the growth factor of farm 2.

This equation ensures that the farm size change of farm 1 compensates the farm size change in farm 2. The same applies to farms 3 and 4, 5 and 6, 7 and 8, 9 and 10, 14 and 15, 19 and 20, 21 and 22, and 23 and 24 (see Appendix A).

Farm 12 and 17 have two neighbours to which they are capable of leasing land (farms 11 and 13 and farms 16 and 18, respectively). In these cases, the following two equations are needed to make sure that the lease of land among the three farms in line happens correctly:

$$
\begin{aligned}
&(GWNNSIZE_{'farm11'} \cdot gwnn\_size_{'farm11'} - GWNNSIZE_{'farm11'}) + \\
&(GWNNSIZE_{'farm12'} \cdot gwnn\_size_{'farm12'} - GWNNSIZE_{'farm12'}) + \\
&\qquad (GWNNSIZE_{'farm13'} \cdot gwnn\_size_{'farm13'} - GWNNSIZE_{'farm13'}) \leq 0
\end{aligned} \tag{4.29}
$$

and

$$
\begin{aligned}
&GWNNSIZE_{'farm11'} + GWNNSIZE_{'farm12'} + GWNNSIZE_{'farm13'} \geq \\
&\qquad GWNNSIZE_{'farm11'} \cdot gwnn\_size_{'farm11'} + \\
&\qquad GWNNSIZE_{'farm12'} \cdot gwnn\_size_{'farm12'} + \\
&\qquad\qquad GWNNSIZE_{'farm13'} \cdot gwnn\_size_{'farm13'}
\end{aligned} \tag{4.30}
$$

where

- $GWNNSIZE_{'farm11'}$ is the farm size of farm 11 [ha].
- $gwnn\_size_{'farm11'}$ is the growth factor of farm 11.
- $GWNNSIZE_{'farm12'}$ is the farm size of farm 12 [ha].
- $gwnn\_size_{'farm12'}$ is the growth factor of farm 12.
- $GWNNSIZE_{'farm13'}$ is the farm size of farm 13 [ha].
- $gwnn\_size_{'farm13'}$ is the growth factor of farm 13.

4.2.4. Alternate political incentive schemes

In the baseline model described above, CAP reform 2003 is implemented according to the legal conditions of the calibration year 2011. The model includes DP per acre and restrictions on permanent grassland conversions. The budget for AES is taken from public data [18] for the year 2014 and is set as an upper limit. Altogether, governmental payments of our case study total around 342,000 Euros.

[18] Available online due to the regulation from 10.12.2008 (eBAnz. 2008, AT147 V1).

The aim of the study is to test the greening component of the new CAP. It ought to contribute to the biodiversity targets of the EC (European Commission, 2013). In our study, we develop three different alternate political incentive schemes. Two of these alternate policy designs are oriented towards the currently implemented new CAP reform and therefore are closely related to the 'real world' situation. In the following, they are called CAP I and CAP II. The other political incentive scheme contains elements of the 'refocus' scenario of the European Commission (2011) as well as newer propositions of the 'Naturschutzbund' (NABU) promoting the abolition of direct payments (DP). We call it the 'nature-focused' scheme. Detailed political framework conditions of the baseline model and the three alternate political incentive schemes (policy designs) are summarized in table 4.3.

In CAP I we try to model the greening architecture of the current CAP. This includes a separation of the DP into two components: the regulatory cross compliance and the mandatory greening financial support[19]. In the model, the greening support is coupled with a designation of 5% ecological focus areas (EFAs) per farm. This can be achieved by farmers through legume cultivation, hedges, or fallow land. Each of these activities are balanced due to their weighting factor as fixed by the European Parliament. For legumes, this is 0.7, for fallow land 1, and the weighting factor for hedges is 2[20]. Additionally, the model accounts for crop diversification as it is obligatory in the new CAP reform. It is not allowed to convert permanent grassland into arable land. As another feature of the new CAP, redistributive payments to support small and medium-sized farms are attributed to the first 30 ha and the following 16 ha farm land. This is taken into account as well. According to BMEL (2015), second pillar payments of the new CAP for Hesse add up to 651 million Euro. This is 9% lower than for the former reform period (HMUKLV, 2011). Due to that, the budget for AES in CAP I is reduced by this amount compared to the baseline model. In CAP I, bioenergy production will probably increase by the end of 2020 due to the ambitious German energy targets. Therefore, the capacity of biogas plantations (which is relatively low in the study area) is assumed to double. In summary, governmental expenditures are higher compared to the baseline model with around 372,000 Euros for the whole study site[21].

CAP II contains the same political framework conditions as the CAP I scenario but with stricter greening requirements. The EFAs increase up to 7% by the end of the year 2018 (European

[19]Both payments will be adjusted to a nationwide limit by the end of the reform period in 2020 and will reach a level of 176 EUR/ha (cross compliance) or 87 EUR/ha (greening).
[20]Annex X of the supplementing Regulation (EU) No 1307/2013.
[21]This arises out of the redistributive payments since DP are lower and the budget for AES.

Commission, 2013). On top of that, crop diversification rules are tightened in such a way that the production area of the two main crops needs to be lower than 75%. The total governmental payments are the same as in CAP I.

The 'nature-focused' scheme covers some main aspects of a new study that was conducted by Oppermann et al. (2016). In this study, a complete new payment structure is proposed, albeit the governmental expenses stay the same as in the first period of the new CAP. The main changes are the abolition of DP and the implementation of an incentive-based AES scheme, which can be seen as payment reallocation from the first into the second pillar. The authors conclude that this would lead to a better achievement of the European biodiversity strategy and to a 'fairer' treatment of tax-payers since all payments are linked to environmentally friendly measures of the farmers (Oppermann et al., 2016). The reform stipulates 10% EFA and a restriction on the conversion of permanent grassland area for each farm in order to get a sustainability premium (SP) of 150 Euros per ha. For our model we assume 7% EFA, the prohibition of permanent grassland conversion into arable land, and crop diversification as modeled in CAP II in order to get the SP. To simplify, the other part of the EC budget[22] is used for AES[23], which are remunerated through the so-called agri-nature premium (ANP) (as suggested by the authors). For these ANPs, a much higher amount of money (1,350 EUR/ha) is paid in order to make participation in environmentally friendly schemes more attractive for farmers. The budget for ANPs in the 'nature-focused' scheme is calculated from the total GAP 2020 budget minus the SP paid for the whole cropping area of the municipality. These two payment schemes both contribute approximately the same quantity to the total governmental budget of the municipality.

Oppermann et al. (2016, Tab. 5, page 33) propose four agri-environmental measures for arable land:

- extensively produced grain crops (no pesticides and higher distance between rows).
- flowering fields.
- fallow.
- buffer strips along hedges, water bodies, and forests.

[22]Which is the same for CAP I and CAP II.
[23]The payment structure of the NABU study is staggered depending on the participation in ecologically valuable measurements.

Table 4.3.: Policy designs with respective parameter dimensions.

	Baseline	CAP I	CAP II	Nature-focused
Payment scheme	- Mostly DP - Low budget for AES (0.3%)	- 30% of DP coupled with greening conditions - Lower budget for AES (0.2%)	- 30% of DP coupled with greening conditions - Lower budget for AES (0.2%)	- SP coupled with greening conditions - Remaining budget for ANP
Ecological focus areas	no	5%	7%	7%
Crop diversification	no	Two main crops $\leq$ 95% of the cropping area	Two main crops $\leq$ 75% of the cropping area	Two main crops $\leq$ 75% of the cropping area
Small-sized farm payment	no	yes	yes	no
Grassland conversion	no	no	no	no
Energy	Silo maize for biogas covers $\leq$ 1% of the area	Silo maize for biogas covers $\leq$ 2% of the area	Silo maize for biogas covers $\leq$ 2% of the area	Silo maize for biogas covers $\leq$ 2% of the area

To cover the ANP schemes, the model is extended so that in addition to flowering strips, entire flowering fields can be managed according to ANP conditions. The maximum utilization level for these kinds of measurements are 25% of the arable land per farm. In the NABU study, as well as in the 'nature-focused' scheme, redistributive payments are not taken into account within the new payment scheme. Energy targets remain the same as in CAP I and CAP II above.

4.2.4.1. Mathematical implementation

Initial efforts to implement greening mechanisms into programming models already exist. Apart from crop diversity constraints for farmers, Waş et al. (2014) implemented EFAs within the framework of the CAPRI model as a set-aside area. EFAs were not further differentiated into possible land use activities such as fallow land, legumes, or other landscape features. The set-aside area was implemented using respective constraints per farmer. They force them to dedicate 5% of arable land to set-aside areas. Solazzo et al. (2015) investigated the impact of the new CAP reform for a smaller region. The authors developed a positive mathematical programming (PMP) model based on Howitt (1995) and applied it to a region in northern Italy. They focused on the greening component as well as on redistribution payments[24] and simulated different states of policy proposals. Mathematically, they emphasized greening requirements and farm exclusions.

[24]Convergence payments due to regional and national (historical) differences.

Similarly to Waş et al. (2014), EFAs were included as percentage constraints for farmers (see also Ahmadi et al. (2015) and Cortignani and Dono (2015)). Based on individual farm models, Louhichi et al. (2017) tested the new crop diversity strategies of the EU by implementing the respective policy constraints. However, other practices of the CAP reform are not explicitly modeled. Cortignani et al. (2017) were the first who considered several production choices of farmers in order to comply mathematically with the greening regulations (EFAs). In doing so, they considered all greening mechanisms at once: crop diversification strategies, EFAs, and penalties (if farmers do not comply). They used PMP and designed it as a mixed-integer model in order to capture binary variables for penalty implementation. The model was applied to farm samples of three regions in Italy. Moreover, they used environmental indicators to evaluate the environmental effects of the greening measures.

Until now, current literature does not capture the contribution of the potential EFA activities. As summarized in Annex X of the supplementing Regulation (EU) No 1307/2013 of the European Parliament, chargeable land use activities are weighted and do not contribute to the same extent to EFAs. The actual area of EFAs differs from the minimum required area depending on the respective land use activity. For example, legumes have a factor of 0.7, while hedges or trees have a factor of 2. In FOLAS we try to capture these differences by considering weighing factors.

We simulated the greening aspect of the new CAP in separating direct payments (DP) into two payment components: area-related payments and mandatory 'greening' payments. The greening payment needs to be considered in the objective function. Similar to Waş et al. (2014), Solazzo et al. (2015) or Ahmadi et al. (2015), we force greening-effective activities such as legume production, hedges, or fallow land to be implemented at a minimum amount as summarized in table 4.3 although they are mandatory. In order to take into account the weighting factors of mandatory greening measures, we included the following equation. For CAP I, this equation leads to at least 5% greening activities per farm:

$$\begin{aligned}\sum_{f}(GREENINGCAP_{'summerpea'} \cdot x_{g,f,'summerpea'} + \\ GREENINGCAP_{'fallow'} \cdot x_{g,f,'fallow'} + \\ GREENINGCAP_{'hedge'} \cdot x_{g,f,'hedge'}) \\ \geq GWNNSIZE_{g} \cdot 0.05 \end{aligned} \tag{4.31}$$

where

- $GREENINGCAP_{'...'}$ are the weighting factors for the respective activity.
- $x_{g,f,'...'}$ is the area of greening-effective activity per field f and farm g [ha].
- $GWNNSIZE_g$ is the size of each farm g [ha].

A similar equation is implemented for CAP II, although 7% instead of 5% EFAs are assumed. With this kind of mathematical formulation, we consider detailed aspects of EFAs and try to contribute to the current literature.

For CAP I and CAP II, small sized farm payments are calculated for each farm depending on the farm size. It is added to the governmental payments in the objective function of the linear programming iterations and does not influence farming decisions with respect to farm size. It might be possible though, that this payment effects farm size decisions. Nonetheless, modeling the small-sized farming support payment within the non-linear programming iteration causes crashes with the GAMS solver since endogenous variables (farm size) cannot be used as conditionals[25]. Considering the support of small-sized farmers at least in the linear programming seems to be a first step.

For the 'nature-focused' scheme, AES need to be restricted to 20% of the farm land area, which is implemented with the following equation:

$$\sum_f aes_{g,f} \leq GWNNSIZE_g \cdot 0.2 \tag{4.32}$$

where

- $aes_{g,f}$ is the area of flowering strips or fields per field f and farm g [ha].
- $GWNNSIZE_g$ is the size of each farm g [ha].

In the 'nature-focused' scheme, the management of entire flowering fields are funded by the agri-nature premium, which is modeled by deleting the upper bound on the $aes_{g,f}$ variable.

[25]Logical conditionals are used in GAMS in order to judge a value to be 'true' or 'false' (Rosenthal, 2017).

Within alternate policy designs, crop diversify strategies are tightened. Similar to Waş et al. (2014), another legal constraint forces the two main crop types[26] to be lower than a certain limit. For all three policy designs, biogas constraints need to be adapted at double the amount of allowed area dedicated to silo maize production for biogas plantations.

4.2.4.2. Iteration sequences

The execution logic of the political incentive schemes is similar to the model execution steps described in section 4.2.1 and summarized in fig. 4.2. It is also build upon iterations, which are linked to each other via data transfer. However, the iteration runs follow after the 1st iteration of the baseline model. This makes sure that the resulting permanent grassland proportions stay the same for each political scheme. The reference system for all three policy schemes is then the land use activities and farming structure of the baseline calibration run.

For each scenario, the second iteration includes the new political framework conditions summarized in table 4.3 and mathematically implemented as described in the previous section. It simulates the optimal crop and livestock production as well as the optimal allocation of flowering strips. This model output is fixed in the 3rd iteration. Here, new farm sizes are calculated as described in section 4.2.3. The last iteration simulates the resulting cropping and livestock patterns of the remaining farms after structural change processes (farm size changes). Already after the second iteration, we can see how much political incentives influence the consolidation process. This has further effects on the farm sizes (within the 3rd iteration) and the resulting landscape appearance (from the 4th iteration).

4.3. Spatial model application: impacts on biodiversity indicators

Results of the FOLAS model are spatially explicit maps of agricultural farms and fields as well as crops (including flowering strips). We further use them to calculate several biodiversity indicators, which is one possible use of the model results. As an alternative, abiotic environmental indicators such as soil erosion control or nitrate depletion could be conceivable. However, with respect to our research questions, we stick to biotic impact assessments with special attention on pollinator dispersion.

[26]Which is in all cases winter wheat and rapeseed.

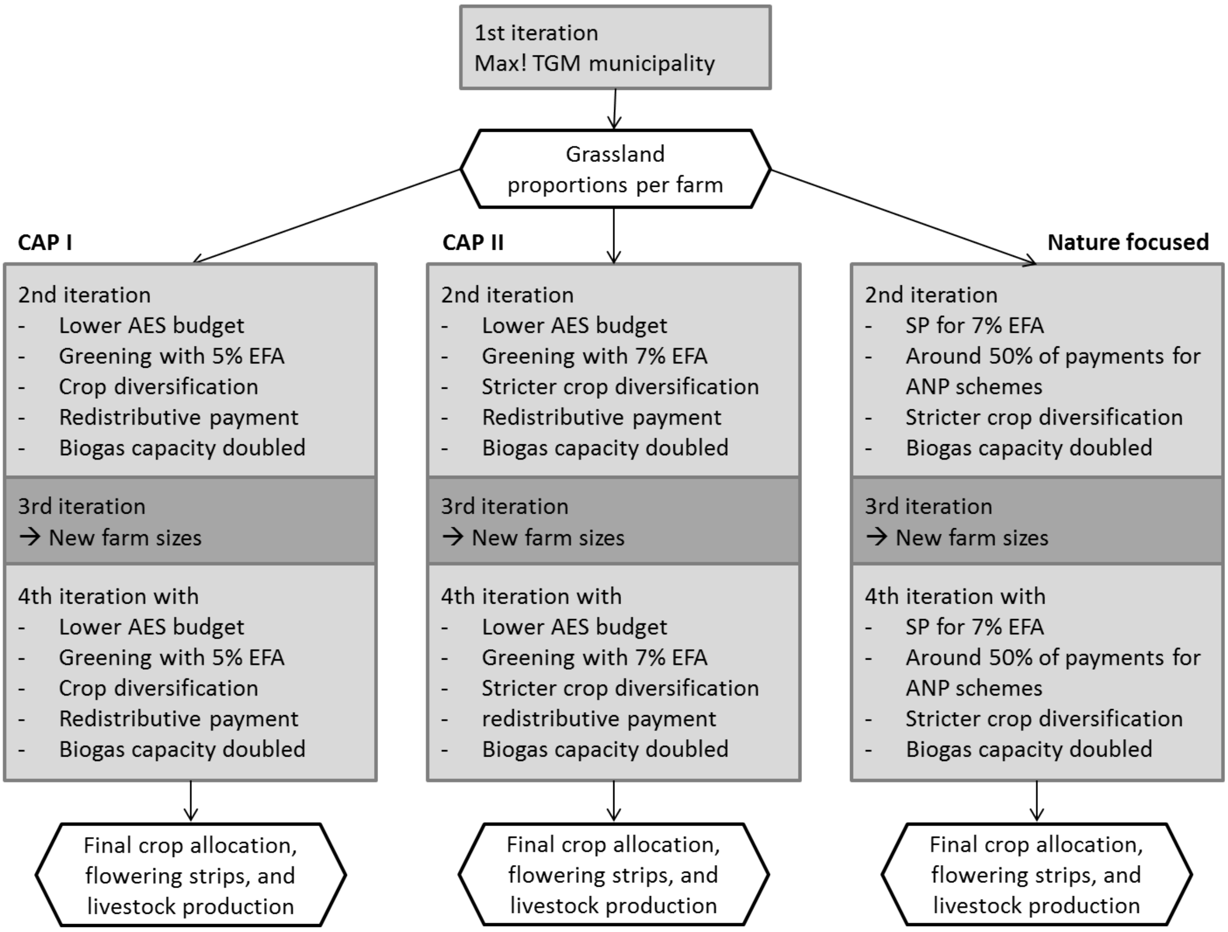

Figure 4.2.: Execution logic for modeling alternate political incentive schemes.

Due to its complexity, more than one hundred biodiversity indicators exist (Pereira et al., 2013). Depending on the indicator, different aspects of biodiversity can be addressed: species diversity, threatened species, habitat quantity/quality or connectivity, habitat diversity, and many more (for a full list, see Gabel et al. (2016)). According to Butchart et al. (2010), these kinds of aspects can be broadly separated into pressure, state, and response indicators. However, trade-offs between them make it difficult to decide on the most appropriate ones (Lindenmayer et al., 2015).

Conservation biologists often use surrogate species such as umbrella species, flagship species, and indicator species for environmental health, population trends, and biodiversity hot spots (Caro and O'Doherty, 1999). These direct measures are helpful for indicating the extent of anthropogenic influences on biodiversity and to tackle conservation problems based on the assumption that fulfilling the requirement of surrogate species would also conserve other species. Another method of biodiversity indication would be the use of landscape structures such as landscape diversity and the extent of certain land use types as the distribution of landscape elements influences the flows of organisms (Forman and Godron, 1986; Dauber et al., 2003). The landscape approach is suitable for studies at large spatial scales and monitoring programs for its feasibility in terms of cost and time. Since it cannot guarantee the presence and/or abundance of species although habitat is available, Chape et al. (2005) recommends using a combination of these two kinds of indicators.

Habitat occurrence, extent, and diversity are indeed main conditions for a rich species diversity (Billeter et al., 2008; Fahrig, 2013). We seek to measure an overall loss of species diversity as well as the abundance of wild bees as pollinators, which occurs due to agricultural landscape patterns within our intensified agricultural production region. Therefore, habitat indicators at the landscape scale have been applied to our case study. Besides bees, we do not look for other surrogate species that are threatened. Of course, applying other habitat models as done in, e.g. Bamière et al. (2013) could also be applied; however, this is not envisaged within the scope of this study.

In order to monitor biodiversity at the farm scale, Herzog et al. (2017) proposed a set of descriptors that evaluate the habitat status of farms and ought to measure the effectiveness of EFAs with respect to the new CAP reform. They put the agricultural landscape into focus. Farmers are able to participate in AES in order to conserve farmland biodiversity. By discarding redundant or misleading descriptors, Herzog et al. (2017) found eight core descriptors addressing the habitat quality of farms and divided them into three groups: (1) structural composition and configuration of farms, (2) specific farm habitats, and (3) interpreted farm habitat descriptors. The first group requires habitat maps, while the latter two can be gathered through interviews with farmers or simple habitat mapping (e.g. number of trees per ha). Since we do not consider trees or shrubs, there are still six indicators left that we could calculate with our approach. These are (1) habitat richness defined as the number of habitat types per ha of farm, (2) habitat diversity, which accounts for both the number of habitat types and the area of each, (3) average patch sizes, (4) crop richness as the number of crops per farm and ha, (5) linear habitats in m per ha, and (6) semi-natural habitats as percentages of farms.

In Herzog et al. (2017), habitat types consist of some general habitat categories such as arable crops and orchards. These habitat categories are further divided into different intensity levels. Our approach covers fewer habitat types since only some of the observed farm types in Herzog et al. (2017) are present in our case study. Habitat richness as the number of habitat types per ha farm in our case does not differ greatly between individual farms. Individual habitat types examined in our study are different types of arable crops, flowering strips, and permanent grassland. This leaves us with four descriptors of the selection above: crop richness, average patch size, linear habitats, and semi-natural habitats.

In our approach we account for crop diversity by using a Simpson's diversity index D, which takes into account the number of crop types present as well as the area of each crop type. Based on Di Falco and Perrings (2003) it is formulated as follows:

$$D = 1 - \sum_{j} x_j^2 \tag{4.33}$$

where x_j is the proportion of farmland allocated to the jth crop. If all land use types n are located within an observed area and have similar proportional distributions, this index approaches one. On the other hand, if only one land use type covers the whole area, D equals 0.

Additionally, the number of patches (fields) within a certain area will be calculated based on the resulting land use and field size maps. Field sizes are basic components of our model and are to be simulated. As our approach stylizes the landscape into rectangular land use units, we drop the linear habitats in m per ha. We extrapolate from actual field shapes, and therefore this descriptor might be misleading. In order to account for the number of semi-natural habitats in our simulated agricultural landscapes, we calculate the share of semi-natural habitats, which are defined as flowering strips and permanent grassland[27]. Mathematically, this is formulated as follows:

$$S = \frac{(x_{grassland} + x_{floweringstrips})}{\sum_{j} x_j} \tag{4.34}$$

[27] According to Herzog et al. (2017), permanent grassland and flowering strips are categorized as semi-natural habitats.

where $x_{grassland}$ is the area of farmland allocated to grassland, $x_{floweringstrips}$ is the area of farmland allocated to flowering strips, and x_j is the area of farmland allocated to the jth crop.

Spatially, our approach differs from Herzog et al. (2017). As they already stated, farm scale descriptors are not suitable for including effects of spatial habitat configurations at the landscape scale. Farm ownership or the question of which farmer manages an area does not influence pollinator dispersion, for example. The dispersion of pollinators depends on the surrounding agricultural matrix, which constitutes of landscape features (Tischendorf and Fahrig, 2000; Hagen et al., 2012). Jauker et al. (2009) investigated pollinator dispersal traits depending on the landscape matrix in an intensively managed agricultural area of the Wetterau region. They found that pollinators such as bees and hoverflies disperse around a radius of 250 m. In order to measure the abundance of bees as pollinators, our approach calculates chosen indicators for circular buffers with a radius of 250 ha as proposed by Jauker et al. (2009). The use of circular buffers is a well-established method in landscape ecology (applications can be found, for example in Berg (2002), Gottschalk et al. (2010), Aue et al. (2014)). We therefore employed the buffer method to measure habitat indicators in our landscapes instead of using farm scale descriptors as done by Herzog et al. (2017). It defines the scale, which is rather relevant for land managers in order to monitor habitat changes in detail. The three indicators are (1) crop diversity, (2) number of patches (fields), and (3) proportion of semi-natural habitat.

In order to calculate the biodiversity indicators at the landscape scale, we are interested in the values of the whole surface. Therefore, the maps need to be converted to pixels in order to reduce computational power [28]. We took a spatial resolution of 1 m^2 so that information loss is low. Then, each pixel is assigned a circular buffer with a radius of 250 m. The buffer zones overlap because each pixel has its own buffer circle. Since land use outside the agricultural area is not modeled, the outer stretch (250 m) of the resulting land use map does not contain reliable information. It will be excluded from indicator calculation. Indicator values are calculated for each pixel, using the land use information within its buffered area.

For example, the Simpson's diversity index is calculated for each 1 m^2 pixel with respect to the circular buffer around it. The same applies to the semi-natural habitat indicator: the share of pixels within a buffer cultivated by semi-natural habitats is calculated as a percentage of the whole buffer area for each pixel separately. In order to account for the landscape structure with respect to the field sizes, the third biodiversity indicator is the number of fields. The modeled land use and field size map is again divided into buffer plots with a radius of 250 m. This time, however, the

[28] This is also done by Jauker et al. (2009).

number of patches/fields within circular buffers was calculated using sampling points that lie 250 m apart from each other. We then estimated the values using a spline function that extrapolates the patterns into non-sampled areas. Closer sampling points need too much computational power and are not workable.

Apart from choosing an appropriate indicator, the reference system against which indicators are compared is required. This can either be (modeled) potential natural vegetation or the current land use (Koellner et al., 2013). Since there is no unified concept about which reference point to take, it depends on the context of the biodiversity assessment. Gabel et al. (2016) suggest at least comparing different scenarios with each other: one of the actual land use and a second alternative land use mixture. With our baseline model (1st iteration), we consider the current land use map as a reference system. Furthermore, we are going to simulate several political incentive schemes that affect landscape appearance. These varying landscapes lead to certain changes in biodiversity indicators, which we are going to compare to each other.

5 Chapter 5. Results

We examined alternate political incentive schemes with respect to the resilience of the socio-ecological system of our case study. In section 2.3 we clarified our analytical framework and defined critical thresholds of different domain-scale combinations in order to test whether certain regime shifts occur due to structural change processes. Regime shifts indicate possible future developments of our case study, which we are able to elaborate before they actually happen. We further seek to assess how political incentives may alter species diversity. Therefore, our model is used (1) to show possible future scenarios of the case study and (2) to test the achievement of political biodiversity protection goals.

The present chapter is arranged according to the three defined thresholds of the driving variables: (1) Amount of farm types, (2) amount of farms, and (3) field sizes. There is then an overview of the model results, which are linked to those respective thresholds. Afterward, the produced stylized maps of our case study are presented. They are used in order to calculate our three chosen biodiversity indicators. The last section of the chapter is dedicated to the political impact assessment with respect to biodiversity targets. Here, we will show results of our first model application, which seeks to measure spatial biodiversity indicators as previously described.

5.1. Critical thresholds

5.1.1. Farm type losses

The critical threshold within the economic domain at the farm scale is crossed if one of the three farm types quits the agricultural sector. According to our baseline model results, after structural change processes, no dairy farm was left, and the critical threshold was reached. This also refers to each alternate policy design. Pig-fattening farms remained in the sector and even increased their acreage[1]. Arable farms still formed the largest part with respect to farm types.

Similar results can be found in Happe et al. (2011), who examined structural change processes and measured their indirect effects on nitrogen loss. With AgriPolis, structural change was modeled, and results were transferred to farm models in order to estimate nitrate depletion. They tested the impacts of two different policy designs on farm structure and nitrogen effects in Denmark. Their results showed that cattle farms, and the number of cattle decreased. Only large-scale dairy farms survived since they were able to use economies of scale. Pig farms increased their acreage and became classified as mixed farms. The reduction in manure production due to the change in livestock numbers could reduce ammonia emission.

In our case study, cattle farms are relatively small compared to the regional average (Hessisches Statistisches Landesamt, 2012a). We can therefore assume that in the long run only arable and mixed pig farms will remain within our research area. In contrast to the impacts on nitrogen depletion, our ecological focus lies with spatial agricultural patterns in order to assess changes in species diversity. Besides field sizes and semi-natural habitats, in our study, landscape structure is represented by land use patterns in the mode of crops. Therefore, we will take a closer look at cultivated crops.

The loss of dairy-producing farms led to less diverse cropping patterns at the landscape level. Table 5.1 depicts the 1st iteration of the baseline model, which comprises our reference point since it calibrates the cropping patterns of 2011. In comparison to these results, we included the cropping results after structural change processes for the baseline model and for the alternate policy designs[2]. In the 4th iteration, grasses and silage maize were no longer produced compared to the 1st iteration of the baseline model in which farmers still keep cattle. The management of permanent grassland was streamlined and only hay was produced at lowest management costs[3], which can be sold to other farmers within the region. Through the obligatory ecological focus areas (EFAs) of the political incentive schemes (5% in CAP I and 7% in CAP II and 'Nature-focused') summer pea production was added at the expense of rapeseed production.

[1]Except for the results of the sensitivity analysis on lower land rents, where one pig-fattening farm closed down.
[2]Land use results of the sensitivity analyses are summarized in Appendix L.
[3]Hay production is cheaper and requires less labour than grass silage.

Table 5.1.: Comparison between the land use results of the baseline model and the policy design models with respect to the reference point and the 4th model iterations. Values are written as percentages of the agricultural area in Wöllstadt.

Crop type [%]	1st iteration baseline	4th iteration baseline	4th iteration CAP I	4th iteration CAP II	4th iteration ‘Nature-focused’
Wheat	50.0	50.0	50.0	50.0	50.0
Sugar beets	16.4	16.4	16.4	16.4	16.4
Rapeseed	22.3	24.2	16.3	13.6	2.4
Barley	0.7	0.7	0.7	0.5	0.7
Potatoes	2.3	2.3	2.3	2.3	2.3
Rye	-	-	-	-	-
Silage maize	0.5	-	-	-	-
Silage maize (biogas)	1.0	1.0	2.0	2.0	2.0
Corn maize	0.9	0.3	0.03	0.06	0.06
Triticale	-	-	-	-	-
Corn cob mix	-	-	-	-	-
Summer peas	-	-	-	-	-
Grass silage (permanent grassland)	1.1	-	-	-	-
Hay (permanent grassland)	3.9	5.0	5.0	5.0	5.0
Weed/grass silage	0.8	-	-	-	-
Summer peas	-	-	7.1	10.0	10.0
Flowering strips	0.11	0.11	0.09	0.09	11.1

In accordance with our analytical framework, regime shifts at the farm scale can trigger regime shifts at the ‘patch’ scale (see figure 2.2). Our results confirm this since crop diversity is obviously affected by farm specialization processes. Apart from the ‘patch’ scale, farm scale regime shifts are also related to the regional scale. As a consequence of dairy farm exits, our model showed that the labour market can be affected as well.

Dairy keeping is strongly linked with farm labour patterns since it requires much more labour input compared to purely arable farming. Actually, all our model results showed that only dairy farmers require additional labour (see table P.1 in the Appendix), which needs to be hired at great cost from the labour market. After field amalgamation hired labour hours could be reduced. Due to bigger fields, labour productivity increased, and more labour became available for other sectors; off-farm labour increased due to free labour capacities (see table P.2). After dairy farms were displaced, there was no more need for hired labour. We can assume that dairy farmers as employers will become rare in Wöllstadt. Depending on the new farm sizes after land exchanges, family labour requirements differed between farms, and off-farm work increased or decreased. Due to farm size enlargements, one farm already worked full time. It was the largest farm after land exchanges (see farm 20 in table P.3). Other larger farms (farms 10, 12, and 15) decreased off-farm labour, while some other farmers worked more outside the agricultural sector; this is a result of EoS effects. At the municipality level, 48 % of the labour capacity was used for agricultural activities within the 1st iteration. After farm size optimization, only 37% labour force was used, with the rest being off-farm labour[4]. Reducing hired labour in order to save costs and shifting from part-time to full-time farms are typical features of the structural change process (Happe et al., 2011).

This trend was even stronger in our sensitivity analysis of lower grain prices. Prices for grain crops are usually volatile and therefore often subject to change. In order to check for differences and deviations within our model results, a sensitivity analysis on grain prices was conducted. Therefore, the median values of time series on grain prices (see section 3.2.2) were exchanged with values of the baseline model, while all other model parameters stayed the same (c.p.). Since grain prices were high in the calibration year, median values of the time series were around one standard deviation lower than the baseline model prices. Results showed that within the 1st iteration, dairy farmers were already reducing their cattle stocking rate by around 25% at the municipality level. Due to lower grain prices, farmers’ incomes were reduced[5]. Therefore, in

[4]These results are similar to the results of the sensitivity analysis on lower and higher land rents as well as to the results of the CAP I and CAP II policy designs.

[5]Detailed model results on income are presented in the next section.

order to reduce costs, farmers minimized dairy keeping and hired less additional labour (see table P.4). On average, off-farm labour was higher than the baseline model results. Farmers worked even more outside the agricultural sector, since less money could be earned within agriculture. After field amalgamation, total dairy capacity was fully utilized (as it was in the baseline model). The use of EoS could reduce farmers' costs and make dairy keeping more competitive meaning that hiring labour would again be worthwhile (see table P.5). This indicates that field consolidation might mitigate cattle losses. However, after additional land exchanges (due to land leases) all dairy farmers quit the sector. At the municipality level, only around 20% of the labour force was used for farming activities; the rest was off-farm work (see table P.6). In the 'nature-focused' policy design, 25% of the labour force was used after land leases at the municipality level, which is around ten percent less than for other policy designs[6]. This deviation was caused by the cultivation of less labour-intensive flowering fields instead of conventional crop production.

Happe et al. (2008) also showed that cost savings first take place in the form of labour-saving technologies and by laying off hired farm workers. Labour saving production technologies are especially preferred since off-farm work is an attractive alternative. The lower the required labour input, the more favourable the agricultural production. These findings are particularly relevant with respect to technical progress and labour-saving technologies, but also regarding job alternatives. The more interesting and lucrative alternative job possibilities are, the more farmers quit the agricultural sector. It shows the importance of off-farm opportunities as drivers for structural change (Happe et al., 2011). With our study, we can confirm that labour alternatives are central conditions for agricultural production.

Agricultural labour movements are linked with population movements (Eberhardt and Vollrath, 2016) and indicate demographic changes within rural areas (Knickel et al., 2017). Depending on the dimension of out-migration and overaging, another regime shift within the socio-cultural domain could follow. According to our conceptual framework, there is a link between the economic farm scale and the socio-cultural regional scale. The proportion of agricultural employees within Wöllstadt is relatively low though[7], and the effect might be marginal for our case study. However, in regions where agricultural production provides jobs for a larger part of the population, this link becomes more significant (Manos et al., 2013). With respect to the debate on promoting strong rural areas within Hesse (HMUKLV, 2015a), it is remarkable that structural change and the use of EoS might foster out-migration due to fewer agricultural labour alternatives.

[6] Off-farm labour patterns of the 4th iteration for alternate policy designs are summarized in table P.7.
[7] 2.3% in 2010 according to Hessisches Statistisches Landesamt (2012a).

Another characteristic of farm type loss worth mentioning here seems to be the participation in agri-environmental schemes (AES). Before modeling land leases within the 3rd iterations, only dairy farmers planted flowering strips[8]. Also Pufahl and Weiss (2009) as well as Zimmermann and Britz (2016) found that low-intensity cattle keeping is related to the participation in AES. After dairy farmers quit the sector, our model showed that the largest farms with respect to area (mainly arable farms) provided flowering strips instead. These findings can be supported by Pavlis et al. (2016), who found that mainly large farms are attracted to AES.

Nevertheless, at the landscape scale, the amount of flowering strips stayed the same within each model run. Cultivation of flowering strips has been modeled through available subsidies of the respective political incentive schemes. In CAP I and II, less money is available for AES (see section 4.2). Therefore, the amount of flowering strips was lower. In the 'nature-focused' scenario though, a lot more AES in the form of flowering strips and fields were realized (see table 5.1). Flowering strips are categorized as semi-natural habitats with a high potential to increase habitat quality in especially homogeneous landscapes (Tscharnke et al., 2012). Therefore, biodiversity is likely to be affected within our case study. Since other landscape characteristics such as field sizes as well as spatial allocation patterns of semi-natural habitats also play an important role for species diversity (Hendrickx et al., 2007; Fahrig et al., 2011; Fahrig, 2013), we will show changes in biodiversity indicators on the basis of our spatial explicit land use maps in section 5.3.

In summary, the farm scale threshold within the economic domain (changes in the amount of farm types) has been crossed within each model run. According to our conceptual framework, this induces shifts within other domain-scale combinations. In our case, cascading regime shifts have been the economic 'patch' scale in the mode of a less diverse cropping patterns, as well as the regional scale through reduced labour requirements. Depending on the importance of the sector, the labour market may influence the socio-cultural domain via out-migration (Eberhardt and Vollrath, 2016). Such cascading effects might lead to new regimes, which are less desired from the social perspective and less resilient from an ecological point of view (Walker et al., 2004; Walker and Meyers, 2004; Matthews and Selman, 2006). Although, before elaborating on the ecological impacts, in the next section we are going to have a look at farm numbers as another driving threshold variable.

[8] Apart from the 'nature-focused' model, where many more farmers took part in environmental measurements.

5.1.2. Farm closures

The 3rd iteration simulates farm sizes and their amount within the total agricultural area of the study site. In our baseline model, 17 of formerly 24 farms remained in the agricultural sector; around 30% stopped agricultural production. The critical threshold of one third fewer farms was almost reached within the baseline model, although not crossed.

In order to understand farm closures, we need to dig more deeply into farm income patterns. Each farmer receives a different total gross margin (TGM), which depends on crop production, farm size, type, and general model conditions. In appendix N, TGMs of each iteration of the baseline model are summarized in tabular form. At the individual farm level, TGMs either decreased or increased from one to the other model iteration. Between the first and the second iterations, farm sizes stay the same for each farmer and only field sizes change due to field consolidation processes. Since field consolidation reduces costs, TGMs are expected to increase. For several reasons, this was not the case for every farm. For example, farm 2 had a TGM of 70,337 Euros in the 1st iteration and a TGM of 47,772 Euros in the 2nd iteration (see appendix N.1). Whereas in the 1st iteration, farm 2 planted almost four ha potatoes, among other crops, in the 2nd iteration, it no longer produced potatoes, producing more sugar beets instead. Potato production delivers the highest TGMs, which is why the TGM strongly decreased. At a first glance, it is not realistic for potato production to change between farmers, since potato production is linked to contracts and specified production machines. A technical solution to control potato cultivation would be to fix potato production results per farm in the 1st iteration of the baseline model and transfer them to the 2nd iteration as it was done with the grassland proportions. This would ensure that potato production stays with the same farmers. However, under free market conditions, farmers are flexible with respect to their potato and sugar beet production[9] and for the harvest, it is possible to borrow machines or pay a service company. When considering a long-time period as it is done within this study it is indeed possible that the production patterns of farmers change. Therefore, the production patterns of potatoes and sugar beets are deliberately not restricted at the farm scale within our model. In the current study, a more important perspective sheds light on changes in field and farm sizes, here at the municipality level, after exploiting EoS. Since the impacts on biodiversity are to be addressed, fields, crops, and technology must interact. A more precise image of production patterns at the farm level requires much more farm data, which is not available without elaborate and expensive data acquisition. Note that the average TGMs at the municipality level increased (see table 5.2), and the model therefore delivers satisfying results. In the following

[9]For sugar beets this trend is even stronger since sugar contingents of the EU end at the end of the year 2017.

paragraphs, we focus on the structural implications at the landscape scale.

At the municipality level, TGM results of the 1st iteration of the baseline model totaled 1,392,101 Euros and the average TGM was around 58,000 Euros per farm. With an average farm size of 51.5 ha, this led to an average TGM of 1,125 Euros/ha (see table 5.2). After field consolidation, the TGM at the municipality level and the average TGM per farm slightly increased. In the 4th iteration, the TGM at the municipality level was lower again due to the loss of dairy production. However, the average TGM was around 78,000 Euros per farm, which is much higher now. Since seven farms closed down, the total agricultural area was allocated to fewer farms, which generated more money per farm. Compared to the 1st and 2nd iterations, the average TGM per ha decreased at the municipality level. When considering much higher farm sizes, though, the average TGM per ha is still higher[10].

Considering the trajectories of the common agricultural policy reforms, the CAP goals of increasing production efficiency through economic rationalities are approached if some farmers quit the sector (Burrell, 2009). Rickard (2015) argues that the new CAP 2020 reform slows down farm growth due to direct payments (DP). He suggests phasing out DP in order to support a more rapid change to larger-scale, capital-intensive, and highly technological farms using precision technology and EoS. When looking at our results, interestingly, in both CAP I and CAP II, one farm less than the baseline model remained. In these cases, the critical threshold levels were crossed. This indicates another regime shift within the socio-cultural domain at the regional scale (compare figure 2.2). According to that, the implemented policy designs inspired by the CAP reform 2020 would accelerate structural change within our research site. We will therefore have another look at farmers' incomes in the CAP I and II policy regimes.

Compared to the baseline model, average TGMs per farm were 1.3% or 2.7% lower after the 2nd iteration (see table 5.2). Waş et al. (2014), Ahmadi et al. (2015), Cortignani et al. (2017) and Louhichi et al. (2017) also found marginal losses in farmers' incomes due to CAP 2020. They showed that income losses were not the same for all farm types. For example, beef finishing farms had higher income reductions (3%) (Ahmadi et al., 2015). However, if structural change is taken into account, what none of the studies above did, a different picture emerged from our results. Based on production patterns of the 2nd iteration, in our case, new farm sizes were simulated. After land leases only 16 farms remained and average TGMs per farm were 3.1% (in CAP I) and 1.6% (in CAP II) higher than in the baseline model results. Our results indicate that even small

[10]EoS might be even higher, if respective investments were modeled.

deviations in the income level may lead to farm closures in the long run.

The spatial scale and the location of the previously mentioned studies vary significantly from our case study. Moreover, apart from Cortignani et al. (2017), they did not cover all greening mechanisms of the new CAP reform. While some considered only crop diversification regulations, others purely focused on EFAs as part of their greening measures. Nevertheless, income may not be the only driver of farm closures. Mandryk et al. (2012) applied regression analyses to analyze impacts on structural change processes. Based on historical data, the authors found that both technology and market prices have strong influences on farm sizes and farm specialization; the lower the wheat price, the larger the farm sizes. Our sensitivity analysis on lower grain prices confirmed the impacts of market prices on farm sizes; only 13 farms remained with a higher average size. Low grain prices led to the highest income reductions at the municipality and average farm levels (see table 5.2). Structural change seems to be accelerated due to lower market prices.

According to Mandryk et al. (2012), climate change and policy change have no impact on the farm size but positively affect specialization. In our study, climate change variables were not explicitly taken into account, although, climate change has an influence on crop prices (Ewert et al., 2011). If crop prices change due to climatic conditions, our sensitivity analysis on grain prices showed that farm sizes would also be expected to change. Therefore, climate change may indeed have significant impacts on farm sizes. Impacts on farm types related to policy changes could not be found in our case. In each model run, dairy farms quit the sector independently of the implemented political incentives. However, in contrast to the results of Mandryk et al. (2012), farm sizes have changed due to alternate policy regimes. Within the 'nature-focused' policy design after land leases, 18 farms remained in the agricultural sector. Since more farms than in CAP I and CAP II remained, the average TGM per farm was lower than in other policy designs. TGMs of the respective iterations were also lower compared to the baseline model (-9.6% compared to the 2nd iteration and -15% compared to the 4th iteration). However, if we take the calibration results of the baseline model as a reference system, after structural change, the average TGM of the 'nature-focused' policy design increases by around 14%. Therefore, the importance of determining the reference point becomes apparent. Nevertheless, if this policy design is chosen, it seems like structural change would be needed in order to compensate farmers' income losses.

Table 5.2.: Comparison of all model runs with respect to the total gross margins (TGMs) at the municipality level in Euros.

	Baseline model	CAP I	CAP II	Nature-focus	Lower grain prices	Higher land rent	Lower land rent
1st iteration							
TGM municipality	1,392,010				1,240,545	1,343,805	1,437,229
Average TGM	58,004				51,689	55,992	59,878
Average TGM/ha	1,125				1,003	1,042	1,162
2nd iteration							
TGM municipality	1,415,929	1,397,744	1,378,160	1,280,189	1,286,951	1,367,633	1,461,058
Average TGM	58,997	58,239	57,423	53,341	53,623	56,984	60,872
Average TGM/ha	1,145	1,130	1,114	1,035	1,040	1,106	1,181
4th iteration							
TGM municipality	1,326,317	1,286,640	1,268,571	1,193,871	1,175,674	1,289,406	1,344,085
Average TGM	78,019	80,415	79,286	66,326	90,436	71,634	79,819
Average TGM/ha	1,072	1,040	1,025	965	950	1,042	1,086

If we compare the outcomes of the 'nature-focused' policy design and the sensitivity analysis for lower grain prices, it becomes more apparent that income variables alone are not sufficient to predict farm closures. In both model runs, the average TGMs of the 2nd iteration reached a comparable level. However, there are major differences in number of farms remaining in the sector after land leases. Due to political incentives, farmers planted more flowering fields instead of cash crops. This diminished the need to exploit EoS to their full extent, and land leases were lower. In contrast to cash crops, EoS are not typically exploited for flowering strips and fields. Instead, farmers need less labour and reduce their production costs. In a situation with low grain prices farmers also face income losses, though they do not have the same alternatives; there is too little money available for environmental measures. The optimal strategy seems to be to intensively exploit EoS, resulting in much fewer farms of bigger sizes. In contrast to Mandryk et al. (2012), our investigation showed that policy may indeed have impacts on farm sizes; at least for our case study. The authors noted that they addressed neither landscape characteristics nor spatial attributes, although these are important for scenario implementations.

Land demand and therefore, farm size is also driven by agricultural land rent prices (Ciaian and Swinnen, 2006; Bartolini and Viaggi, 2013). In our sensitivity analyses, lower land rent prices led to 15 farms remaining, while higher rent prices led to 18 farms remaining. At the municipality level, TGMs of higher rent prices were lower than in the baseline model, and TGMs of lower rent prices led to higher TGMs in each iteration (see table 5.2). According to the statistics, in our model 64% of the agricultural area has been rented (see section 3.2.10). If rent prices increase, farmers have higher costs, and their TGMs are lower; more money needs to be spent in order to produce crops on more expensive land. This seems to alleviate structural change, since more farms than in the baseline model remain. On the contrary, lower land rents seems to accelerate structural change: in the sensitivity analysis on lower land rents, only 15 farms remained. Farming seemed to be more profitable since TGMs were higher and farmers had more incentives to increase farm sizes. If land prices were lower, farmers were willing to rent more land than if land prices were higher, meaning fewer but bigger farms remained within our case study.

Similar results can be found in Happe et al. (2008). They investigated the effects of political changes on farm structures and found that land prices were influenced by political reforms. Land prices affected farm profits to such an extent that structural change was accelerated or slowed down. If rental rates were lower, structural change was faster and vice versa. Since the remaining farms were able to lease land at lower prices, they could implement economies of scale more easily. This is why Happe and Balmann (2002) claimed that DPs increase land rents artificially and thereby slow down structural change. This of course also depends on the initial state of the system. If farms are already big and most scale effects are exploited, the effects on structural change are lower. The authors further stated that livestock farms suffer from lower land prices, since land also serves as security for the high investment requirements. However, the authors did not model each farm type separately, so these interpretations need to be treated with caution. In our sensitivity analysis on lower land rents, we found that apart from all dairy keeping farms, one mixed pig-fattening farm also closed down. Therefore, livestock-keeping farms may indeed be more affected by lower land rents.

To summarize, farm closures are driven by complex interrelations. There seems to be a trade-off between income pressures that push farmers to increase farm sizes (see the example of lower grain prices) and the profitability of farming per se (see the example of higher land rents). If land prices are too high, farms rather stop expanding in size. On the contrary, farms get larger if land rents are low since profitability increases. As we showed with the 'nature-focused' policy design, policy also has the power to control farm closures through incentives.

Within CAP I and CAP II, the critical thresholds of farm closures have been reached. This might influence other scales and domains, too. For example, a reduction in farms may induce a reduction in the rural population (Eberhardt and Vollrath, 2016). After farm size enlargement mainly small part-time farms quit the sector. The obligation to stay in the rural area is lower and migration to urban centers becomes more attractive (Knickel et al., 2017). Even labour markets are influenced by farm closures since fewer farms as potential employers remain (as already outlined above). Larger farms have larger fields (Cong et al., 2016); therefore, also the patch scale is influenced by regime shifts of the economic farm scale. This leads us to our last critical threshold, which we are going to discuss in the following section.

5.1.3. Increasing field sizes

The vast majority of agricultural system models simulated land use variables for the prevailing field structures. Therefore, some authors converted landscape features into raster data and allocated economic land use results randomly to the respective pixels albeit they considered spatially geographic data (e.g. Rounsevell et al. (2006); Weinmann (2002); Dabbert et al. (1999)). Others used Markov chains to allocate land use results (Castellazzi et al., 2010; Aurbacher and Dabbert, 2011). Kapfer et al. (2015), Schönhart et al. (2010), Aurbacher (2010), and Kächele and Dabbert (2002) simulated land use results by working at the field level and referred to each existing field; due to computational power and time restrictions, only small case studies could be investigated.

Table 5.3.: Average field sizes of the baseline model per farm in ha.

Farm	1st iteration	2nd iteration	4th iteration
1	3.1	4.2	10.6
2	3.3	7.0	3.0
3	3.1	6.5	10.9
4	3.1	6.5	-
5	3.2	8.6	16.4
6	3.4	7.8	-
7	3.4	5.9	6.2
8	2.8	5.2	4.9
9	3.0	7.2	-
10	2.9	6.4	13.6
11	2.8	4.6	2.6
12	3.0	6.5	10.7
13	2.7	5.5	4.1
14	3.2	8.6	-
15	3.2	10.4	19.1
16	2.9	4.6	10.5
17	2.5	4.1	-
18	3.1	6.6	6.9
19	3.1	6.8	-
20	3.5	15.9	21.5
21	2.7	4.1	11.4
22	2.9	5.8	-
23	3.3	7.5	7.9
24	3.2	6.8	6.4
Total average	*3.1*	*6.7*	*9.9*

We simulated field sizes as a part of farming decisions. They are subject to change throughout every iteration due to new model conditions (e.g. field amalgamation or farm size changes). Our results showed that the average field size at the municipality level increased after each iteration. The average field size of the baseline model increased from 3.1 ha in the 1st iteration to 6.7 ha in the 2nd iteration and 9.9 ha in 4th iteration after land exchanges[11]. At the farm level, field sizes developed differently as summarized in table 5.3. From the 1st to the 2nd iteration, due to field consolidation, average field sizes increased for every farm. However, in the 4th iteration for some farms, the average field size decreased, whereas they increased for other farms. Negative field size developments occurred on the farms getting smaller during the 3rd iteration. For example, the average field sizes of farm 2 decreased while its size also decreased from around 49 ha to 14 ha (see tables P.1 and P.3). This farm is a typical case, where even more farms become part-time enterprises with a higher degree of off-farm labour. Despite having less land, crop rotation still needs to occur, which is why the field sizes cannot go beyond a certain minimum or maximum, depending on the farm size.

In the sensitivity analysis on lower grain prices, the average field size within the calibration step was the same as in the baseline model (see table 5.4). However, in the 2nd iteration, the average field size at the municipality level was higher with 7.1 ha. The difference is relatively low and stems from changes in the grassland allocation patterns within the 1st iteration[12]. Corresponding to higher farm sizes, field sizes were also higher after land leases than in the baseline model (13.3 ha). Field size results of the sensitivity analysis for higher land rents were almost the same as the baseline model. The only deviation was the average field size of the last iteration, which was lower when putting higher land rents at the base (9.4 ha). The reason is that more farms remained after farm size optimization than in the baseline model. In the sensitivity analysis of lower land prices, field sizes increased from 3.1 ha in the 1st iteration to 6.7 ha in the 2nd iteration and 11.0 ha in the 4th iteration. Compared to the baseline model, these values are the same for the 1st and 2nd iteration but higher than in the last iteration due to there being fewer farms after land leases. The sensitivity analyses again showed, that the model reacted plausibly.

If we compare our results of the different policy designs, the highest average field size after land leases occurred in CAP II with 10.8 ha. Although in both the CAP I and II policy schemes, the same amount of farms remained, the average field sizes in CAP I was lower in both iterations (see table 5.4). In CAP II farmers faced stricter crop diversification requirements and more land

[11] Field size changes occurred similarly to Brady et al. (2012).
[12] Allocation patterns of grassland to farms are summarized in Appendix M.

Table 5.4.: Amount of farms and average farm and field sizes of all model runs at the municipality level in ha.

Model run	Amount of farms	∅ farm size (ha)	∅ field size (ha)
Baseline			
1st iteration	24	51.5	3.1
2nd iteration	24	51.5	6.7
4th iteration	17	72.8	9.9
Sensitivity analysis lower grain prices			
1st iteration	24	51.5	3.1
2nd iteration	24	51.5	7.1
4th iteration	13	95.2	13.3
Sensitivity analysis higher land rents			
1st iteration	24	51.5	3.1
2nd iteration	24	51.5	7.1
4th iteration	18	68.7	9.4
Sensitivity analysis lower land rents			
1st iteration	24	51.5	3.1
2nd iteration	24	51.5	6.7
4th iteration	15	82.5	11.0
CAP I			
2nd iteration	24	51.5	6.6
4th iteration	16	77.3	10.2
CAP II			
2nd iteration	24	51.5	7.2
4th iteration	16	77.3	10.8
'Nature-focused'			
2nd iteration	24	51.5	7.0
4th iteration	18	68.7	9.5

needed to be reassigned to EFAs. Therefore, farmers balanced income losses by using EoS to increase their agricultural fields. In the 'nature-focused' policy design after land exchanges, the lowest average field sizes at the municipality level remained (even compared to the baseline model). Farmers' incentives to increase their field sizes seemed to be lower since alternative income possibilities from flowering fields were not driven as much by EoS measures as cash crops were (see previous section).

In all model runs, the critical field size threshold was crossed. According to our definition, it is reached if field sizes more than double. Since in the 1st iteration of the baseline model (the model calibration step) the average field size was 3.1 ha, this threshold was heavily crossed by all ensuing model runs. This indicates another regime shift, which takes place at the 'patch' scale within the economic domain. Field size changes at the 'patch' scale also effect landscapes and therefore species diversity (Billeter et al., 2008). Field sizes determine the distance between potential habitats and the location where beneficial insects can have a positive effect on pollination

(Ricketts et al., 2008). Crop pollination is densely edged to marginal areas, and valuable ES might get lost (Pufal et al., 2017; Potts et al., 2010). On the other hand, bigger fields lead to higher homogeneity since fewer crop types are planted within the same area. As a result, aesthetic values might get lost (Van Zanten et al., 2016), too.

In our study, three spatial indicators capturing these phenomenons will be presented. Indicator calculations were applied based on the land use and field size maps of the baseline model as well as the three alternate policy regimes. However, before elaborating on our biodiversity indicator results, land use and field size maps of the respective model runs are summarized and visualized in the following section.

5.2. Stylized maps of agricultural lands in Wöllstadt

Cropping and field size results were translated into spatial units as described in section 2.4.2. Therefore, spatial information of the study site was taken in order to depict agricultural farms as rectangles lying adjacent to each other. By simplifying our spatial outlay (see section 2.4.2.3), fields also have rectangular shapes and are located next to each other. Based on spatial information from the case study, the economic model was built, and the results could be allocated to the respective farms (rectangles).

Figure 5.1 visualizes the calibration results of the baseline model as a stylized map of the case studies. The assumption behind this iteration (and stylized map) is a scattered allocation of farmers' fields[13]. Fields cannot exceed a certain size since farmers did not yet consolidate them.

Compared to HLUG data on land use from 2011, the calibration results of the cropping activities (as derived from the 1st iteration of the baseline model) delivered a coarse picture of the actual land use patterns of Wöllstadt (see table 5.5). Wheat production mirrored the current condition quite good, although the model overestimated rapeseed production. Barley production in the actual land use in 2011 was higher than the model calculated. Sugar beet and potato production were adapted to the observed level of cultivation and therefore had the same array as the statistical data. Silo maize production in the model was split into silage maize for fodder and for biogas plantations. The observed level of silage maize in 2011 was 0.6 ha; no information about utilization was available. In the model, silage maize for biogas production was limited to a maximum level of 1% due to plantation capacities (see section 3.2.9). Silage maize as fodder was with 0.5% similar to

[13]This means there is no allocation of fields to farmers.

Figure 5.1.: Stylized land use and field size results of the 1st iteration of the baseline model. Farmers' fields are scattered within the agricultural area of the study site.

Table 5.5.: Baseline model cropping results as a % of the agricultural area at the municipality level compared to HLUG data of the calibration year (2011).

Crop type [%]	1st iteration	*Statistics 2011*	2nd iteration	4th iteration
Wheat	50.0	*51.7*	50.0	50.0
Sugar beets	16.4	*16.4*	16.4	16.4
Rapeseed	22.3	*10.6*	22.3	24.2
Barley	0.7	*7.9*	0.7	0.7
Potatoes	2.3	*2.3*	2.3	2.3
Rye	-	*1.3*	0	0
Silage maize	0.5	*0.6*	0.5	-
Silage maize (biogas)	1.0	*0.6 (n.d)*	1.0	1.0
Corn maize	0.9	*0.4*	0.9	0.3
Triticale	-	*0.2*	-	-
Corn cob mix	-	-	-	-
Summer peas	-	-	-	-
Grass silage (permanent grassland)	1.1	*n.d (5 %)*	1.1	-
Hay (permanent grassland)	3.9	*n.d (5 %)*	3.9	5.0
Weed silage	0.8	-	0.8	-
Summer peas	-	-	-	-
Flowering strips	0.11	*n.d*	0.11	0.11

the observed amount of 0.6%. Corn maize production was also similar to the observed level with 0.9% compared to 0.4% from statistical data. It is sold and not used for private consumption. In the model as well as in the statistics, no corn cob mix or summer peas were produced. Although rye and triticale were cultivated in the calibration year, they were not in the solution vector of the model results, although these crops have been included as cropping options. According to statistics, Wöllstadt had 5% permanent grassland in 2011. However, information about grassland management was not available in public statistics. Due to our model results, 1% of permanent grassland was managed as grass silage and 4% as hay. Model results further simulated 0.8% weed silage, although this crop type was not observed in the calibration year of the study site. Within the statistical land use data received from HLUG, no information on participation in AES existed. Therefore, the amount of flowering strips was calculated based on monetary information about payments for AES in Wöllstadt (see section 3.2.8). The simulated value of 0.11% seemed to be appropriate since studies showed similar percentages of arable land managed as AES within Germany (Oppermann et al., 2012, e.g).

Within the next iterations, crop proportions changed if ever only in a minor manner for most crops. Potato, sugar beet, and silo maize (biogas) production did not change at all due to contractual model restrictions. Rapeseed proportions only increased from the 1st to the 2nd iteration, whereas the proportions of wheat and barley stayed the same. In the last iteration, silage maize as fodder crops, as well as weed silage and grass silage from permanent grassland, disappeared from the landscape while corn maize decreased. These changes came from the loss of dairy farms since there was no more cattle that required fodder. Permanent grassland was managed as hay, bearing lower management input requirements. Notably, the proportion of flowering strips stayed the same since the budget for AES was not supposed to change within the baseline model run.

Although crop proportions at the landscape level did not change significantly throughout the model iterations, field sizes did change. How much farming decisions about field sizes spatially influenced landscape appearance becomes apparent when looking at our stylized maps of the 2nd and 4th model iterations (figure 5.2). In the 2nd iteration, field sizes had already doubled for most of the farms (cf table 5.3). The assumption behind this iteration is perfect consolidation of fields for any farm. Therefore, farmers increased their fields in order to use EoS. The numbers in figure 5.2 represent the farms of Wöllstadt. After land leases (right map), only 17 farms remained, and average field sizes even increased. However, there were still some small farms left having smaller field sizes. These farms reduced their acreage but still needed to meet crop rotation requirements implemented as model restrictions (e.g. farm 2 or 11). More labour is spent outside the agricultural sector, and those farms are managed part-time.

High deviations in the cropping patterns of farmers occurred due to different market conditions. The sensitivity analysis on grain prices revealed quite different cropping results. The main differences were much higher corn maize production (almost 50%) and a maximum proportion of rapeseed (25% due to biophysical constraints) at the expense of wheat production, whereas the other crops had similar values compared to the baseline model (see table L.2 and table 5.5). This shows how sensitive farmers react on changes in crop prices. In contrast to the sensitivity analysis on grain prices, cropping patterns in both sensitivity analyses on different land rents did not differ much from the baseline model, nor between each other (see table L.3). The sensitivity analyses showed that our model reacted flexibly to changes in crop prices and land rents. However, the results were not stylized as maps.

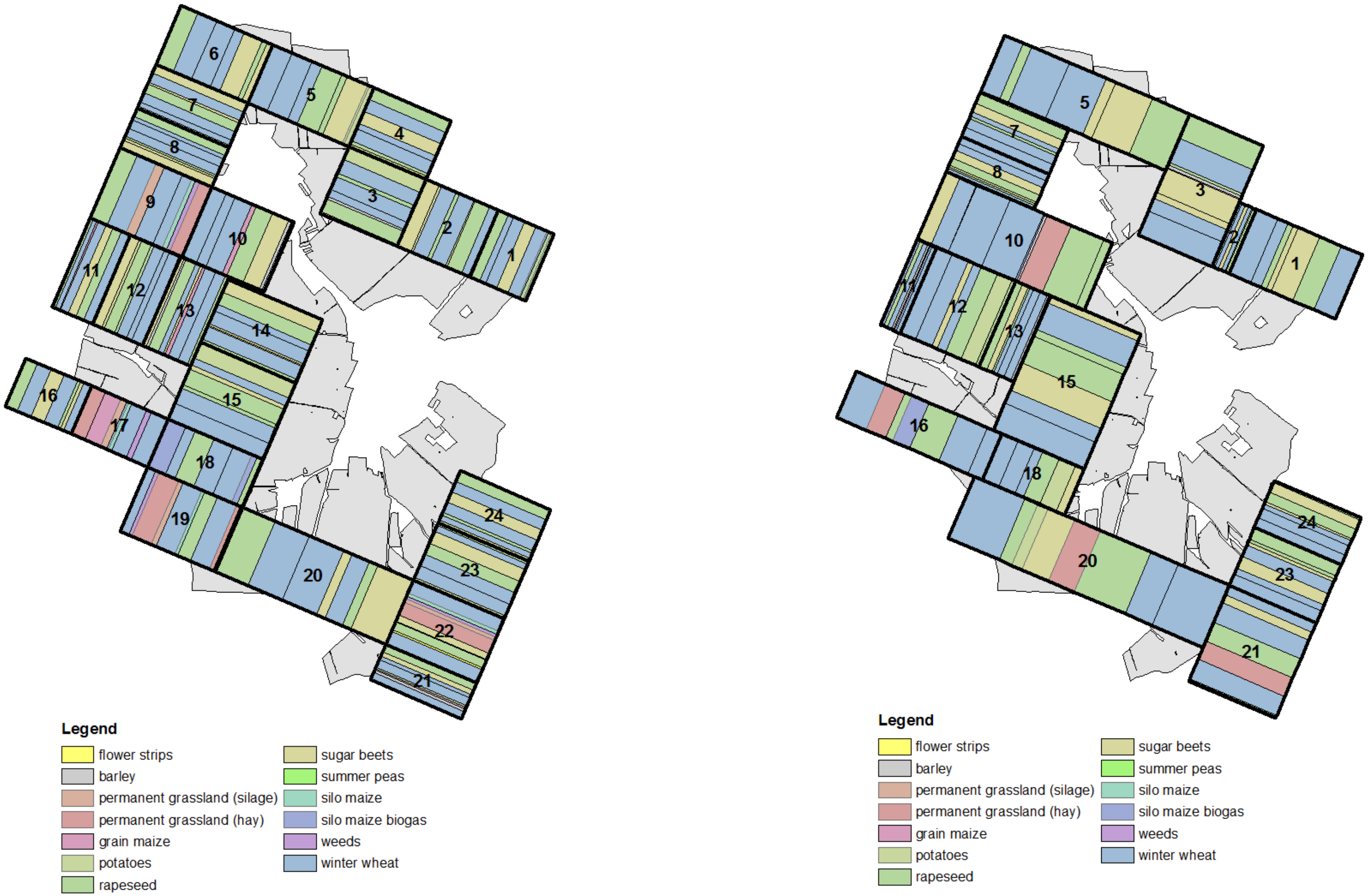

Figure 5.2.: Stylized land use and field size results of the 2nd (left) and 4th (right) iterations of the baseline model. Fields are perfectly consolidated; numbers mark each farm.

Our alternate policy schemes, though, were visualized as stylized maps. A comparison of the 2nd and 4th iterations of each political incentive scheme can be found in figures 5.3 and 5.4. Field sizes are comparable with the baseline model as well as among policy designs. However, cropping patterns differ between the CAP schemes and the 'nature-focused' policy design. Wheat production was relatively even for all three political schemes (see table L.4); it was lowest with 46.8% in the 'nature-focused' scheme. Similar to the baseline model, sugar beet and potato production was the same due to contractual conditions. Rapeseed production in CAP I was with 14.7% in the 2nd iteration and 16.3% in the 4th iteration much lower than in the baseline model (22% and 24%). It further decreased within the CAP II and 'nature-focused' policy designs. There were almost no changes for barley, rye, triticale, or corn cob mix production. Silage maize, grass silage, and weeds were only cultivated within the 2nd iteration steps since after farm size optimization, dairy farms and fodder crops disappeared from the landscape. Again, after farm closures, only hay management remained on permanent grassland since there was no more use for grass silage as a potential fodder crop. Silage maize for biogas was produced to a higher extent (2%) than in the baseline model since the political incentive schemes assume that policy sticks to its ambitious energy targets. Corn maize production was relatively low for all three policy designs within the 2nd iteration and even lower within the last iteration, with always less than 0.5% of the total agricultural area. Compared to the baseline model, summer pea production was now another significant crop. Due to the new CAP reform, farmers needed to meet a certain amount of EFAs in order to receive a greening premium in CAP I and II, respectively a sustainability premium (SP) in the 'nature-focused' design. In CAP I, summer pea production was lower than in the other two policy designs since only five instead of seven percent agricultural area needed such 'ecological treatment'. The proportions of flowering strips in CAP I and II are notably even lower than in the baseline model since the budget for agri-environmental schemes of the new CAP reform is lower (see section 3.2.8). In the 'nature-focused' policy design, many more flowering strips (11.1%), which can now cover entire fields (flowering fields), adorn the landscape.

The stylized land use and field size maps show certain points in time. While the 2nd iteration indicates agricultural landscape after land amalgamation, the 4th iteration shows the resulting land use and field size patterns after land exchange processes through land leases. Based on the demonstrated stylized maps, three different spatially explicit biodiversity indicators were calculated in order to investigate the potential impacts of structural change processes on biodiversity. We are going to present the biodiversity indicator results in the following section.

Figure 5.3.: Stylized land use and field size results of the 2nd iterations of the alternate policy designs: CAP I (left), CAP II (middle), and 'nature-focused' (right). Fields are perfectly consolidated; numbers mark each farm.

Figure 5.4.: Stylized land use and field size results of the 4th iterations of the alternate policy designs: CAP I (left), CAP II (middle), and ‘nature-focused’ (right). Fields are perfectly consolidated; numbers mark each farm.

5.3. Biodiversity indicators

The farmland indicator for biotic diversity of the German national indicator report consists of stock values of ten main birds taken as indicator species. For the calculation, averaged stock values were set in relation to the target value (BMBU, 2010). According BMBU (2015), only 56% of the target value had been reached in 2011. In order to test the political achievements of three alternate policy designs with respect to the biodiversity targets, we applied three different biodiversity indicators to our simulated and stylized agricultural landscape maps. Our indicators covered (1) crop diversity in the mode of a Simpson's diversity index, which takes into account the number and area of different crops, (2) proportions of semi-natural habitats consisting of permanent grassland and flowering strips/fields, and (3) number of patches/fields. These indicators are targeted to measure landscape heterogeneity. They are not adapted to the habitat requirements of birds or mammals. They rather indicate the abundance of insects such as bees since our study also covers flowering strips as semi-natural habitats and allocates them spatially. With this procedure we seek to get comparable figures to the indicator value of the national indicator report since insects form an important part of the food web of birds. Indeed, Valerio et al. (2016) found that the species richness of surrogate bird species protected by the European Birds Directive[14] corresponds to landscape heterogeneity. At a spatial scale of 125-250 m, the correlation between bird diversity and landscape heterogeneity is especially significant (Morelli et al., 2013).

Landscape matrices determine pollinator dispersions (Tischendorf and Fahrig, 2000). As we focused on pollination as an ecosystem service, the surrounding agricultural landscape matrix was considered for the indicator calculation. For each sample point, land use features of a buffer zone with a radius of 250 m served as the calculation base. In doing so, for each indicator a value map for each iteration was produced. In the following, descriptive statistics of these maps are analyzed. However since space matters, descriptive statistics alone cannot cover the significance of the indicator calculation. This is why biodiversity indicator maps need to be considered as well.

5.3.1. Crop diversity

Crop diversity is directly associated with agricultural production and influences soil degradation, fertility, and erosion (Bullock, 1992; Thrupp, 2000). It therefore can alter the performance of ecosystem services (Naeem et al., 1994) and becomes important for food security issues (Smale and King, 2005) in terms of improving the resilience of agricultural systems (FAO, 2011). It also

[14] And also used for the calculation of the biodiversity indicator of the German national indicator report.

Table 5.6.: Simpson's diversity indicators of the baseline model and alternate policy designs: descriptive statistics of stylized maps based on pixel data.

Simpson's Diversity Indicators	Min	Max	Mean	Std Dev.
Baseline model				
1st iteration	0.26	0.85	0.63	0.09
2nd iteration	0.0	0.82	0.57	0.12
4th iteration	0.0	0.79	0.47	0.19
CAP I				
2nd iteration	0.03	0.78	0.60	0.13
4th iteration	0.0	0.82	0.56	0.18
CAP II				
2nd iteration	0.02	0.78	0.61	0.12
4th iteration	0.0	0.79	0.49	0.19
'Nature-focused'				
2nd iteration	0.0	0.83	0.62	0.14
4th iteration	0.0	0.80	0.51	0.18

helps one understand how landscapes change over time (Herzog et al., 2017).

The Simpson's diversity index (Simpson, 1949) (SD) reflects the richness and evenness of species within an observed area (Magurran, 2004). It therefore can be used as a proxy for spatial crop diversity within a research area and has also been applied in several newer studies (Palmu et al., 2014; Conrad et al., 2017, e.g.). Depending on the spatial resolutions, SD indicators vary. For example, Conrad et al. (2017) used circular buffers of 1.5 km and 5 km radii. Comparisons between studies with different spatial resolutions need to be interpreted with caution. This indicator, though, has a relatively intuitive interpretation. It represents the probability that two randomly picked observations would be different crops (Hurlbert, 1971).

Table 5.6 summarizes descriptive statistics for each stylized map. The respective maps for each iteration of the baseline model can be found in figure 5.5. Note that for the outer stretch (250 m distance from the mapped border) no indicator values were calculated. We only modeled the agricultural area within Wöllstadt, and no reliable data for the surrounding area, which serves as information for the indicator calculation, was available.

In the 1st iteration, where fields of farmers have been scattered, the mean SD indicator at the landscape level was 0.63. The highest crop diversity values reached a level of 0.85. In only a few spots, SD indicators fell below 0.4 (see left map in figure 5.5). After field amalgamation (2nd iteration), the average SD indicator decreased. More spots showed values below 0.4, while crop diversity of those spots with already low SD values became even lower. The lowest values could be found after land leases (4th iteration) with an average mean SD indicator of 0.47.

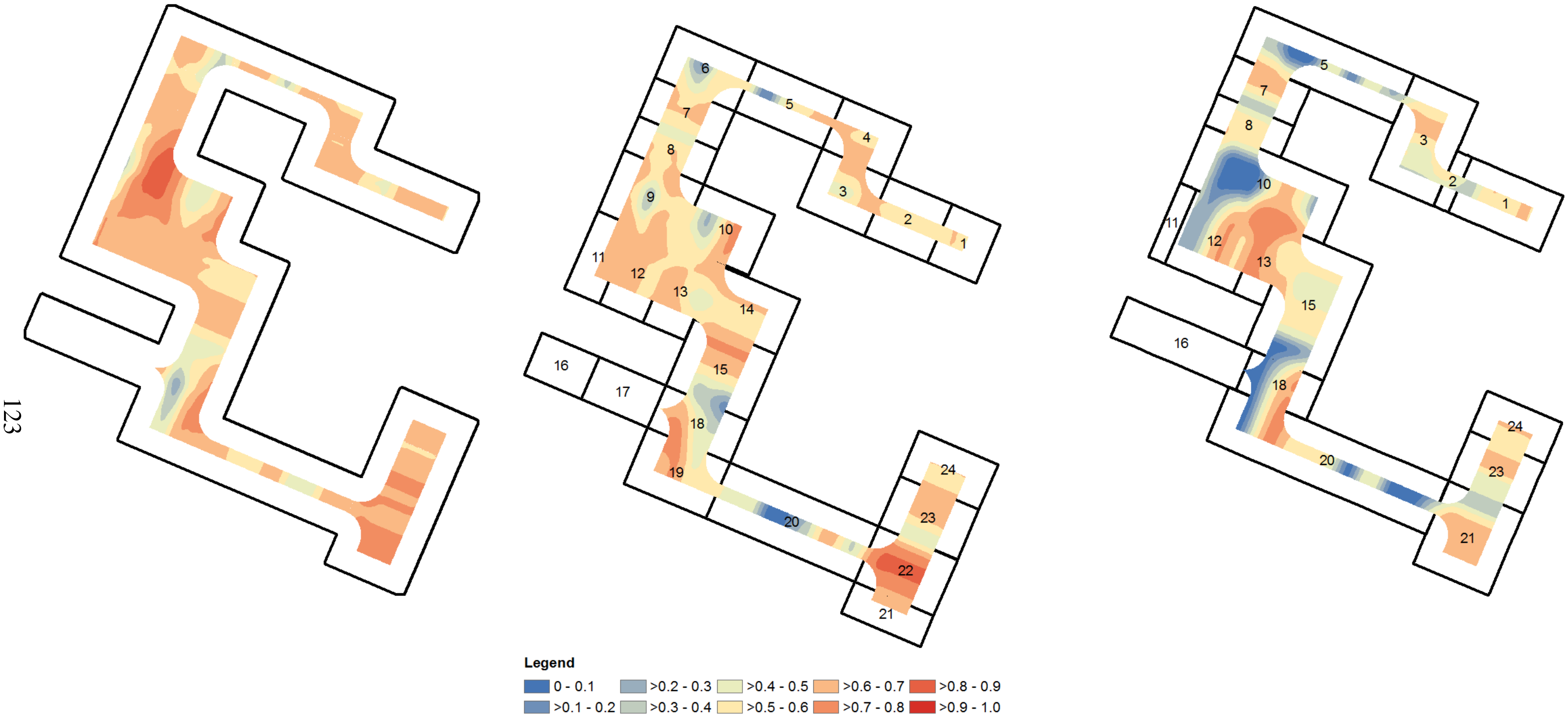

Figure 5.5.: Spatially explicit Simpson's diversity index calculation for each iteration of the baseline model (*left:* 1st iteration, *middle:* 2nd iteration, *right:* 4th iteration). Source: author's own results.

According to our results, crop diversity also changed due to different political incentive schemes. Compared with the current situation (1st iteration of the baseline model), no policy design could compensate crop diversity losses due to structural change processes and always led to lower SD indicator values after land leases (4th iterations). Taking the mean value of the 1st iteration of the baseline model (0.63) as a basis, all alternate policy designs were associated with slightly lower SD indicators within the 2nd iteration (see table 5.6)[15]. However, if taking the last iteration of the baseline model as a basis, SD indicators of the policy designs were higher. All alternate policy regimes led to an improvement with respect to crop diversity in comparison to the 'business as usual' policy design. CAP I led to highest mean values (0.56), followed by the 'nature-focused' policy (0.51), CAP II (0.49), and the baseline model (0.47).

In summary, due to structural change processes, crop diversity of our case study decreased independently from the policy design. But what does this tell us about biodiversity? Indicators for habitat diversity are linked to species richness (Billeter et al., 2008). The Simpson's diversity indicator evaluates heterogeneity of landscapes in general and does not emphasize rare species, for example. In landscape ecology, it indicates species diversity since the higher the heterogeneity, the higher the variety of resources available for species; it does not guarantee the abundance of certain species (Valerio et al., 2016). Therefore, landscape heterogeneity needs to be defined according to its functions to serve as a habitat for target species (Fahrig et al., 2011). If all species living within a landscape need to be captured, the Simpson's diversity index is a popular measure, although universal statements about biodiversity or indicator thresholds in the sense of species richness, for example, are not possible (Kadoya and Washitani, 2011). This is why Chape et al. (2005) recommends a combination of different biodiversity measures.

5.3.2. Semi-natural habitats

In contrast to the Simpson's diversity index, which is often used to show general biodiversity trends, the abundance of certain habitats also indicates the abundance of target species (Fahrig et al., 2011). In fact, the extent of habitats is much more critical for organisms than land use diversity is (Rüdisser et al., 2015). This is why habitat indicators are more useful for conservation purposes, than, for example, crop diversity indicators (Kadoya and Washitani, 2011).

[15] Spatially explicit SD indicator maps for the alternate policy designs can be found in Appendix Q.

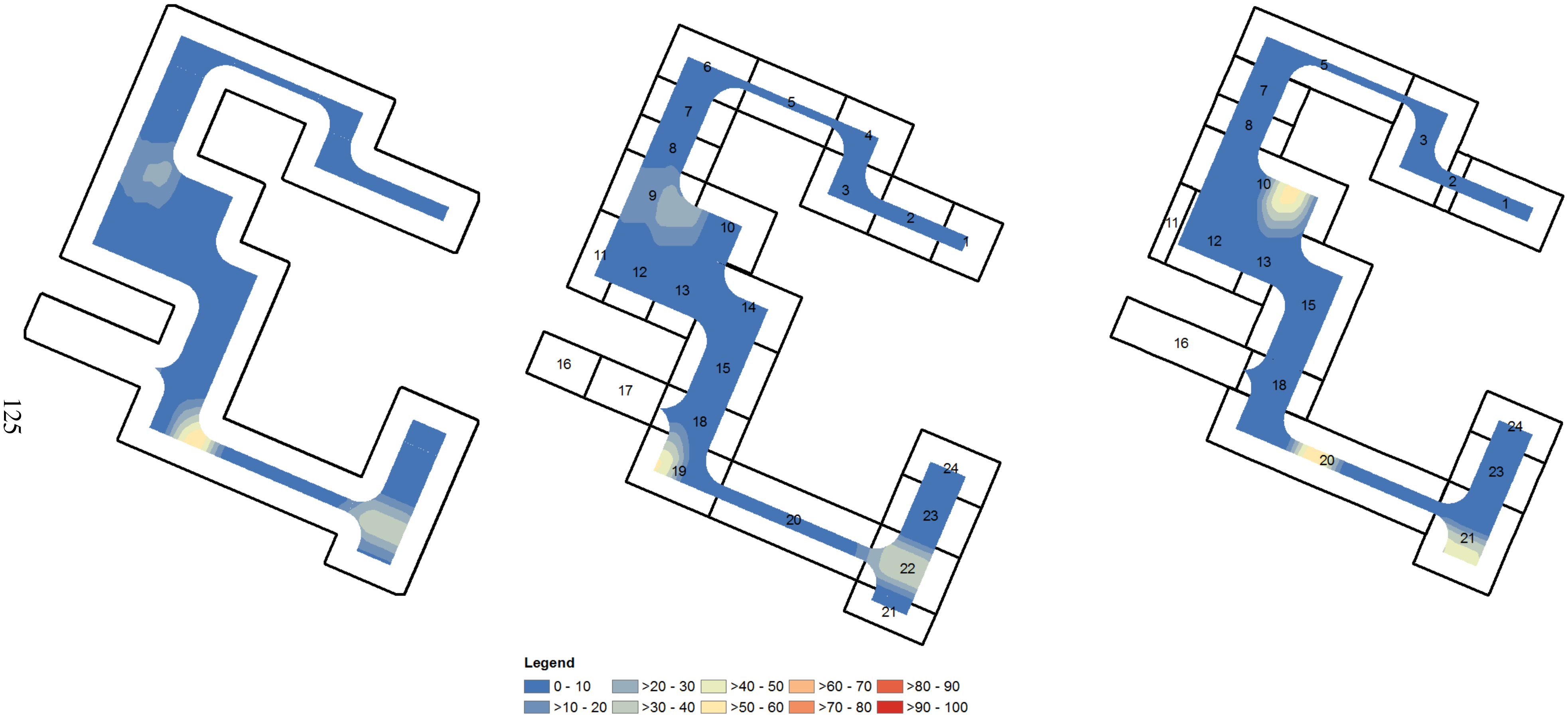

Figure 5.6.: Spatially explicit semi-natural habitat index calculation for each iteration of the baseline model (*left:* 1st iteration, *middle:* 2nd iteration, *right:* 4th iteration). Source: author's own results.

Table 5.7.: Semi-natural habitats of the baseline model and alternate policy designs (%): descriptive statistics of stylized maps based on pixel data.

Semi-natural habitats	Min	Max	Mean	Std Dev.
Baseline model				
1st iteration	0	57	5.5	10.5
2nd iteration	0	53	5.6	10.1
4th iteration	0	58	4.5	11.8
CAP I				
2nd iteration	0	50	6.1	11.6
4th iteration	0	55	4.3	10.6
CAP II				
2nd iteration	0	57	6.4	12.2
4th iteration	0	58	4.8	11.6
'Nature-focused'				
2nd iteration	0	88	16.8	18.3
4th iteration	0	66	15.8	18.0

In our study we included another landscape indicator that measures semi-natural habitats. These influence the flow of organisms (Forman and Godron, 1986; Dauber et al., 2003) and are important for the overall species richness within agricultural areas (Hendrickx et al., 2007; Billeter et al., 2008). Species richness in turn preserves the flow of ecosystem services such as pollination (Klein et al., 2007). Literature shows that semi-natural habitats serve as sources for the dispersion of pollinators, which contribute to a higher species richness within adjacent areas (Duelli and Obrist, 2003; Öckinger and Smith, 2007).

We focused especially on pollinators such as wild bees. For pollinators, flowering strips and permanent grassland can serve as habitats and determine the matrix quality of agriculturally characterized lands (Jauker et al., 2009). Results for our baseline model showed, that semi-natural habitat values have been relatively stable throughout the iterations; they changed only within a range of 1%. In each iteration there have been samples that contained no semi-natural elements at all (see minimum values in table 5.7). In these areas crop pollination might be critical. However, there have been some spots with more than 50% semi-natural habitats. Those spots cover the areas of dairy farms (see farms 9, 17, 19, 22 in figure 5.6). Dairy farms manage permanent grassland and additionally often participate in AES (Pufahl and Weiss, 2009; Zimmermann and Britz, 2016), which could also be confirmed with our results (see section 5.1.1). With respect to habitat quality, our case study seemed to be already in bad condition, which might have become even worse with the policy design at the time. Mean indicator values were very low in each modeling step, with 5.6% in the 2nd iteration and 4.5% in the 4th iteration (see table 5.7). According to Jauker et al. (2009), 10% semi-natural habitat is regarded as low. Our results further showed that semi-natural habitats conglomerated within certain areas where higher values had already occurred (see figure

5.6). The highest maximal values could be found in the 4th iteration (cf table 5.7). After dairy farms closed down, the biggest farms that rented in the land of dairy farms planted flowering strips. Semi-natural habitat aggregation remained unchanged and even increased after structural change processes.

The results for CAP I and II delivered comparably low values for the semi-natural habitat indicator (see table 5.7). In the shorter run, the CAP reform seemed to have some positive effects with respect to potential semi-natural habitats. In both policy designs initially, after the 2nd iteration, mean values were higher than the reference point (1st iteration of the baseline model). However, these small deviations might have stemmed from random field allocations within the rectangular farms. For example, if permanent grassland had been allocated to an outer border of the indicator map, higher values might have gotten lost and vice versa. After land leases, lower average values for the semi-natural habitat indicators occurred. In CAP I, the mean value (4.3%) was even lower than in the baseline model, while in CAP II the mean index value was slightly higher (4.8%). Similar to the baseline model, in each policy scenario, the index for semi-natural habitats had minimum values of 0. There were still areas where some pollinators such as bees might not have found enough living space. However, pollinator dispersion strongly depends on the life history traits of the observed taxon (Jauker et al., 2009). In CAP I and II, maximal index values increased from the 2nd to the 4th iteration, while mean values decreased. Similar to the baseline model, this and the reduction of standard deviations indicated that areas that already had enough semi-natural habitat conglomerated, whereas the habitat quality of other areas decreased. Merely a few grasslands and even fewer flowering strips along field margins are not enough to support diverse bee communities and do not allow adequate levels of dispersion (Steffan-Dewenter, 2003).

The 'nature-focused' policy scheme delivered totally different results. Farmers participated in ANP schemes and planted many more flowering fields and strips. Mean semi-natural habitat values of the 2nd (16.8%) and 4th (15.8%) iterations were much higher. Maximal values also reached a much higher level of 88% and 66%, respectively. There were still areas with no semi-natural elements since the minimum value was 0. However, when looking at the value maps of the semi-natural habitat indicator summarized in Appendix Q, these areas occurred less often. Improvements in the matrix quality of agricultural areas (semi-natural habitats well above 10%) lead to a lower decay in species numbers, especially for wild bees, and supports species dispersion (Jauker et al., 2009).

Table 5.8.: Number of patches in the baseline model and alternate policy designs: descriptive statistics of stylized maps based on pixel data.

Number of patches	Min	Max	Mean	Std Dev.
Baseline model				
1st iteration	8	31	17.2	5.0
2nd iteration	2	16	8.6	2.8
4th iteration	1	15	6.4	2.9
CAP I				
2nd iteration	2	20	8.7	3.2
4th iteration	1	19	6.7	3.8
CAP II				
2nd iteration	2	17	8.0	3.0
4th iteration	2	12	5.3	2.0
'Nature-focused'				
2nd iteration	2	18	7.9	2.9
4th iteration	1	14	5.8	2.5

5.3.3. Number of fields

Another indicator we analyzed in our study was the number of fields. In landscape ecology, this indicator usually counts the number of patches and is related to the number of habitat patches (Jepsen et al., 2005; Dramstad et al., 2001, e.g.). However, we applied it to our agricultural fields to show changes in field sizes and related them to pollination services by choosing the adequate dispersion radius of potential pollinators. Field sizes determine field margins within agricultural landscapes. Field margins can serve as corridors and enhance species migration to and from nesting and/or food resources (Collinge, 2000; Cane, 2001). In general, movements between several habitats would be lower without corridors (Mabry and Barrett, 2002). On the other hand, field sizes determine the distance between potential habitat and required pollinator services (Ricketts et al., 2008; Pufal et al., 2017).

Our results for the baseline model showed that the mean values, as well as the minimum and maximum values, decreased from one iteration to the next iteration (see table 5.8). The standard deviation was lowest for the 2nd iteration; however, it did not vary much from the 4th iteration. While the current farming structure exhibited at least 8 agricultural fields within a radius of 250 m, after structural change, only one field could be found in some areas. Mean values reduced to around one third of the reference scenario. The highest changes occurred after field amalgamation; differences in the patch sizes were relatively low between the 2nd and 4th iterations. This became more obvious when comparing the stylized maps of figure 5.7.

Similar to the baseline model, the mean as well as maximum, minimum, and standard deviation values in all alternate policy designs decreased after farm size optimization. A slightly higher mean value than in the baseline model could be found in CAP I, with 8.7 in the 2nd iteration and 6.7 in the 4th iteration. The CAP II and 'nature-focused' policy design delivered lower values than the baseline model (see table 5.8). Each model run showed a declining trend in the number of patches[16]. Field sizes increased while field margins that may serve as corridors became fewer. Therefore, we expect species' movements to be lower after structural change processes, and crop pollination might become difficult for internal areas since fields are bigger. However, this depends on the pollinator species and matrix quality (Jauker et al., 2009).

Measuring habitat composition always needs a set of several habitat indicators since species richness is driven by a variety of factors (Bailey et al., 2007). The lower Simpson's diversity index of the 'nature-focused' policy design compared to CAP I, for example, is driven by a relatively low number of patches after structural change processes (see table 5.8). While the minimum values for the number of patches indicator did not differ between these policy designs, the maximum values of the CAP I scenario were higher than in the 'nature-focused' design. Furthermore, the standard deviations were higher in CAP I than in the 'nature-focused' design and even increased in the 4th iteration of CAP I. Field sizes in the 'nature-focused' policy design were lower and distributed more evenly than in CAP I. This contributed to lower Simpson's diversity indicators for the 'nature-focused' incentive scheme. However, since the SD indicator covers only the average crop type diversity within the buffer zones and no habitat quality, the low deviation in the 'nature-focused' policy regime did not necessarily indicate reductions in biodiversity. In fact, species diversity most probably increased due to many more semi-natural habitats (Hagen et al., 2012; Hendrickx et al., 2007; Steffan-Dewenter, 2003).

The current state of our case study seems to already be very poor with respect to semi-natural habitats and will get worse without significant changes in agricultural policy reforms. In both CAP designs, a lower abundance of semi-natural habitats, lower cropping diversity, and a lower number of patches led to landscape homogeneity, probably with adverse effects on biodiversity and pollination. For our case study, the new CAP reform was not able to compensate biodiversity losses linked to structural change processes.

[16] Indicator maps are summarized in Appendix Q.

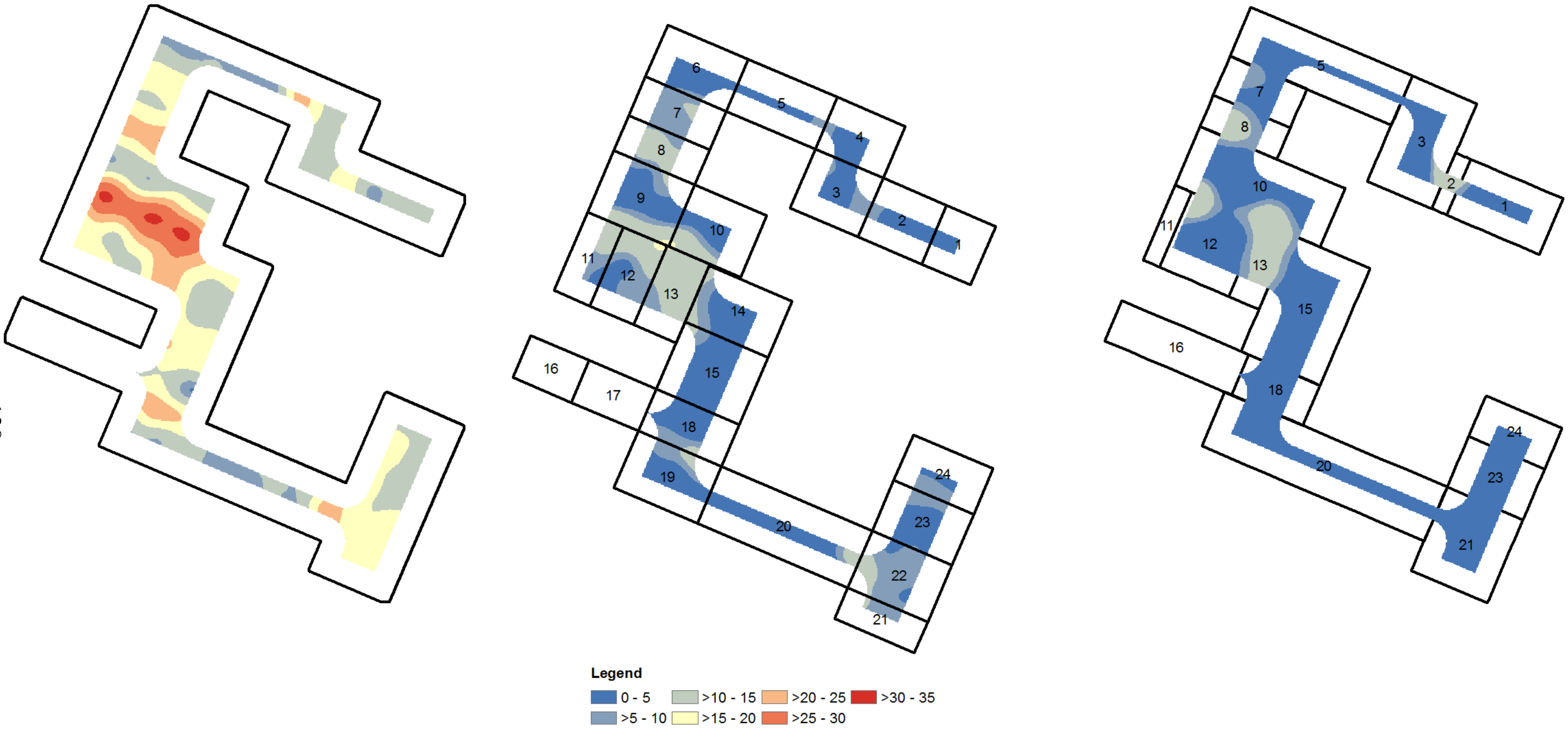

Figure 5.7.: Spatially explicit number of patches index calculation for each iteration of the baseline model (*left:* 1st iteration, *middle:* 2nd iteration, *right:* 4th iteration). Source: author's own results.

6 Chapter 6. Socio-ecological resilience of the case study

In the first part of the chapter, a picture of the socio-ecological resilience of our case study is drawn, and possible future developments under the new CAP reform are envisaged. According to these trends, political implications to enhance biodiversity within agricultural lands are delineated. Afterward, conditions under which the research questions can be answered are elaborated. This contains critical examinations with respect to methodology and the spatial outlay. At the same time, we debate possible model extensions.

6.1. The new basin of attraction

The conceptual framework of the study at hand follows a holistic view where humans play a crucial role in landscape appearance (Berkes et al., 2003). It seeks to measure the socio-ecological resilience (Walker et al., 2004) of our system. Interactions between components of a system occur at several scales and domains (Carpenter, 2003). Therefore the economic, ecological, and socio-cultural domains of our case study were investigated regarding several scales. Within the economic domain, all three scales were addressed: (1) the small scale in the form of agricultural fields, (2) the medium scale represented by farms as decision units, and (3) the large scale because interactions between farmers at municipality level were modeled. Within the ecological scale, large-scale effects in mode of biodiversity changes as being the consequence of drivers were investigated. The socio-cultural domain takes into account large-scale interactions within the labour market. Since temporal aspects are crucial for investigating socio-ecological systems (Kinzig et al., 2006), we considered structural change processes.

The identified variables that might drive the socio-ecological system into another basin of attraction were the amount of farms, field sizes, and farm types. Our results demonstrated that structural change within the study site can be modeled and traced back to these drivers. According to our analytical framework, thresholds were defined for the driving variables, which tie together different domains and scales. These are used to show possible future developments of our case study.

In the baseline model, two of the three critical thresholds were crossed: field sizes more than doubled at the 'patch' scale, and farm type loss occurred at the farm scale. In both CAP policy designs, within each scale, critical thresholds were reached. Similar to the baseline model, two critical thresholds were crossed in the 'nature-focused' policy design. Here, more farms remained in the agricultural sector. Structural change processes were prevalent in all model runs, where the amount of farms were reduced at the expense of dairy farms. Field sizes sharply increased due to field consolidations and the reduction of farm numbers.

Each time a threshold is crossed, regime shifts are induced. Since often one single regime shift triggers a whole system to change it is most probable, that several regime shifts in combination (cascading effects) lead the system into another basin of attraction (Walker et al., 2004). In fact, cascading domain-scale regime shifts within investigated case studies always led to new system regimes that were more resilient to changes and often even irreversible (Kinzig et al., 2006). These new regimes are usually less popular from the societal perspective (Matthews and Selman, 2006). The authors further note that desired landscapes need to be assessed and depend on cultural and ecological values as well as many other aspects such as current needs or conditions.

The regime shifts outlined above had different effects on biodiversity indicators, however. All policy designs led to lower biodiversity indicator values except the 'nature-focused' incentive scheme. There, more semi-natural habitats occurred despite structural change processes. Since semi-natural habitats predict species richness more securely than landscape indicators such as the Simpson's diversity index (Rüdisser et al., 2015), we assume that the 'nature-focused' policy design has potential to fulfill the desired biodiversity targets of the EC. Within our case study, CAP policy designs, which were aligned to the new CAP reform, were not able to enhance biodiversity. However, a rich species diversity is a declared goal of the political and hence societal domain. We can conclude that in our case study, a new basin of attraction induced by the 'CAP towards 2020' reform will follow. This new basin of attraction will be worse with respect to ecological desires. It might even be irreversible. With the CAP policy design, the cultural landscape of our case seems to change into a 'pure' food and energy production site. Only few semi-natural habitats will remain, whereas small and scattered fields might be displaced by huge and uniform production units. Agricultural production might be specialized in a mode of arable and pig-fattening farms, while dairy farming will disappear. The loss of dairy farms leads to lower crop diversity, which can only partly be offset by the simulated greening measures of the new CAP reform. Furthermore, the trend to cost-saving, full-time farms probably holds, and only a few family farms working part-time will stay in the agricultural sector. Running parallel to the insecure question of farm succession and demographic changes within rural areas (Mawick et al., 2011), the rural population might further decrease due to fewer job alternatives within the agricultural sector and changing landscapes that no longer mirror cultural identities (Eberhardt and Vollrath, 2016; Dramstad et al., 2001).

Semi-natural habitat loss and aggregation through structural change processes could not be compensated through incentives schemes of the CAP designs. As outlined above, pollinator dispersion strongly depends on potential nesting and fodder habitats, as well as on the agricultural landscape matrix (Tischendorf and Fahrig, 2000). Therefore, the new basin of attraction might be fragile with respect to the pollination on which farmers rely. Either farmers need to replace these missing ES by bearing high costs as a result of paying pollination service companies, or adequate habitats need to be established. As we showed with our biodiversity indicators, this could be the establishment of semi-natural habitats or reduction of field sizes in order to increase landscape mosaic and the amount of field margins. However, as already altered systems might get stuck within their new basin of attraction (Kinzig et al., 2006), it might not be enough to restore earlier landscape structures (hysteresis effect). More landscape structures are possibly required in order to establish new populations since a lot of knowledge and time is needed to restore ecosystems (Bengtsson et al., 2003). For that reason it seems to be fundamental to maintain already existing habitats within agriculturally intensified areas. Otherwise costs for re-establishing pollinator populations might explode.

Our results further showed that cost pressures influence the amount of farmers remaining in the agricultural sector. Costs for declining pollinators are indeed high (see Bauer and Wing (2016) or Breeze et al. (2016) for a review). If farmers need to replace missing ES through, for example, self-pollination, high costs might further accelerate structural change. Small-scale farmers might simply not be able to bear those costs, and buying pollination services only pays off for large farms (Geslin et al., 2017).

6.2. Political implications

As the aim of the study is to assess policy impacts on landscape changes and on biodiversity, within a given modeling framework, the hypothesis is that structural change subverts political measurements. In order to protect ecosystems and therein living species active management is needed. Three political scenarios were built with varying degrees of environmental support. (1) The CAP I policy design, which mirrors the current ('CAP towards 2020') legal framework conditions, (2) the CAP II policy design with strengthened greening conditions, and (3) the 'nature-focused' policy design where DP are replaced by other premiums linked to environmental measurements based on ideas of the 'refocus' scenario of the European Commission (2011) and the NABU as described in Oppermann et al. (2016). Detailed political framework conditions of the baseline model and all three alternate political incentive schemes can be found in table 4.3.

The results of the first two policy designs confirmed the hypothesis and led to increasing landscape homogeneity accompanied with less species diversity. Fewer ES such as pollination and pest control can be expected within the research area. For an improved scenario, policy advise is required in order to prevent biodiversity declines. Quite the contrary, improving the conditions for species diversity is envisaged. This is why we included a third policy relevant incentive scheme. The results for the 'nature-focused' policy revealed that biodiversity and pollination services can be improved within our case study. However, what are the costs for farmers?

Farmers did not have significant monetary losses through the new CAP designs. Model results delivered comparable values for farmers' income than without the reform. In fact, farmers even profited from the CAP designs, especially after land leases took place. The average income level per farm increased since less farms than without the reform remained. However, all other farmers need to find job alternatives.

Regarding the economic performance of agricultural farms within the 'nature-focused' policy design, farmers' income was lower. In the 2nd iteration, the average income per farm was lower than in our reference point and could only be compensated through farm size optimization. Although, compared to CAP I and II, the income after structural change was still lower (-15%). This example shows the importance of where to put the reference point especially in political planning and decision making. If structural change is assumed in any case, farmers would suffer income losses under the governmental measurements of the 'nature-focused' policy design. However, as long as the income situation after structural change processes is not yet reached, any income improvement might be experienced positively. From a social perspective, the required farm and field size enlargements in order to compensate farmers' income losses might be easier to accept if biodiversity also profits from such political measurements.

In literature, the abolition of DP as it is also part of our 'nature-focused' policy design is discussed controversial (Rickard, 2015). There were other studies simulating farm exits due to reductions in DP. For example, Brady et al. (2012) found that under this policy regime structural change was strongest with highest shares of farm exits. This consequently resulted in negative impacts on landscape heterogeneity and biodiversity[1]. Similar results can be found in Uthes et al. (2011), where the abolition of DP led to high income losses, which further led to land abandonment and farm closings especially of small-scale and livestock keeping farms. In contrast to our policy scheme, in these studies, DP were reduced without cushioning farmers' income losses with alternative payment schemes.

In our case study, abolishing DP and replace them by well paid and ecologically effective environmental measures, led to structural change processes with less farms and larger fields, but with higher species diversity and pollination services. Structural change might probably be mitigated through this incentive scheme since compared to all other tested policy regimes more farms remained and fewer critical thresholds were reached. This has influences on other domains and scales. The socio-ecological system might be more stable and this might prevent undesired initiations of regime shifts at other domains and scales.

In all modeled policy designs crop diversity decreased. Crop diversity is linked with soil quality and soil erosion issues (Bullock, 1992; Thrupp, 2000). It needs to be considered in questions of food security and contributes to a high resilience of agricultural systems (FAO, 2011). Our model showed, that keeping dairy farms at place leads to a higher crop diversity. Dairy farmers could get additional governmental support, such that cattle keeping becomes more viable again. Another possibility could be to strengthen crop diversification regulations. The current crop diversification requirements do not effectively increase crop diversity (European Commission, 2016; Lakner and Bosse, 2016) and needs to be better targeted (European Parliament, 2016). Louhichi et al. (2017) found that especially livestock keeping farms were more affected than arable farms. Our results suggest to relieve dairy farms and enforce stricter crop diversification requirements for arable farms. They have more options to diversify their cropping patterns than livestock keeping farms have (Louhichi et al., 2017).

Especially in structural poor regions hedges have a positive impact on biodiversity (Batàry et al., 2010; Tscharnke et al., 2012). However, farmers would rather choose nitrogen-fixing crops than ecologically more beneficially landscape features such as hedges (Pe'er et al., 2016). According to Zinngrebe et al. (2017), farmers' EFA choices are motivated by several reasons such as economic issues (e.g. implementation costs), administrative, locational, and ecological considerations. Our

[1]They used a Shannon-Wiener index as biodiversity indicator.

model results showed that in any model only summer pea production as one of three alternative greening measures was chosen. Farmers did not decide to leave some of their land fallow or plant hedges. The two activities seemed to be of low attraction. Summer peas can be sold or used as fodder, while the other two alternatives not. Fallow land has the advantage that it saves labour, though this incentive was too low. Depending upon the hedge type, this greening measure can be used for wood or berries for own consumption. However, compared to all other land use activities, planting and maintaining hedges bears income losses and high costs. Hedges need to be maintained especially if they are young and in the growing period. Even if some labour would be invested instead of paying someone else, the costs would just get displaced. A possibility to support planting hedgerows could be to initiate programes, in which farmers just need to release some of their lands and dedicate it to hedge land. The farmer, though, is not obliged to contribute to the establishment and maintenance of hedges and at least does not need to bear any costs. Compensations are needed and perhaps the farmer gets, similar to the current AES for flowering strips, compensation payments for the losses of farmland (BMBU, 2010).

Within our case study and under the given model framework, the greening measurements of CAP 2020 were not able to reach pursued biodiversity targets of the European Commission. On contrary, due to structural change processes, biodiversity indicators showed to be worse. However, without the new CAP reform, biodiversity might have been even lower. There is a high political scope to control farm closures with incentive-based environmental measurements that compete with the risk of income loss due to volatile market prices. In the 'nature-focused' policy, for example, the average field sizes were lower and the amount of farms higher since alternative income possibilities were not driven as much by EoS measures. Either regulations to establish more semi-natural habitats such as hedges, fallow land, flowering fields, or a complete restructure of governmental incentive schemes could improve the conditions for a richer species diversity within our case study. Restructured political incentive schemes might include the abolition of DP. This however, needs to be compensated by incentive-based environmental measurements having positive effects on the habitat quality and farmers' incomes. Incentives to support dairy farms would increase the crop diversity of agricultural landscapes.

6.3. Modeling restrictions and development chances

6.3.1. Structural change

In order to assess the effects of structural change on agricultural land use patterns and landscape mosaics, the present model has taken account of farm and field sizes. With a methodological combination of linear and non-linear programming, linked together through iterations, the temporal scale of structural change has been addressed. Since the conceptual framework was comparatively static, it provided three theoretical points in time: (1) current cropping conditions, (2) after full land consolidation, and (3) after farm size optimization through land leases; the speed of structural change was not investigated. One possibility to explicitly account for the speed of structural change would be to include empirical knowledge about the annual change in farm sizes as a model constraint in the non-linear optimization model step. However, the methodological uncertainties of empirical operations would be an issue (Heckelei and Wolff, 2001). Structural change depends on several conditions that are difficult to predict even if statistical information exists (Happe et al., 2008; Mandryk et al., 2012). Empirical uncertainties paired with less model flexibility due to a higher amount of constraints would need to be brought in. Nevertheless, determination of the speed of structural change would be a meaningful task, since the time horizon plays a crucial role in setting political goals (Lai et al., 2017). The new CAP reform, for example, has a time frame until 2020. However, farm size changes calculated by our model may take longer. The model results for the CAP policy designs might be more reliable, i.e. if these policy designs will not change significantly for a longer time period than until 2020. Our results at least indicate directions of structural change and corresponding regime shifts under the ‘CAP towards 2020’ reform.

Other framework conditions might be changes in output or input prices, as well as climatic conditions affecting yield levels (Ewert et al., 2011). With the sensitivity analysis on grain prices, we already showed that through lower grain prices structural change will be faster. This is in line with findings of Mandryk et al. (2012), who measured the impact of price trends on structural change processes.

Ewert et al. (2011) estimated the price effects of climate and market changes on crop prices and yields for the EU member states. After initial assessments, however, the extent to which yield levels have changed within our study region has not yet been fully explored. Basically, in order to capture insecure climatic change possibilities, several scenarios exist but were not considered in this work. According to the authors, grain prices will increase mainly due to macroeconomic variables (e.g. GDP and world demand). Yield effects play only a minor role.

Commodity prices might also increase due to boosting energy prices (Wang et al., 2014). If energetic aspects are taken into account, questions about thresholds where the use of energy-intensive machinery will be replaced with more labour arise. Under such framework conditions and a declining agricultural energy productivity (Cleveland, 1995; Rydberg and Haden, 2006), it might be interesting if labour subsidies affect structural change. Furthermore, catch crops might become more competitive than fertilizer. In general, it might be interesting to explore under which input prices (labour and energy) healthy ecosystems become superior. According to McArthur and McCord (2017), agricultural inputs drive structural change and have strong impacts on agricultural labour patterns. The extent to which macroeconomic variables and international price developments impact farm size changes of several farm types and sizes within different regions needs to be further investigated (Landi et al., 2016).

Based on the background of yield and production uncertainties, it might be important to model risk aversion of farmers, especially in combination with farmland biodiversity assessments (Mouysset et al., 2013; Baumgärtner and Quaas, 2006). The argument for this is that farmers try to reduce their risk through diversification of land use. Mouysset et al. (2013) showed that the stronger the risk aversion is, the stronger the habitat heterogeneity becomes. However, the correlation between habitat heterogeneity and the ecological performance was weaker. The model was very coarse with respect to landscape characteristics. Nevertheless, it showed that incorporating risk into the decision model significantly altered results.

Crop rotations are also linked to risk mitigation strategies (Hauk et al., 2017). Including them would additionally equip our model with temporal aspects. Taking into account several crop rotation schemes may lead to different optimal field and farm sizes by diminishing the utilization of scale economies. Finely tuned crop rotation schemes might also increase yields (Schönhardt et al., 2009).

Especially with regard to uncertainties due to climatic change, farmers' risk aversion might also change (Woods et al., 2017). Farmers might want to diversify their cropping patterns through suitable crop rotation schemes. Again, reaching critical thresholds within the ecological domain might trigger regime shifts within the economic domain. Therefore, modeling risk aversion might reveal further insights into the systems' dynamics.

Structural change processes are driven by dynamic interactions between human population growth and improved technologies (Boserup, 1965). If the competitive pressure increases, farmers will be tempted to either abandon their agricultural business or expand it through modernization and investments. Technologies that are part of farmers' investment decisions, though, are not modeled within the scope of this work. Usually, farms are differently equipped with respect to machinery. In our model, though, all farmers had the same engine power of 120 kW. Higher machine power might shift labour requirements outwards, but only to the extent of the available production units. Our results showed that if a certain farm size was reached, no more off-farm labour was chosen by farmers[2]. If farms could improve their technologies, off-farm labour might become available again, or farms might get even larger (Romano and Traù, 2017; Eberhardt and Vollrath, 2016). Therefore our results need to be viewed under these bounded framework conditions of a certain unchangeable machine power. Investment decisions determine to a large extent the speed of farm size growth as well as the farm type development (Chavas, 2001; Boehlje, 1992). Increasing farm sizes are often accompanied with field consolidation in order to reduce production costs (Burton, 1988). Intensified agricultural production is highly mechanized and depends on advanced and expensive machinery (Pimentel, 2009). Therefore, large investments are needed in order to keep pace with free market developments that confront farmers nowadays. Once a farmer invests a large amount of money and takes the risk of an uncertain business, he needs to use the machinery in an optimal manner until amortization. In order to utilize the potential of big machinery in an adequate way, fields must become bigger and easier to work with (Zimmermann and Heckelei, 2012). This reduces the abundance of e.g. field margins, which often fulfill important ecological functions (Lanz et al., 2018). These kinds of negative impacts on ecosystem services are linked to lower plant productivity (Cardinale et al., 2012). Therefore, field and farm size decisions need to be captured by the model and related to investment decisions. Regime shifts at the 'patch' scale induces regime shifts at the economic farm scale and vise versa.

6.3.2. Land market

After land leases, our model showed that most farms had already reached maximal possible sizes since they fully took over neighbouring farms. On the one hand this means that the amount of new land a farm could rent was well defined: based on the considered machinery power, labour capacities were sufficient for new farm sizes under the condition of economies of size. However, predefined renting possibilities make the model less flexible and less realistic. There are different programming techniques for land leasings.

For example, Kantelhardt (2003) or Moosburger (1999) simulated rental markets based on farm models. Therefore, iterative processes were applied. Single farms and their shadow prices for land

[2]See for example farm 20 after farm size optimization.

were calculated independently from each other. In a second iteration, an equilibrium situation was modeled, where pre-calculated shadow prices determined the new tenant of the land that was set free. The new situation served as a starting point for further potential land rental processes. A similar approach was applied by Kapfer et al. (2015), who ranked farmers according to their economic strength and land according to its site quality. However, those approaches lack spatial references, as opposed to agent-based models.

A famous modeling technique applied in agent-based models includes auctions as simulated in Happe et al. (2004). Similar to a free market situation, an auctioneer (the market) allocates free land from each farmer who offers it to those who have intentions to rent additional land. In such a case, the shadow price for land and spatial attributes (e.g. distance between fields and farms) determine farmers' bids. This kind of land market where farmers compete individually for land is often found in ABM models (Berger, 2001; Schnicke et al., 2007; Schreinemachers and Berger, 2011, e.g.). If the land market would have been modeled using auction schemes, potential land rentals would have needed to be allocated to adjacent fields of the tenderers, or restrictions would be necessary so that tenderers only bid for land that lies next to their own fields. Otherwise, field size changes would be difficult to simulate since they are only possible if farmers cultivate a coherent area. If this auction mechanism were achieved, though, renting possibilities could also include the whole neighbourhood instead of only one neighbouring farm. Auction schemes require less assumptions implemented as constraints and would make the model more flexible. Neighbourhood relationships would not need to be defined explicitly prior to each iteration, which is time consuming.

We did not model land auctions due to our conceptual and methodological framework. Instead of modeling for yearly changes in farm sizes through taking into account shadow prices reflecting the current cropping patterns of each farm, longer time steps caused by long-term processes such as the question of finding a successor were addressed. These underlying processes were reflected in the farm size optimization process of a whole bunch of farms; it all depends on the market situation being at the mercy of hard competition conditions, and the farming community within our study area is seen in the long run as a group of collaborating farms with unbounded rationalities. Therefore, the total gross margin at the municipality level is maximized, and reductions in farm sizes cause increases in other farm sizes as it is done in the non-linear modeling part (3rd iteration). Within the linear modeling parts (short-term farming decisions), no farmer intends to rent out land. In our case therefore, an auction based land market would lead to no changes in the farm structure at all. As an alternative, though, it is conceivable that after the 4th iteration more iterations follow with different neighbourhood relationships (e.g. the opposite farm).

6.3.3. Spatial aspects

Our model delivers a novel form of spatial visualization in order to improve the interdisciplinary work within landscape research. Up until now, either raster-based landscape models (Happe et al., 2004; Weinmann, 2002, e.g.) or heavily stylized landscapes (Havlík et al., 2008; Cong et al., 2016, e.g.) exist. The former approach often lacks the possibility to include important agricultural landscape elements such as field margins or flowering strips, while the latter approach is rather technical and has not yet been applied to real agricultural regions.

One important feature of our model is the optimization of field sizes. Field sizes were conceptualized as driving variables inducing potential regime shifts. Within each linear model iteration, field sizes were simulated. This assumes that farmers have free choices on where to put their field boundaries. Through including some constraints due to biophysical and legal preconditions, they at least need to have a minimum number of fields. Results showed that farmers decided on fewer and bigger fields instead of many small fields. This decision is demanding since it is only possible to decide on the field sizes if the assumption of full consolidation holds true. At the moment, this type of spatial condition does not exist. However, there are political and technical endeavors to reach such a spatial allocation of agricultural land. Gritzmann et al. (2014) developed a mathematical software that ought to help farmers exchange fields in order to reach full consolidation. It already takes into account physical aspects such as soil qualities. Based on applications of the ProLand model, field consolidation is proposed in order to reduce production costs (Möller et al., 2002; Schroers, 2005). Historical observations also confirm the assumption of a trend to full consolidation. It can be observed in many parts of the world (Luo and Timothy, 2017). A pioneer example is the wheat belt region of Western Australia (Walker and Salt, 2006) as already mentioned in section 2.3.

If farmers lease agricultural lands, they lease whole fields instead of certain hectares. Similarly to Lauber (2006), the model should only allow for entire fields to be rented out. Therefore, field sizes would need a predefined value that requires a lot more data, programming, and working memory. The present study deliberately kept field sizes as variables to be simulated in order to optimize them. This procedure seemed to be plausible since a longer time span in which spatial processes can take place was covered. Furthermore, field consolidation and farm size optimization might occur simultaneously. In the long run, farmers who cultivate a huge agricultural area might indeed determine their field units. Field shapes, though, are not taken into account in our model. At the moment, all fields of a farmer have the same field length. They therefore are relatively slim. If it might be possible to at least divide the agricultural land of a farmer into two 'rows', our stylized land use maps would become more realistic since they would mirror the actual field shapes more acurately. An additional benefit would be that management possibilities with respect to the location of flowering strips could be realized. The minimum spanning tree calculation as it was done in Wossink et al. (1998), for example, might then be another interesting feature of our model. In doing so, also the allocation of fields within farms can be adapted. At the moment, FOLAS distributes fields within farms randomly.

The spatial visualization in FOLAS is rather restricted to intensively used agricultural lands. Our results do not necessarily apply to extensively used areas. In order to transfer our model to other regions, the landscape structures need to be observed. If, for example, the research site lies in an extensively used agricultural area with a high proportion of natural habitat (e.g. forest or conservation areas), physical or geographical landscape conditions might not allow perfect consolidation and the realization of scale economies would not be possible to a similar extent as in intensively used lands. In other words, income losses initiated by e.g. greening measures might not easily be compensated with technological progress. As field structures become more irregular with unused gaps inbetween, spatial simplification becomes more complex.

However, the stylized landscape visualization proves the usability of the model for other disciplines within intensively used agricultural areas. Spatial attributes play a crucial role in land management, especially when addressing ecological questions (Bailey et al., 2007; Hendrickx et al., 2007; Tscharnke et al., 2012; Hagen et al., 2012; Fahrig et al., 2011). With our spatially explicit modeling design, ecological information in terms of spatial indicators for biodiversity assessment was gained without abstracting from the farmer as the primary land user and decision maker.

6.3.4. Policy designs

In order to assess political impacts on farming decisions and the resulting landscape appearance, several scenarios were built (see section 4.2 for more details). We simulated effects of the greening mechanism of the new CAP reform using a different payment scheme than in the baseline model. The baseline model simulated the current policy of the calibration year (2011). In order to model the greening component of CAP 2020, a part of the total DP was linked to management practices in order to reach a given amount of EFAs. We made first attempts to simulate ecological management practices according to their weighting factors as fixed in the EU Regulation No. 1307/2013. Management practices were hedges, summer peas (as legumes), and fallow land.

Furthermore, within 'CAP towards 2020', redistributive payments ought to support small holders in allocating money to the first 46 ha. In our model this mechanism is included only in the linear modeling parts where no decision on farm size takes place. In order to measure the farm size effects of redistributive payments, this has to be modeled in the the non-linear part (3rd iteration). However, this causes problems with the GAMS solver used since endogenous variables (farm sizes) cannot be used as conditionals. As-if statements can only be used for exogenous variables (such as given parameters). Hence, the incentive scheme of the new CAP reform to support small holders is not fully taken into account within our model. This, however, might be important and alter our model results. To our knowledge, in the current literature, no other modeling example exists that takes into account redistributive small-scale support payments as they were implemented in Germany. Nevertheless, as the redistributive payments are relatively low, high influences are not expected.

Our 'nature-focused' policy design is based on the newest NABU study conducted by Oppermann et al. (2016). There, they considered participation in basic environmental measures[3] as voluntary and showed that 75% of all farms participated. Due to their results, ten percent of the arable and 20% of the grassland area was dedicated to such environmental measures. Our study assumes that all farms participate obligatorily with 10% of their areas in order to get a basic environmental premium. The same applies to the 'greening' component of CAP I and CAP II: in our model the participation in EFAs is an obligation, whereas in reality, it is voluntary. Additionally, in the NABU study, the cultivation of legumes and inter-cropping are not accepted in order to receive environmental premiums. In this study, however, legume production is accepted. Due to our results, it actually was the single land use type chosen by farmers in order to fulfill the requirements. The result of the NABU study further showed that farms that engaged the most in higher-value environmental measures[4] were better off than those that participated with less than five percent of their agricultural area. In contrast, our results revealed that small farms with a lower TGM rather participated in higher-value environmental measures than big farms with high TGMs. However, in contrast to the present work, the NABU study did not consider farm sizes, and the results are not directly comparable.

[3]In order to get the sustainability premium; these measurements are comparable with the EFA measurements of CAP 2020.

[4]Comparable to AES.

6.3.5. Ecological model interlinkages

With our stylized agricultural maps, we delivered the basis for the calculation of biodiversity indicators. However, we did not consider other non-agricultural landscape elements such as trees, conservation areas, forest edges, or ditches. These landscape elements are already part of the landscape and have high impacts on pollinator dispersion (Ricketts et al., 2008). When spatially explicit biodiversity indicators were calculated, already existing natural and semi-natural habitats need to be taken into account since biodiversity indicator assessments need to consider the space around agricultural fields in order to deliver reliable values (Dauber et al., 2003).

Peripheral areas of our land use maps lack information and therefore cannot be assessed with respect to indicator values. Therefore, surrounding landscape characteristics need to be explored as well. They might form the outer border of a research area with influences on biodiversity but be unchangeable geographical items (e.g. forests). Non-agricultural landscape characteristics might also be roads or other infrastructure that could have negative impacts on species habitats. In future work, more detailed non-agricultural landscape features need to be taken into account.

Moreover, different soil parameters could be included for the study area. Soil quality influence cropping decisions due to differences in yield levels. These could be modeled via yield functions that depend on several components such as precipitation, soils, and temperature as, for example, in Sheridan (2010). In addition to the classical yield function, however, ecosystem services such as pollination may also influence certain crop yields (Isaacs et al., 2017). If other farming practices such as drilling or soil conversion measurements were covered by the model, soil and water erosion could also be addressed. Then, the decision on a certain soil treatment feeds back on the production function. This can be reflected in the form of a nature production function influencing income and hence, farmers' decision making. Feedback loops that influence crop yields are linked to the economic domain via the 'patch' scale (see Figure 6.1). If at the ecological 'patch' scale, ES are low due to low biodiversity grounded in low landscape heterogeneity, low crop diversity, few semi-natural landscape elements, or other (missing) landscape features, yield levels might decrease. This in turn affects the economic domain since farm viability might also decrease due to lower and/or insecure income. Similar to other domain-scale combinations, such a regime shift might trigger changes in other domain-scale combinations at the regional scale. For example, more farms might quit the agricultural sector with further impacts on the labour market and on the rural population as already described above. Assessments of how the absence of ES at the patch scale affects the economic farm scale are urgently needed in order to estimate future regional landscape pattern developments (Cong et al., 2016).

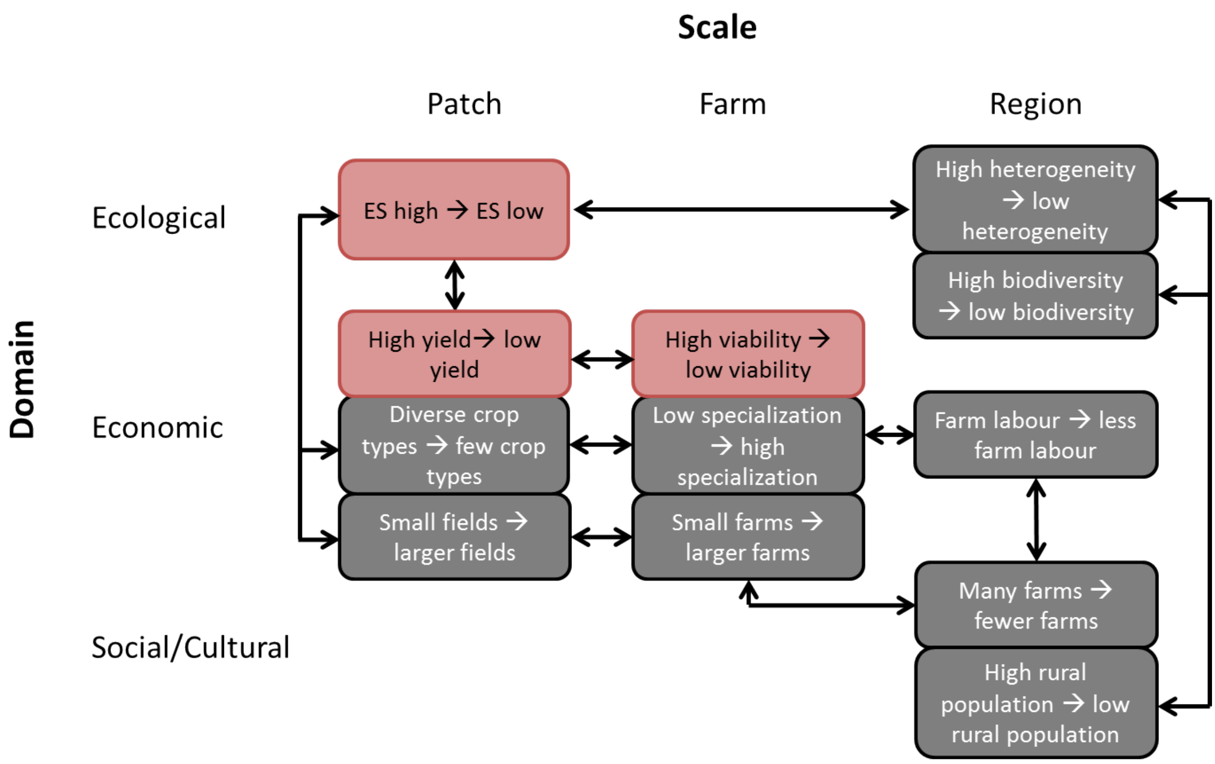

Figure 6.1.: Main variables of the possible interactions between different domains and scales including a feedback loop at the ecological patch scale. Figure adapted from Kinzig et al. (2006).

In any case, our model is predestined to implement feedback loops of flowering strips in order to design a landscape pattern bio-economic model including a nature production function. It is further capable to compare different policy designs and reveals insights into changes in the matrix quality and pollination services over time. Furthermore, in order to design spatially targeted policies for habitat conservation, spatially explicit habitat functions can be coupled with our economic model. If ecological feedbacks are integrated into the model, the following research questions can be raised:

- Is there a natural threshold regarding structural change?
- How can political measures be designed in order to improve spatial targeting?

Consequently, the spatially explicit modeling of landscape structures can result in win-win situations since optimal distributions in the sense of cost-saving implementations can be achieved (Wossink et al., 1998; Havlík et al., 2008; Bamière et al., 2011).

7 Chapter 7. Summary

Agro-ecosystems form a major part of today's habitats worldwide (FAO, 2017). Their main function is to deliver food and fibers in order to feed the growing population (Alexandratos and Bruinsma, 2012). However, agricultural land management is expected to provide multiple functions such as biomass production, the provision of cultural heritage, or mitigation of climate-relevant gases (De Groot, 2006; Kremen, 2005). Furthermore, land use within agro-ecosystems determines biodiversity and is driven by agricultural activities (Millenium Ecosystem Assessment, 2005). Agricultural ecosystems underlie permanent physical changes that affect species distribution, whilst political targets to protect biodiversity and ecosystem services fail (Pe'er et al., 2014; Leadley et al., 2014; European Commission, 2015). With its newest nature conservation offensive, the German federal agency for environmental protection acknowledged the malfunctions of reaching the German and European biodiversity targets (BMBU, 2015). Similarly, the mid-term review of the European Commission confirms the poor situation of ecosystems within the whole EU and especially in agriculturally related habitats (European Commission, 2015). This is why the BMBU (2015) proposed proofing the new CAP with respect to its efficacy in protecting ecosystems. They doubt that the new greening measures and cross-compliance obligations of the new CAP reform 2020 can contribute in order to reach the biodiversity goals and therefore urge changing the current governmental payment system.

For a political impact assessment on biodiversity issues, the whole landscape including its geological diversity (Gray, 2005) and biological characteristics needs to be considered (Parks and Mulligan, 2010). This comprises not only agricultural and silvicultural areas but also non-agricultural areas such as natural landscape elements or infrastructure. On top of these spatial scales, assessing political impacts on agricultural production and cultural landscape appearance needs to explore temporal and spatial farming decision scales (Cohen and Crowder, 2017). With respect to the past failures in meeting several political goals within the agricultural sector, a better interdisciplinary collaboration, especially between agronomists and landscape-oriented researchers is required (Banks, 2004). Therefore, ecologists and geographers, as well as economists interested in agro-ecosystems, need to work together since it is central to understand the spatial organization of farmers' decision making (Rizzo et al., 2013).

Collaboration, though, requires a systemic approach that considers all patterns and processes of socio-ecological systems. The concept of socio-ecological resilience shifts the way of thinking to a broader perspective of system interactions at multiple scales (Anderies et al., 2006). It considers all scales and domains of a socio-ecological system: the field, farm, and landscape scale, as well as the economic, ecological, and socio-cultural domain. Changes within one scale-domain combination can trigger other scale shifts (Kinzig et al., 2006), which might lead to a highly resistant and undesired new 'basins of attraction' (Matthews and Selman, 2006). Taking into account several domains and scales might improve political impact analyses and the assessment of future landscape developments (Kinzig et al., 2006; Anderies et al., 2006). For example, structural change processes within the agricultural sector take place slowly but most irreversibly influence the landscape mosaic in the form of field sizes, natural and semi-natural habitats, and cropping heterogeneity (Jones et al., 2016; Khoury et al., 2014; Hötker and Leuschner, 2014).

Most agro-environmental studies have not considered important structural change processes within the agricultural sector, although this had massive influences on landscape constitution during the last decades (Kenny, 2017). Long-term farming decisions contain not only cropping decisions or the question of participation in environmental measures, they also include investment decisions, branch considerations, and the difficulty finding a farm successor. Against the background of demographic changes and urban migration (Knickel et al., 2017), farming takeovers became a lifestyle decision (Manos et al., 2013).

FOLAS combined the agricultural land use decisions of farmers with economical principles of cost reductions, namely economies of size being a driver of structural change. We defined structural change as the increase in field sizes, the reduction in the amount of farms, and a less diverse farm type structure due to the reduction of farm types. We further used the concept of socio-ecological resilience in order to apply an impact assessment of the new CAP reform 2020 for a small research area. As described above, a switch from one to another basin of attraction is determined by the resilience of a SES. We applied a conceptual framework based on Kinzig et al. (2006), which allowed us to operationalize resilience thinking by defining critical threshold levels for three domain-scale combinations and tested, if the thresholds were crossed within the model runs. According to studies, threshold crossings indicated regime shifts and altered whole socio-ecological systems (Kinzig et al., 2006). Our specified thresholds were (1) the loss of farm types, (2) a defined reduction of farms, and (3) a certain level of field sizes. In addition, our study captures spatial aspects of farmers' decision making regarding farm and field sizes combined with land use changes affecting the landscape appearance. With our novel spatial approach we seek to improve the interdisciplinary work within landscape research. It simulates farmers' decisions at several scales. At the field scale it takes into account field sizes. Each farmer is represented by a certain consolidated cropping area on which field size and cropping decisions take place (the farm scale). Neighbouring farmers are linked together via land leases. Modeling farm interactions allowed us to simulate landscape changes at higher scales. Adapted to statistical information of the case study, several farm types and sizes were differentiated. In an iterative process, short and long-term farming decisions were simulated using a combination of linear mixed integer and non-linear programming models that underlie certain spatial and temporal assumptions (see section 2.4.2.2). The model results were visualized with a certain spatial outlay in a stylized rectangular form using GIS. Based on these maps, three different biodiversity indicators linked to landscape heterogeneity were applied (see section 4.3). This spatial model component forms a valuable interface between economic and ecological models.

FOLAS was applied to a small municipality of around 15 km^2 lying in the heart of one of the most intensively used agricultural area in Wetterau county in Hesse, Germany. Three different policy designs were tested: (1) CAP I, which is adapted to the new CAP 2020 reform; (2) CAP II with a stricter CAP 2020 reform, probably entering into force in 2018; and (3) a 'nature-focused' policy design with a completely different governmental payment structure based on incentive-based payments for environmental measurements. In order to test in how far the socio-ecological system of our case study was influenced by the CAP 2020 reform, all three scenarios were analyzed with respect to potential regime shifts within several domains and scales according to Kinzig et al. (2006). In a second step, we analyzed the political impacts on species diversity within our case study.

Results showed that in the CAP I and II policy designs, all critical thresholds indicating a scale-domain regime shift were crossed. A shift into another new regime seems to be inevitable, although it is less desired from a social and cultural perspective. All three biodiversity indicators had negative trends. The new 'CAP towards 2020' reform seems to be ineffective in reaching the biodiversity targets of the EC within our case study. On the contrary, it accelerated structural change and led to a more homogeneous landscape due to fewer farms and bigger fields. It has to be noted that the Simpson's diversity index improved compared to no policy reform. However, semi-natural habitats conglomerated in areas where already a larger amount of semi-natural habitats occurred. Since the abundance and distribution of semi-natural habitats determine pollinator dispersion (Tischendorf and Fahrig, 2000), species diversity will most probably decrease within our study area. In the CAP II policy design, resulting field sizes were highest. Therefore, we expect species' movements and the dispersion of pollinators to be lower after structural change processes since with increasing field sizes fewer field margins remained. Crop pollination might become difficult for some areas.

Within the 'nature-focused' policy design, two of three critical thresholds were reached. This policy scheme is less labour intensive and structural change processes were slower since fewer farms closed down. It had significant positive impacts on our chosen biodiversity indicators and seemed to be superior with respect to achieving the EC biodiversity targets. Due to incentive-based environmental measurements, many more semi-natural habitats in the form of flowering strips and fields have emerged. However, in order to compensate income losses of the 'nature-focused' policy design, structural change seem to be required (although of lesser magnitude) so that fewer farms use EoS and share the available amount of land.

In all policy designs, dairy farmers quit the agricultural sector, which led to a less diverse cropping pattern at the municipality level. Farm type losses and the resulting cropping homogeneity could not be compensated by either the crop diversity strategy of the new CAP reform or the implementation of EFAs. Since dairy farming requires more labour, the labour market at the regional scale could be affected. In consequence, the rural population might further shrink and rural areas become less attractive (Knickel et al., 2017). Therefore, it is advisable to maintain dairy farms. If this is pursued, though, particular support is needed. We found out that already small income reductions led to farm closures in the long run. More investigations are needed to assess the requirements to maintain dairy farming, if this is politically pursued. This is why we even more need a long-term perspective in political impact assessments.

The single environmental measurement for EFAs chosen by farmers was the cultivation of legumes. Planting hedges or leaving land fallow were not profitable. However, this unified picture may also be related to methodological issues regarding space. Soil quality determines cropping decisions as well. If, for example, biophysical information had been included, farming decisions might have been more diverse. Furthermore, hedges that already exist would most probably have been maintained. Therefore, fixed landscape elements lying adjacent to or within the agricultural area need to be considered in farming decisions and in the calculation of biodiversity indicators. But also spatial scales can be improved. In order to capture the temporal scale more explicitly, (1) implementing investment decisions, and (2) modeling crop rotation schemes are highly suggested.

Agricultural policy instruments are crucial in determining future ecosystem health and species diversity (Leventon et al., 2017; Bartolini and Viaggi, 2013; Gomez y Paloma et al., 2013). However, political measures and reforms are often spatially ill targeted since they need to cover a huge variety of different landscape structures (Marcos-Martinez et al., 2017; Leventon et al., 2017). A wise use of political regulations and incentives, as well as a clear idea of socio-cultural values has the potential to control future developments (Manos et al., 2013; Petrick and Zier, 2012). In order to understand the dynamics of agro-ecosystems, several scales and domains need to be considered so that interacting single regime-shifts are identified (Anderies et al., 2006; Berkes et al., 2003; Folke, 2006; Walker and Salt, 2006). FOLAS captures a long time frame and is adapted to the local and spatial production conditions. It therefore lays an important cornerstone for the development of spatially and temporally explicit bio-economic models.

How the agricultural area in Wöllstadt would look in the future will strongly depend on governmental incentives and legal framework conditions. These are affected by socio-cultural aspects that influence the political sphere and new institutions. In most studies, however, such aspects are neglected, and often detrimental one-way paths are detected too late (Matthews and Selman, 2006).

The introductory song lyric ought to remind us of the socio-cultural context of socio-ecological systems. After flowers, girls, men, and soldiers were gone, also their graveyards disappeared. The song asks *where* they went, and the answer is 'long time passing.' It seems that there is no reason to ask *why* they went, because the reason is known and somehow endured. Each verse ends with the beginning of the next one. Here, the scale-domain interrelations become obvious: slow changes in the reference systems of the socio-cultural domain might have the power to induce encompassing regime shifts of the whole socio-ecological system. Socio-ecological systems are dynamic, and their parts are interconnected. Changes in one part bring about changes in other parts, and seemingly insignificant shifts can trigger a whole system to change. Some changes within a society run very slowly and in the background, while others are obvious and fast (Kinzig et al., 2006). The structural change in agriculture is one of those developments that happens slowly with less notice from a large part of society. Structural change has impacts on the landscape and has already had severe impacts on species diversity (Khoury et al., 2014). The environment creates feedback if certain thresholds are crossed or if the ecosystem is degraded (Pearce, 2007). However, action might be too late, which is why we need spatially and temporally in-depth political impact assessments in order to protect public goods.

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A Appendix A.

Data: Landscape transformation

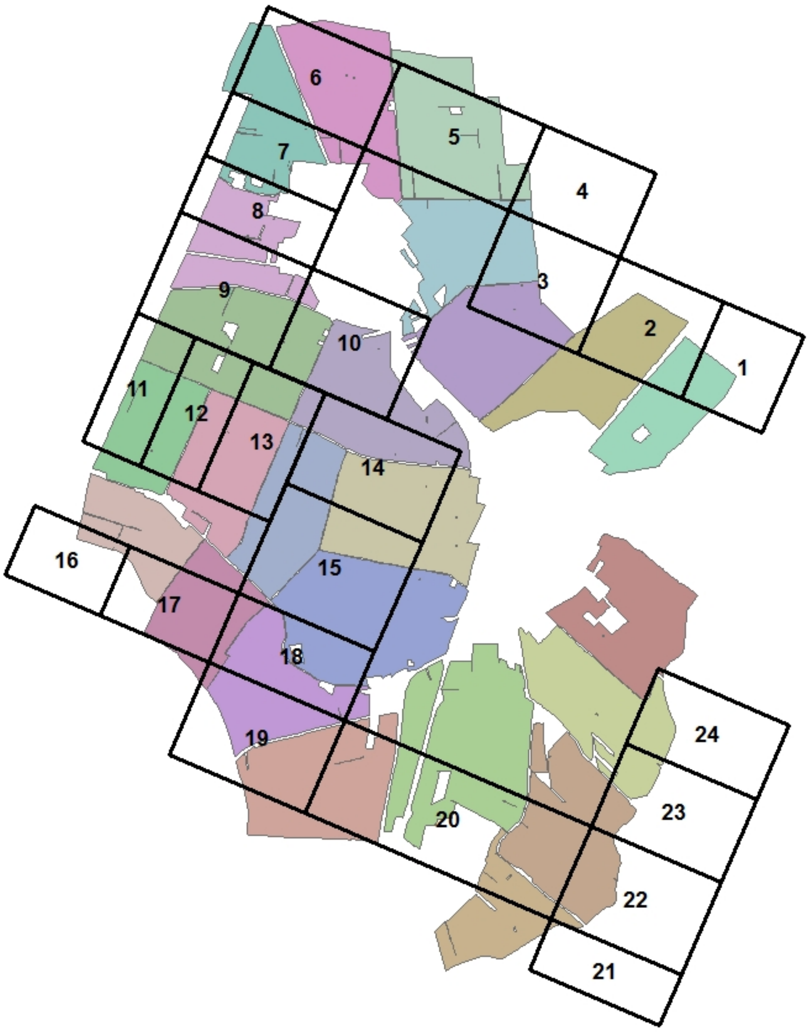


Figure A.1.: Landscape transformation via GIS: aggregated farms and rectangular stylized decision units within the study area.

Appendix B.

B Data: Model parameters

Table B.1.: Model parameters: scalars and their units, values, and references.

Model parameter	Unit	Value	Source
Off-farm wage	Euros per hour	9.8	Hessisches Statistisches Landesamt (2014b)
Family labour	hours per year	997	Hessisches Statistisches Landesamt (2012i)
Farm wage (permanent)	Euros per hour	15.5	LLH (2015)
Farm wage (seasonal)	Euros per hour	7.4	LLH (2015)
AES budget	Euros per year	1,027	BLE (2014)
Payments for AES	Euros per ha	750	HMUKLV (2015b)
Variable costs AES	Euros per ha	160	Calibration
Labour input coefficient AES	Euros per ha	2.78	Ktbl (2013a)
Capital input coefficient AES	Euros per ha	10.41	Ktbl (2013a)
Variable costs hedges	Euros per ha	5,407	Calculated on the base of Beiersdorf (2012)
Direct payments	Euros per ha	275.6	Hessisches Statistisches Landesamt (2012i)
Land rent	Euros per ha	214	Hessisches Statistisches Landesamt (2012c)
Rented land	Percentage	64	Hessisches Statistisches Landesamt (2012j)
Dry matter requirements cows	dt per year and stable place	60.6	Meyer (2005)
Energy requirements pigs	GJ per year and stable place	10.4	GfE (2006)
Protein requirement pigs	kg per year and stable place	78.5	GfE (2006)

Appendix C. Data: Farm sizes

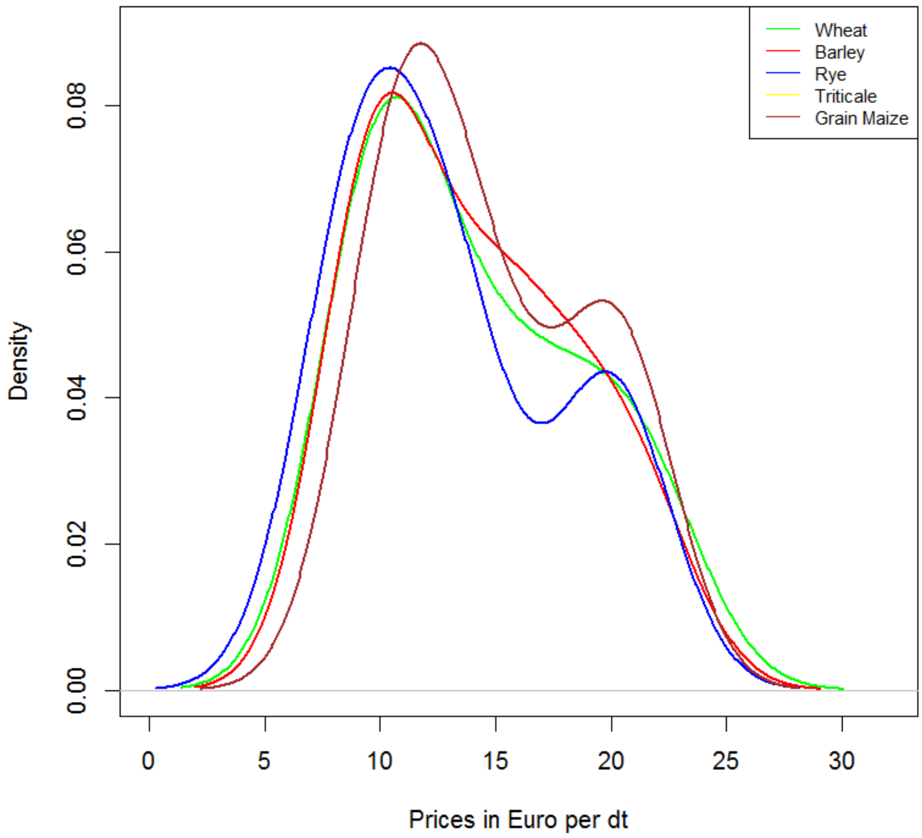
Table C.1.: Initial farm sizes of the baseline model and field lengths of the stylized rectangular farms.

Farm	Field length [m]	Farm size [ha]
1	671.63	37.64
2	671.63	48.82
3	774.68	52.03
4	774.68	45.81
5	591.34	59.95
6	591.34	54.54
7	922.30	40.98
8	922.30	36.68
9	702.50	64.79
10	702.50	57.7
11	900.00	36.41
12	900.00	45.3
13	900.00	43.71
14	950.00	60.48
15	950.00	72.92
16	482.63	32.03
17	482.63	37.02
18	482.63	45.85
19	647.05	61.47
20	647.05	111.31
21	910.90	32.49
22	910.90	58.94
23	910.90	52.80
24	910.90	47.41

D

Appendix D.

Data: Grain price volatilities



H

Figure D.1.: Density functions for grain prices: Darmstadt district (2000/01 - 2014/15). Source: Ktbl (2016).

E

Appendix E.

Data: Crop prices

Table E.1.: Crop prices used as parameters for the model in Euros/dt, their year of collection, regional scale, and references.

Activity	Price [Euros/dt]	Year	Regional scale	Source
Winter wheat	20	2010/11	Darmstadt district	Ktbl (2016)
Sugar beets	2.98	2010/11	Darmstadt district	Ktbl (2016)
Rapeseed	40.72	2010/11	Darmstadt district	Ktbl (2016)
Winter barley	17.8	2010/11	Darmstadt district	Ktbl (2016)
Permanent grassland - grass silage	2.594	2011	Hesse	LLH (2012)
Permanent grassland - hay	9	-	-	Internal farm price*
Potatoes	25.08	2010/11	Darmstadt district	Ktbl (2016)
Winter rye	19.15	2010/11	Darmstadt district	Ktbl (2016)
Fallow	0	-	-	-
Silo maize	2.8	2010/11	Darmstadt district	Ktbl (2016)
Silo maize for biogas plants	3.75	2011	Hesse	LLH (2012)
Corn cob mix	8.7	2012	Germany	Ktbl (2013a)
Corn maize	16.2	2015	Hesse	LLH (2015)
Weeds - grass silage	6.8	2011	Hesse	LLH (2012)
Triticale	17.7	2010/11	Darmstadt district	Ktbl (2016)
Summer peas	17.82	2010/11	Darmstadt district	Ktbl (2016)
Dairy	2,368**	2012	Germany	Ktbl (2013b)
Pig fattening	1,047.2***	2012	Germany	Ktbl (2013b)

*calibrated

**revenue (including milk, slaughter, and calves per stable place per year)

***revenue per stable place per year

Appendix F. Data: Fodder crops

Table F.1.: Fodder crops and their nutritional contents. Source: LfL (2017).

Fodder crop	Energy [GJ/dt]	Proteins [kg/dt]	RNB	Dry matter [kg/dt]
Winter wheat	0.7421	14.79	-5	0.87
Winter barley	0.7038	14.18	-6	0.87
Permanent grassland - grass silage	0.2448	5.4	3	0.4
Permanent grassland - hay	0.4764	11.09	2	0.86
Winter rye	0.7325	14.09	-8	0.87
Silo maize	0.2352	4.69	-8	0.35
Corn cob mix	0.4764	9.54	-9	0.6
Corn maize	0.7291	14.62	-10	0.87
Weeds - grass silage	0.1957	4.59	3	0.17
Triticale	0.7212	14.09	-6	0.87
Summer peas	0.7378	16.18	12	0.87

Appendix G.

G Data: Crop yields

Table G.1.: Yield levels used as parameters for the model in dt per ha, their year of collection, regional scale, and references.

Crop	Yield [dt/ha]	Year	Regional scale	Source
Winter wheat	79.6	2011	Wetterau county	Hessisches Statistisches Landesamt (2012d)
Sugar beets	791	2011	Wetterau county	Hessisches Statistisches Landesamt (2012d)
Rapeseed	38.5	2011	Wetterau county	Hessisches Statistisches Landesamt (2012d)
Winter barley	69.8	2011	Wetterau county	Hessisches Statistisches Landesamt (2012d)
Permanent grassland - grass silage	280	2011	Hesse	LLH (2012)
Permanent grassland - hay	57.5	2012	Germany	Ktbl (2013a)
Potatoes	444.8	2011	Wetterau county	Hessisches Statistisches Landesamt (2012d)
Winter rye	58.3	2010/11	Darmstadt district	Ktbl (2016)
Fallow	0	-	-	-
Silo maize	473	2010/11	Darmstadt district	Ktbl (2016)
Silo maize for biogas plants	500	2011	Hesse	LLH (2012)
Corn cob mix	94.8	2011	Hesse	Hessisches Statistisches Landesamt (2012d)
Corn maize	93.7	2010/11	Darmstadt district	Ktbl (2016)
Weeds - grass silage	360	2011	Hesse	LLH (2012)
Triticale	60	2010/11	Darmstadt district	Ktbl (2016)
Summer peas	30.9	2011	Hesse	Hessisches Statistisches Landesamt (2012d)

Appendix H.
Data: Labour coefficients

Table H.1.: Labour coefficients of the model activities in Akh/ha or stable place and year for different field sizes and livestock keeping activities. Source: Ktbl (2013a,b)

Model activity	2 ha	5 ha	10 ha	per stable place and year
Winter wheat	7.67	5.75	5.13	-
Sugar beets	5.91	4.56	4.06	-
Rapeseed	6.64	4.96	4.39	-
Winter barley	7.13	5.41	4.83	-
Permanent grassland - grass silage	12.09	10.42	10.42	-
Permanent grassland - hay	10.13	8.47	7.62	-
Potatoes	28.1	24.57	23.31	-
Winter rye	6.58	4.97	4.44	-
Fallow*	0.66	0.66	0.66	-
Silo maize	10.58	9.33	8.52	-
Silo maize for biogas plants	9.79	8.57	7.78	-
Corn cob mix	7.07	6.04	5.65	-
Corn maize	6.66	5.39	4.93	-
Weeds - grass silage	19.2	17.03	16.23	-
Triticale	6.69	5.06	4.53	-
Summer peas	5.9	4.51	3.99	-
Dairy	-	-	-	44.33
Pig fattening	-	-	-	1.02

*maintenance: mulching (5.5 m working width)

Appendix I.

Data: Proximity functions

Table I.1.: Proximity functions for each crop: machine costs per labour hour. Calculated based on data from Ktbl (2013a).

Crop	Machine costs per labour hour	R^2
Winter wheat	$f(x) = 12.975x + 106.97$	0.9998
Sugar beets	$f(x) = 18.365x + 147.93$	0.9988
Rapeseed	$f(x) = 13.386x + 117.98$	0.9999
Barley	$f(x) = 13.1x + 103.84$	0.9999
Potatoes	$f(x) = 12.465x + 302.02$	0.9995
Maize (silage)	$f(x) = 11.036x + 142.41$	0.9945
Corn cob mix	$f(x) = 13.673x + 115.61$	0.9998
Permanent grassland (silage)	$f(x) = 10.625x + 319.16$	0.9997
Permanent grassland (hay)	$f(x) = 10.865x + 161.16$	0.9998
Grasses (silage)	$f(x) = 11.535x + 549.36$	0.9995
Triticale	$f(x) = 13.64x + 97.961$	0.9992
Rye	$f(x) = 13.731x + 99.393$	0.9993
Maize (silage biogas)	$f(x) = 10.714x + 152.57$	0.9748
Corn maize	$f(x) = 12.649x + 375.21$	0.9989
Summer peas	$f(x) = 14.408x + 98.955$	0.9996

Table I.2.: Logarithmic proximity functions for each crop: labour hours per field size. Calculated based on data from Ktbl (2013a).

Crop	Labour hours per field size	R^2
Winter wheat	$f(x) = -1.833ln(x) + 9.5183$	0.8956
Sugar beets	$f(x) = -1.257ln(x) + 7.1173$	0.9053
Rapeseed	$f(x) = -1.58ln(x) + 8.2004$	0.899
Barley	$f(x) = -1.637ln(x) + 8.7727$	0.8952
Potatoes (permanent employees)	$f(x) = -2.733ln(x) + 20.646$	0.8892
Potatoes (seasonal workers)	$f(x) = -0.926ln(x) + 11.3$	0.928
Maize (silage)	$f(x) = -1.382ln(x) + 12.061$	0.872
Corn cob mix	$f(x) = -0.988ln(x) + 8.0582$	0.8944
Permanent grassland (silage)	$f(x) = -1.631ln(x) + 13.832$	0.8657
Permanent grassland (hay)	$f(x) = -1.58ln(x) + 11.715$	0.893
Grasses (silage)	$f(x) = -2.14ln(x) + 21.487$	0.8661
Triticale	$f(x) = -1.534ln(x) + 8.2179$	0.8962
Rye	$f(x) = -1.515ln(x) + 8.0837$	0.8979
Maize (biogas)	$f(x) = -1.349ln(x) + 11.253$	0.8637
Corn maize	$f(x) = -1.241ln(x) + 7.9384$	0.8871
Summer peas	$f(x) = -1.334ln(x) + 7.2316$	0.8962

J

Appendix J.

Data: EoS Approximations

Table J.1.: Labour requirement equations for each crop: labour hours per ha approximated for an average field size of 3 ha. Calculations based on data from Ktbl (2013a).

Crop	equation
Winter wheat	$labreq\_wheat_{g,f} \geq 5.67154367 + 5.499 \div Xfix_{g,f,'wheat'} \cdot gwnn\_size_g$
Sugar beets	$labreq\_sugarbeets_{g,f} \geq 4.479344353 + 3.771 \div Xfix_{g,f,'sugarbeets'} \cdot gwnn\_size_g$
Rapeseed	$labreq\_rapeseeds_{g,f} \geq 4.88459258 + 4.74 \div Xfix_{g,f,'rapeseeds'} \cdot gwnn\_size_g$
Barley	$labreq\_barley_{g,f} \geq 5.33727168 + 4.911 \div Xfix_{g,f,'barley'} \cdot gwnn\_size_g$
Potatoes (permanent employees)	$labreq\_pP_{g,f} \geq 14.9104926 + 8.199 \div Xfix_{g,f,'pP'} \cdot gwnn\_size_g$
Potatoes (seasonal workers)	$labreq\_sP_{g,f} \geq 11.391315 + 2.778 \div Xfix_{g,f,'sP'} \cdot gwnn\_size_g$
Maize (silage)	$labreq\_maizesilage_{g,f} \geq 9.16071782 + 4.146 \div Xfix_{g,f,'maizesilage'} \cdot gwnn\_size_g$
Corn cob mix	$labreq\_ccm_{g,f} \geq 4.741644014 + 4.635 \div Xfix_{g,f,'ccm'} \cdot gwnn\_size_g$
Permanent grassland (silage)	$labreq\_pGsilage_{g,f} \geq 10.9904571 + 3.15 \div Xfix_{g,f,'pGsilage'} \cdot gwnn\_size_g$
Permanent grassland (hay)	$labreq\_pGhay_{g,f} \geq 8.93065432 + 3.144 \div Xfix_{g,f,'pGhay'} \cdot gwnn\_size_g$
Grasses (silage)	$labreq\_grasses_{g,f} \geq 17.7990568 + 4.014 \div Xfix_{g,f,'grasses'} \cdot gwnn\_size_g$
Triticale	$labreq\_triticale_{g,f} \geq 5.50782379 + 3.078 \div Xfix_{g,f,'triticale'} \cdot gwnn\_size_g$
Rye	$labreq\_rye_{g,f} \geq 5.40770575 + 3.039 \div Xfix_{g,f,'rye'} \cdot gwnn\_size_g$
Maize (silage biogas)	$labreq\_maizeBG_{g,f} \geq 8.9979838 + 2.343 \div Xfix_{g,f,'maizeBG'} \cdot gwnn\_size_g$
Corn maize	$labreq\_corn_{g,f} \geq 5.73777123 + 2.532 \div Xfix_{g,f,'corn'} \cdot gwnn\_size_g$
Summer peas	$labreq\_peas_{g,f} \geq 4.87443923 + 2.679 \div Xfix_{g,f,'peas'} \cdot gwnn\_size_g$

K

Appendix K.

Data: Financial and cost coefficients

Table K.1.: Financial and cost coefficients of the model activities. Source: Ktbl (2013a,b)

Activity	Financial coefficients [Euro/ha]	Variable costs [Euro/ha]	Direct costs [Euro/ha]	Machine cost coefficients [Euro/ha]
Winter wheat	64.76	992.08	766.47	12.975
Sugar beets	61.14	1371.84	1096.10	18.365
Rapeseed	61.21	987.64	759.45	13.386
Winter barley	58.38	799.08	593.03	13.100
Permanent grassland - grass silage	50.5	834.25	386.22	10.625
Permanent grassland - hay	32.14	295.37	100.37	10.865
Potatoes	328.88	3270.81	2584.28	12.465
Winter rye	56.32	740	536.19	13.731
Fallow	2.97	638.04*	638.04	-
Silo maize	97.22	867.5	620.45	11.036
Silo maize for biogas plants	63.86	714.58	435.15	10.714
Corn cob mix	68.68	834.53	506.58	13.673
Corn maize	61.69	951.85	488.32	12.649
Weeds - grass silage	79.85	915.67	501.94	11.535
Triticale	56.32	737.86	532.13	13.640
Summer peas	54.4	535.21	349.36	14.408
Dairy	59.11	1032	-	-
Pig fattening	3.77	1016.4	-	-

*Opportunity costs for wheat production plus machine costs for mulching

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Appendix L.

Results: Land use patterns

Table L.1.: Baseline land use results of the linear programming iterations as a % of the cropping area at the municipality level compared to statistical data of the calibration year (received from HLUG).

Crop [%]	1st iteration	*Statistics 2011*	2nd iteration	4th iteration
Wheat	50.0	*51.7*	50.0	50.0
Sugar beets	16.4	*16.4*	16.4	16.4
Rapeseed	22.3	*10.6*	22.3	24.2
Barley	0.7	*7.9*	0.7	0.7
Potatoes	2.3	*2.3*	2.3	2.3
Rye	-	*1.3*	0	0
Silage maize	0.5	*0.6*	0.5	-
Silage maize (biogas)	1.0	*0.6 (n.d)*	1.0	1.0
Corn maize	0.9	*0.4*	0.9	0.3
Triticale	-	*0.2*	-	-
Corn cob mix	-	-	-	-
Summer peas	-	-	-	-
Grass silage (permanent grassland)	1.1	*n.d (5 %)*	1.1	-
Hay (permanent grassland)	3.9	*n.d (5 %)*	3.9	5.0
Weed silage	0.8	-	0.8	-
Summer peas	-	-	-	-
Flowering strips	0.11	*n.d*	0.11	0.11

Table L.2.: Land use results of the linear programming iterations as a % of the cropping area at the municipality level for the sensitivity analysis on grain prices.

Crop [%]	1st iteration	2nd iteration	4th iteration
Wheat	0.2	0.2	0.1
Sugar beets	16.4	16.4	16.4
Rapeseed	25.0	25.0	25.0
Barley	0.7	0.7	0.4
Potatoes	2.3	2.3	2.3
Silage maize	0.4	0.4	-
Silage maize (biogas)	1.0	1.0	1.0
Corn maize	46.7	48.1	49.6
Corn cob mix	-	-	-
Grass silage (permanent grassland)	0.8	1.2	-
Hay (permanent grassland)	4.2	3.8	5.0
Weed silage	0.4	0.8	-
Summer peas	1.7	-	-
Flowering strips	0.11	0.11	0.11

Table L.3.: Land use results of the linear programming iterations as a % of the cropping area at the municipality level for the sensitivity analysis on higher and lower land rent prices.

Crop [%]	1st iteration		2nd iteration		4th iteration	
	High	Low	High	Low	High	Low
Wheat	50	50	50	50	50	50
Sugar beets	16.4	16.4	16.4	16.4	16.4	16.4
Rape seeds	22.3	22.3	22.3	22.3	24.2	24.5
Barley	0.7	0.7	0.7	0.7	0.7	0.5
Potatoes	2.3	2.3	2.3	2.3	2.3	2.3
Silage maize	0.5	0.5	0.5	0.5	-	-
Silage maize (biogas)	1.0	1.0	1.0	1.0	1.0	1.0
Corn maize	0.9	0.9	0.9	0.9	0.3	0.2
Corn-Cobb mix	-	-	-	-	-	-
Grass silage (permanent grassland)	1.1	1.1	1.1	1.1	-	-
Hay (permanent grassland)	3.9	3.9	3.9	3.9	5	5
Weed silage	0.8	0.8	0.8	0.8	-	-
Summer peas	-	-	-	-	-	-
Flowering strips	0.11	0.11	0.11	0.11	0.11	0.11

Table L.4.: Land use results of the 2nd and 4th iterations for all three scenarios as a percentage of the agricultural area at the municipality level.

	CAP I		CAP II		'Nature-focus'	
Crop type	2nd iteration	4th iteration	2nd iteration	4th iteration	2nd iteration	4th iteration
Wheat	49.8	50.0	49.8	50	46.8	50
Sugar beets	16.4	16.4	16.4	16.4	16.4	16.4
Rape seeds	14.7	16.3	12.0	13.6	3.9	2.4
Barley	0.7	0.7	0.7	0.5	0.7	0.7
Potatoes	2.3	2.3	2.3	2.3	2.3	2.3
Rye	-	-	-	-	-	-
Silage maize	0.6	-	0.6	-	0.6	-
Silage maize (biogas)	2.0	2.0	2.0	2.0	2.0	2.0
Corn maize	0.4	0.03	0.3	0.06	0.3	0.06
Triticale	-	-	-	-	-	-
Corn-Cobb mix	-	-	-	-	-	-
Grass silage (permanent grassland)	0.9	-	0.8	-	0.8	-
Hay (permanent grassland)	4.1	5.0	4.1	5.0	4.2	5.0
Weed silage	0.8	-	0.8	-	0.8	-
Summer peas	7.1	7.1	10.0	10.0	10.0	10.0
Flowering strips/fields	0.09	0.09	0.09	0.09	11.1	11.1

Appendix M.

Results: Grassland farms

Table M.1.: Farms with permanent grassland in ha and distribution to farm types. Results are the same for the baseline model as well as for the sensitivity analyses on lower and higher land rents.

Farm	Grassland area [ha]	Farm type
Farm 9	17.4	Dairy producing farm
Farm 17	9.7	Dairy producing farm
Farm 19	17.4	Dairy producing farm
Farm 22	17.4	Dairy producing farm

Table M.2.: Farms with permanent grassland in ha from the 1st iteration of the sensitivity analysis on grain prices and distribution to farm types.

Farm	Grassland area [ha]	Farm type
Farm 2	1.3	Arable farm
Farm 6	2.4	Arable farm
Farm 9	10.5	Dairy producing farm
Farm 17	14.6	Dairy producing farm
Farm 19	14.0	Dairy producing farm
Farm 21	1.1	Pig farm
Farm 22	17.9	Dairy producing farm

Appendix N.

Results: Total gross margins

Table N.1.: Total gross margins (TGMs) of the 1st and 2nd iterations of the baseline model per farm in Euros.

Farm	Farm size [ha]	TGM 1st iteration [Euros]	TGM 2nd iteration [Euros]
1	37.7	38,435	42,942
2	48.8	70,337	47,772
3	52.0	48,625	118,199
4	45.8	44,657	45,428
5	60.0	70,947	76,529
6	54.5	51,306	52,224
7	41.0	40,979	41,668
8	36.7	37,013	38,321
9	64.8	66,526	66,871
10	57.7	52,944	53,920
11	36.4	79,109	37,347
12	45.3	66,473	45,031
13	43.7	50,090	43,030
14	60.5	63,314	56,848
15	72.9	65,305	164,294
16	32.0	34,162	34,701
17	37.0	50,528	51,537
18	45.9	68,075	41,571
19	61.5	63,010	64,727
20	111.3	93,419	96,427
21	32.5	33,743	34,294
22	58.9	61,654	64,710
23	52.8	96,755	50,870
24	47.4	44,693	46,674

Table N.2.: Total gross margins (TGMs) of the 4th iteration of the baseline model per farm in Euros.

Farm	TGM [Euros]	Farm size [ha]
1	73,147	74.4
2	14,182	12.1
3	93,124	97.8
4	0	0
5	107,011	114.5
6	0	0
7	43,869	43.1
8	36,560	34.5
9	0	0
10	100,023	122.5
11	20,152	18.1
12	163,357	74.8
13	32,961	32.5
14	0	0
15	122,652	133.4
16	62,471	73.6
17	0	0
18	97,889	41.3
19	0	0
20	183,990	172.8
21	76,820	91.43
22	0	0
23	53,637	55.5
24	44,474	44.7

Appendix O.

O Results: Feeding practices

Table O.1.: Feeding practices of all pig fattening farms for the linear programming iterations.

Farm	Wheat [dt]	Barley [dt]	Corn maize [dt]	Hay [dt]
1st iteration				
3	210.3	118.3	99.7	-
10	210.3	118.3	99.7	-
11	210.3	118.3	99.7	-
13	210.3	118.3	99.7	-
21	210.3	118.3	99.7	-
2nd iteration				
3	210.3	118.3	99.7	-
10	210.3	118.3	99.7	-
11	210.3	118.3	99.7	-
13	210.3	118.3	99.7	-
21	210.3	118.3	99.7	-
4th iteration				
3	210.3	118.3	99.7	-
10	210.3	118.3	-	152.7
11	210.3	118.3	99.7	-
13	210.3	118.3	99.7	-
21	210.3	118.3	-	152.7

Table O.2.: Feeding practices of all dairy farms for the linear programming iterations.

Farm	Wheat [dt]	Silage maize [dt]	Grass silage [dt]	Hay [dt]	Weed silage [dt]	Corn maize [dt]
1st iteration						
9	34.7	820.9	796.7	835.9	891.2	-
17	34.7	195.1	1,344.3	281.8	891.2	554.1
19	34.7	820.9	796.7	835.9	891.2	-
22	34.7	820.9	796.7	835.9	891.2	-
2nd iteration						
9	34.7	820.9	796.7	835.9	891.2	-
17	34.7	195.1	1,344.3	281.8	891.2	554.1
19	34.7	820.9	796.7	835.9	891.2	-
22	34.7	820.9	796.7	835.9	891.2	-
4th iteration						
9	-	-	-	-	-	-
17	-	-	-	-	-	-
19	-	-	-	-	-	-
22	-	-	-	-	-	-

Appendix P. Results: Labour

Table P.1.: Off-farm and hired labour per farm for the 1st iteration of the baseline model.

Farm	Farm size [ha]	Off-farm labour [hours]	Hired labour [hours]
1	37.64	734	-
2	48.82	579	-
3	52.03	600	-
4	45.81	677	-
5	59.95	527	-
6	54.54	616	-
7	40.98	711	-
8	36.68	733	-
9	64.79	-	673
10	57.7	561	-
11	36.41	569	-
12	45.3	590	-
13	43.71	633	-
14	60.48	550	-
15	72.92	488	-
16	32.03	773	-
17	37.02	-	453
18	45.85	599	-
19	61.47	-	656
20	111.31	208	-
21	32.49	737	-
22	58.94	-	635
23	52.80	473	-
24	47.41	653	-

Table P.2.: Off-farm and hired labour per farm for the 2nd iteration of the baseline model.

Farm	Farm size [ha]	Off-farm labour [hours]	Hired labour [hours]
1	37.64	783	-
2	48.82	740	-
3	52.03	485	-
4	45.81	756	-
5	59.95	622	-
6	54.54	710	-
7	40.98	781	-
8	36.68	804	-
9	64.79	-	563
10	57.7	661	-
11	36.41	772	-
12	45.3	759	-
13	43.71	734	-
14	60.48	679	-
15	72.92	322	-
16	32.03	828	-
17	37.02	-	387
18	45.85	710	-
19	61.47	-	545
20	111.31	412	-
21	32.49	793	-
22	58.94	-	527
23	52.80	719	-
24	47.41	748	-

Table P.3.: Off-farm and hired labour per farm for the 4th iteration of the baseline model.

Farm	Farm size [ha]	Off-farm labour [hours]	Hired labour [hours]
1	73,147	647	-
2	14,182	940	-
3	93,124	507	-
4	0	-	-
5	107,011	461	-
6	0	-	-
7	43,869	795	-
8	36,560	835	-
9	0	-	-
10	100,023	327	-
11	20,152	870	-
12	163,357	386	-
13	32,961	812	-
14	0	-	-
15	122,652	373	-
16	62,471	586	-
17	0	-	-
18	97,889	645	-
19	0	-	-
20	183,990	0	-
21	76,820	476	-
22	0	-	-
23	53,637	737	-
24	44,474	787	-

Table P.4.: Off-farm and hired labour per farm for the 1st iteration of the sensitivity analysis on grain prices.

Farm	Farm size [ha]	Off-farm labour [hours]	Hired labour [hours]
1	37.64	749	-
2	48.82	671	-
3	52.03	621	-
4	45.81	697	-
5	59.95	587	-
6	54.54	620	-
7	40.98	731	-
8	36.68	753	-
9	64.79	-	7
10	57.7	309	-
11	36.41	566	-
12	45.3	695	-
13	43.71	679	-
14	60.48	599	-
15	72.92	470	-
16	32.03	789	-
17	37.02	107	-
18	45.85	648	-
19	61.47	-	600
20	111.31	266	-
21	32.49	675	-
22	58.94	-	593
23	52.80	651	-
24	47.41	690	-

Table P.5.: Off-farm and hired labour per farm for the 2nd iteration of the sensitivity analysis on grain prices.

Farm	Farm size [ha]	Off-farm labour [hours]	Hired labour [hours]
1	37.64	798	-
2	48.82	506	-
3	52.03	700	-
4	45.81	762	-
5	59.95	690	-
6	54.54	701	-
7	40.98	780	-
8	36.68	809	-
9	64.79	-	509
10	57.7	671	-
11	36.41	731	-
12	45.3	765	-
13	43.71	735	-
14	60.48	677	-
15	72.92	624	-
16	32.03	827	-
17	37.02	-	387
18	45.85	586	-
19	61.47	-	507
20	111.31	427	-
21	32.49	673	-
22	58.94	-	507
23	52.80	727	-
24	47.41	754	-

Table P.6.: Off-farm and hired labour per farm for the 4th iteration of the sensitivity analysis on grain prices.

Farm	Farm size [ha]	Off-farm labour [hours]	Hired labour [hours]
1	-	-	-
2	86.46	329	-
3	23.51	856	-
4	74.33	650	-
5	114.49	461	-
6	-	-	-
7	-	-	-
8	77.66	634	-
9	-	-	-
10	122.49	202	-
11	125.42	387	-
12	-	-	-
13	-	-	-
14	-	-	-
15	133.4	279	-
16	-	-	-
17	-	-	-
18	114.9	406	-
19	-	-	-
20	172.78	127	-
21	91.43	496	-
22	-	-	-
23	55.48	739	-
24	44.73	790	-

Table P.7.: Farm sizes in ha and off-farm labour in hours of the 4th iteration of the alternate policy designs.

	CAP I		CAP II		'nature-focused'	
Farm	Size [ha]	Off-farm [hours]	Size [ha]	Off-farm [hours]	Size [ha]	Off-farm [hours]
1	-	-	86.5	350	37.8	641
2	86.5	573	-	-	48.7	723
3	97.8	259	97.8	454	56.7	719
4	-	-	-	-	41.2	610
5	-	-	62.6	704	114.5	411
6	114.5	463	51.8	559	-	-
7	-	-	77.7	635	43.1	809
8	77.7	635	-	-	34.5	847
9	-	-	-	-	-	-
10	122.5	156	122.5	341	122.5	341
11	39.1	783	8.2	926	8.3	927
12	53.2	749	59.1	722	80.5	648
13	33.1	811	58.2	694	36.6	698
14	54.4	608	133.4	377	133.4	419
15	79.0	629	-	-	-	-
16	54.1	744	37.7	822	37.7	833
17	-	-	-	-	-	-
18	60.8	679	77.2	601	77.2	628
19	-	-	-	-	-	-
20	172.8	127	172.8	-	172.8	110
21	91.4	483	91.4	483	91.4	494
22	-	-	-	-	-	-
23	45.1	786	74.8	650	85.3	622
24	55.1	697	25.4	878	14.9	931

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Appendix Q.

Results: Biodiversity indicators

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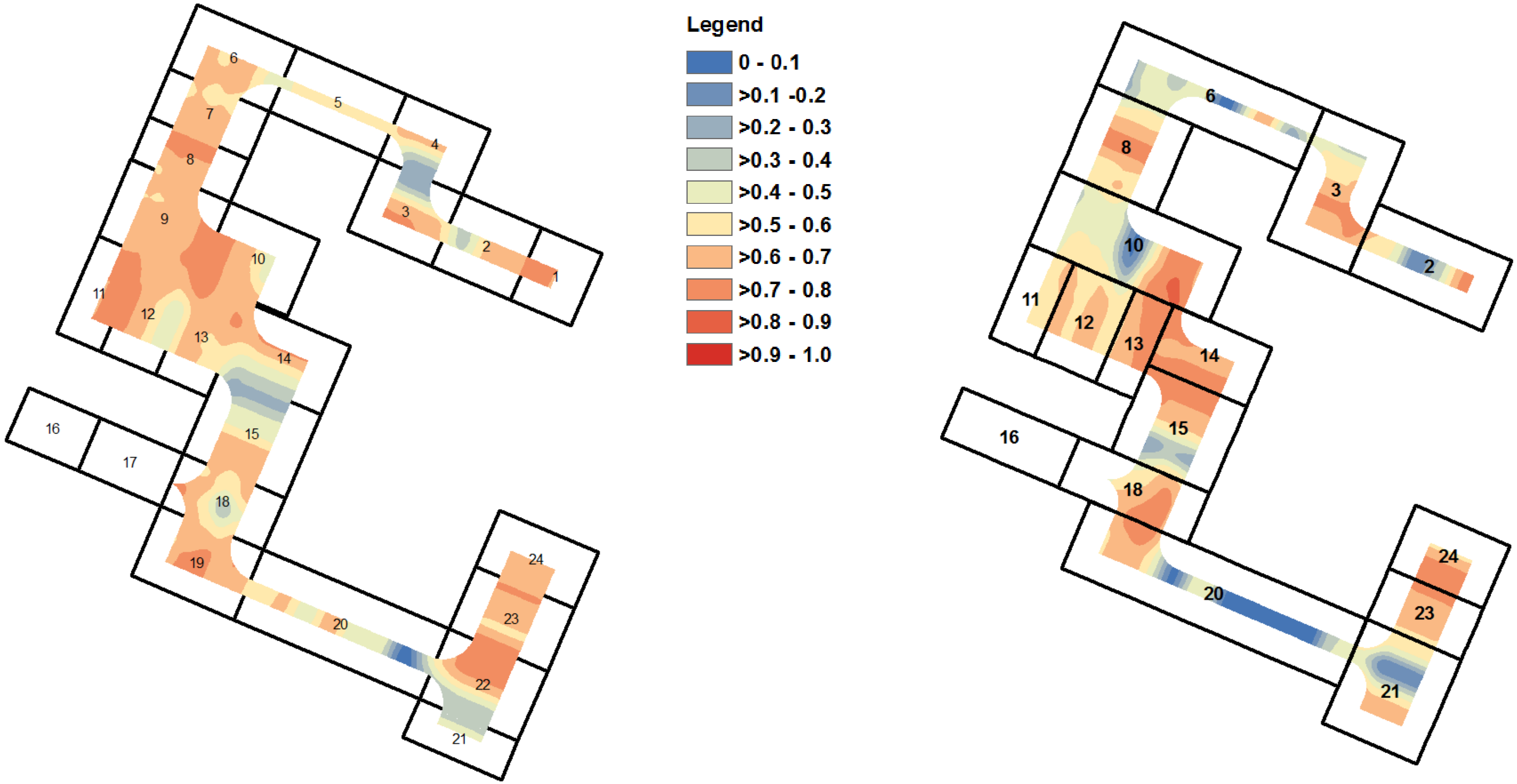

Figure Q.1.: Spatially explicit Simpson's diversity index calculation for the 2nd (*left*) and 4th iterations (*right*) of CAP I.

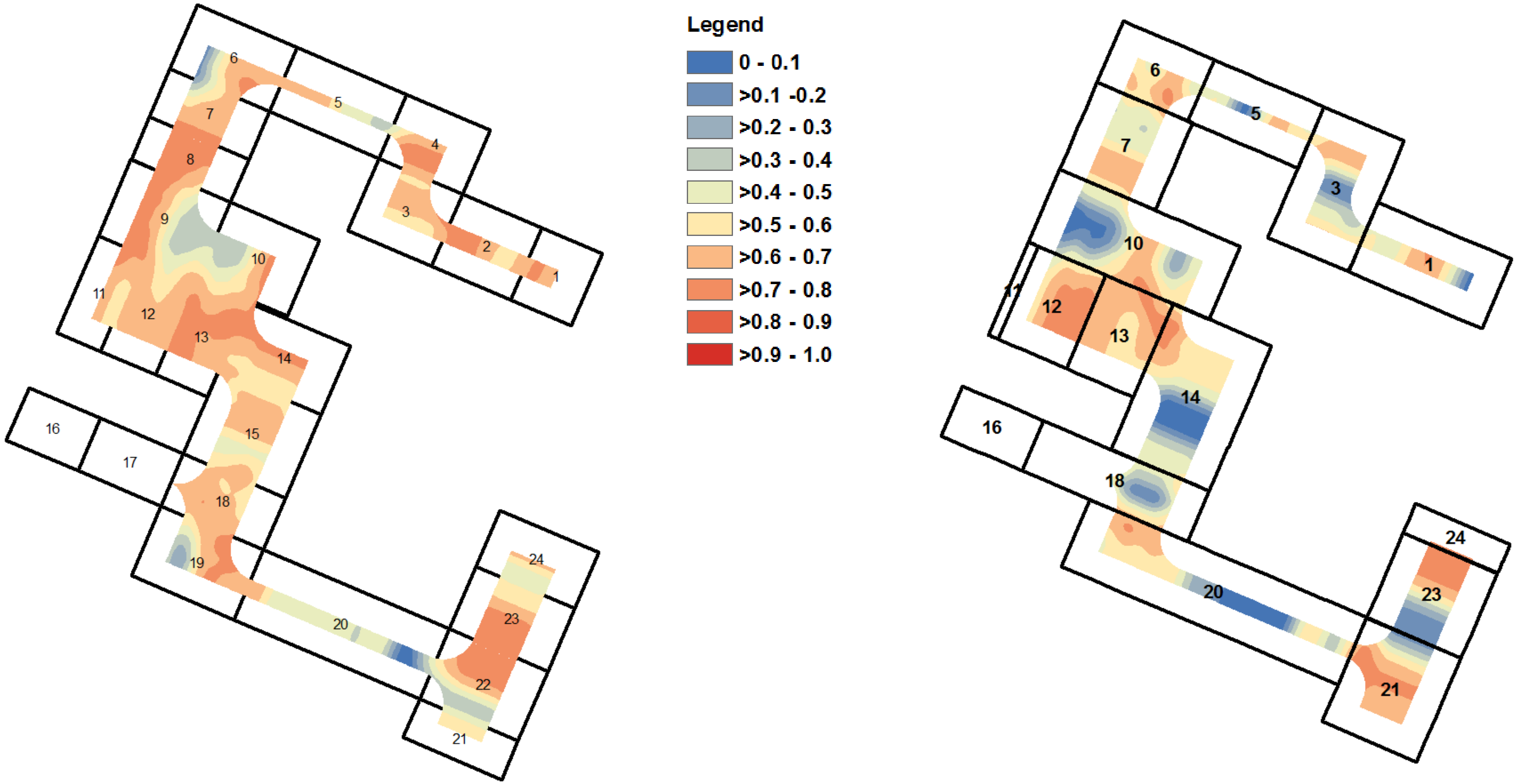

Figure Q.2.: Spatially explicit Simpson's diversity index calculation for the 2nd (*left*) and 4th iterations (*right*) of CAP II.

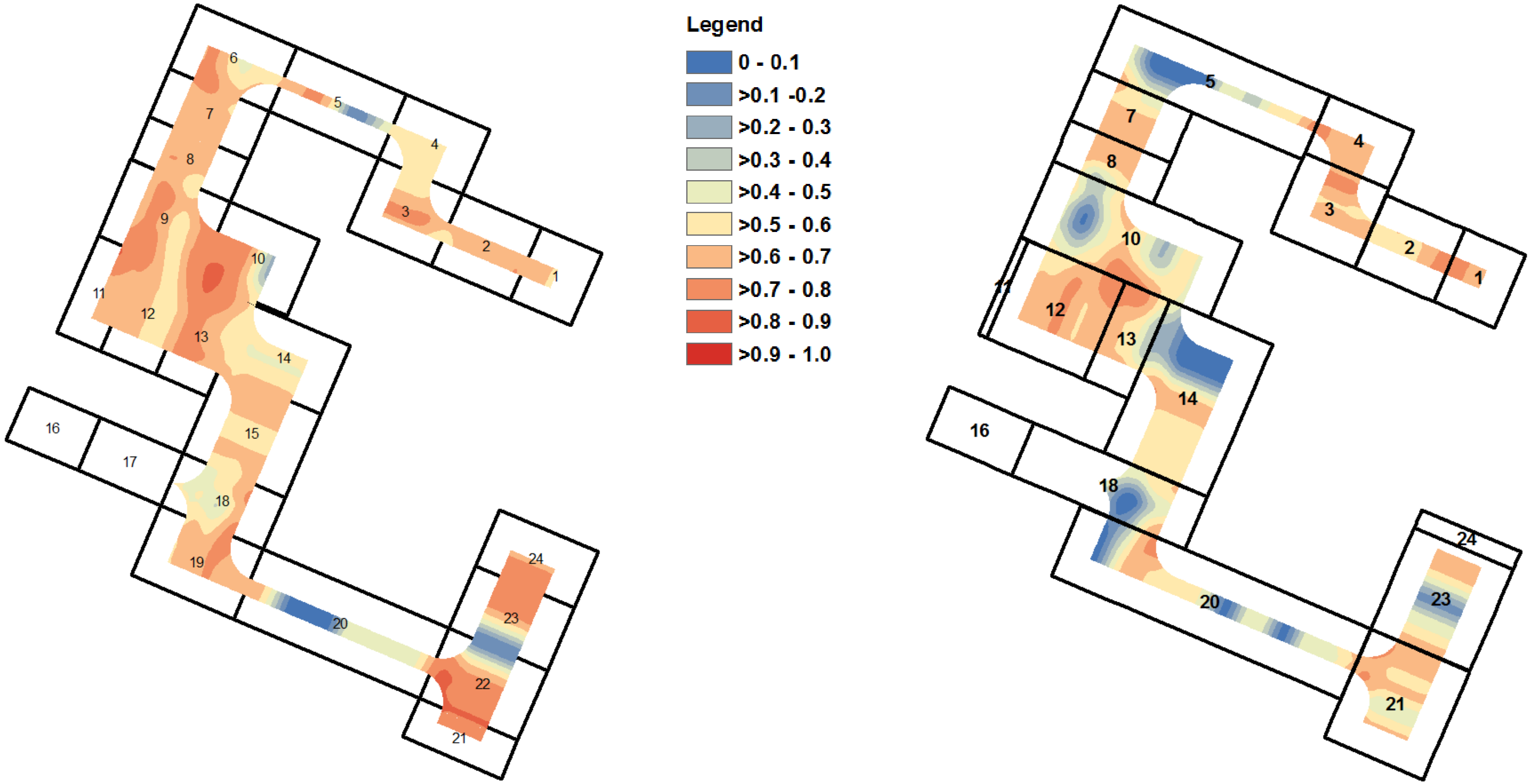

Figure Q.3.: Spatially explicit Simpson's diversity index calculation for the 2nd (*left*) and 4th iterations (*right*) of the 'nature-focused' policy design.

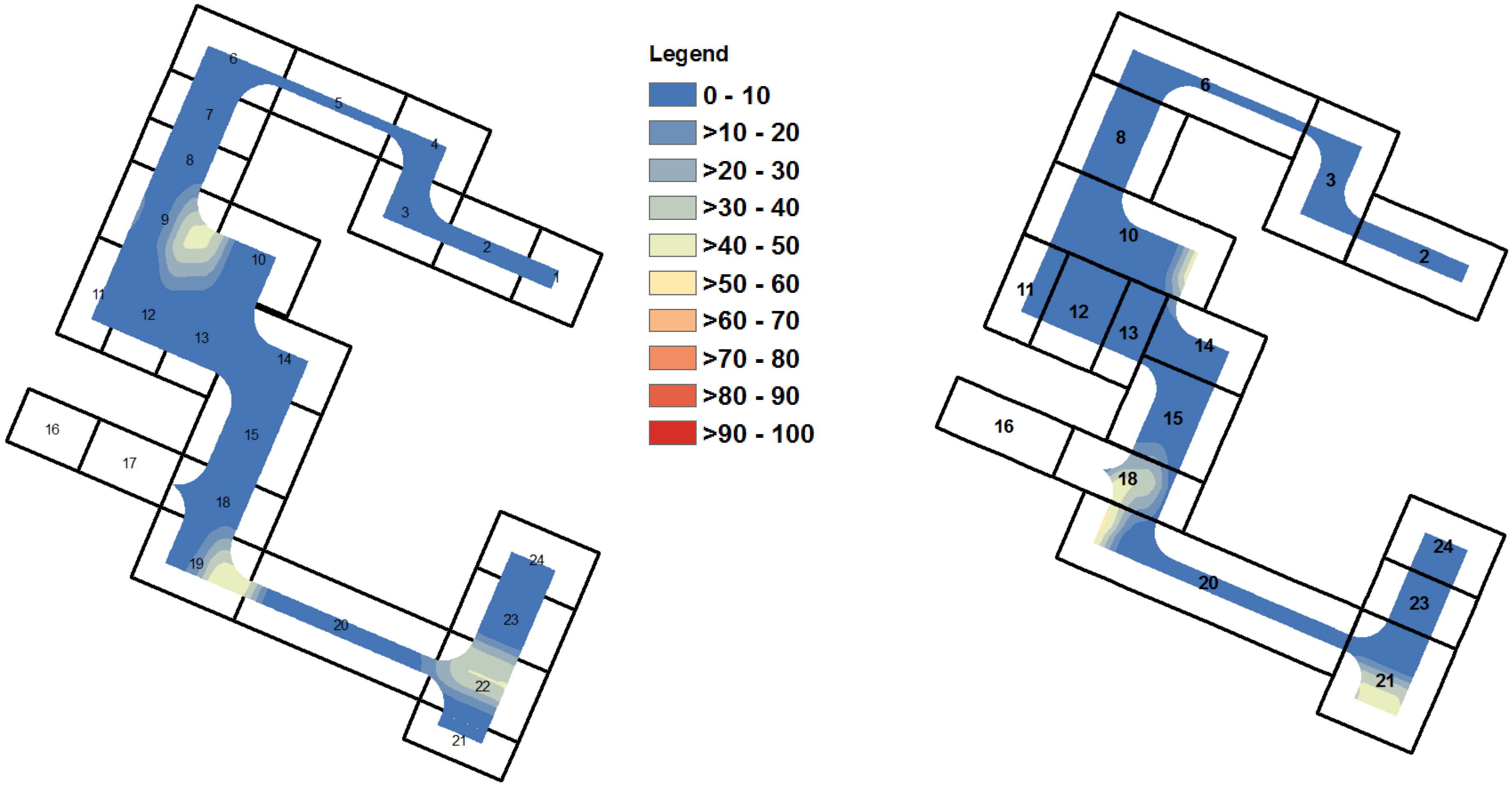

Figure Q.4.: Spatially explicit semi-natural habitat index calculation for the 2nd (*left*) and 4th iterations (*right*) of CAP I.

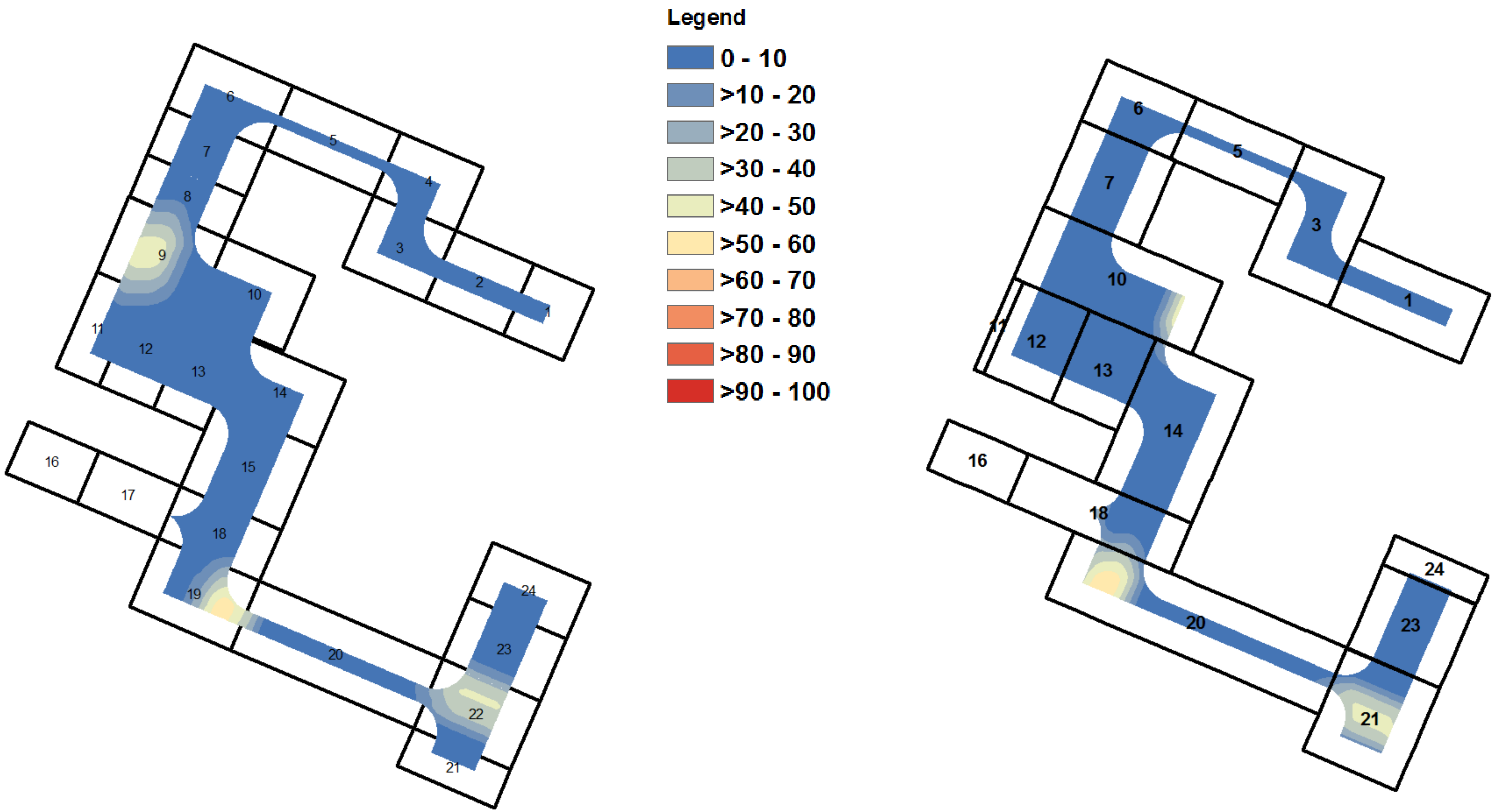

Figure Q.5.: Spatially explicit semi-natural habitat index calculation for the 2nd (*left*) and 4th iterations (*right*) of CAP II.

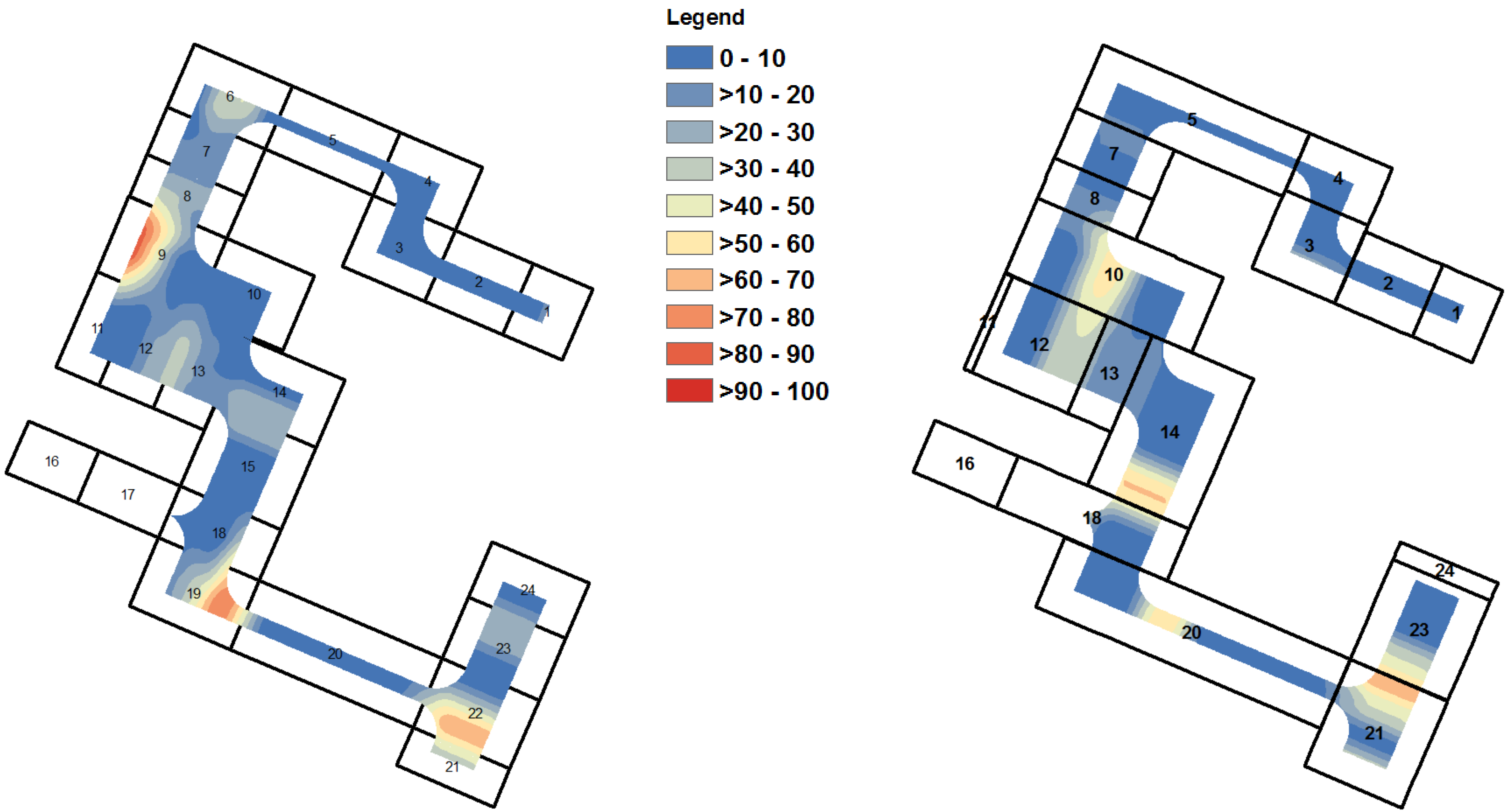

Figure Q.6.: Spatially explicit semi-natural habitat index calculation for the 2nd (*left*) and 4th iterations (*right*) of the 'nature-focused' policy design.

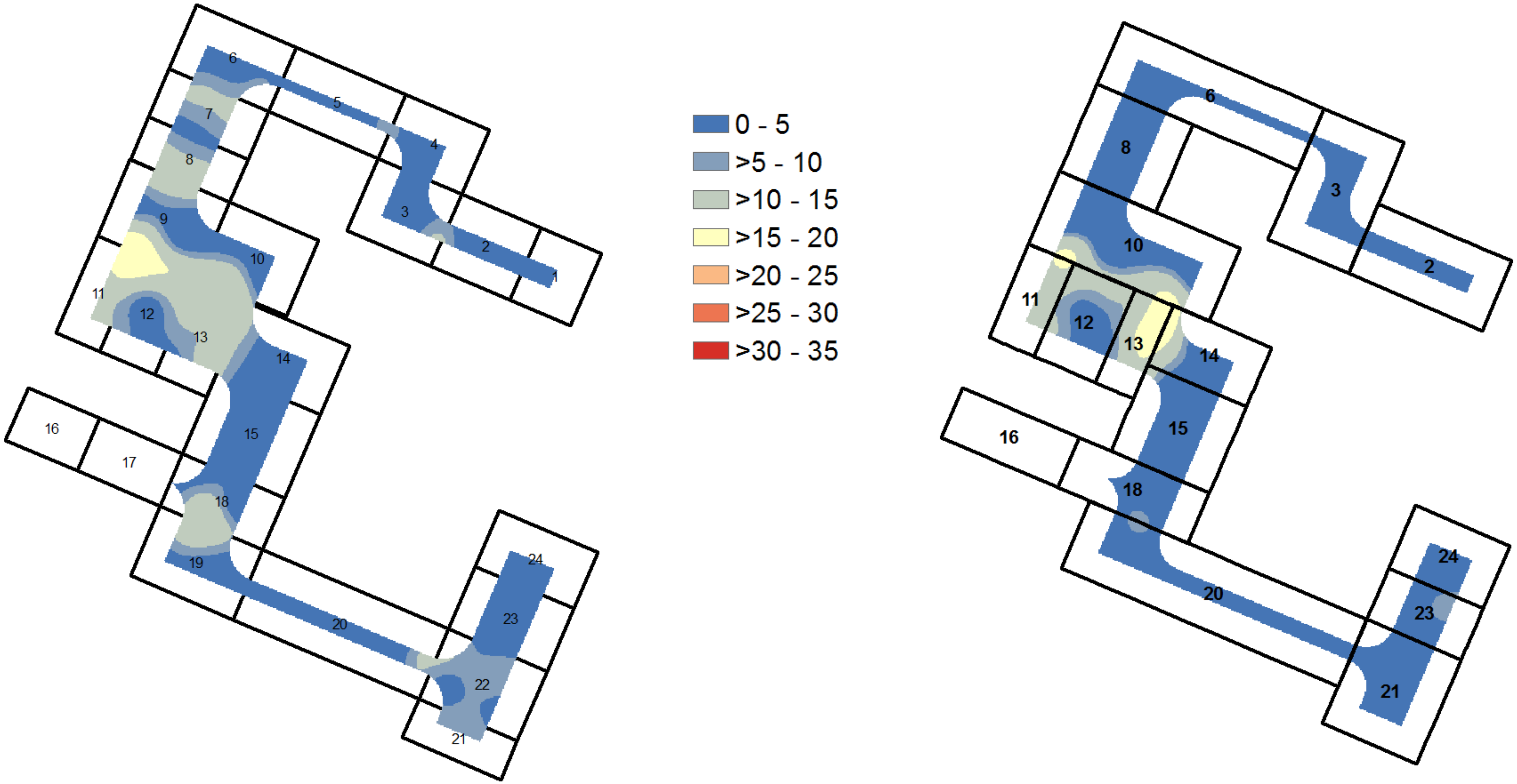

Figure Q.7.: Spatially explicit number of patches index calculation for the 2nd (*left*) and 4th iterations (*right*) of CAP I.

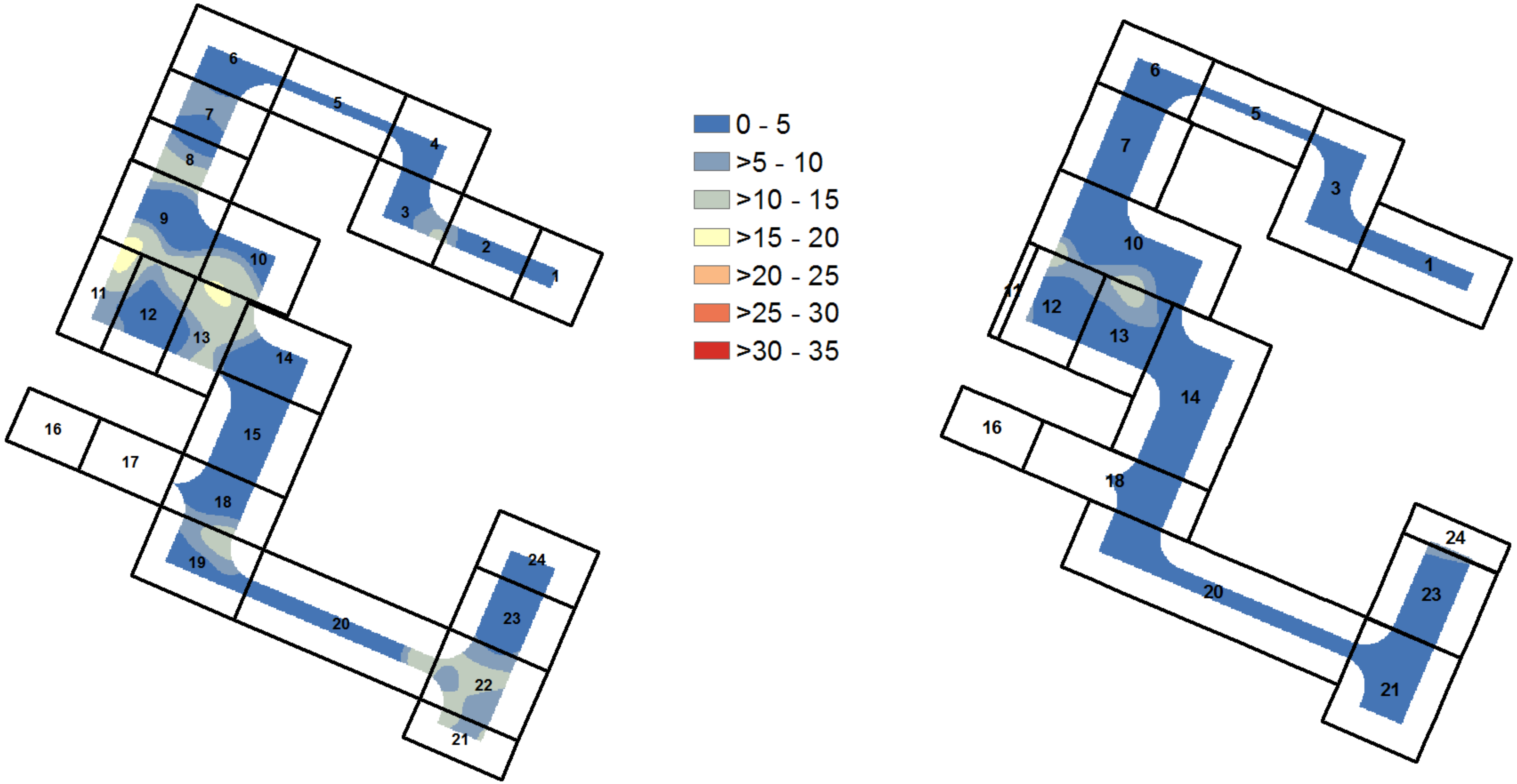

Figure Q.8.: Spatially explicit number of patches index calculation for the 2nd (*left*) and 4th iterations (*right*) of CAP II.

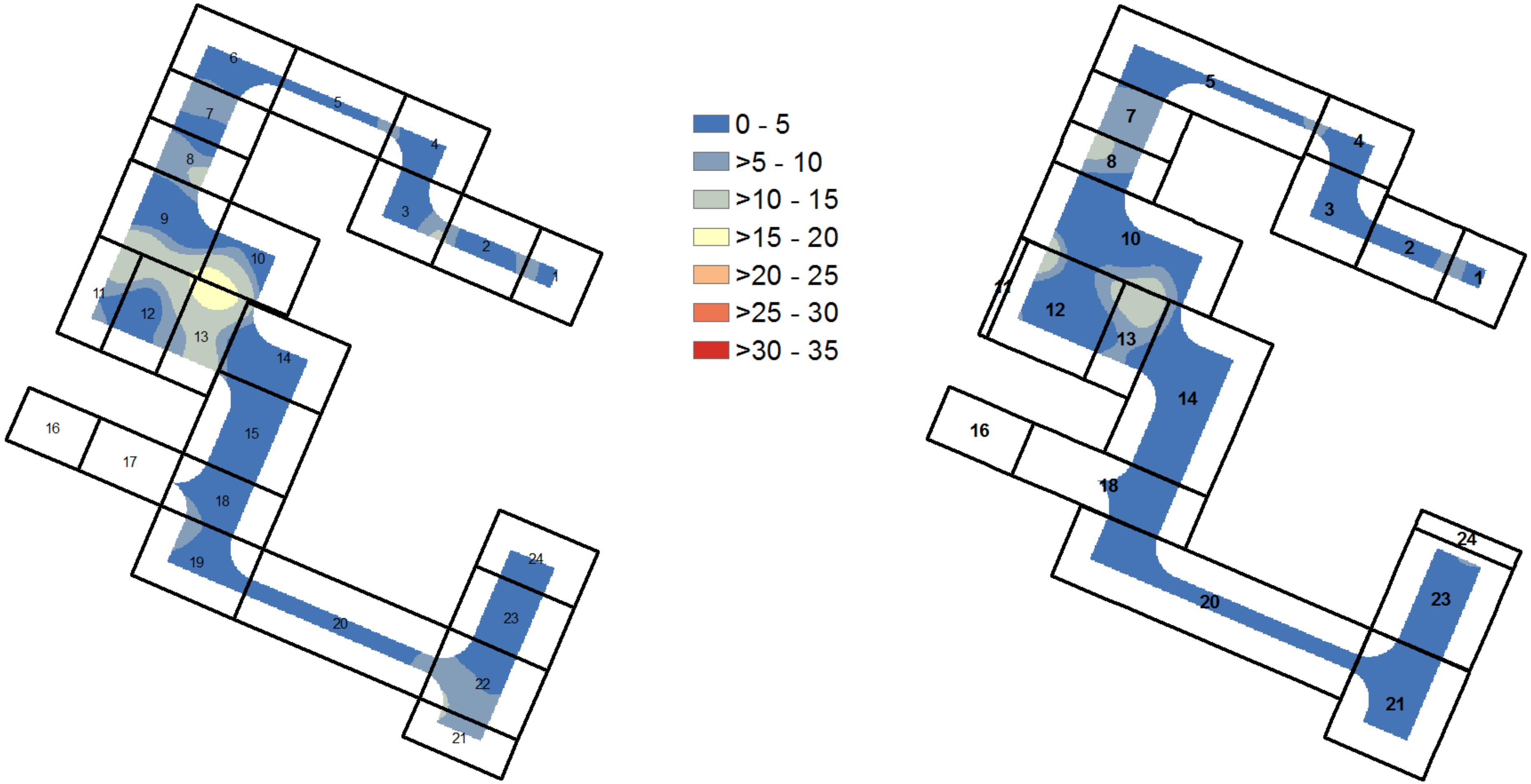

Figure Q.9.: Spatially explicit number of patches index calculation for the 2nd (*left*) and 4th iterations (*right*) of the ‘nature-focused’ policy design.