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**Bastian Bertsch-Hörmann**

# Effects of agroforestry on the carbon dynamics of an agroecological landscape: Human Appropriation of Net Primary Production in two land use scenarios in the Eisenwurzen region, Austria

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# **Effects of agroforestry on the carbon dynamics of an agroecological landscape:**
Human Appropriation of Net Primary Production in two land use scenarios in the Eisenwurzen region, Austria*

Bastian Bertsch-Hörmann

* Masterarbeit verfasst am Institut für Soziale Ökologie, Studium der Sozial- und Humanökologie. Diese Arbeit wurde von Priv.-Doz. Dr. Gingrich und Dr. Gaube betreut. (Die vorliegende Fassung ist eine geringfügig überarbeitete Version der Masterarbeit.)

## Abstract

Land use is a major driver of global environmental change and further intensification of agriculture will be necessary to satisfy the increasing global demand for biomass production. Land systems are required to minimize greenhouse gas (GHG) emissions and other negative environmental impacts as well as to maximize carbon (C) stocks and enhance other ecosystem services. Agroforestry – the combination of crops and trees on the same unit of land – is often handled as a strategy for agroecological intensification because it has the potential to simultaneously increase productivity, restore ecosystem services and enhance resilience. Nevertheless, there still exists a knowledge gap concerning the trade-offs between different functions of an agroforestry system, in particular between provisioning services and climate change mitigation in temperate regions and at landscape-level. This study contributes to the discussion by quantifying the C dynamics of a hypothetical transition to agroforestry in the Austrian *Eisenwurzen* region between 2020–2080, enabling the assessment of trade-offs between carbon sequestration (CS) and biomass harvest. A landscape-level modelling approach included development of two land use scenarios and integration of data from two distinct land use models. The socio-ecological indicator framework Human Appropriation of Net Primary Production (HANPP) was slightly extended and used to quantify C dynamics. Results show that the implementation of agroforestry has profound impacts on the carbon flows in the agroecosystem. Main dynamics relate to a high rate of CS (of 1.1 t C $ha^{-1}$ $yr^{-1}$ between 2020–2080), a strong decrease of annual biomass harvest (of up to -70% for combined crop and grass yields from 2020–2050) and a small increase of accumulated net primary productivity (of 3–8% between 2020–2050). Combined effects lead to a strong decrease of total HANPP, suggesting a relieve from human-induced pressure on the ecosystem while simultaneously increasing landscape-level productivity. The calculated climate change mitigation and yield potentials are discussed in the context of methodological limitations, Austria's GHG emissions and climate strategy, food security, self-sufficiency as well as other socio-economic and temporal dimensions. In conclusion, multi-functional agroecosystems such as agroforestry can – if done right – benefit from the complementarity of resource use and potentially provide a wide range of ecosystem services, but negative effects on yields must be considered carefully. Future research in this regard should concentrate on balancing the provision of different ecosystem services within a region's larger socio-ecological dynamics and socio-economic demands.

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## List of Abbreviations

| | |
|---|---|
| °C | Degree Celsius |
| ABM | Agent-based model |
| ACS | Accumulated carbon stock |
| AFS | Agroforestry scenario |
| AFS-GRAD | Gradual agroforestry scenario |
| AFS-MAX | Maximum agroforestry scenario |
| AGES | Agentur für Gesundheit und Ernährungssicherheit |
| AGR | Agricultural scenario |
| BAU | Business-as-usual |
| BFW | Bundesforschungs- und Ausbildungszentrum für Wald, Naturgefahren und Landschaft |
| BMNT | Bundesministerium für Nachhaltigkeit und Tourismus |
| C | Carbon |
| CAP | Common Agricultural Policy |
| CC | Carbon content |
| CCC | Carbon carrying capacity |
| CCM | Corn-cob-mix |
| CMIP5 | Coupled Model Intercomparison Project – Phase 5 |
| $CO_2$ | Carbon dioxide |
| CS | Carbon sequestration |
| DBH | Diameter at breast height |
| DGVM | Dynamic global vegetation model |
| DM | Dry matter |
| DOY | Day of year |
| EU | Eurpoean Union |
| EU-27 | 27 member states of the EU |
| EUR | Euro (monetary) |
| GeoTIFF | Geocoded Tagged-Image File Format |
| GHG | Greenhouse gas |
| GIS | Geographical information system |
| ha | Hectare |
| HANPP | Human Appropriation of Net Primary Production |
| aHANPP | Above-ground HANPP |
| aHANPP-harv | Harvest share of aHANPP |
| aHANPP-luc | Land use change share of aHANPP |
| HI | Harvest index |
| IACS | Integrated Administration and Control System |
| ICRAF | International Centre for Research in Agroforestry |
| ID | Identification |

| | |
|---|---|
| IPBES | Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services |
| IPCC | Intergovernmental Panel on Climate Change |
| km | Kilometer |
| $km^2$ | Square kilometer |
| kt | Kiloton |
| LEADER | Liason entre Actions de Developement de l'Economie Rurale |
| LER | Land equivalent ratio |
| LI | Low-input factor |
| LTSER | Long-Term Socio-Ecological Research |
| LUBIO | Land use, climate change and biodiversity in agricultural landscapes |
| LULMC | Land use and land management change |
| LULUCF | Land use, land use change and forestry |
| m | Meter |
| $m^3$ | Cubic meter |
| Mha | Mega hectare |
| mm | Millimeter |
| NPP | Net primary production |
| aNPP | Above-ground NPP |
| aNPP-act | Actual aNPP |
| aNPP-eco | Remaining aNPP in the ecosystem |
| aNPP-pot | Potential aNPP |
| ÖPUL | Österreichisches Programm für umweltgerechte Landwirtschaft |
| ÖROK | Österreichische Raumordnungskonferenz |
| Pg | Petagram |
| PHL | Pre-harvest loss factor |
| PJ | Petajoule |
| RAB | Remaining annual biomass |
| RCP | Representative concentration pathway |
| RPB | Remaining perennial biomass |
| RR | Recovery rate |
| SSP | Share socio-economic pathway |
| t | Ton (metric) |
| UNFCCC | United Nations Framework Convention on Climate Change |
| USA | United States of America |
| WC | Water content |
| yr | Year |

# 1 Introduction

Land use is a major driver of environmental change (Ellis et al., 2013; Turner et al., 2007). At present, more than 75% of Earth's ice-free land show evidence of human alteration and less than 25% remain as wildlands being "embedded within anthropogenic mosaics of land use and land cover" (Ellis and Ramankutty, 2008). Agriculture in particular makes use of up to 50% of earth's terrestrial surface (Tilman et al. 2002; Foley et al. 2011), containing the most fertile and suitable lands and representing the largest type of land use on the planet. Increasing land use intensity affects global carbon (C), water and nutrient cycles, contributing to climate change and the detriment of ecosystems across the globe (Erb et al., 2009; Foley, 2005; Lanz et al., 2018; Tilman, 2001). With population growth, rising per capita demand of land-based products and a trend towards the substitution of fossil fuels with biomass, land use intensity and the accompanying pressures on the Earth system are expected to rise further (Coelho et al., 2012; Tilman et al., 2002). Agricultural expansion inevitably leads to land conversion from forest, shrub- and grassland to cropland, resulting in C release from biomass and soils that could – at least in part – be avoided by further yield improvements (Burney et al., 2010), i.e. increasing the agricultural output per unit of land. A farmer's capacity to close a given yield gap either depends on external inputs (with all known implications) and/or ensuring the *integrity and abundance* of supporting and regulating services of the agroecosystem and its surrounding landscape (Bommarco et al., 2013). The latter concept, known as ecological intensification, refers to the intensive and smart use of supporting and regulating ecosystem services and functionalities (in multifunctional agroecosystems) to ensure efficiency and resilience (Tittonell, 2014). Hence, research on agroecological intensification is directly related to sustainable development, especially by offering a pathway to approach one important question in current sustainability debate: *How to meet future biomass demand for food, feed, fiber and fuel while simultaneously providing mitigation of and adaptation to climate change as well as restoring and enhancing ecosystems functioning and services?*

While historic land use intensification, specialization and industrialization of agriculture (and in particular the Green Revolution) allowed humanity to overcome the "Malthusian trap" time and again – that is supporting enduring population growth and the transition to an industrial and consumerist lifestyle – it also led to the maximization of yields over the incremental neglect of supporting, regulating and cultural ecosystem services (Ellis et al., 2013; Erb et al., 2016; Tilman, 1999). Unprecedented gains in output per unit area – especially through mechanization, application of agrochemicals and the use of enhanced species varieties – were only made possible by the introduction of fossil energy (Pellegrini and Fernández, 2018). The transition from a biomass- to a fossil-based energy regime also resolved land competition between human food and *technical energy* such as fire wood or feed for draft animals (Krausmann et al., 2016). As a consequence, energy efficiency plummeted drastically, with energy return on investment of agriculture sometimes even dropping to <1 (Krausmann et al., 2003). Tittonell (2014) points out that "one thing is known for certain: the current model of agricultural intensification is not sustainable (socially and thermodynamically), it is neither ecological nor eco-efficient, it is ineffective at feeding the world, it is harmful for the environment and contributes to biodiversity loss."

Since the inception of the term "sustainable development" in the so-called *Brundtland Report* (United Nations, 1987), calls for alternative and sustainable land use strategies are becoming louder, aiming to ensure economic, ecological and social well-being for current and future generations (Aznar-Sánchez et al., 2019; Bryan et al., 2018; IPBES, 2019; IPCC, 2019). Land systems are increasingly scrutinized to preserve and promote other ecosystem services than the pure provisioning of food, feed, fiber and fuel – in particular the many regulating and supporting services such as climate regulation, water purification or soil formation (Millennium Ecosystem Assessment, 2005). The assessment of trade-offs and synergies between such services is important to guide the formulation of sustainable land use strategies. In the face of global warming and the Paris Agreement (UNFCCC, 2015), the potential of agroecosystems to mitigate climate change gained in relevance. Besides reducing energy-intensive inputs and greenhouse gas (GHG) emissions of agricultural systems, carbon sequestration (CS) in soils and biomass can be a significant mitigation measure (Bossio et al., 2020; Kay et al., 2019; Lal, 2004; Smith et al., 2008; P. Smith et al., 2013) as well as an addition to technical carbon dioxide ($CO_2$) capture, utilization and storage systems that still face many challenges and uncertainties (Bachu, 2008; Li et al., 2012; Liang et al., 2011; Stigson et al., 2012). The question of how to simultaneously maximize CS and biomass harvest while reducing environmental pressures is paramount.

The theoretical framework of social ecology (Haberl et al., 2016) is a useful lens through which to analyze land use change and the impacts it has on the socioeconomic and ecological sub-systems. The two core concepts at its base are *social metabolism* and *colonization of natural systems*. *Social metabolism* applies the biological concept of metabolism to a society's material and energetic interactions with the natural system and serves as an important paradigm for the interdisciplinary research on society-nature-interaction (Fischer-Kowalski, 1998; Fischer-Kowalski and Hüttler, 1998). *Colonization of natural systems* denotes the practice of continuous intentional intervention into ecosystems in order to render them more useful or productive for human society (Fischer-Kowalski and Erb, 2016). Colonization thus provides the means for the preservation and growth of human populations by supplying provisioning and other ecosystem services. As ecosystems as well as social systems are complex and autopoietic, intended interventions always create unintended side effects (Fischer-Kowalski and Weisz, 2016). Figure 1 summarizes the process of land use change being embedded into the concept of social metabolism as elaborately described in Erb et al. (2016). Based on cost-benefit relationships and other socioeconomic factors, inputs into the ecosystem define land cover modification, and thus the variety, quality and quantity of outputs back into society. These outputs can come in many forms related to ecosystem services, but are often reduced to provisioning of food and other biomass products. Unintended consequences affect ecosystems as well as societies, and can hence produce feedback loops which again alter the composition and properties of inputs and outputs.

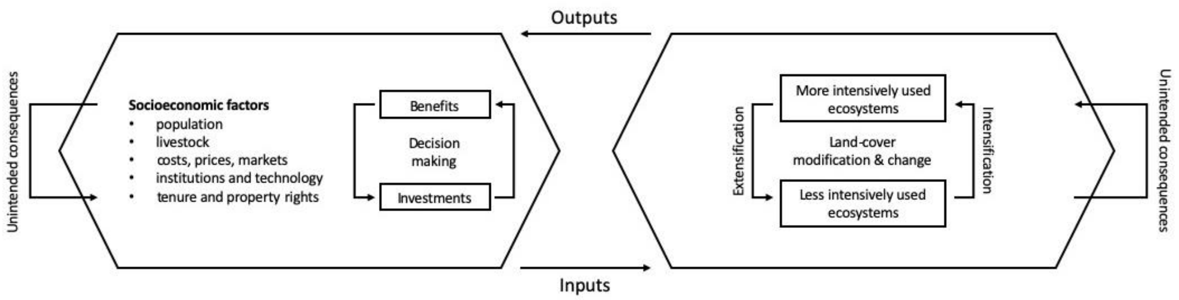


**Figure 1: A socioecological metabolism perspective on land use change and land use intensity. Adapted from Erb et al. (2016).**

On the one hand, this theoretical framework allows to explain the long history of anthropogenic land use as well as the associated shifts in socio-metabolic regimes and land use intensification (Krausmann et al., 2016). On the other hand, it provides a conceptual framework to approach the question of sustainable agriculture and, to be more specific, to evaluate the trade-offs between two outputs to society in the form of ecosystem services: (i) the regulating service of CS to mitigate climate change, (ii) the provisioning service of food and fiber for the maintenance of human population and its livestock. While the concept of social metabolism enables analysis of a variety of socio-economic and ecological facets, this study concentrates on the quantification of C dynamics.

This study assesses the impacts of two different land use systems on harvest and climate change mitigation at a landscape level. Analysis will include a conventional agriculture, which is characterized by the strict segregation of crops and trees, as well as agroforestry, which is the intentional combination of the same.

Agroforestry has the potential to provide positive environmental effects while simultaneously maximizing the use of radiation, water and nutrients and as such is often handled as a possible solution to address climate change, food security and environmental degradation (Kay et al., 2019; Ong et al., 1996; Ramachandran Nair et al., 2010). While it was shown that agroforestry can sequester a significant amount of carbon in biomass and soils, it can also have negative effects on individual provisioning service elements (i.e. the woody or the non-woody elements of a system) (Torralba et al., 2016). But, because the overall productivity (measured as the Land Equivalent Ratio) in agroforestry systems is often shown to be higher than in agricultural systems, acute assessments of the trade-offs between environmental benefits and crop yields are rare and can appear biased towards an environmentalist perspective (J. Smith et al., 2013).

## 1.1 Objectives and research questions

To enable the formulation of sustainable long-term land use strategies, policy-relevant aspects concerning benefits and constraints to pressing issues such as food security and climate change mitigation need to be addressed in a systematical manner. This study aims to contribute to the discussion by analyzing and comparing the carbon dynamics of two hypothetical land use scenarios, (i) the agricultural scenario (AGR) and (ii) the agroforestry scenario (AFS), from 2020–2050 and beyond and on a landscape level. The quantification of carbon dynamics allows for the assessment of changes in biomass production, biomass harvest and biomass accumulation, and thus informs on the overall productivity of the system as well as the relationship between provisioning of food, feed, fiber and fuel and climate change mitigation through CS. Results add to the assessment of trade-offs and synergies between different ecosystem services.

The following research questions will be addressed in the study:

1) *What are the differences in the carbon dynamics of the agricultural and agroforestry scenarios in the Eisenwurzen between 2020–2050 and beyond?*
2) *How does the hypothetical transition to agroforestry affect the relationship between net primary production, biomass harvest and carbon sequestration?*

The structure of this work is as follows: Chapter 2 provides an introduction into agroforestry systems with a focus on temperate agroforestry. Chapter 3 describes the methodological approach of this study, including the study region, model integration, formulation of the land use scenarios as well as data and methods used for the calculations. Chapters 4 and 5 subsequently present the results and conclude with a discussion in the context of relevant aspects.

## 2 Agroforestry

Many definitions of agroforestry exist, reflecting the broad spectrum of perspectives, practices and scales at which it takes place around the world. It can be defined in rather simple terms (Nair et al., 2008, p. 101):

> *"Agroforestry is the relatively new name for the age-old practice of growing trees and shrubs with crops and/or animals in interacting combinations on the same unit of land."*

A wider description by the *World Agroforestry Center* highlights the different spatial scales that can be involved (taken from the website of the ICRAF, 2020):

> *"[Agroforestry] comprises trees on farms and in agricultural landscapes, farming in forests and along forest margins and tree-crop production. […] Interactions between trees and other components of agriculture may be important at a range of scales: in fields (where trees and crops are grown together), on farms (where trees may provide fodder for livestock, fuel, food, shelter or income from products including timber) and landscapes (where agricultural and forest land uses combine in determining the provision of ecosystem services)."*

The concept can also be formulated by integrating temporal dynamics and focusing more on the ecological and social interactions (Leakey, 2017, p. 5):

> *"[Agroforestry] practices can be seen as phases in the development of a productive agroecosystem, akin to the normal dynamics of natural ecosystems. Over time, the increasing integration of trees into land-use systems through agroforestry can be seen as the passage toward a mature agroforest of increasing ecological integrity. By the same token, with increasing scale, the integration of various agroforestry practices into a landscape is like the formation of a complex mosaic of patches in an ecosystem, each of which is composed of many niches. […] Within this ecological framework, farmers can manipulate and manage their trees to take advantage of the benefits of the processes in ecosystem services or products, by breaking the process of agradation – or ecosystem development – at any point, or by allowing a mature agroforest to develop."*

This notion of intentionally shaping an agroecosystem over space and time by purposeful human action can be elaborated, emphasizing the importance of specific knowledge about tree-crop interactions and combinations (Mosquera-Losada et al., 2008, p. 4):

> *"All types of agroforestry systems integrate people as part of the system as they are artificial systems to a higher (i.e. domestic animals) or lower degree (i.e. wild animals in natural or national parks), where one component can be promoted over the other, or both at the same time trying to reach equilibrium between the different components. The promotion of one component over the other can be modified as the tree develops. Man, through traditional experience and practice or new knowledge, should promote the positive*

*interactions between the two components, by an initial knowledge-based selection of the tree species and later by adequate management […].”*

This small selection of definitions already hints at the complexity of the field. Taking into account all the socio-economic and ecological effects emanating from the implementation of agroforestry systems reveals the many diverse scientific disciplines that might be involved in its exploration (such as forestry, agronomy, biogeosciences, sociology, economics, political science and systems theory) as well as the need for interdisciplinary research approaches. The present study focuses on C dynamics in agroforestry, contributing to the nexus of biogeosciences and socio-economic studies.

## 2.1 Agroforestry in Europe

While the majority of traditional as well as innovative agroforestry is practiced in tropical regions and research was focused thereupon for many years, the interest in temperate agroforestry has increased considerably in the recent past (Mosquera-Losada et al., 2012; Ramachandran Nair, 2014; Rigueiro Rodríguez et al., 2009).
Due to the broad spectrum of different agroforestry practices, classifications can be made according to different criteria, such as a system’s components, predominant land use, spatial and temporal structure, agroecological zone, socio-economic status or function (McAdam et al., 2008). The primary and simplest classification is according to a system’s components (Ramachandran Nair, 2014):

- ***Agrisilviculture:*** *trees and crops*
- ***Silvopasture:*** *trees and animals*
- ***Agrosilvopasture:*** *trees, crops and animals*

This classification, however, cannot reflect the large diversity of agroforestry practices and was revised many times since its inception in the 1980s. Mosquera-Losada et al. (2008) classified agroforestry presently existing in Europe into six basic types (Table 1): silvoarable, silvopasture, forest farming, riparian buffers, improved fallow and multipurpose trees.
Wood pastures have existed in Europe for thousands of years and other practices, such as hedgerows, windbreaks or intercropped and grazed orchards (*Streuobst*) were widely used at least throughout the last centuries (Herzog, 1998; Mosquera-Losada et al., 2008; Nerlich et al., 2013). In the course of the specialization of land use during the Middle Ages and later through mechanization and industrialization of agriculture in the 20$^{th}$ century, trees were progressively removed from arable land and traditional agroforestry systems gradually disappeared (Eichhorn et al., 2006; Mosquera-Losada et al., 2012; Nerlich et al., 2013; Von Maydell, 1995). This was, among other things, accelerated by policies such as the European Common Agricultural Policy (CAP) promoting the specialization of agriculture and forestry and its focus on the production function (McAdam et al., 2008; van Zanten et al., 2014). Mosquera-Losada et al. (2012), however, argue that since the mid-1990s European policies generally encourage land use that combines production, environmental services and social benefits, but “simplification of the number of measures to promote agroforestry practices is needed to better follow up the implementation and to evaluate and provide future policies more adapted at European levels” (2018). In opposition, Swiss agricultural policy, for example, directly promotes agroforestry by cross-compliance, agri-environmental schemes and landscape quality payments (Herzog et al., 2018).

**Table 1: Agroforestry practices in Europe. Source: Mosquera-Losada et al. (2008).**

| *Agroforestry practice* | *Description* |
|---|---|
| *Silvoarable agroforestry* | *Widely spaced trees inter-cropped with annual or perennial crops. It comprises alley cropping, scattered trees and line belts.* |
| *Silvopasture* | *Combining trees with forage and animal production. It comprises forest or woodland grazing and open forest trees.* |
| *Forest farming* | *Forested areas used for production or harvest of natural standing specialty crops for medicinal, ornamental or culinary uses.* |
| *Riparian buffer strips* | *Strips of perennial vegetation (tree/shrub/grass) natural or planted between croplands/pastures and water sources such as streams, lakes, wetlands, and ponds to protect water quality.* |
| *Improved fallow* | *Fast growing, preferably leguminous woody species planted during the fallow phase of shifting cultivation; the woody species improve soil fertility and may yield economic products.* |
| *Multipurpose trees* | *Fruit and other trees randomly or systematically planted in cropland or pasture for the purpose of providing fruit, fuelwood, fodder and timber, among other services, on farms and rangelands* |

Even though agroforestry in Europe still exists, “the lack of European data, and a narrow definition of agroforestry, has led in the past to the misconception that agroforestry is unimportant in the European context” (den Herder et al., 2017). In their study, den Herder et al. (2017) estimated the current extent of agroforestry in the European Union (EU) by quantifying and mapping its distribution. They found that presently about 8.8% of the utilized agricultural area in the EU are under agroforestry management, comprising wood pastures, hedgerows, windbreaks, riparian buffer strips, intercropped and grazed orchards, grazed forests, forest farming and more innovative silvoarable and silvopastoral systems such as alley cropping, short rotation alley coppice and woodland chicken. The large majority of agroforestry in Europe, however, is silvopastoral covering 3.5% of the territorial area in the EU, while arable and high value tree systems cover only 0.1 and 0.2%, respectively. They furthermore showed that the distribution is skewed towards the south of Europe, with clusters exhibiting a high density of agroforestry points found throughout the Mediterranean biogeographic region in Spain, Portugal, France, Italy, Greece and Bulgaria. In northern Europe, there exist only silvopastoral systems (such as reindeer husbandry), while high value tree agroforestry and to a lesser extent arable agroforestry are scattered throughout atlantic and continental Europe. Nevertheless, den Herder et al. (2017, p. 121) conclude:

> *“Because agroforestry covers a considerable part of the agricultural land in the EU, it is crucial that it gets a more prominent and clearer place in EU statistical reporting in order to provide decision makers with more reliable information on the extent and nature of agroforestry. Reliable information, in turn, should help to guide policy development and implementation, and the evaluation of the impact of agricultural and other policies on agroforestry.” (den Herder et al., 2017)*

While the extent of agroforestry systems in Europe is hence not to be underestimated, further adoption is hesitant, also due to other than purely policy-related factors. Transitions from annual monoculture to perennial polyculture systems involve a variety of uncertainties and risks due to the lack of knowledge and expert support concerning the many species composition and management options in varying climatic and geographic conditions, as well as farmers' concerns about socio-economic viability and the lack of financial support (Graves et al., 2008; Hernández-Morcillo et al., 2018; Rois-Díaz et al., 2018).

## 2.2 Provision of ecosystem services

Agroforestry influences ecosystem functions and services in a variety of ways. The tree-crop interactions in agroforestry system depend on many factors such as the agroforestry type, species composition, system design, climatic and site-specific conditions as well as management. Tree-crop interactions can also have positive and negative effects on the agroecosystem at the same time, highlighting the high complexity of interactions taking place within agroforestry systems. For example, trees and crops compete for water, but at the same time evaporation is reduced due to shade and wind protection, hydraulic lift from deep roots enables exudation of water to drier surface areas during the night, leave litter and root exudates enhance soil organic matter which in turn leads to higher water storage capacity and less run-off, and mycorrhizal symbionts improve nutrition and water supply etc. (Lawson et al., 2019).

The concept of ecosystem services can generally be used to analyze these interactions. According to the Millennium Ecosystem Assessment (2005) and the Common International Classification of Ecosystem Services (Haines-Young and Potschin, 2013), ecosystem services are classified into the following categories:

- ***Provisioning Services:*** *food, water, materials and energy;*
- ***Regulation and Maintenance Services:*** *climate regulation, bioremediation and filtration, flood and storm protection, pest and disease control, erosion control, habitat and genepool protection, soil formation, composition and nutrient cycling;*
- ***Cultural Services:*** *physical and intellectual interactions (educational and scientific, recreational, aesthetic), spiritual, symbolic and other interactions (religious, cultural).*

Agroforestry systems can without a doubt provide a wide range of products, including but not limited to crops and vegetables, meat and dairy, oils, nuts and leaves, forage and fodder, timber, firewood and biofuels, wood pulp, rubber and cork, wool, leather, horn, silk, cotton and linen, gums and resins, medicine, honey and herbs (Lawson et al., 2019; J. Smith et al., 2013). Due to the complementarity and more efficient overall use of radiation, water and nutrients, agroforestry systems can have a higher overall productivity than sole cropping or forestry (Graves et al., 2007; Quinkenstein et al., 2009; Seserman et al., 2018; Sharrow and Ismail, 2004), but effects on individual yield components tend to be negative (Artru et al., 2017; Jose et al., 2004; Torralba et al., 2016). "In North-Western Europe, light is likely to be the principal limiting resource for understorey crops, and most agronomic studies show a systematic reduction of final yield as shade increases" (Artru et al., 2017).

Agroforestry systems can fulfil a variety of important regulatory and maintenance functions. Palma et al. (2007), for example, modelled the environmental effects of silvoarable agroforestry in three test sites in Spain, France and The Netherlands over the period of 60 years. Their model predicted reductions in soil erosion by up to 70%, reductions in nitrogen leaching by 20–30%, an increase of CS over 60 years by up to 140 tons C $ha^{-1}$ and an increase of landscape diversity by up to four times. Agroforestry can contribute to soil productivity and conservation (Ilany et al., 2010; Lawson et al., 2019; Nair et al., 2008; J. H. N. Palma et al., 2007; Pardon et al., 2017; Torralba et al., 2016; Tsonkova et al., 2012), pest, disease and weed control (Pumariño et al., 2015) due to increased habitat and species diversity (Lawson et al., 2019; Montagnini, 2017; J. H. N. Palma et al., 2007; Quinkenstein et al., 2009; Torralba et al., 2016; Tsonkova et al., 2012), as well as increased water infiltration and reduction in the intensity and size of floods (Carroll et al., 2006; Lawson et al., 2019; Mencuccini and Moncrieff, 2004). Furthermore, agroforestry has climate regulating effects in the tree's immediate vicinity as well as on a plot or on a landscape-scale (Lawson et al., 2019). Relating to the microclimate, trees can cause lower daytime and higher nighttime temperatures, lower wind speed and higher humidity (Gosme et al., 2016; Monteith et al., 1991). The magnitude of effects is very much dependent on the specific tree and crop species involved and can potentially increase or decrease yields (Grimaldi et al., 2016; Kanzler and Mirck, 2016). Relating to the mesoclimate, distribution of snow and rainfall can be influenced (Lawson et al., 2019).

In an evaluation of ten agroforestry systems in Europe termed *high nature and cultural value*, Moreno et al. (2018) identified a variety of cultural services, ranging from provision of historically important areas for cultural gatherings to the preservation of unique European cultural heritages with a high aesthetic value. Many of these systems additionally provide a focus for tourism and recreation, education and leisure activities.

### 2.2.1 Climate change mitigation in agroforestry systems

Agroforestry has the potential to mitigate climate change through CS in biomass and soils as well as through reduction of other GHG emissions such as methane and nitrous oxide[1]. CS is defined as the process of removing and storing C from the atmosphere in C sinks through physical or biological processes (Jose, 2009).

> *"Carbon sequestration occurs in two major segments of the agroforestry ecosystem: aboveground and belowground. Each can be partitioned into subsegments: the former into specific plant parts (stem, leaves, etc. of trees and herbaceous components), and the latter into living biomass such as roots and other belowground plant parts, soil organisms and C stored in various soil horizons. The total amount sequestered in each part differs greatly depending on a number of factors, including the region, the type of system (and the nature of components and age of perennials such as trees), site quality, and previous land-use." (Ramachandran Nair et al., 2010, p. 247)*

There exists a relatively large variety of studies that observed CS in above- and belowground biomass as well as soil organic C for a range of different agroforestry systems, climatic conditions and world regions (Aertsens et al., 2013; Feliciano et al., 2018; Lawson et al., 2019; Ramachandran Nair et al., 2010). The majority of studies are from tropical regions in Africa, Latin America and Asia, while a smaller part is focused on temperate regions in North America and Europe.

[1] This section will focus on CS; mitigation of other GHG emissions in agroforestry systems is for example covered in Lawson et al. (2019), Palma et al. (2017) or Hernández-Morcillo et al. (2018).

Dixon (1995) estimated the global CS potential of agroforestry in vegetation and soils from 12–228 t C $ha^{-1}$ with a median value of 95 t C $ha^{-1}$ (of which 70 t C $ha^{-1}$ in above-ground storage in vegetation) over a period of 50 years, corresponding to 1.9 (and 1.4) t C $ha^{-1}$ $yr^{-1}$. Multiplied by a range of 585–1,215 Mha of land that are technically suitable, Dixon estimates a global C storage of 1.1–2.2 Pg over 50 years.
In a review by Lawson et al. (2019), the CS rates in temperate agroforestry systems (with tree age ranging from 6–41 years) vary from 1–12 t C $ha^{-1}$ $yr^{-1}$, depending on species, climate, soil, management, and rotation. Kay et al (2019) estimated CS rates for a wide range of agroforestry practices specifically selected for a variety of European biogeographical regions (Atlantic, Mediterranean, Continental, Steppic) between 0.09 and 7.29 t C $ha^{-1}$ $yr^{-1}$ (again corresponding to a wide range of tree harvest and rotation cycles between 2 and 90 years).
Aertsens et al. (2013) estimated a mean CS rate of 2.75 t C $ha^{-1}$ $yr^{-1}$ (without stating a reference time frame) for agroforestry on arable land and pastures in the EU-27 and assumed 90 and 50 Mha of potentially productive land for silvoarable and silvopasture agroforestry, respectively. This resulted in an increase of C-stocks of 248 Mt C $yr^{-1}$ on arable land and 138 Mt C $yr^{-1}$ on pastures. Adding C storage by implementing hedgerows, cover crops and low/no tillage, a total estimate of 428 Mt C $yr^{-1}$ equals roughly 37% of $CO_2$ emissions in the EU in 2007.
On the one hand, these studies collectively show that CS rates can be substantial in agroforestry systems (although reference time frames vary widely or are not always clearly reported). On the other hand, simultaneous effects on biomass harvest and yields are not consistently captured and accounted for.

## 2.3 Modelling agroforestry

To meet the research demands necessary for the improvement of agroforestry policy and practice, field experiments and modelling are the main options. Field experiments generally are restricted by temporal and spatial dynamics, variability in type, number and manner of tree, crop and animal species as well as the large spectrum of management options involved (Burgess et al., 2019; Luedeling et al., 2016). In particular, temporal variation between individual agroforestry components is often a problem: most trees have harvest cycles of several decades up to 100 years whereas annual crops – and even some animals – need no more than a couple of months to mature (Burgess et al., 2019). The trees' interactions with other components of a system thereby changes with time and all life cycle stages from seeds, seedlings or cuttings to mature trees should be considered (Luedeling et al., 2016). Therefore, empirical data on long-term interactions of agroforestry components are scarce and experiments are time-consuming and expensive. An alternative method "is the use of dynamic computer simulations that predict the effect of climate, tree and crop species, soil type and management choices on tree and crop production, economics and the environment" (van der Werf et al., 2007).

Agroforestry models can be classified into six major model types (Burgess et al., 2019):

1. **Allometric or regression models** use past observed data to describe the relationship between an organism's properties with changes in size. They are for example used in detailed growth models or to relate changes in tree dry weights to height, diameter and timber volume.

2. **Non-plant growth models to determine agroforestry impact** use tree and crop yields to describe the environmental impacts of agroforestry, for example changes in soil C, soil nutrients or water flow.
3. **Plot-based mechanistic models of tree and crop growth** predict tree, crop and environmental interactions over time based on physical and biological principles, for example that the growth of a crop can generally be determined by the interception of light by a canopy and the uptake of water by a root system. They generally need to be dynamic to capture changes of a system's components over time, and their complexity can range from simple one-dimensional to very complex three-dimensional models.
4. **Architectural models of tree growth** for example describe canopy or root architecture to assess light availability and root bulkiness.
5. **Farm-scale management decision models** are used to improve decision-making by farmers. They range from qualitative and conceptual models to financial, economic and social welfare models.
6. **Landscape models** often aggregate results of bio-physical and/or financial models to the landscape-scale and link them to a geographical information system (GIS). As such they are able to model a range of different arable, forestry and agroforestry systems under different environmental and management conditions and sum up the outputs to a regional or national scale.

In this study, a landscape modelling approach was applied by aggregating data from the plot-based mechanistic agroforestry model Yield-SAFE (van der Werf et al., 2007). Yield-SAFE is regarded as one of the simplest mechanistic agroforestry models to simulate biomass growth. The main limitation thereby concerns the relatively low level of detail that can be captured by one-dimensional models such as Yield-SAFE, for example not being able to grasp effects of distance between trees and crops (unlike two dimensional models such as *WaNuLCAS*) or effects of row-alignment on radiation interception (such as three dimensional models like *Hybrid*) (Burgess et al., 2019). Palma et al. (2007) discuss that higher model complexity can be associated with an increase in *input error* due to additional data requirements and a decrease in *specification error* due to gains in accuracy. At the landscape level, however, data of high thematic and temporal resolution may not be available and algorithms which are limited to existing knowledge and its main governing factors are hence more appropriate (J. H. N. Palma et al., 2007).

# 3 Materials and methods

This study combines data from two distinct modelling approaches and regional-level agricultural statistics to assess trade-offs of biomass production and CS in a regional case study in Austria. The methodological approach chosen for this task can be described as a three-stage process covering input, processing and output (Figure 2).

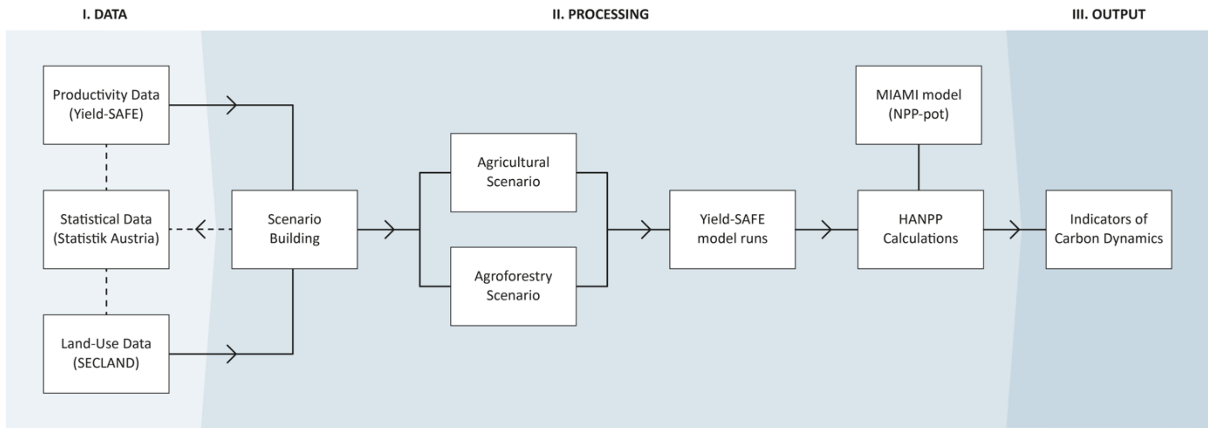


**Figure 2: Overview of the methodological approach**

Input data required to perform the necessary calculations were obtained from different models. The agroforestry model Yield-SAFE (van der Werf et al., 2007) simulates net primary productivity of crops and/or trees on a plot scale. Here it is used to compute biomass production from various agricultural and agroforestry configurations and on two modelling sites. Yield-SAFE esults are then aggregated to the landscape level. The agent-based model SECLAND (Dullinger et al., 2020), on the other hand, provides a simulation of future land use change in the study region. Land use datasets obtained from SECLAND thereby define the upscaling process of the productivity data to the landscape level. The models and simulated data are described in more detail in chapters 3.2.2 and 3.2.3, respectively. This chapter also includes sensitivity testing of specific Yield-SAFE input parameters as well as evaluation of specific Yield-SAFE output data with agricultural statistics. Agricultural statistics, subsequently also used to formulate the land use scenarios, were obtained from Statistik Austria (STATcube database) and are described in chapter 3.2.1.

Data processing included the formulation of the agricultural and agroforestry land use scenarios, described in chapter 3.3. Scenario building was an iterative process that involved above-mentioned sensitivity and evaluation procedures. Once the scenarios were formulated, the necessary Yield-SAFE model runs were performed to quantify C dynamics.

For the quantification of C dynamics the Human Appropriation of Net Primary Production (HANPP) framework (Haberl et al., 2007) was used and modified in this study, see chapter 3.4. At this stage, the MIAMI model provided additional data on the potential net primary productivity (chapter 3.4.1). Finally, various indicators depicting land use intensity, biomass extraction and CS were compiled and used to interpret the outcomes. Results are presented and discussed in chapters 4 and 5.

The chosen methods enable the formulation and assessment of a hypothetical option space concerning the CS potential of an agroforestry scenario and its trade-offs with food production. To my knowledge, it is the first attempt to systematically analyze these trade-offs on a landscape level and on the basis of regional land use data. Application of the HANPP framework thereby enables the comparability of the two scenarios as well as an in-depth analysis by decomposition of inherent C dynamics. This study thus contributes to the debate on the possibilities of agroecological intensification with agroforestry systems.

## 3.1 Description of study area

The study area is part of the Austrian Eisenwurzen, a loosely bounded region in the Northern Limestone Alps situated in parts of Upper Austria, Lower Austria and Styria. The name Eisenwurzen is derived from its history as an important region for metal mining and metallurgy. Although the beginning of mining activities dates back to the 4$^{th}$ century AD, the region regained in importance only in the Middle Ages and was in its prime in the 16$^{th}$ century (Brodda and Heintel, 2009; Peterseil et al., 2013). Provision of food and timber for the industrial workers and industry resulted in widespread deforestation and environmental change (Peterseil et al., 2013). The region's industry, however, was increasingly dismantled since the mid 19$^{th}$ century and, after World War II, the region was stricken by economic decline, rising unemployment and depopulation (Brodda and Heintel, 2009; Peterseil et al., 2013). At the same time, land use in the region underwent a profound transformation from an agrarian to an industrial socio-metabolic regime with the typical characteristics of land use intensification[2], although leading to very distinct changes in the agroecosystems within the region (Gingrich et al., 2013): a study site located in the northern low-lands turned to the specialization on high-yielding crops as well as chicken and pig farming and abandoned most grasslands and cattle stocks; another study site, located only 30km further south in the Alpine foothills, kept the share of grasslands stable and concentrated on cattle and pig farming with very high livestock densities, while abandoning low-yielding cropland in favor of new forest land. The socio-ecological changes that resulted in these distinct agroecosystems are valid until today and can, for example, be clearly seen on the land cover map in Figure 3-b.

To counteract socio-economic decline in the region, a variety of inter-sectoral measures and regional development and governance programs were launched since the 1980s, such as founding the cultural and tourism project *Eisenstraße*, hosting an Austrian State Exhibition, initiating a long-term European subsidy-program (LEADER–Liason entre Actions de Developement de l'Economie Rurale) as well as the establishment of a network of two national parks and two nature protection sites (Brodda and Heintel, 2009). Today, forestry, agriculture and tourism form the main sources of income in the region, making the socio-ecological systems also vulnerable to the negative effects of climate change (Peterseil et al., 2013).

To investigate the interconnectedness of environmental change, the history and future of land use, the socio-economic framework conditions as well as global change, a long-term socio-ecological research (LTSER) platform was established in the Eisenwurzen in 2004 (Peterseil et al., 2013). LTSER in general aims at understanding the patterns and transitions of socio-ecological systems over time to inform sustainability research and policy (Gingrich et al., 2016). It requires an interdisciplinary approach and examination of the three central themes of socio-ecological metabolism, land use and landscapes as well as governance and decision making (Haberl et al., 2006). Research activities in the LTSER platform Eisenwurzen integrate

[2] Industrialized intensification of agriculture is typically characterized by strong increases in yields and livestock numbers, mechanization, dependency on external inputs (of fossil fuels, agri-chemicals, animal feed and litter) as well as increased outputs to other socio-economic systems (Krausmann et al., 2016).

natural and social sciences with the humanities as well as local stakeholders and hence link interdisciplinary research to its regional non-academic context (Gingrich et al., 2016).

The part of the Eisenwurzen under investigation in this study (Figure 3) covers an area of 1,425 $km^2$ in the Northern Limestone Alps. It follows the central and lower course of the river Enns from the village of Admont (Styria) in the south-west to its estuary into the Danube in Enns (Upper Austria) and features a varied topography and land use. It can be roughly described as a gradient of elevation, climate and land use from the northern lowlands to the southern alpine mountains.

The region's northern part, with an elevation of around 250–400 m a.s.l. forms an elongated corridor of about 80 $km^2$ between Enns and Steyr. Geissler et al. (2003) elaborate that the area is characterized by good soil conditions, mostly made up of unconsolidated brown earth sediment. The climate is moderate with 8–9°C mean annual temperature and 800–1,000 mm annual precipitation. The majority of the area is arable farmland, primarily cultivated with intensively managed cereals and maize. Pastures and meadows are only found on the less favorable lands. Only small forest islands exist, which are mainly made up of spruce (Picea abies) in monoculture stands or semi-natural broad-leaved species. This situation results in pressures on the ecosystem, mainly concerning biodiversity loss and groundwater contamination (Geissler et al., 2003). There exist a variety of measures to tackle these problems within the Austrian Agri-Environmental Programme (ÖPUL). Accordingly, one potential for agro-ecological development is the establishment and expansion of site-typical broad-leaved forest stands to enhance ecological connectivity and reduce nitrate leeching. Site-typical broad-leaved trees in this area include sycamore maple (Acer pseudoplatanus), ash (Fraxinus), oak (Quercus) and beech (Fagus sylvatica) as well as poplar (Populus) and willow (Salix) in the floodplains of the rivers Steyr and Enns. The introduction of agroforestry systems could contribute to the realization of this potential.

The area between Steyr, Ternberg and Großraming is characterized by hilly landscape with an elevation of 300–1,000 m a.s.l. As Helga et al. (2005) point out, soils are mostly made up of shallow to medium, fine-grained and densely packed brown loam and brown earth as well as pseudogleys, exhibiting only average productivity for crops and being better suited to grassland. The climate is cool to moderate with 6–9°C mean annual temperature and humid with 1,166–1,560 mm annual precipitation. Almost two thirds of precipitation occur between April and September. Accordingly, the region south of Steyr is dominated by grassland, mostly found in the form of intensively managed meadows while pastures are only found on steeper terrain. Forests play a minor role in this area – deciduous mixed forests dominated by beech have thereby in part been replaced by intensively managed spruce stands. Nature conservation objectives include the restoration of biodiversity in these structurally poor areas (Draschan et al., 2003). Extensive meadow orchards (Streuobstwiesen), which qualify as agroforestry systems, are characteristic of the landscape and can be found close to the farms as well as to a larger extent in the lower lying areas between Steyr and Ternberg (Helga et al., 2005). These systems create a structural- and species-rich landscape with a high cultural value. Nevertheless, their continued existence is uncertain and primarily based on economic feasibility.

South of the Ternberg–Großraming axis, the landscape is dominated by a rugged topography and elevation increases continually with peaks between 1,200–1,500. Towards the south, the climate is generally colder and wetter, but varies widely due to the strong relief (differences range, for example, from 5.8°C annual mean temperature in Admont to 7.9°C in Altenmarkt, or from 1,423 mm annual precipitation in Admont to 1,841 mm in Hieflau). A small area at the western border of the study area forms part of the *Kalkalpen* national park. The southernmost

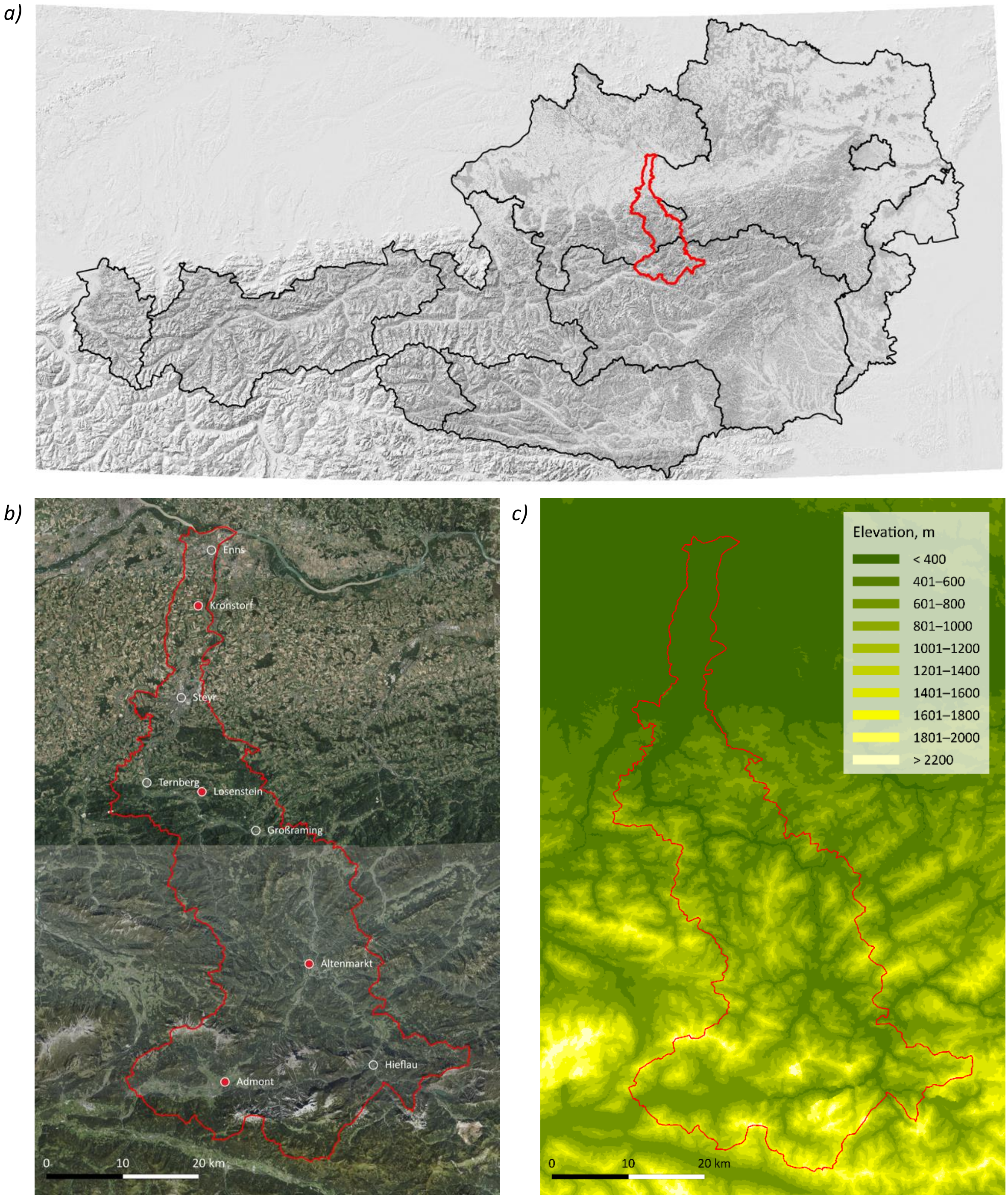


**Figure 3: The study region (red line) within the LTSER platform Eisenwurzen, presented on different geodata layers: [a] shows its location within Austria on a digital surface and terrain model (source: basemap.at); [b] shows a digital orthophoto indicating the prevailing land cover (source: geoland.at), with red dots marking the modelling sites (Kronstorf, Losenstein) and sensitivity test sites (Altenmarkt, Admont) and the white dashed line depicting the border between Upper Austria (north) and Styria (south); [c] shows the elevation gradient from the northern lowlands to the southern high peaks (source: geoland.at).**

area, which lies in Styria, forms part of the nature- and geopark *Steirische Eisenwurzen*. This high-alpine landscape is dominated by the rocky peaks of the *Gesäuse*, also a national park, with the highest elevation in the study area of 2,369 m (*Hochtor*). As stated by Draschan et al. (2003), the area is covered with forests (>85%), which are mostly made up of spruce and beech as well as stands of sycamore maple and ash on slopes and in gorges. While there is no commercial forestry taking place in the national parks besides re-naturalisation and control measures against the bark beetle (Höbinger and Kreiner, 2017), silviculture outside the national park tends towards spruce-dominated intensification on productive sites, while marginal yielding areas are abandoned. The occurrence of extensive meadow orchards declines towards the south and is mainly found in close vicinity to farms. They nevertheless form an important habitat and gene pool of old and rare fruit tree varieties and their continued existence is supported by various regional initiatives and cooperations (Styrian Eisenwurzen Nature & Geopark, 2020). Soils in this region are primarily made up of brown earth and brown loam; around the town of Admont there are also gleys, fluvial sediments and rendzina. Agricultural usability in the region is generally reduced by the steep and rocky terrain. In the valleys and flatter areas, which are not very well suited to cropland, grasslands are managed mostly intensively. Extensively managed humid grasslands exist only to a small extent. Alpine pastures are mostly managed extensively too, but dairy farmers are continually diminishing due to the increasingly tough socio-economic conditions (Draschan et al., 2003). Consequently, these species-rich habitats valuable to phytodiversity, are prone to forest encroachment with various implications for the region's biodiversity (Plank, 2017). This situation, however, is not unambiguous. ÖKOTEAM (2013), for example, found in a study covering three sites in the *Gesäuse*, that, from the viewpoint of zoological biodiversity, further significant extensification would be necessary, otherwise (semi-)natural forests could be preferable to the current state of the pastures.

## 3.2 Input data

### *3.2.1 Statistical data*

Statistical data was used in this study to evaluate Yield-SAFE model performance (chapter 3.2.3.2) as well as to formulate the land use scenarios (chapter 3.3). Relevant datasets were obtained from STATcube database and comprise the total production (in tons year$^{-1}$) and total area (in hectares year$^{-1}$) of field crops and fruit production (cherries, sour cherries) in Upper Austria and Styria (STATcube, 2019; Statistik Austria, 2019). Standard methodology for the registration of the production of plant-based products in Austria (Statistik Austria, 2014) applies under the Regulation (EG) No 543/2009 of 18 June 2009 concerning crop statistics; Federal Statistics Act 2000 (BGBl I No 163/1999 as amended).

### *3.2.2 SECLAND model*

The agent-based model (ABM) SECLAND provides geo-referenced simulations of future land use in the study region. It acts as one of the two central sources of input data in this study. The generated datasets, which are provided in a yearly time step from 2020–2050 as rasterized GeoTIFF files, form the basis for all landscape-scale calculations herein. This chapter briefly describes the modelling environment, structure and logic, as well as the simulated land use datasets applied in this study.

SECLAND forms part of a larger modelling environment called LUBIO (Land use, climate change and biodiversity in agricultural landscapes). LUBIO itself is designed to simulate the interactive effects of various climatic and socio-economic changes on land use and subsequently plant diversity in the study region from 2014–2050 (Dullinger et al., 2020; Mayer et al., 2019, 2018). It is made up of the ABM, which simulates land owner's decisions on land use, a GIS to map the simulated land use change, and a species-distribution model to assess changes in biodiversity patterns. Within this modelling framework, climate change affects societal, economic and political conditions, established along the narratives of the Shared Socioeconomic Pathways (SSPs) by O'Neill et al. (2017, 2014) and translated into input factors to the ABM. SECLAND's structure and logic are described in detail in Dullinger et al. (2020) and Mayer et al. (2019, 2018) and are briefly summarized here.

SECLAND consists of 1,329 agents who are all farm owners (except for two national parks). Farm owners invest labor to generate income. On this basis, they evaluate their "happiness" on account of set thresholds (i.e. labor should not exceed 1,800 hours year$^{-1}$; and income should not fall short of 20,000 EUR year$^{-1}$ or the average income of all farmers of the same type). Additional inputs (yields, prices, subsidies), derived from the narratives of the SSPs, provide the context for evaluation. According to their "happiness", agents decide for one of a predefined set of actions (i.e. termination, intensification, extensification, land use change).

Decisions are probabilistic and additionally influenced by farming type and farming style. The latter were identified in qualitative interviews and randomly assigned to each farm (i.e. traditionalist, yield optimizer, support optimizer, idealist, innovative). Subsequently, decisions and associated land use change translates into the GIS. The model outputs geo-referenced raster files with a 25-meter resolution, with each grid cell's value depicting its specific land use class in a given year. Due to the probabilistic modelling approach, 100 model runs per scenario were performed to integrate stochastic variation. Out of those, five runs were selected to represent the full spectrum of possible land use change.

For initialization, land use information was classified into 22 land use classes and assigned to 12,498 plots. The land use classes were harmonized from field recordings of 5,897 plots and supplemented by the Austrian Vegetation Database (Willner et al., 2012). A land use map for the year 2014 was generated based on data from the Integrated Administration and Control System (IACS) providing type (cropland, processing, livestock) and intensity (extensive meadows: number of cuts ≤2; extensive pastures: stocking density <1.5 livestock units per hectare; low-input cropland: farmer receives subsidies for organic farming).

Three SSP-based scenarios were modelled with SECLAND: a sustainability scenario (SSP1), a growth scenario (SSP5) and a business-as-usual scenario (BAU). In this study, datasets from the "SSP1A-centroid" simulation were used (hereinafter simply referred to as SSP1). The decision to use the SSP1 scenario was primarily based on the assumption that the implementation of agroforestry systems is more likely in a sustainability narrative (e.g. characterized by higher subsidies for low-input agriculture and agri-environmental measures, as well as an increasing environmental awareness of consumers (Mayer et al., 2019)). Additionally, application of this scenario offers a chance to compare agroforestry systems against a supposedly sustainable but nevertheless conventional agricultural system, thereby allowing for a differentiated discussion of sustainable land use.

The scenario suffix "A" refers to the standard SSP1 simulation, while "B" and "C" are SSP1-variations favoring energy crops and low-input agriculture, respectively. These variations were not used in this study. The scenario suffix "centroid" depicts the one simulation out of the 100 stochastic repetitions that is closest to the center of a bounding rectangle, formed by the two indices "homogeneity of land use classes in terms of area" and "total area of intensively used land". Further information on the selection of stochastic simulations can be found in Dullinger et al. (2020).

#### 3.2.2.1 Land use datasets

Land use datasets for 2020 to 2050 at annual resolution were derived from SECLAND. Table 2 provides an overview of the relevant land use classes[3] and their respective area extent in 2020 and 2050. Because of this study's specific focus on the implementation of agroforestry on agricultural lands, all classes depicting other land uses (including different types of forests, alpine habitats and infrastructure) are of no direct relevance and are thus excluded from analysis hereinafter.

Figure 4 shows the development of agricultural land use classes over time, while Figure 5 and Figure 6 show the geo-referenced datasets for 2020 and 2050, respectively. According to the SSP1 scenario, changes to agriculture in the region will be characterized by gradual abandonment of grassland, a trend towards the production of energy crops as well as an intensity shift to extensive management and production. The dynamics behind these grander transformations are quite complex. For a detailed Sankey visualization of the flows between the individual land use classes between 2014, 2031 and 2050 see Mayer et al. (2019).

By 2050, SECLAND computes a decrease of intensively managed grassland by 3 kha. This is mainly effected by the conversion of high-yielding plots to cropland for the production of energy crops and low-yielding plots to extensive meadows or broad-leaved forest. Extensive grasslands simultaneously decrease by approximately the same area extent, which is primarily converted to broad-leaved forest, too. Within the cropland category, the most noticeable change pertains to the conversion of non-cereal crops, and to a lesser degree cereal crops, to energy crops. Meanwhile, low-input production of cereal and non-cereal crops remains relatively constant or increases slightly, depicting a slight shift in land use intensity (although energy crops are primarily cultivated intensively). Miscellaneous arable land and fallow land remains relatively constant over the whole study period. Combination of all the changes leads to a slight increase of cropland and a strong decrease of grassland, ultimately resulting in a 25% reduction of total agricultural area (and subsequently a significant increase of broad-leaved forest).

As mentioned above, due to this study's focus on the potentials of agroforestry on agricultural lands, forests are not accounted for in the calculations. To solve the issue of conversion of agricultural land to forest, it was therefore decided that the amount of C sequestered in tree biomass drops out of the calculations the year the conversion from cropland or grassland to forest takes place. This decision naturally affects the results of the calculations and will be considered when interpreting the results.

[3] Unfortunately, extensive orchard meadows are not identified separately but contained within the extensive meadow category.

**Table 2: Relevant land use classes and their respective area extent in hectares, 2020 and 2050, as well as the change in area in percent.**

| *Land use class* | *Area extent in ha* | *2020* | *2050* | *Change in %* |
|---|---|---|---|---|
| *Energy crop* | | *724.5* | *2,208.4* | *205%* |
| *Energy crop low-input* | | *115.1* | *824.9* | *617%* |
| *Non-cereal crop* | | *2,180.8* | *294.4* | *-87%* |
| *Non-cereal low-input* | | *1,836.5* | *1,917.1* | *4%* |
| *Cereal* | | *1,528.7* | *628.8* | *-59%* |
| *Cereal low-input* | | *996.4* | *928.4* | *-7%* |
| *Misc. arable land* | | *595.5* | *589.5* | *-1%* |
| *Cropland fallow* | | *2,070.1* | *2,166.4* | *5%* |
| *Extensive pasture* | | *10,149.6* | *8,271.9* | *-19%* |
| *Extensive meadow* | | *4,534.9* | *3,341.9* | *-26%* |
| *Intensive pasture* | | *1,732.7* | *196.2* | *-89%* |
| *Intensive meadow* | | *1,879.8* | *269.7* | *-86%* |

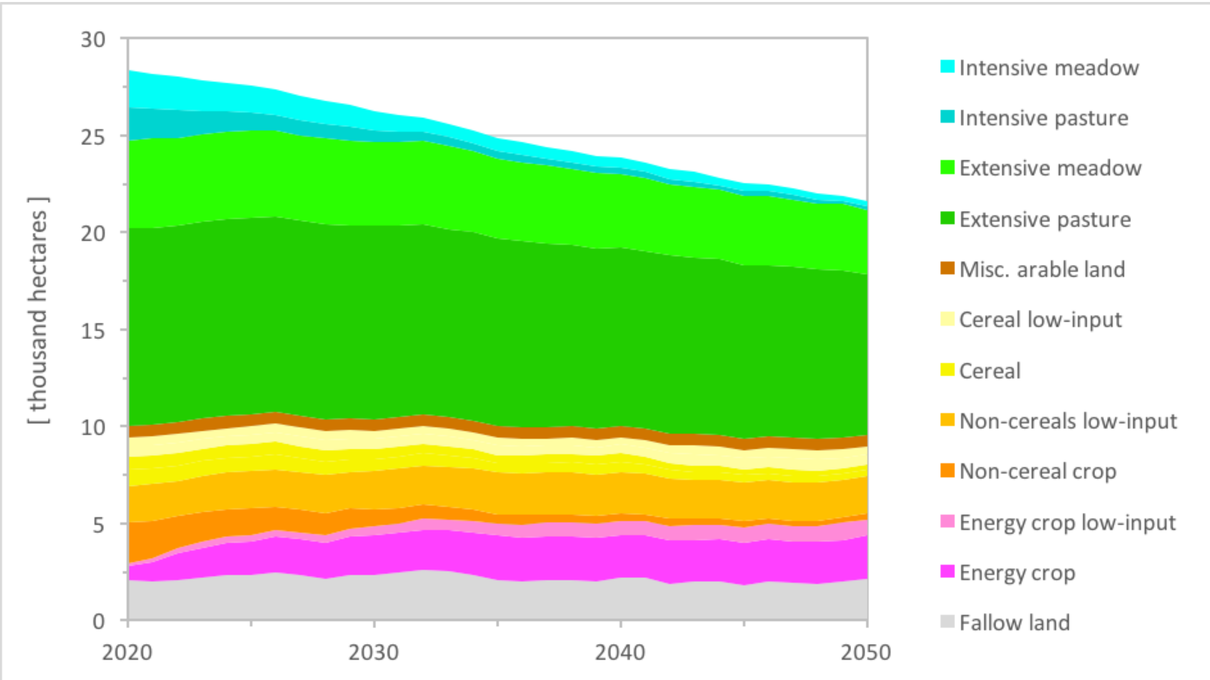


**Figure 4: Development of agricultural land use classes in thousand hectares, 2020–2050.**

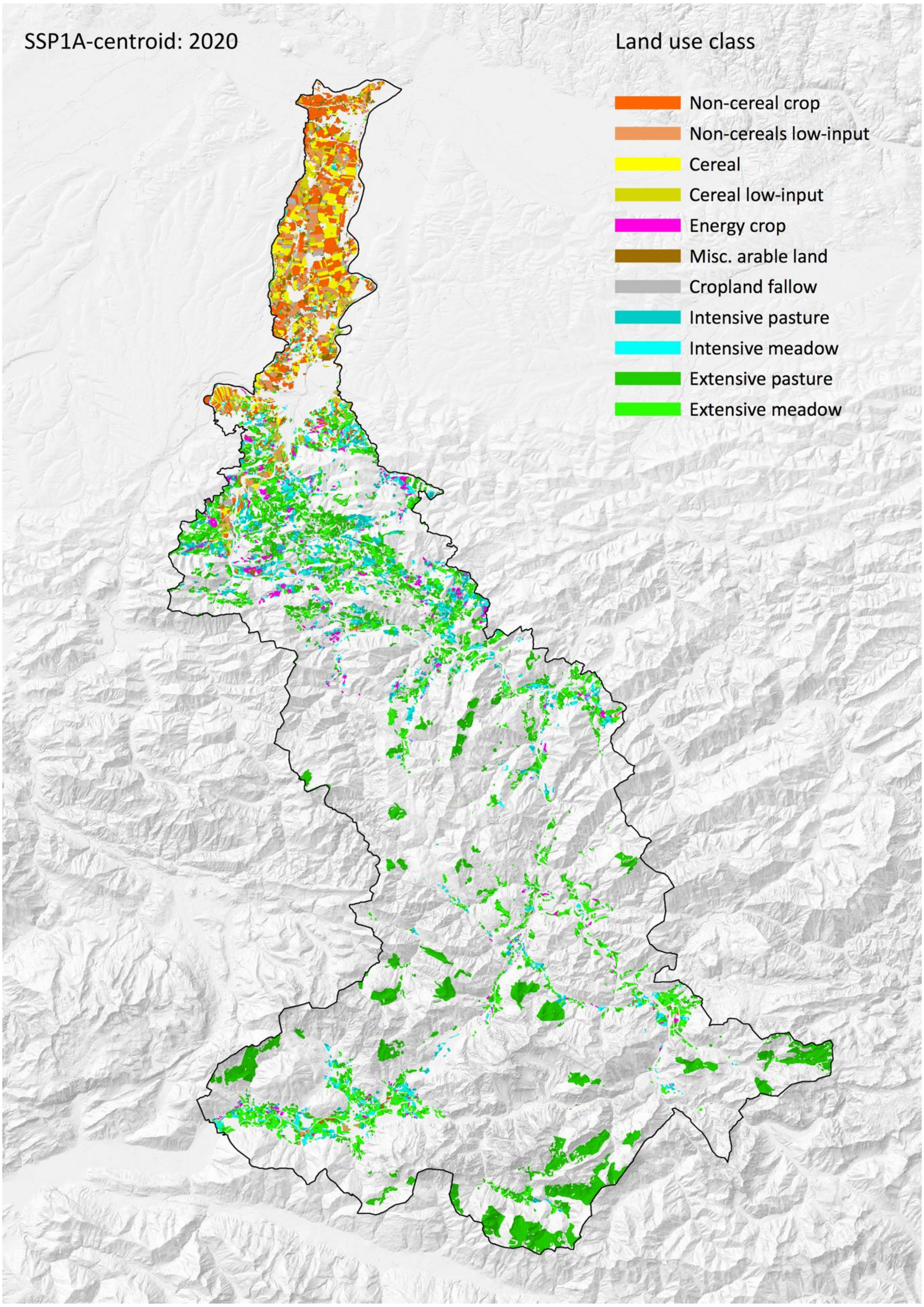


**Figure 5: Land use dataset of the SSP1 model run for the initial year 2020.**

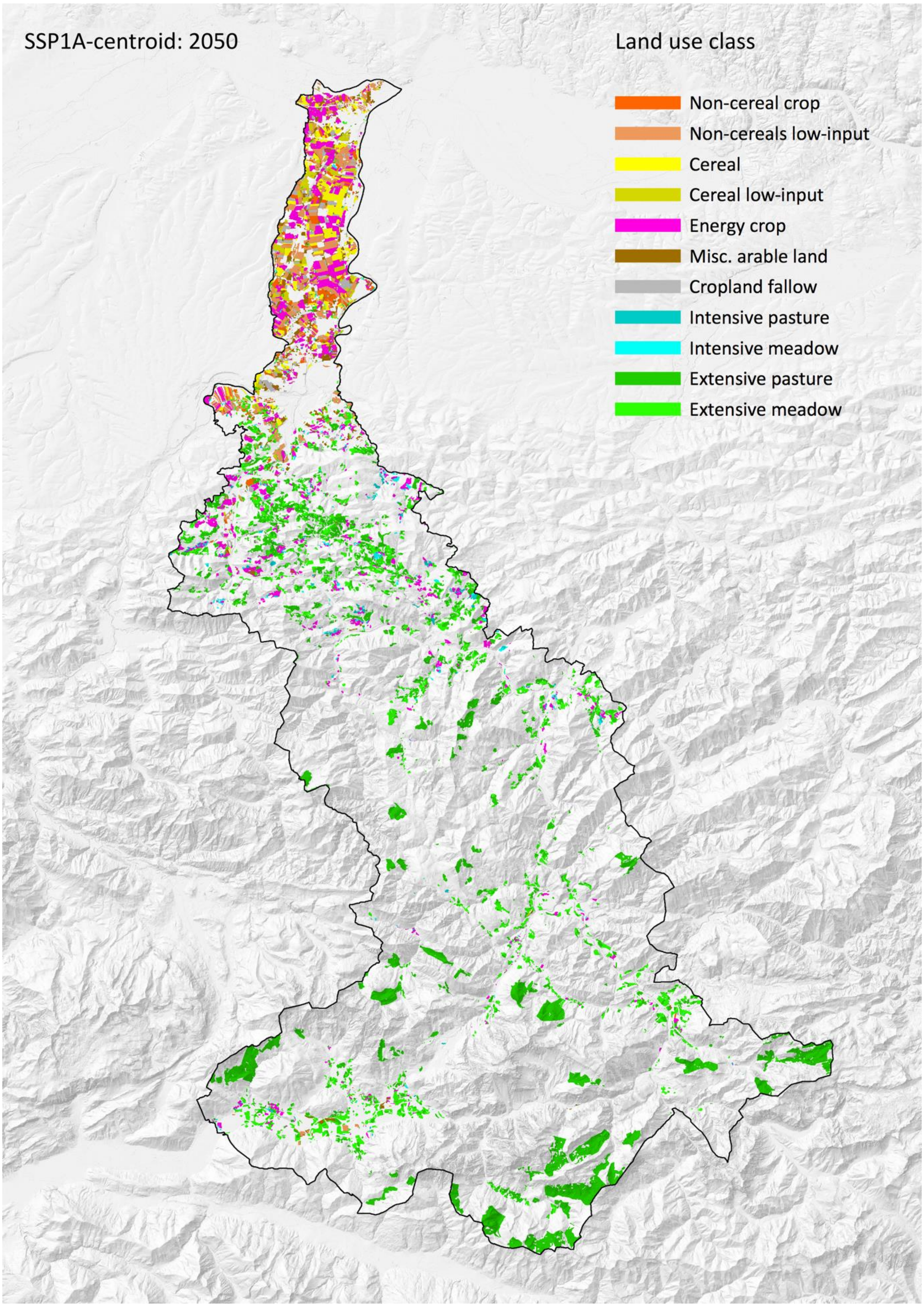


**Figure 6: Land use dataset of the SSP1 model run for the final year 2050.**

### 3.2.3 Yield-SAFE model

Yield-SAFE (van der Werf et al., 2007) is the second central data source in this study and the major modelling environment through which results were generated. Required model inputs are site- and system-specific relating to climate, soil and management factors. Outputs are a variety of data concerning the productivity and behavior of the systems' components. In this study, the biomass production of trees and crops are used for further processing. As such they determine the actual net primary production (NPP-act) of a system, which is a central HANPP indicator that forms the starting point for the calculation of the C dynamics. This chapter briefly describes the model's logic as well as the specific input parameters defined in this study. It furthermore includes sensitivity tests for a variety of input parameters and evaluation of output data with relevant yield statistics.

Yield-SAFE is a parameter-sparse, process-based dynamic model simulating resource capture, growth and production of forestry, arable and agroforestry systems in temperate climates. The model was conceptualized and implemented as part of the SAFE project (Silvoarable Agroforestry For Europe) (Dupraz et al., 2005). It has been calibrated, parameterized and revised continuously since its inception (Graves et al., 2010; Palma et al., 2014; Dupraz et al., 2005; van der Werf et al., 2007), on the one hand based on regional yield statistics and experimental data from test sites, on the other hand by application of modelled data from the agroforestry model HyPAR (derived from "Hybrid" and "Predicting Arable Resource Capture in Hostile environments") (Mobbs et al., 2001, 1999) and the crop model STICS (Simulateur mulTIdisciplinaire pour les Cultures Standard) (Brisson et al., 2003). Yield-SAFE was also improved and extended during the AGROFORWARD (AGroFORestry that Will Advance Rural Development) project between 2014–2016 (J. H. N. Palma et al., 2016; J.H.N. Palma et al., 2016) concerning various tree crop and environmental components.

The model is based on a simplistic, mechanistic eco-physiological simulation of tree-crop-environment interactions, governed by seven state equations for (1) tree biomass, (2) tree leaf area, (3) number of shoots per tree, (4) crop biomass, (5) crop leaf area index, (6) soil water content and (7) heat sum (van der Werf et al., 2007). These equations, together with a range of soil-, tree-, crop- and site-specific parameters and forcing functions (Palma et al., 2017), ultimately express resource acquisition and dry matter accumulation of trees and crops under competition for light and water (Graves et al., 2010). For each given day, the model thus calculates the radiation intercepted by trees and crops and derives predictions for tree and crop yields under consideration of water availability, which is deduced by a water balance model. The model considers the system to be spatially homogenous, meaning that light and water competition is simulated as an average over the whole plot (Dupraz et al., 2005). It is important to note, that nutrient competition is not accounted for in the model, resulting in nutrient non-limited biomass predictions (van der Werf et al., 2007). Other site-specific limitations such as weeds, pests, diseases and management shortcomings are not accounted for either. Daily inputs are mean temperature, incoming solar radiation and precipitation.

To facilitate application, Yield-SAFE was implemented as a web interface called EcoYield-SAFE[4] (J. H. N. Palma et al., 2016; J.H.N. Palma et al., 2016), which was used in this study. This interface was integrated with CliPick[5] (Palma, 2017), another web interface allowing access to regional climate change datasets for Western Europe from the Coupled Model Intercomparison Project – Phase 5 (CMIP5), allowing for simulations under the two Representative Concentration Pathways RCP4.5 and RCP8.5 (van Meijgaard et al., 2012).

[4] The EcoYield-SAFE web interface is available at: http://www.isa.ulisboa.pt/proj/ecoyieldsafe

[5] The CliPick web interface is available at: http://www.isa.ulisboa.pt/proj/clipick

Three limitations to the application of Yield-SAFE pertain to the limited scope of this master thesis. (1) The model has not been parameterized and calibrated specifically for the various model runs because of the lack of appropriate experimental data sets. Due to the extent and structure of the modelling exercise, i.e. the inclusion of six crop and one tree species as well as two modelling sites, acquisition of relevant data was not feasible. Instead, standard model calibration (Palma et al., 2017) for the selected crop and tree species, available through EcoYield-SAFE, was used. (2) When operating on a landscape-scale, integration of a plot-based model theoretically enables the differentiation of soil and climate variables at a very fine spatial resolution, simply by performing individual model runs for every site in question. Considering the size of the study region and the resolution of the land use dataset, this procedure would, however, result in an unfeasible amount of individual model runs. Additionally, this approach would also be limited by the availability of soil and climate data at the same resolution. Instead, model runs were performed using input data for one representative location for cropland and grassland, respectively. This approach ensures practicability but neglects spatial heterogeneities of soil and climate variables across the study region. Additionally, land use change leading to concentrations of specific land use classes on favorable or unfavorable plots cannot be accounted for either. (3) Tree mortality and loss of branch wood through windfall or disease is not accounted for.

#### 3.2.3.1 Application of Yield-SAFE

This chapter describes all input specifications made to the model, as well as the output parameters applied in the calculations.

Input parameters

Various input parameters must be specified to run the model, others are optional. A compilation of all input parameters and their specifications used in this study is shown in **Fehler! Verweisquelle konnte nicht gefunden werden.**. Additional input parameters available in Yield-SAFE but not listed explicitly hereinafter were not used in this study. Information on parameters was retrieved from (J.H.N. Palma et al., 2016). *Global Parameters* apply to all model runs, with the exception of *Site* and *Soil ID*, which vary for the cropland and grassland modelling sites (independent of the particular scenario), as well as *Years to model* and *Crop area*, which vary for the agriculture and agroforestry scenarios (independent of the particular modelling site). The most recent model version *May 2018* was selected. Model runs for the agricultural scenario (AGR) were performed for a period of 31 years from 2020–2050. For the agroforestry scenario (AFS), the period spans 61 years from 2020–2080. Although the study period extends no further than 2050, prolongation of the agroforestry scenario enables an outlook over the whole rotation period of the trees and to identify temporalities of CS and maximum potential levels. The climatic dataset used was *Future climate (RCP4.5)*. This dataset was the obvious choice representing the middle ground between the alternatives *Current climate* and *Future climate RCP8.5*, and thus being the best match to the SSP1 scenario used in SECLAND (Riahi et al., 2017; van Vuuren et al., 2011). *Water limitation* was set to *Yes*, assuming no irrigation throughout the study region. One modelling site was defined for cropland and grassland, respectively, indicated in Figure 3-b. The location of the *cropland modelling site* was set to *Latitude 48,160073* and *Longitude 14,451603*. The selected site is around the town of Kronstorf situated between the cities of Steyr and Enns. This is where the vast majority of cropland within the study region is concentrated on a relatively small and uniform patch of land with relatively constant climate variables. Nevertheless, the extent of cropland south of Steyr (mainly used for the production of energy

**Table 3: Yield-SAFE input parameters specified for the model runs in this study. Some parameters vary for the different scenarios and modelling sites, indicated in parenthesis.**

| ***Global Parameters*** | ***Value*** |
|---|---|
| *Model* | *May 2018* |
| *Start year* | *2020* |
| *Years to model (AGR)* | *31* |
| *Years to model (AFS)* | *61* |
| *Climatic dataset* | *Future climate (RCP4.5)* |
| *Water limitation* | *Yes* |
| *Latitude (Cropland)* | *48,160073* |
| *Longitude (Cropland)* | *14,451603* |
| *Latitude (Grassland)* | *47,945322* |
| *Longitude (Grassland)* | *14,443828* |
| *Soil ID (Cropland)* | *Medium-Fine* |
| *Soil ID (Grassland)* | *Medium* |
| *Soil depth (mm)* | *700* |
| *Crop species* | *according to land use class* |
| *Crop area (%) (AGR)* | *1* |
| *Crop area (%) (AFS)* | *0.84* |

| ***Agroforestry Parameters*** | ***Value*** |
|---|---|
| *Tree ID* | *Wild Cherry (Prunus Avium) (fruit)* |
| *Plant density (trees ha$^{-1}$)* | *80* |
| *DOY to plant* | *60* |
| *Alley width* | *2* |
| *Fruit production* | *Yes* |
| *Canopy effect on evapotranspiration* | *On* |
| *Tree height for temperature effect on evapotranspiration* | *4* |
| *Maximum difference in temperature in summer* | *5* |
| *Maximum difference in temperature in winter* | *2* |
| *Tree wind effect on evapotranspiration* | *On* |
| *Tree height for wind effect on evapotranspiration* | *1* |

crops) increases throughout the study period. As described in more detail in chapter 3.1, these southerly locations show stronger climatic variation, especially lower temperatures due to differences in elevation and topography. It is thus to be expected, that modelling results overestimate productivity of energy crops. Meadows and pastures are scattered throughout the alpine and prealpine parts south of Steyr. The *grassland modelling site*, located near the town of Losenstein, was selected because it features a relatively high share of meadows and pastures and a relatively moderate climate, avoiding extremes. The parameter was set accordingly to *Latitude 47,945322* and *Longitude 14,443828*. While this placement ideally


**Figure 7: Distribution of Soil ID (Wösten et al., 1999) with 1-km resolution and overlapping land use types (SSP1–2050) with 25-m resolution.**

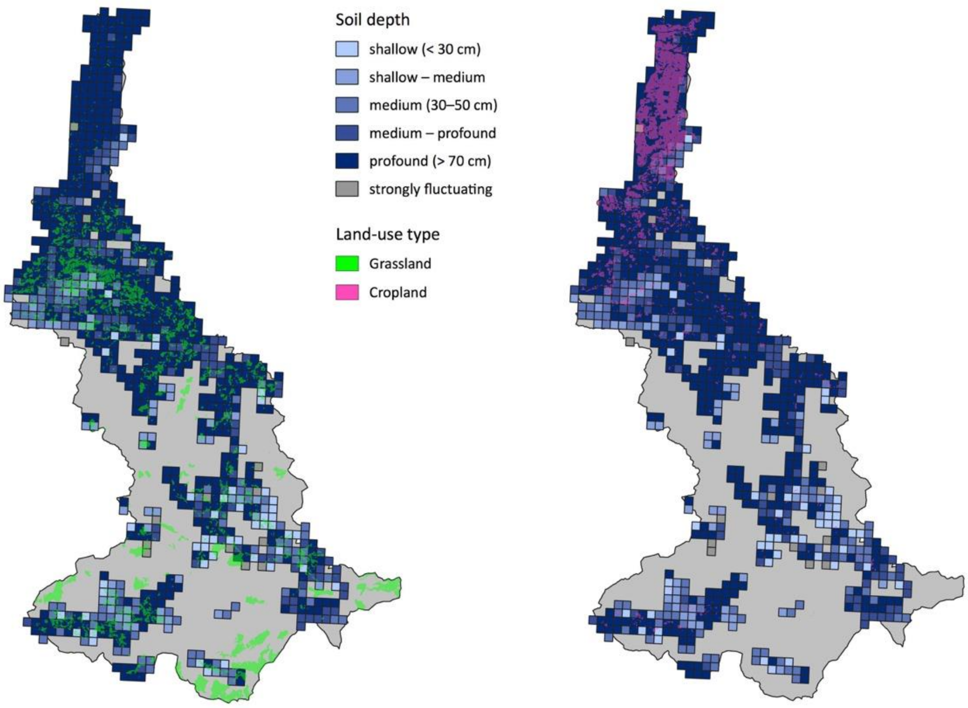


**Figure 8: Distribution of Soil depth with 1-km resolution and overlapping land use types (SSP1–2050) with 25-m resolution.**

ensures to acquire average productivity simulations, it is also expected to result in an overestimation of modelled grassland productivity, as large extensive meadows and pastures are situated at the southern end of the study region featuring the harshest conditions. This limitation, arising from reduction of coverage to two modelling sites, is accepted within the scope of this study and was already adhered to in chapter 3.2.3. Sensitivity tests for changing modelling sites to assess expected deviations are described in the next chapter 3.2.3.2. The *Soil ID* for cropland was set to *Medium-Fine* and for grassland to *Medium. Soil depth* was set to 700 mm for all model runs. These values do not exactly reflect the actual values found at the effective sites, but rather refer to the conditions found in the majority of the datasets. Sensitivity tests performed for soil parameters are described in chapter 3.2.3.2. Figure 7 and Figure 8 show the respective distribution of *Soil ID* and *Soil depth* (with 1-km resolution) in combination with the relevant land use types (with 25-m resolution). Analysis was based on soil data from *eBOD* (BFW – Bundesforschungs- und Ausbildungszentrum für Wald, Naturgefahren und Landschaft, 2016) and carried out in QGIS software. Soil IDs are derived from (Wösten et al., 1999). *Crop species* are set for each individual model run according to the specific land use class and independent of the modelling scenario. Species-equivalents for each land use class are specified in chapter 3.3.1. *Crop area (%)* for the AGR is set to 1 (i.e. 100% crops) and for the AFS to 0.84 (as defined by tree density and number of rows per hectare, specified in chapter 3.3.2).

*Agroforestry Parameters* concern all parameters relating to the trees and thus apply to the AFSs only. Further information concerning those parameters associated with the design of the agroforestry system are found in chapter 3.3.2. *Tree ID* is set to Wild Cherry (Prunus avium) (fruit). *Plant density* is set to 80 trees ha$^{-1}$. *Day of year to plant* is set to 60, as rooted cuttings with a new shoot should be planted out at the end of winter when the shoot is fully lignified (Ducci et al., 2013). *Alley width* is set to 2 meters. *Fruit production* is set to *Yes*. As stated by Palma et al. (2016), further developments are needed for this module. The inherent C flow should nevertheless not be omitted in this study. Values for canopy and tree wind effects on evapotranspiration were taken from model presets of a cherry orchard and pasture in Switzerland (Palma et al., 2017): *Canopy effect on evapotranspiration* was turned on. *Tree height for temperature effect on evapotranspiration* was set to 4m. *Maximum difference in temperature in summer and winter* was set to 5°C and 2°C, respectively. *Tree wind effect on evapotranspiration* was turned on. *Tree height for wind effect on evapotranspiration* was set to 1 m.

<u>Output parameters</u>

Model outputs used in this study are summarized in Table 4. Yield-SAFE predicts above-ground biomass production, provided in monthly time steps. A wide variety of output parameters concerning specific sub-compartments of plants, soils and livestock is available from the model. This study applies three of those parameters depicting the biomass production of crops and trees. Specifically, parameters were chosen that represent plant growth on the highest level, i.e. total biomass production, instead of using already downscaled parameters such as yield data (which are effected by application of harvest indices). This decision ensured more control and transparency over the acquired data.

The output parameter used for crops is “Yc1Ac”, measured in t DM ha$^{-1}$. This output refers to the accumulated crop biomass standing on one hectare of land in a given month, irrespective of how much of that area is in fact occupied by crops. Depending on the crop species, values peak between July–October each year, corresponding to harvest dates. Factors to calculate pre-harvest losses, yields and recovery rates were subsequently applied to crop data. The output parameter used for trees is “Bt_tonha”, measured in t DM $ha^{-1}$. This output refers to the accumulated stand biomass on one hectare of land in a given month. It allows for the derivation of yearly biomass production and accumulation as well as leave fall. The output parameter used for fruits is “ProductionFruitSum_Kghayear”, measured in kg $ha^{-1}$. This output refers to the yearly accumulated fresh weight of fruits on one hectare of land in a given month. It was converted to dry weight by subtraction of the water content. For a detailed description of all factors and conversions see chapter 3.4.

**Table 4: Yield-SAFE output parameters used in this study.**

| *Output family* | *Parameter* | *Description* | *Unit* |
|---|---|---|---|
| *Crop* | *Yc1Ac* | *Total crop biomass per total area* | *t DM $ha^{-1}$* |
| *Tree* | *Bt_tonha* | *Stand biomass* | *t DM $ha^{-1}$* |
| *Fruit* | *ProductionFruitSum_Kghayear* | *Yearly accumulated fruit production* | *kg $ha^{-1}$* |

#### 3.2.3.2 Model sensitivity and evaluation

A number of model runs was performed to acquire a sense of the behavior of the model with regard to certain input parameters, as well as a degree of evaluation by assessing the model’s output data. The baseline model runs were generally performed according to the input parameters specified in chapter 3.2.3.1, except for climate data that was set to “current climate”. Evaluation and sensitivity tests for crop and grass cultivars were performed on the basis of a 15-year mean (2000–2014), applicable to the modelled as well as the statistical data to compensate for singular climate extremes affecting year-to-year productivity. Sensitivity tests for wild cherry were performed over the whole study period of 31 years (2020–2050). Evaluation of wild cherry growth was performed by comparison to data from literature on the basis of the standard model runs in this study with an extended period until 2099 as well as one additional model run according to parameters of one case study. For the evaluation of fruit yield simulations, data from literature was used.

Sensitivity

Sensitivity tests were performed for the most important cultivar of the grassland and cropland categories in terms of area in the year 2050, as well as for the tree species wild cherry. Tests involved varying inputs of soil ID, soil depth and modelling site. Inputs were thereby varied by relatively large degrees, as to acquire significant results. Table 5 summarizes the relevant baseline input parameters and their variations.

**Table 5: Baseline input parameters and variations for sensitivity tests. Input variations of Soil ID, Soil depth and Modelling site apply for all three cultivars, except for the modelling site variation Kronstorf, which only applies to Wild cherry.**

| | *Cultivar* | *Soil ID* | *Soil depth* | *Modelling site* |
|---|---|---|---|---|
| *Grassland Baseline* | *Grass extensive (Switzerland)* | *Medium* | *700* | *Losenstein* |
| *Cropland Baseline* | *Oilseed* | *Medium-fine* | *700* | *Kronstorf* |
| *Tree Baseline* | *Wild cherry* | *Medium* | *700* | *Losenstein* |
| *Input variations* | *All cultivars* | *Very fine* | *100* | *Altenmarkt* |
| | | *Coarse* | *1,400* | *Admont* |
| | | | | *Kronstorf* |

Figure 9 shows the results of the sensitivity tests. The effect of changes to soil ID are generally within a minimal range of <1.5%. Changes are generally larger with “very fine” soil (i.e. >60% clay), resulting in reduced yields of -0.9% for grass and -1.3% for oilseed, while “coarse” soils (i.e. >65% sand) increase yields by up to 0.2%. No changes to tree biomass were detected for different soil IDs.
Changes to soil depth have stronger effects, whereby a drastic reduction to 100 mm results in reduced yields of -3.3% for grass, -1.6% for oilseed and -2.3% for wild cherry. A significant increase of soil depth to 1,400 mm only leads to an increase in yields of 0.4% for grass, 0.6% for oilseed and no change for wild cherry.
Changes to the modelling site show much larger effects on biomass production, primarily driven by changes in temperature. Compared to Kronstorf (the most northerly modelling site with the mildest climate), average temperatures decline by 2–5 °C at the respective sites, according to their geographic position (the further south, the colder)[6]. Biomass production of grass and oilseed declined significantly by 15–40%, with oilseed having the largest productivity losses in Admont. While warmer temperatures in Kronstorf led to an increase of tree biomass production of around 4%, the colder temperatures in Altenmarkt and Admont resulted in a decrease of 14% and 19%, respectively. From these test results the following conclusions can be drawn with respect to the selected input criteria.
*Soil ID:* Although there exist some fine soils in the study region, the relevant area is limited and mostly used for grasslands, which show less sensitivity to changes in soil ID than oilseed. Additionally, there is a tendency to convert parts of these areas to forest or leave them to succession by 2050. Yield deviations due to variations in soil ID of around or well below 1% can thus be neglected.
*Soil depth:* While reductions in biomass production reach over 3% (for grass on very shallow soil), the overall situation is similar to that of soil ID. There exist very little such shallow soils in the study region, which also adhere to the tendency of being converted to forest or left to succession by 2050. Furthermore, the tested value of 100 mm is very low, even for shallow soils (which are <300 mm). Most soils in the study region are profound, that is >700 mm. The increase to 1,400 mm only led to an increase in biomass production of below 1%. Therefore, deviations due to changes in soil depth can be neglected, too.
*Site:* As expected, deviations in biomass production due to climatic conditions occurring at different modelling sites are much more significant, and they are becoming more pronounced the further south a location lies. When looking at the spatial distribution of agricultural lands throughout the study region, most cropland is however situated north of Steyr, and even the large majority of grasslands is found in the northern parts of the alpine region. Therefore, yield deviations due to changes in modelling site and subsequently temperature are acknowledged, but the loss in accuracy is accepted within the scope of this study.

Evaluation

Comparison of land use systems in this study is based on simulated data alone. The model’s over- and undermodulation of biomass production does subsequently not factor directly into the study results, because deviations hence occur to the same degree in all of the examined systems. Nevertheless, evaluation was necessary for the formulation of land use scenarios as well as a useful reference point for the interpretation of results, especially when comparing them to other HANPP studies.

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[6] The location of the modelling sites and sensitivity test sites are indicated in Figure 3-b.

Evaluation of model performance for crops was accomplished by comparison of simulated productivity data produced in Yield-SAFE to actually reported yield statistics from the region. To make the data comparable, pre-harvest losses and harvest indices were applied to the simulated data (see chapters 3.4.2 and 3.4.3 for a detailed description), as well as dry matter factors to the reported yield statistics[7]. Comparison involved only those cultivars, which were eligible to represent a SECLAND land use class (and as such were identified during the scenario building process described in chapter 3.3). Yield statistics from Upper Austria were applied for cropland model runs, whereas grassland model runs were validated with averaged values from yield statistics from Upper Austria and Styria.[8]

Performed simulations produced substantial deviations between 119% and -65% from respective yield statistics (Figure 10). When assessing these results, it needs to be considered that the model was not specifically calibrated for the cultivars in the study region, but standard calibrations were used. This might explain why the model overmodulates for some species and undermodulates for others.

Even though, as stated at the outset, deviations from reality are not an intrinsic problem to this study, a threshold for excessive deviation was set anyhow to avoid too large a distortion of HANPP results[9]. In doing so, and simultaneously being aware of the fact that this range might be regarded as too large in other contexts, deviations of >30% were considered unsatisfactory. Simulations for Barley (-28.5%), Grain maize (-10.9%), Grass (80% Dactylus) (-8.4%) and Grass (extensive CH) (-9.8%) were inside the threshold, but it was exceeded by Forage maize (-64.9%), Oats (119.2%), Oilseed (57.5%), Sugar beet (56.4%) and Winter wheat (44.4%). Forage maize, Oats and Sugar beet were subsequently eliminated from further processing. Even though Oilseed and Winter wheat surpassed the threshold too, these cultivars still had to be included in the following calculations. This decision is owed to the fact, that these two cultivars are considered indispensable in the formation of the land use scenario, as they are fundamental to the representation of their respective land use classes. *Indispensable* thereby refers to the fact that Yield-SAFE does not offer alternatives to these land use class-equivalent crop species. Consequently, if deviations would have been considered to be too high, the land use class “energy crops” would have had to be eliminated completely from further processing, and the land use class “cereal” would have lost their primary species-equivalent. For further information on the formulation of the scenarios see chapter 3.3.1.

Due to the lack of appropriate growth statistics for wild cherry in Austria, model simulations were first compared to data found in literature (

[7] Applied factors for the water content of grains and grasses was 14% and for oilseed 10% (Gingrich et al., 2015, SOM).

[8] The vast majority of cropland within the study region is situated in Upper Austria, whereas grassland is located in Upper Austria as well as in Styria (see chapters 3.1 and3.2.2).

[9] HANPP is determined by subtraction of actual NPP from potential NPP (see chapter 3.4). Thus, excessive over- or underestimation of actual NPP results in the distortion of HANPP indicators.

Table 6). Generally, dimensions of wild cherry growth from literature have a relatively wide range. This could be explained by the different sources reporting values from natural growth as well as silvicultural management, as well as the fact that wild cherry growth depends strongly on climatic and soil conditions (Ducci et al., 2013; Evans, 1984; Welk et al., 2016).

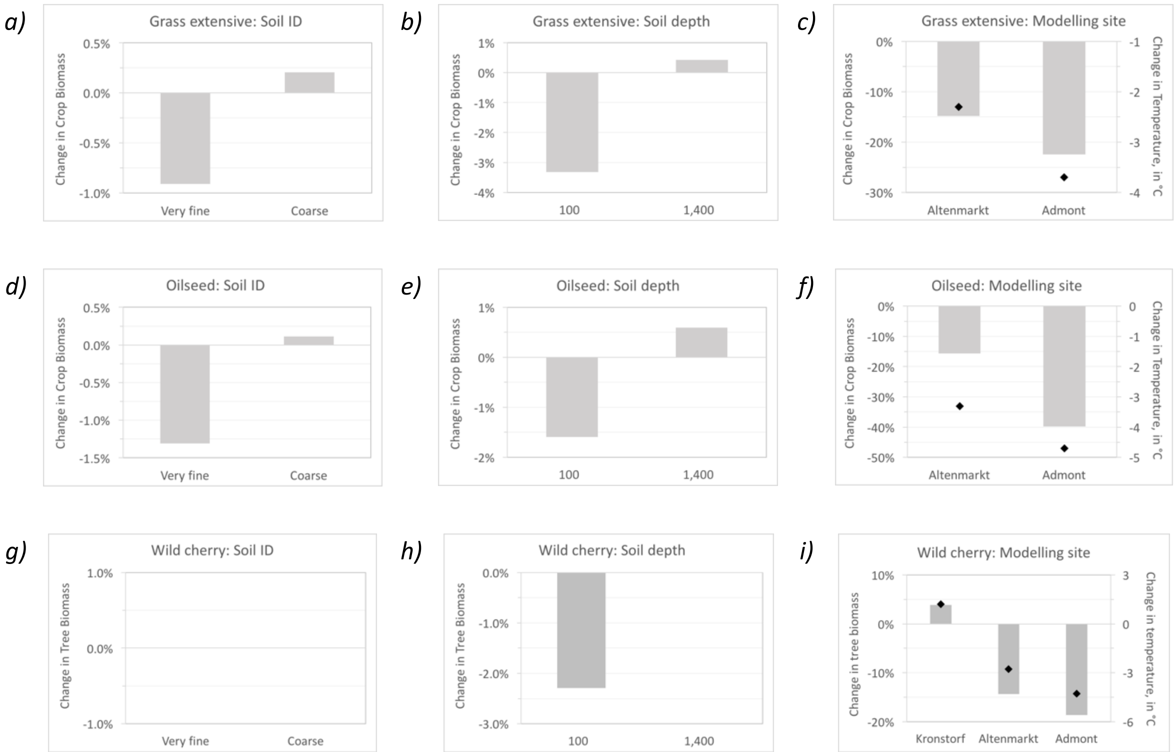



**Figure 9: Sensitivity tests of the input parameters Soil ID, Soil depth and Modelling site, for [a–c] Grass extensive (Switzerland), [d–f] Oilseed and [g–i] Wild cherry. Changes refer to the deviation from the respective baseline model run. Changes to total crop biomass and total tree biomass (corresponding to actual NPP) are given in percent, changes to temperature in degree Celsius.**

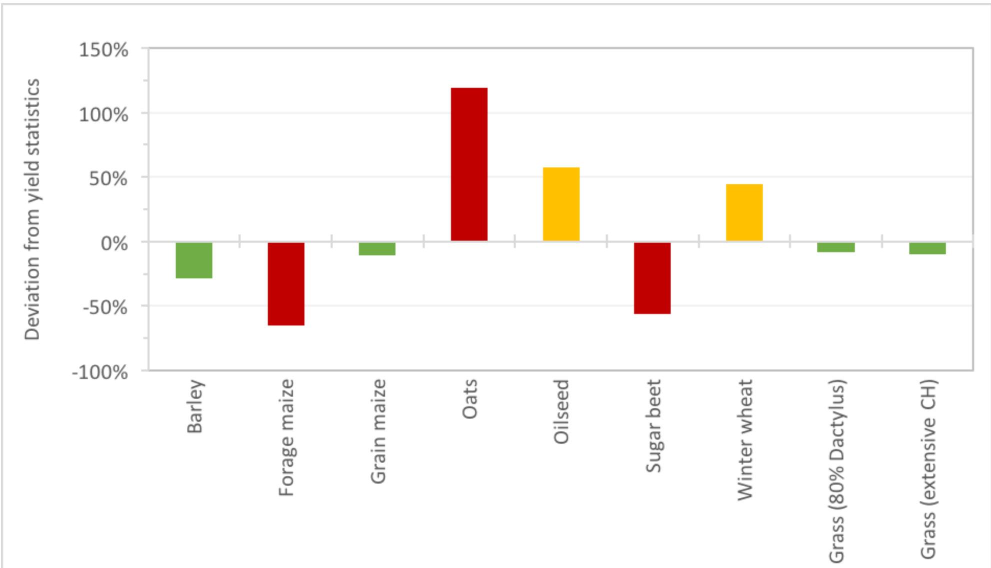


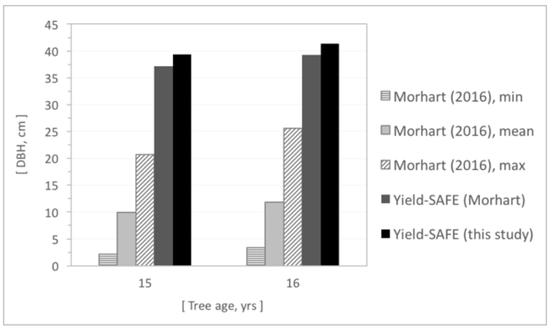

**Figure 10: Evaluation of Yield-SAFE model performance for selected cultivars. Green bars represent cultivars with an acceptable range of deviation (<30%), yellow bars represent cultivars outside this range (>30%) but considered indispensable and thus included for further processing (see chapter 3.3.1), and red bars represent cultivars eliminated from further processing.**

**Table 6: Range of reported and simulated growth dimensions of wild cherry.**

| *Data source* | *Tree age, yrs* | *Height, m* | *DBH, cm* |
|---|---|---|---|
| *Ducci et al. (2013)* | *70–80* | *28–28* | *50–90* |
| *Welk et al. (2016)* | *100–150* | *15–32* | *90–120* |
| *Evans (1984)* | *50–60* | *20* | *60* |
| *Yield-SAFE* | *60* | *12* | *80* |
| *Yield-SAFE* | *80* | *13* | *88* |

Ducci et al. (2013) states tree height at maturity of 25–28 m and DBH of 50–90 cm in best situations, with a natural life span of 70–80 years. Welk at al. (2016) report a much larger range and longer life span with heights of 15–32 m, DBH of 90–120 and lifespan from 100–150 years. Evans (1984) describes wild cherries with 20 m height and 60 cm DBH with 50–60 years of age. In this study, simulated wild cherry growth after 60 and 80 years reached approximately 12 and 13 m in height as well as 80 and 88 cm in DBH, respectively. Yield-SAFE simulated tree heights are clearly below the reported range and even below the reported minimum of 15 m. Simulated DBH values, on the other hand, are clearly above or at the upper limit of the reported range, when considering tree age in the comparison.
A second comparison was performed using data from a research test site. Morhart et al. (2016) sampled a total number of 39 unmanaged wild cherry trees in south-west Germany to assess above-ground growth 15–16 years after planting[10]. Comparison was made with a standard Yield-SAFE model run as applied in this study as well as with an additional model run with input parameters corresponding to Morhart et al. (2016)'s study-specific timeframe, soil and stand characteristics. Simulated data deviated substantially from the observed data with the same tendency as identified before (Figure 11). Observed results showed a mean DBH of 9.9–11.8 cm and a mean tree length[11] of 8.5–10.2 m, whereas Yield-SAFE model runs showed results for DBH of 37–41 cm and tree height of 5.6–6.2 m. Simulated wild cherry growth hence results in thicker stems but smaller trees.

*a)*




*b)*

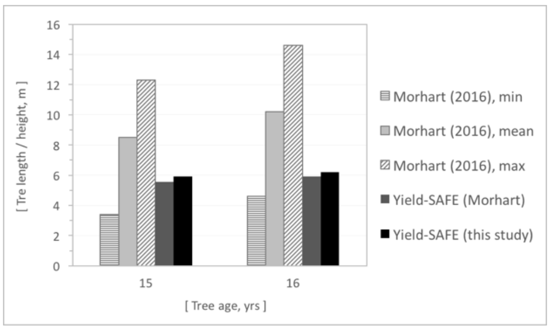



**Figure 11: Comparison of observed and simulated wild cherry growth data of [a] DBH in cm and [b] tree length/height in m. Observed data is from Morhart et al. (2016), simulated data from a Yield-SAFE model run corresponding to the specifications of Morhart et al. (2016) and a standard model run as applied in this study.**

[10] Trees were planted in 1997 and measured in 2012 and 2013.

[11] In contrast to tree height, tree length refers to the total length of the stem when partitioned into smaller sections. Tree length is hence larger than tree height.

The sampled trees on the experimental site included dominant and co-dominant individuals as well as dominated and suppressed individuals (with additional augmentation of the allometric curve at the lower end). In contrast, simulated tree growth will be much closer to potential growth (within the model's performance) if climate and soil parameters permit. This might attenuate differences in DBH but accentuate differences in height.
The behavior of Yield-SAFE to simulate rather small trees with broad stems can be explained by the specific parameterization and calibration of wild cherry. Palma et al. (2017, p. 103) state the following:

> *"The main output from the Yield-SAFE model is tree biomass. From the biomass value, volume, diameter and height of the average tree are calculated. Whenever a pruning is made, the amount of biomass is reduced and consequently, the values of volume, height and diameter. Due to this model structure, there is a need to calibrate the model with two sets of different parameters for the same species when the same tree can be conducted to produce timber or fruit. The types of management options are different and so are the results in the tree growth. When the tree is managed to fruit production, the prunings are made to increase leaf area and fruit productivity, so the calibration has to be made in order to respond that way. When the main output is timber, initial planting density is higher (800 to 1000 trees per hectare) and all the operations made are to insure straight and tall trees."*

Measured data for calibration of wild cherry for fruit production was taken from 22 cherry trees of a grazed orchard consisting of young and old trees for cherry production in Switzerland (Palma et al., 2017). The life span of the trees was about 60 years and the density was 80 trees hectare$^{-1}$. Model predictions of tree height and DBH showed a good fit with measured data.
During the conception and methodological development of this study, both variations of wild cherry (for fruit and timber production) were available for modelling. Unfortunately, the option of wild cherry for timber production was not available anymore in the Yield-SAFE version finally applied in this study. The reason for this ex post exclusion is unknown.
Evaluation of fruit production was performed again on the basis of data from literature. Data from sweet cherry production orchards (for example Lanauskas et al., 2012; Raduni et al., 2011) do not qualify for comparison here due to the vast differences in management techniques of intensive training systems as well as the application of grafted high yielding sweet cherry varieties. Sereke et al. (2015) refer to cherry yields of 41 kg tree$^{-1}$ in the lowlands of Switzerland planted in similar densities and agroforestry stands as applied in this study. For stands of 40 and 70 trees ha$^{-1}$ they accordingly calculated annual cherry yields of 1.8 and 2.9 t ha$^{-1}$. In comparison, annual cherry yields as modelled in this study are lower. 30 and 60 years after planting, annual cherry yields amounted to 24 and 28 kg tree$^{-1}$ as well as 1.9 and 2.3 t ha$^{-1}$, respectively.

## 3.3 Land use scenarios

Two distinct land-use scenarios – the agricultural scenario (AGR) and the agroforestry scenario (AFS) – were developed to assess the effects of the implementation of agroforestry on agricultural lands in the study region. AFS is furthermore implemented in two variations: (i) AFS-MAX depicts the maximum potential, and (ii) AFS-GRAD a more realistic approach. All scenarios assume land use change from SECLAND's SSP1 model run between 2020–2050, but differentiate in terms of internal structure, implementation and modelling period. Table 7 provides an overview.

**Table 7: Overview of the implementation of land use scenarios**

| *Scenario* | *Land use system* | *Modelling period* | *Agroforestry implementation* |
|---|---|---|---|
| *AGR* | *Agriculture* | *2020–2050* | *None* |
| *AFS-MAX* | *Agroforestry* | *2020–2080* | *Abrupt, 100% in 2020* |
| *AFS-GRAD* | *Mixed* | *2020–2080* | *Gradual, 100% by 2045* |

AGR depicts the current form of agricultural land use in the region, i.e. agriculture without trees (except for some areas exhibiting extensive orchard meadows, which are subsumed in the extensive grassland category in SECLAND, see chapter 3.2.2). AGR is modelled for the full study period from 2020–2050.

AFS depicts the integration of trees on agricultural lands. The allocation procedure of crop species to land use classes applied to AGR thereby forms the basis for AFS. The structure and design of the prototypical agroforestry system is described in chapter 3.3.2. It includes the choice of tree species, spacing and arrangement on the plot. Implementation of AFS is additionally calculated in two variations: AFS-MAX assumes an abrupt transition to agroforestry on 100% of the available land in the initial year 2020. It represents the maximum potential of this scenario. AFS-GRAD, on the other hand, depicts a more realistic – if still purely hypothetical – scenario, which assumes a gradual transition to agroforestry on 100% of the available land in 5-year time steps until 2045, resulting in a delayed manifestation of effects. To assess saturation effects of CS in agroforestry systems, modelling of AFS scenarios is furthermore extended to the year 2080. The last available areal distribution of land use classes from the SECLAND dataset in the year 2050 is thereby held constant. Extending the modelling period over a period of 61 years – corresponding to an assumed full harvest cycle of the trees – enables the calculation of the potential carbon carrying capacity (CCC) of AFS. This represents the amount of C stored in the system, if the age structure of the trees were equally distributed throughout a full harvest cycle. Considering the importance of temporality when dealing with climate change mitigation (Röder and Thornley, 2016), this serves as an important indicator for assessment.

### *3.3.1 Agricultural scenario (AGR)*

AGR assumes the continuation of current land use systems on agricultural lands in the study region, i.e. the production of crops and grasses in the absence of trees, and serves as a baseline against which to compare AFS. Building on SECLAND's land use classes and Yield-SAFE's productivity simulations, AGR and AFS are made comparable.

SECLAND's land use classes are thereby strongly aggregated, including a variety of different species each, and Yield-SAFE also provides a choice of different crop species to model. To allocate the most representative Yield-SAFE crop species to each SECLAND land use class, a systematic approach was chosen. Table 8 shows the outcome of this scenario building process. Thereby, the most relevant crop species of each land use class was identified from production statistics, according to each cultivar's acreage and share within its land use class. All cultivars

with minor importance in the region were eliminated from the scenarios, with cut-off values of <30% share in area. The ranking was based on the area extent provided by agricultural statistics from Upper Austria in the year 2014. Then the corresponding Yield-SAFE species could be allocated accordingly, while all cultivars without a direct land use class-equivalent were eliminated, too.
While this synchronization and reduction procedure provided a vastly reduced list of cultivars, some of those were further eliminated due to large deviations of model performance from statistical yields. These include Forage maize, Oats and Sugar beet. Despite distinctive deviations for Oilseed and Winter wheat these cultivars were considered indispensable and were thus incorporated into the land use scenario (see chapter 3.2.3.2 for further details).

**Table 8: Compilation of SECLAND's land use classes included in the scenarios, corresponding representative species identified from production statistics and available equivalents in Yield-SAFE.**

| ***Land use class*** | ***Statistics*** | ***Yield-SAFE*** |
|---|---|---|
| *Energy Crop* | *Winter oilseed* | *Oilseed* |
| *Non-cereal crop* | *Grain maize incl. CCM* | *Grain maize* |
| *Cereal* | *Winter barley* | *Barley* |
| *Cereal* | *Winter wheat* | *Winter wheat* |
| *Misc. arable land* | *Egart* | *Grass (extensive CH)* |
| *Fallow land* | *n.d.* | *Grass (extensive CH)* |
| *Extensive pasture / meadow* | *Meadow, one and two mowings* | *Grass (extensive CH)* |
| *Intensive pasture / meadow* | *Meadow, three and more mowings* | *Grass (80% Dactylus)* |

In the case of grassland, differentiation between the individual land use classes "pasture" and "meadow" was not possible. Although there exists a livestock module in Yield-SAFE, this module primarily aims at calculating carrying capacities through the conversion of biomass to energy and does not provide any changes to plant productivity per se. Consequently, consolidation into a single land use class seemed favorable in terms of modelling biomass production. Nevertheless, differences in land use intensity should not be lost, despite obvious differences in terms of classification: while intensity levels for pastures are defined by the stocking density (with a threshold of 1.5 livestock units $ha^{-1}$), intensity levels for meadows are defined by the number of mowings (with a threshold of two mowings $year^{-1}$). While multiple mowings cannot be reproduced in Yield-SAFE, there exists an appropriate equivalent for modelling extensive grassland, i.e. Grass (extensive CH), which was calibrated with appropriate experimental data. For intensive grassland, however, the crop species Grass (80% Dactylus) was used as a proxy instead of introducing a factor to depict the associated increase in yields. As shown in chapter 2.2.2.2, simulated biomass values for these two cultivars are well within the acceptable range of deviation. Grass (extensive CH) furthermore represents the land use classes Miscellaneous arable land and Fallow land.
SECLAND's land use classes depicting extensive crop management, i.e. "Energy crop low-input", "Cereal low-input" and "Non-cereal low-input", adhere to the same species-equivalents as their conventional counterparts. Further information on the conversion factors from conventional to low-input yields can be found in chapter 3.4.3.

### 3.3.2 *Agroforestry scenario (AFS)*

The AFS scenario was formulated in the form of a silvoarable alley cropping system to potentially maximize CS and food production. AFS was directly derived from AGR, using the same land use classes and corresponding crop species and complementing them with a suitable tree species. The choice of tree in this study was thereby determined by the focus of this study to assess the trade-off between long-term CS and biomass harvest as well as the options provided in Yield-SAFE. On this basis, relevant tree species had to fulfil the following criteria: (i) trees must be hardy enough and suitable for an alpine climate; (ii) trees must provide some form of agricultural product and/or long-lived timber (in contrast to trees primarily grown as a source for bio-energy, e.g. in short rotation coppice systems, or for short-lived wood products); (iii) trees must fit into assumed regional development policies concerning climate change and other ecological factors (e.g. vulnerability to extreme weather, invasive neophytes) as well as cultural/aesthetical reasons. The one tree species that fits all above-mentioned criteria is wild cherry (Prunus avium L.) and was thus chosen for AFS. The remaining tree species that at least fulfilled criterion (i) were poplar (Populus L.), norway spruce (Picea abies) and black locust (Robinia pseudoacacia). Poplar (Burger, 2006; Grosser, 2006; Reeg et al., 2009), however, did not fulfil criteria (ii), norway spruce (Hagen-Thorn et al., 2004; Lexer et al., 2001; Lindner et al., 2010; Mayer, 2013) as well as black locust (Schulz, 2017; Vítková et al., 2017) did not fulfil criteria (iii).

#### 3.3.2.1 Wild cherry (Prunus avium L.)

Wild cherry is a fast growing medium-sized deciduous tree that grows to 15–32 m height with a stem DBH of 50–120 cm, developing a straight trunk; the first erect-pyramidal crown shape becomes more rounded on old trees (Ducci et al., 2013; Evans, 1984; Welk et al., 2016). Wild cherry was brought to Europe from Asia Minor and was spread throughout the Roman Empire around the first century AD (Raftopoulo in Schmidt, 2010). It occurs naturally throughout European temperate forests and is found at colline to submontane altitudes up to an elevation of 1,700 m a.s.l. in the Northern and Central Alps (Ducci et al., 2013; Welk et al., 2016). In Europe it often occurs in several mixed deciduous forests type alliances of the class Querco-Fagetea, together with ash (Fraxinus excelsior), sycamore maple (Acer pseudoplatanus), elm (Ulmus glabra), beech (Fagus sylvatica), hornbeam (Carpinus betulus) and oak (Quercus petraea and Quercus robur) (Welk et al., 2016).

Wild cherry is a valuable hardwood and fruit tree species, and as such provides long-lived timber products that store C far beyond the harvest cycle of the tree. Wild cherry wood is in high demand throughout Europe (Welk et al., 2016) as it is increasingly substituting tropical hardwood, resulting in stable market prices (Ducci et al., 2013) and the sparing of tropical forests. Additionally, wild cherry provides agricultural yields in the form of cherries, serving humans as a source of food for thousands of years and today being cultivated as a fruit tree in temperate regions all over the world (Welk et al., 2016). Wild cherry is also beneficial to biodiversity, making it a good fit to the general aspiration of a sustainable land use scenario. Flowers are pollinated mainly by honeybees, wild bees and bumblebees (Welk et al., 2016), and fruits are eaten by 48 bird species and the trees are habitat to over 100 beetle species (Bußler, Schmidt in Schmidt, 2010). Furthermore, wild cherry has a high aesthetic, cultural and touristic value, with a very good fit into the region's characteristic *image* marked by existing extensive orchard meadows (Styrian Eisenwurzen Nature & Geopark, 2020).

Target regions for silvoarable agroforestry systems with wild cherry were estimated in a study by Reisner et al. (2007). Target regions were defined by overlaying arable landscapes

(excluding pastures and heterogeneous agricultural land) with regions allowing for productive tree growth as well as areas with environmental problems that could be helped to solve with agroforestry (e.g. erosion, nitrogen leeching, landscape diversity). Target regions coincide with the northern lowlands of the Eisenwurzen, whereas the southern parts of the study region were not accounted for as arable land in that study.
Overall productivity depends strongly on the site conditions. On silvicultural experimental sites in Bavaria, Germany, high growth rates suitable for timber production (i.e. bole height > 3 m and DBH > 40 cm after 70 years) occurred only on well-drained and nutrient-rich soils with good water availability (Pretzsch et al. in Schmidt, 2010). Accordingly, acidic, dense, waterlogged and nutrient-poor soils are not suitable for wild cherry, which applies only to small areas in the very south of the study region. General risks for wild cherry include late frost, windfall, various bacterial and fungal diseases as well as browsing and fraying (Albrecht in Schmidt, 2010), all of which potentially exist in the study region, especially late frosts in higher altitudes and windfall due to extreme weather events.
Sheppard and Spiecker (2015) suggest that there is low potential for cherry yields on timber-optimized trees and within forested systems. On the other hand, veneer production from timber requires log-lengths of 3m and a DBH of 40cm, not easily achieved in fruit-optimized trees. Although wild cherry in Yield-SAFE was calibrated for fruit production with experimental data from a grazed cherry orchard in Switzerland (see chapter 3.2.3.2), developments of tree DBH suggest that veneer production might still be feasible, although tree height is comparably low. In any case, the discrepancy between fruit and timber production poses a limitation to this study. It becomes clear at this stage that at best only a relatively small proportion of C sequestered in a fruit tree can be continuously stored in long-lasting veneer products after harvest. Remaining timber and branch wood might, nevertheless, be used energetically and as such substitute the use of fossil fuels. This effect is, however, not included in the calculations in this study.

#### 3.3.2.2 Agroforestry system design

In this study, the agroforestry design follows suggestions for silvoarable alley-cropping systems with high value timber and fruit trees in a continental climate (Reeg et al., 2009) and alpine region (Kaeser et al., 2011; Sereke et al., 2015). Respective design parameters are well aligned with this study's climate and general context. Concerning the ratio of tree area to crop area, the design also takes into account a study by Seserman et al. (2019), which found that those systems performed best where either the tree or crop component was dominant (>75% of the land area)[12]. Concerning tree density, a study by (Crous-Duran et al., 2018) found that accumulated energy[13] of a wild cherry and pasture system in Switzerland increased with higher tree densities.
Naturally, the planning of individual agroforestry systems is subject to each farm's specific components, such as a field's characteristics (e.g. size, shape and slope), its aptness for certain species and crop rotations (e.g. due to climate or soil characteristics, or the farmer's decision to transition from crops to pasture when the trees mature), or use of special machinery (e.g. the width of a specific plough or harvester in use). The prototypical design applied in this study, however, had to be based on generalization. It is planned on the basis of a square shape the size of one hectare, depicted in Figure 12.

[12] The study used short rotation coppice systems with poplar and different crops in Germany.
[13] Accumulated energy was converted to MJ $ha^{-1}$ and included cherries, grass, timber and branch wood.

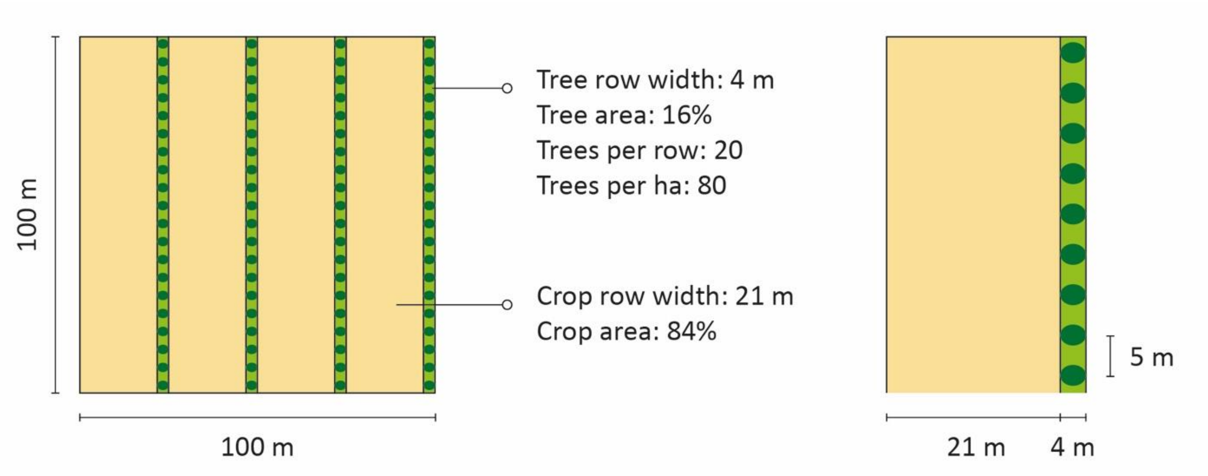


**Figure 12: Prototypical design of the agroforestry system used throughout the study region**

Crop rows feature a width of 21 meters well suited to mechanical cultivation with standard machinery. Tree rows feature a width of 4 meters to allow for mechanical harvesting of fruit. Tillage can thereby be performed up to 1–2 meters close to the stem on either side of the tree, if started immediately after the trees were planted. This prevents the growth of shallow horizontal roots and encourages a deeper rooting system (Kaeser et al., 2010).
Each tree row is planted with 20 trees per 100 meters, resulting in a distance of 5 meters between each tree. This spacing is rather adequate to fruit production and might prove too small when aiming for high DBH in timber production (Spiecker, 2010). The set-up results in a tree density of 80 trees $ha^{-1}$. The tree rows thereby occupy 16% of the area, while crop rows occupy the remaining 84%.

## 3.4 Human Appropriation of Net Primary Production (HANPP)

Trade-offs among productivity, CS and biomass harvest were assessed based on the indicator framework Human Appropriation of Net Primary Production (HANPP) for all scenarios. The HANPP framework is based on the concept of net primary productivity (NPP), a key component of the terrestrial C cycle. NPP is the autotrophic fixation of C (mostly by plants) via photosynthesis minus the C lost through respiration and metabolism (Andersen and Quinn, 2020; Haberl et al., 2014). As such, NPP is a flux that serves as an index for the energy flow through ecosystems and their functioning, as it also represents the energy available to all heterotrophs (Haberl et al., 2014; Zaks et al., 2007).
HANPP quantifies human-induced changes to ecological biomass flows (Haberl et al., 2014). It thus serves as a pressure-indicator for ecosystems by denoting the amount of energy withdrawn from its flow through the trophic levels of the food chain, as compared to the potential NPP (NPP-pot). NPP-pot thereby denotes the amount of NPP assumed to exist without human interference. HANPP, in other words, is a measure of the transformation of land to provide services for humans, which in turn results in less energy being available to the rest of the ecosystem. Using a slightly extended HANPP framework (Figure 13), this study quantifies the above-ground biomass flows (denoted by the prefix "a" to all HANPP parameters).

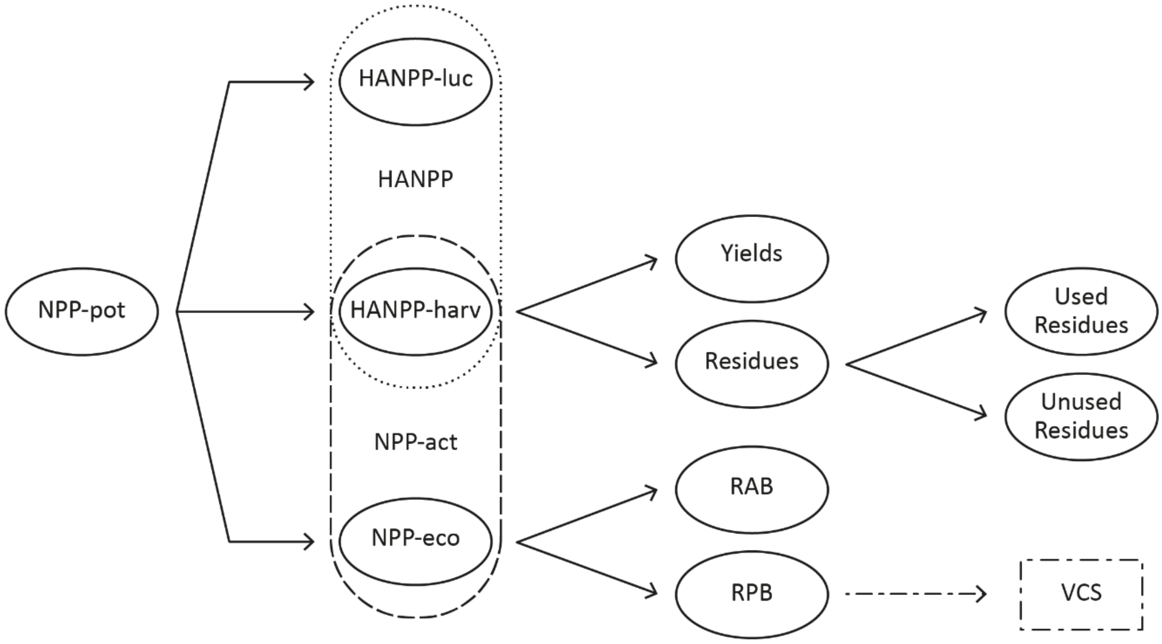



**Figure 13: Definition of the extended HANPP framework used in this study (adapted from Erb et al. 2009), including the various subsets of HANPP indicators. The framework extension refers to one additional subset: NPP-eco is decomposed into remaining annual biomass (RAB) and remaining perennial biomass (RPB). RPB results in the accumulation of C in the vegetation carbon stock (VCS), denoted by the dashed line.**

HANPP is calculated by comparing the amount of NPP that would be available without human intervention (NPP-pot) to the actual NPP (NPP-act) of an area in a given year, accounting for human-induced land use change (HANPP-luc), biomass harvest (HANPP-harv) and the amount of biomass remaining in the ecosystem (NPP-eco). HANPP-harv is composed of *Yields* and *Residues*. Residues, in turn, consist of *Used Residues* and *Unused Residues*, denoting the actual amount of biomass consumed by society (for example grain yields and straw used as litter) and the amount of biomass destroyed during harvest but remaining in the ecosystem (for example unused residues of stalks and roots, which are usually ploughed-in after harvest). In the extended form of the framework, reflecting on the methodology applied in Niedertscheider et al. (2017) and Guzmán et al. (2018), NPP-eco is decomposed into remaining annual biomass (RAB) and remaining perennial biomass (RPB). RPB accumulates every year to build up vegetation C stocks during tree growth. On the basis of these parameters, HANPP can be calculated with the following equation (Erb et al., 2009; Haberl et al., 2014):

$$HANPP = HANPP_{luc} + HANPP_{harv} = NPP_{pot} - NPP_{eco} \quad \text{(Equation 1)}$$

### *3.4.1 Potential net primary productivity (aNPP-pot)*

aNPP-pot relates to the above-ground NPP of vegetation that would prevail without human influence. The hypothetical natural vegetation on agricultural lands in the study region would, depending on altitude, exist of deciduous, coniferous and mixed forests (Haberl, 1995). There exist numerous methods to estimate NPP-pot, ranging from very complex dynamic global vegetation models (DGVM) such as the Lundt-Potsdam-Jena DGVM, which are rather used for global or large-scale estimates (Erb et al., 2007; Gingrich et al., 2015; Haberl et al., 2007; Krausmann et al., 2013), to very simple models such as the MIAMI model (Lieth and Whittaker,

1975). In this study, the MIAMI model was assumed as a feasible and appropriate application due to its simplicity and relative accuracy (Zaks et al., 2007).
The MIAMI model predicts NPP based on the relationship between annual mean temperature (T, in °C) and annual precipitation (P, in mm), by using the van't Hoff rule (which states that productivity doubles every 10°C between -10°C and 20°C) and the Walter ratio (where the NPP for arid regions was observed to increase by 1 g C $m^{-2}$ for each millimeter of precipitation) (Lieth and Whittaker, 1975; Zaks et al., 2007). NPP is assumed to increase with increasing temperature and precipitation, and is thus always limited by either. A saturation value of 3000 g dry matter $m^{-2}$ $yr^{-1}$ cannot be exceeded and the model does not allow for a negative effect of too much rain or too high temperatures (Grieser et al., 2006; Lieth and Whittaker, 1975). The following equations to calculate NPP-pot apply:

$$\mathrm{NPP} = \min(NPP_T,\ NPP_P) \qquad \text{(Equation 2)}$$

$$NPP_T = 3000 \times (1 + \exp(1{,}315 - 0{,}119 \times \mathrm{T})^{-1} \qquad \text{(Equation 3)}$$

$$NPP_P = 3000 \times (1 - \exp(-0{,}000664 \times \mathrm{P}) \qquad \text{(Equation 4)}$$

Since Lieth's formula contains estimates for below-ground NPP of 17% of the total NPP (Haberl, 1995), a corresponding discount is taken into account to determine an estimate of aNPP-pot. The MIAMI model further generates dry matter values, which were converted into C by assuming 50% carbon content (CC) in dry matter biomass (IPCC, 2006). aNPP-pot was calculated as a 5-yr average to reduce the impact of stochastic events such as unusually hot or dry years (Haberl et al., 2007; Krausmann et al., 2008).
Climate data for the modelling sites was acquired from the respective Yield-SAFE model runs. As described in chapters 3.2.3 and 3.2.3.1, in this study climate data from RCP4.5 was applied.

**Table 9: Predicted accumulated annual precipitation in mm (P) and average annual temperature in °C (T) from RCP4.5. Climate data was retrieved from CliPick via the Yield-SAFE interface.**

| | *Kronstorf (cropland)* | | *Losenstein (grassland)* | |
|---|---|---|---|---|
| ***Year*** | ***P (mm)*** | ***T (°C)*** | ***P (mm)*** | ***T (°C)*** |
| *2020* | *1,102.5* | *7.4* | *1,423.0* | *6.1* |
| *2021* | *823.9* | *8.9* | *1,122.8* | *8.0* |
| *2022* | *884.6* | *8.3* | *1,220.3* | *6.8* |
| *2023* | *913.7* | *8.2* | *1,306.2* | *7.0* |
| *2024* | *940.8* | *8.1* | *1,282.6* | *6.7* |
| *2025* | *933.1* | *7.4* | *1,220.5* | *6.0* |
| *2026* | *626.5* | *9.1* | *897.2* | *7.9* |
| *2027* | *1,081.1* | *7.7* | *1,438.2* | *6.6* |
| *2028* | *1,072.9* | *7.5* | *1,401.4* | *6.2* |
| *2029* | *1,028.4* | *7.0* | *1,386.7* | *5.6* |
| *2030* | *965.8* | *8.8* | *1,285.0* | *7.7* |
| *2031* | *863.2* | *6.7* | *1,202.1* | *5.6* |
| *2032* | *651.8* | *8.2* | *957.4* | *7.1* |
| *2033* | *1,027.1* | *8.0* | *1,328.5* | *6.9* |
| *2034* | *944.6* | *8.1* | *1,367.0* | *6.8* |
| *2035* | *666.2* | *7.9* | *926.2* | *7.0* |

| | | | | |
|---|---|---|---|---|
| *2036* | *889.1* | *5.6* | *1,207.2* | *4.5* |
| *2037* | *813.1* | *7.6* | *1,085.8* | *6.5* |
| *2038* | *953.9* | *8.3* | *1,375.1* | *7.1* |
| *2039* | *895.6* | *7.3* | *1,171.1* | *6.3* |
| *2040* | *752.7* | *8.5* | *1,043.4* | *7.3* |
| *2041* | *470.5* | *8.5* | *631.8* | *7.1* |
| *2042* | *579.9* | *7.4* | *811.2* | *5.9* |
| *2043* | *919.2* | *6.7* | *1,221.6* | *5.8* |
| *2044* | *809.4* | *8.9* | *1,096.7* | *7.8* |
| *2045* | *807.8* | *8.5* | *1,085.0* | *7.3* |
| *2046* | *862.8* | *9.4* | *1,122.9* | *8.4* |
| *2047* | *727.1* | *8.1* | *954.9* | *7.1* |
| *2048* | *874.5* | *7.3* | *1,137.6* | *6.4* |
| *2049* | *1,046.3* | *7.7* | *1,489.5* | *6.5* |
| *2050* | *868.6* | *7.6* | *1,251.8* | *6.4* |
| *2051* | *979.8* | *7.9* | *1,373.5* | *6.8* |
| *2052* | *939.6* | *6.7* | *1,237.4* | *5.9* |
| *2053* | *1,008.5* | *6.5* | *1,357.5* | *5.2* |
| *2054* | *862.7* | *6.1* | *1,207.7* | *5.4* |
| *2055* | *744.9* | *9.3* | *971.2* | *8.0* |
| *2056* | *935.3* | *8.4* | *1,277.1* | *7.1* |
| *2057* | *1,057.2* | *7.1* | *1,442.1* | *6.1* |
| *2058* | *795.3* | *7.6* | *1,100.4* | *6.2* |
| *2059* | *946.8* | *7.8* | *1,211.4* | *6.7* |
| *2060* | *957.2* | *7.2* | *1,269.2* | *6.0* |
| *2061* | *679.0* | *7.7* | *944.8* | *6.6* |
| *2062* | *1,077.5* | *7.9* | *1,510.7* | *6.9* |
| *2063* | *812.7* | *7.6* | *1,111.3* | *6.3* |
| *2064* | *947.2* | *8.5* | *1,304.9* | *7.4* |
| *2065* | *911.8* | *7.8* | *1,197.9* | *6.6* |
| *2066* | *814.4* | *8.2* | *1,074.5* | *7.0* |
| *2067* | *802.8* | *7.4* | *1,053.8* | *6.5* |
| *2068* | *835.0* | *8.6* | *1,120.8* | *7.4* |
| *2069* | *811.1* | *9.2* | *1,108.5* | *8.1* |
| *2070* | *887.6* | *8.5* | *1,196.7* | *7.4* |
| *2071* | *842.3* | *7.9* | *1,103.3* | *6.6* |
| *2072* | *932.6* | *9.5* | *1,211.0* | *8.5* |
| *2073* | *810.1* | *9.0* | *1,128.7* | *7.9* |
| *2074* | *827.2* | *9.9* | *1,151.2* | *8.8* |
| *2075* | *716.5* | *9.3* | *915.7* | *8.0* |
| *2076* | *934.0* | *8.3* | *1,259.3* | *6.9* |
| *2077* | *1,090.6* | *9.2* | *1,450.9* | *8.1* |
| *2078* | *847.7* | *9.3* | *1,109.7* | *8.2* |
| *2079* | *649.0* | *9.3* | *873.3* | *8.0* |
| *2080* | *647.4* | *9.5* | *891.0* | *8.4* |

### 3.4.2 Actual net primary productivity (aNPP-act)

aNPP-act on cropland and grazing land is often extrapolated from reported data on crop yields, livestock units and trade balances, using appropriate calculations and coefficients to estimate the share of crop harvest in total biomass as well as the so-called grazing gap, the difference between livestock feed demand and available market and non-market feed (Erb et al., 2009; Haberl et al., 2014, 2007; Krausmann et al., 2008).

**Table 10: Coefficients to calculate aNPP-act and aHANPP-harv components from Yield-SAFE model outputs. CC: carbon content; WC: water content; LI: low-input factor; PHL: pre-harvest loss factor; HI: harvest index; RR: recovery rate.**

| *Cultivars* | *CC* | *WC* | *LI* | *PHL* | *HI* | *RR* |
|---|---|---|---|---|---|---|
| *Cherry fruit* | *0.5* | *0.17* | *-* | *1.09* | *-* | *-* |
| *Winter Wheat* | *0.5* | *-* | *0.6* | *1.14* | *0.5* | *0.7* |
| *Barley* | *0.5* | *-* | *0.6* | *1.14* | *0.45* | *0.7* |
| *Grain Maize* | *0.5* | *-* | *0.65* | *1.14* | *0.45* | *0.7* |
| *Oilseed* | *0.5* | *-* | *0.65* | *1.14* | *0.35* | *0.7* |
| *Grass (ext. CH)* | *0.5* | *-* | *-* | *1.14* | *0.75* | *-* |
| *Grass (80% Dactylus)* | *0.5* | *-* | *-* | *1.14* | *0.75* | *-* |
| *Wild cherry* | *0.5* | *-* | *-* | *-* | *-* | *-* |

In this study, however, aNPP-act was derived from Yield-SAFE output parameters (see chapter 3.2.3.1). aNPP-act of trees was derived from the model output Bt_tonha (Equation 5–(Equation 7), which relates to the accumulated stand biomass reported per month. It was calculated as the difference between the minimum and maximum values per year. Between October and November, values decline, relating to the falling leaves. Yearly leave fall (accounted as RAB) was thus calculated as the difference between the maximum and the second largest value per year. Yearly remaining perennial biomass (RPB) was subsequently calculated as aNPP-act minus leave fall.

$$\mathrm{aNPP}_{act}(\,tree\,) = \max Bt\_tonha - \min Bt\_tonha \qquad \text{(Equation 5)}$$

$$\text{Leave Fall} = \max Bt\_tonha - \frac{\max}{2} Bt\_tonha \qquad \text{(Equation 6)}$$

$$\mathrm{RPB} = \mathrm{aNPP}_{act}(\,tree\,) - Leave\ Fall \qquad \text{(Equation 7)}$$

provides an overview of the coefficients used to calculate aNPP-act (and subsequently aHANPP-harv, see chapter 3.4.3). Yield-SAFE model simulations report above-ground biomass production in dry matter, except for cherries, which are reported in fresh weight. Dry matter conversion to C was based on the same factor (CC) as used to convert aNPP-pot (see chapter 3.4.1). The water content (WC) for cherries was derived from Walker et al. (2011) and Wojdyło et al. (2014) and was assumed to be 0.83. Low-input factors (LI) simulate efficiency losses due to the reduction of artificial fertilizers and pesticides. Factors were based on weighted averages of organic and conventional yields in *occasionally dry areas* in Austria between 2003–2015 and were derived from Resl and Brückler (2016). No factor for oilseed was available and was thus derived from the average of oil sunflower and oil squash. aNPP-act of trees was derived from the model output Bt_tonha (Equation 5–(Equation 7), which relates to the accumulated stand biomass reported per month. It was calculated as the difference between the minimum and maximum values per year. Between October and November, values decline, relating to the falling leaves. Yearly leave fall (accounted as RAB) was thus calculated as the difference between the maximum and the second largest value per year. Yearly remaining perennial biomass (RPB) was subsequently calculated as aNPP-act minus leave fall.

$$\mathrm{aNPP}_{act}(\,tree\,) = \max Bt\_tonha - \min Bt\_tonha \qquad \text{(Equation 5)}$$

$$\text{Leave Fall} = \max Bt\_tonha\; - \frac{\max}{2} Bt\_tonha \qquad \text{(Equation 6)}$$

$$\mathrm{RPB} = \mathrm{aNPP}_{act}(\,tree\,) - Leave\ Fall \qquad \text{(Equation 7)}$$

### 3.4.3 Biomass harvest (aHANPP-harv)

aHANPP-harv relates to above-ground biomass harvest of crops and their residues, as well as harvest of grass through mowing or grazing. aHANPP-harv and its components were derived from aNPP-act by using a series of coefficients presented in  aNPP-act of trees was derived from the model output Bt_tonha (Equation 5–(Equation 7), which relates to the accumulated stand biomass reported per month. It was calculated as the difference between the minimum and maximum values per year. Between October and November, values decline, relating to the falling leaves. Yearly leave fall (accounted as RAB) was thus calculated as the difference between the maximum and the second largest value per year. Yearly remaining perennial biomass (RPB) was subsequently calculated as aNPP-act minus leave fall.


$$\mathrm{aNPP}_{act}(\,tree\,) = \max Bt\_tonha - \min Bt\_tonha \qquad \text{(Equation 5)}$$

$$\text{Leave Fall} = \max Bt\_tonha\; - \frac{\max}{2} Bt\_tonha \qquad \text{(Equation 6)}$$

$$\mathrm{RPB} = \mathrm{aNPP}_{act}(\,tree\,) - Leave\ Fall \qquad \text{(Equation 7)}$$


, listed above.
The pre-harvest loss factor (PHL) for crops relates to losses during the growth period due to herbivory and the NPP of weeds (Haberl et al., 2007, SI; Krausmann, 2001). Factors vary for a country's development stage; the appropriate factor for industrialized countries was chosen. The pre-harvest loss factor for cherries relates to bird damage, bruising, cracking, decaying and physiological disorders and was derived from Öztürk et al. (2010).
Harvest indices (HI) to calculate yields and residues, as well as recovery rates (RR) to calculate the fraction of used and unused residues were taken from Wirsenius (2003, 2000). Harvest on grassland is calculated with a factor of 0.75 as the maximal fraction of aNPP-act assumed to be harvestable through grazing or mowing (Haberl et al., 2007).
The following equations were applied to calculate the various components of aHANPP-harv:

$$\mathrm{aHANPP}_{harv}\ (\,grassland\,) = \ \mathrm{aNPP}_{act}\ /\ PHL \times\ HI \qquad \text{(Equation 8)}$$

$$\mathrm{aHANPP}_{harv}\ (\,cropland\,) = \ \text{Yield} + \text{Residues} \qquad \text{(Equation 9)}$$

$$\text{Yield} = \mathrm{aNPP}_{act}\ /\ PHL \times\ HI \qquad \text{(Equation 10)}$$

$$\text{Residues} = \text{Yield}\ \times\ (1 - \mathrm{HI}) \qquad \text{(Equation 11)}$$

$$\text{Used Residues} = \text{Residues} \times \text{RR} \quad \text{(Equation 12)}$$

$$\text{Unused Residues} = \text{Used Residues} \times (1 - \text{RR}) \quad \text{(Equation 13)}$$

### 3.4.4 Biomass remaining in the ecosystem (aNPP-eco)

aNPP-eco in this study consists of the remaining annual biomass (RAB) and the remaining perennial biomass (RPB).

RAB represents the amount of NPP that naturally dies off every year and remains in the ecosystem for herbivory consumption and decomposition. RAB consists of pre-harvest losses, the fraction of aNPP-act on grassland remaining in the ecosystem, the complete aNPP-act on fallow land and, additionally, in the case of the AFS leave fall.

Total aNPP-act on fallow land is accounted for as aNPP-eco in this study. This is a *maximum assumption,* because usually no harvest takes place on fallow land. According to the regulation of the European Union (Commission Regulation (EC) No 1200/2009, 2009), natural growth may be used as feed or ploughed in when the soil is prepared for the next cultivation cycle. However, no data on grazing or ploughing of these areas are available.

RPB represents the amount of NPP from perennial plants that remains alive in the ecosystem and as such is stored over time (for example roots or woody biomass). RPB in this study thus represents the amount of C sequestered in perennial above-ground biomass and is therefore an addition to the vegetation CS. An increase of CS over time functions as a net C sink, contributing to climate change mitigation.

As the study period extends for 31 years but trees accumulate biomass throughout their lifespans, results would only be able to grasp a certain share of sequestered C. As, in this study, the harvest cycle of wild cherry was assumed to be around 60 years, the study period thus covers only half of this cycle. Additionally, wild cherry is characterized by strong growth during its early stages and growth rates decrease towards maturity. Therefore, yearly RPB cannot be extrapolated linearly over a whole harvest cycle. To nevertheless allow for an assessment of the long-term CS potential, i.e. the maximum CS of an agroforestry stand with an equally distributed tree age until harvest, AFS model runs were performed for 61 years, from 2020–2080, and the gradual implementation scenario (AFS-GRAD) was introduced.

# 4 Results

The following chapter presents the results of the quantification of C dynamics between 2020–2050, and, in the case of the accumulation of woody biomass and its implications in the AFS-scenarios, until the year 2080. The effects on ecosystem dynamics of the implementation of agroforestry systems will be disentangled from effects of land management changes emerging from the SECLAND-land use dataset (i.e. changes in total agricultural area, production intensity and the share of individual cultivars). After describing trends in the potential NPP, each scenario will be discussed, followed by a comparison of relevant indicators.

## 4.1 Potential vegetation in the study region

aNPP-pot is a relevant parameter of the indicator framework HANPP, because it is used to establish aHANPP-luc and thus directly affects aHANPP. Therefore, before discussing the land use scenarios, aNPP-pot results are briefly presented.

Figure 14 [a, b] shows the different developments of aNPP-pot from 2020–2080 for the cropland and grassland modelling sites, as computed through the MIAMI model. As detailed in chapter 3.4.1, the MIAMI model estimates productivity as a function of the limitation of temperature and precipitation, i.e. for every year, the smaller estimate is selected.

*a)* *b)*

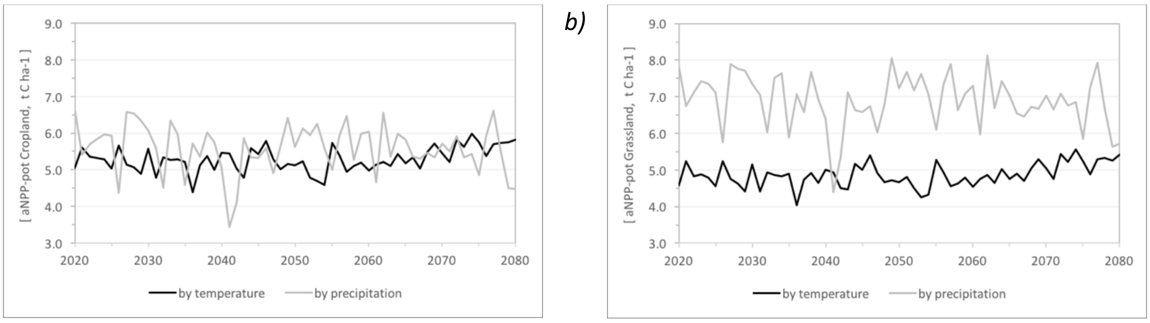



**Figure 14: aNPP-pot on the cropland [a] and grassland [b] modelling sites, calculated on the basis of mean annual temperature in °C and annual precipitation in mm, 2020–2080, in t C ha$^{-1}$. The MIAMI model estimates NPP as a function of the limitation of temperature and precipitation.**

aNPP-pot in the study region is clearly more often limited by temperature than by precipitation, with a ratio of 40:21 at the cropland site and 60:1 at the grassland site. While temperature and precipitation follow the same trends at the two sites, the cropland site is endowed with higher temperatures due to its lower altitude, whereas the grassland site receives significantly more rain due to its geographical particularities. The graphs also show a trend of increasing temperatures after 2060. Yearly climate data (RCP4.5) is listed in chapter 3.4.1.

Figure 15 shows the final aNPP-pot values used as input to the HANPP calculations. aNPP-pot generally ranges between 4.5 and 5.5 t C ha$^{-1}$ at the two modelling sites, with slight fluctuations due to the climate input data. No linear trend can be detected, although rising temperatures after 2060 lead to an increase in productivity. The temperature difference between the cropland and grassland modelling sites results in cropland aNPP-pot being slightly higher than grassland aNPP-pot almost throughout the extended study period. Only the very dry years from 2040–2042 cause a distinctive downward spike in cropland aNPP-pot (even in the 5-year moving average as applied here), resulting in cropland productivity dropping below that of grassland for these three consecutive years. After 2070, cropland is

again affected by little rain for multiple years, causing productivity to decline, while the limitations to grassland productivity decrease due to relatively high temperatures.
The MIAMI model operates on a relatively coarse level based on aggregated yearly climate data, while Yield-SAFE – simulating aNPP-act – operates at a much higher temporal resolution (i.e. on a daily time step) and includes additional variables (such as solar radiation and relevant soil physical characteristics). An assessment of the relative performance of the two models is included in the discussion.

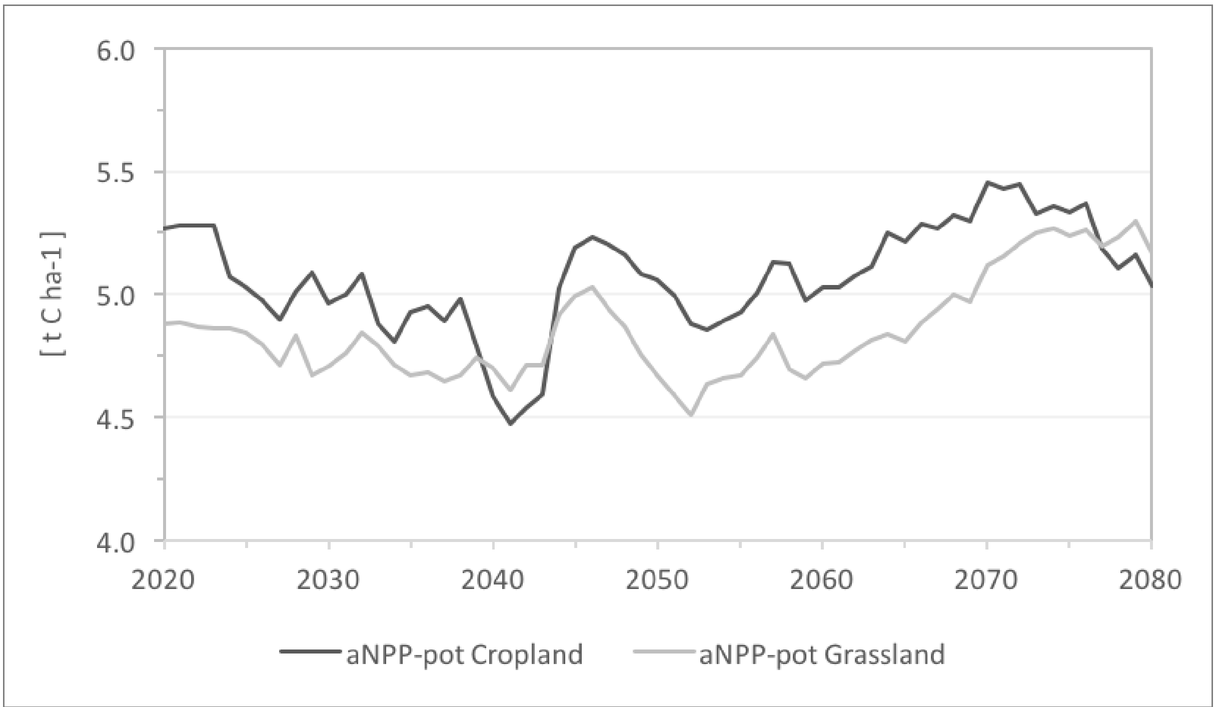



**Figure 15: Final aNPP-pot on the cropland and grassland modelling sites, 2020–2080, 5-year moving average in t C ha$^{-1}$. Values differ because of varying climate input data for the two modelling sites.**

## 4.2 Land use and land management change (LULMC)

This chapter presents a summary of LULMC in the study region and its effects on productivity. LULMC originates from the datasets provided by SECLAND and as such form the basis for all landscape-scale calculations. They are thus of equal consequence to all land use scenarios in this study. LULMC can be separated into changes in area and changes in intensity. Changes in area concern individual cultivars, cropland and grassland categories as well as total agricultural area. Changes in intensity, on the other hand, relate to management practices of conventional/intensive and low-input/extensive production on cropland and grassland, respectively. Changes in area will be discussed first, followed by changes in intensity. LULMC equally affects all scenarios, as shifts between cropland and grassland or intensity shifts in management apply to AGR and AFS.
On the highest level, land use change in the study region between 2020–2050 will pertain to the conversion of agricultural land to forest, resulting in a reduction of total agricultural area by -24% (
Figure 16). While this is a major incidence by its own right with varying effects on the socio-ecological system in the region, it is not subject of the present study. As the performed calculations involve crop- and grasslands only, affected areas are excluded from further

consideration the moment the conversion takes place (see chapter 3.2.2). The reduction of total agricultural area consequently leads to an equivalent reduction of all related parameters (such as the region's accumulated aNPP-act or aNPP-pot) and needs to be considered as such. This effect, however, is factored out by looking at the per hectare values of HANPP indicators. Additional LULMC, described in the following sections, will nevertheless be effective in per hectare values, too.

Changes in the area extent of the cropland and grassland categories are inherent to the reduction of total agricultural area, having varying effects not only on the region's total but also on per hectare productivity. Reductions in area primarily affect (intensive and extensive) grassland, while the area extent of cropland stays almost constant (
Figure 16). This leads to an overall higher share of cropland, which on average is more productive than grassland, resulting in an increase of per hectare productivity.

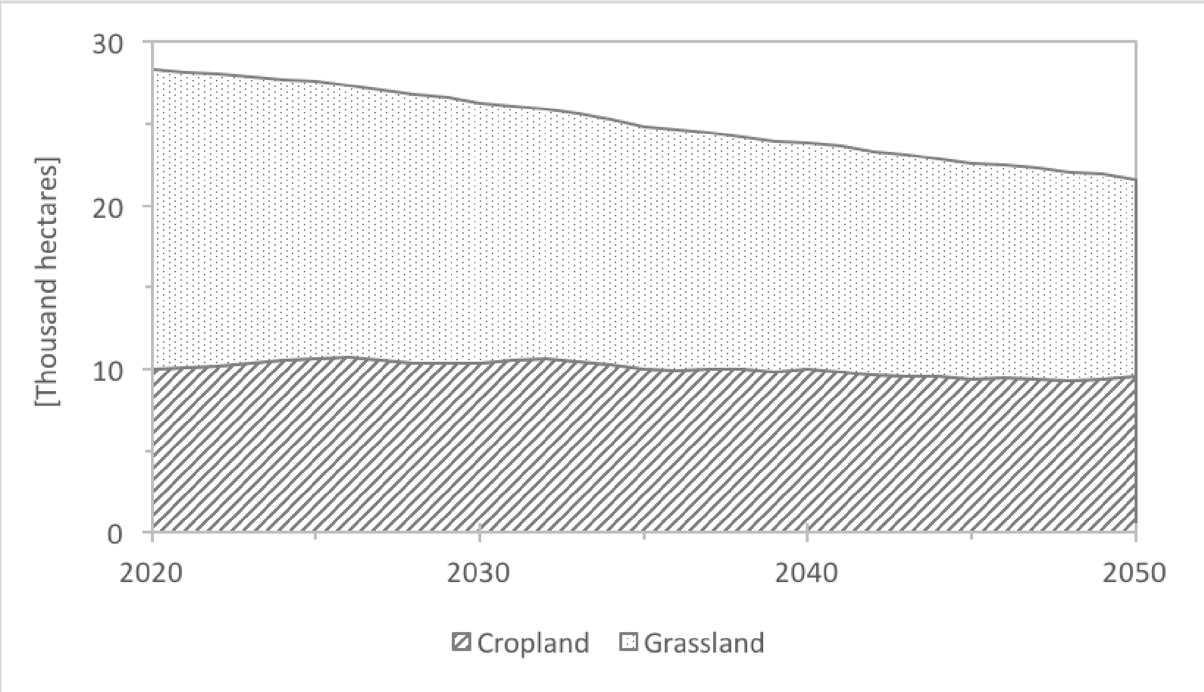



**Figure 16: Development of the area extent of cropland and grassland in thousand hectares.**

Furthermore, a shift in the area extent of individual cultivars within the cropland category takes place (see Figure 4, chapter 3.2.2.1). Most notable is the increase in the share of oilseed (from 8% to 32% of cropland area) and the decrease in the share of maize and cereals (from 40% to 12%, and from 25% to 16% of cropland area, respectively). These changes in the share of individual cultivars have mixed effects on productivity and cannot easily be discerned in the interpretation of results.

Changes in production intensity (Figure 17) affect cropland and grassland, denoting a general shift from intensive/conventional to extensive/low-input production. Grassland is affected more strongly, where intensive pastures and meadows are phased out almost completely (from 20% to 4% of grassland area). In the cropland category, the shift to low-input production affects maize as well as cereals. Oilseed, on the other hand, is primarily managed conventionally. This general shift towards extensive/low-input production leads to a decrease in per hectare productivity in the region.

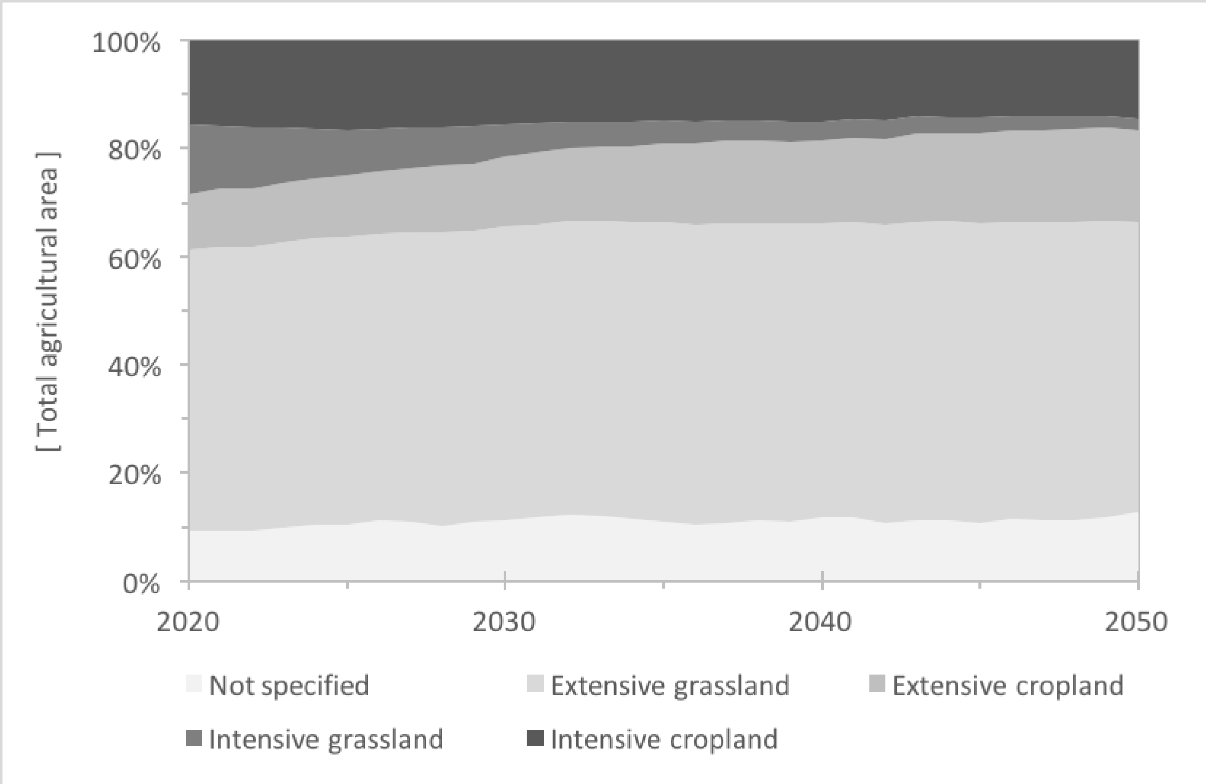



**Figure 17: Development of production intensity in percent of the total agricultural area, 2020–2050.**

## 4.3 Agricultural scenario (AGR)

The AGR scenario depicts *business-as-usual* in the study region. Although the land use dataset predicts land use change within a sustainable socio-economic narrative (as described in more detail in chapter 3.2.2), agricultural practice in AGR is defined by the segregation of crops and trees, as is currently standard practice in Austria. AGR thereby serves as the baseline against which to compare the AFS.

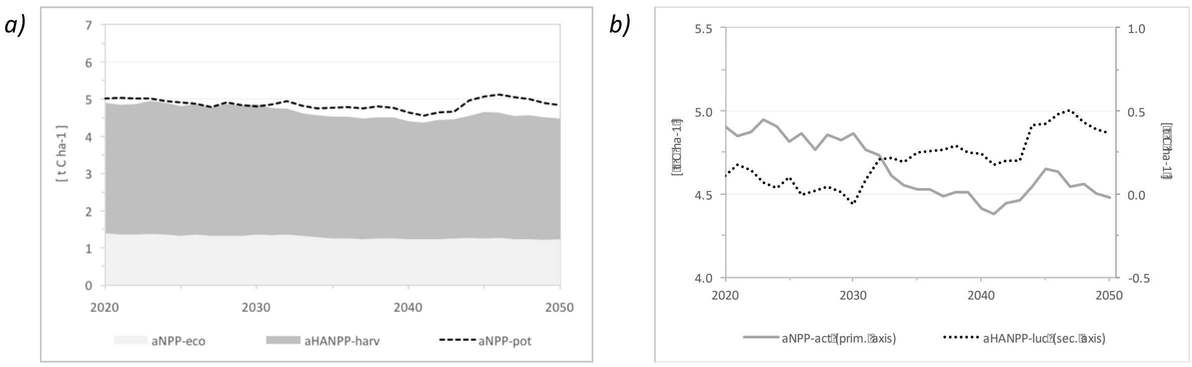



**Figure 18: Development of [a] area-weighted aNPP-eco, aHANPP-harv and aNPP-pot and [b] area-weighted aNPP-act (primary axis) and area-weighted aHANPP-luc (secondary axis) in the AGR scenario, 2020–2050, in t C ha$^{-1}$ yr$^{-1}$. In [a] the sum of aNPP-eco and aHANPP-harv equals aNPP-act, whereas the difference between aNPP-act and aNPP-pot equals aHANPP-luc.**

Figure 18-a shows the development of aNPP-pot as well as aNPP-eco and aHANPP-harv, which in sum equal aNPP-act, expressed in area-weighted average productivity per hectare. As such, values express the combined effects of LULMC (except for the reduction in total area). While aNPP-pot remains relatively constant around 5 t C $ha^{-1}$ $yr^{-1}$, aNPP-act declines from 4.8 t C $ha^{-1}$ $yr^{-1}$ in 2020 to 4.5 t C $ha^{-1}$ $yr^{-1}$ in 2050. aHANPP-luc accordingly follows an inverted trajectory (Figure 18-b) increasing by approximately the same amount. Decreasing production intensity, i.e. the high share of extensive grassland and low-input arable crops, thus outweighs the increase in area of more productive cultivars, leading to an overall reduction in productivity per hectare by about 8% between 2020–2050.

Figure 19 shows aHANPP components expressed in percent of aNPP-pot. The rate of aHANPP is thereby rising slightly from 72% to 74%. Developments of aHANPP-harv and aHANPP-luc, as discussed before, are somewhat more pronounced, decreasing from 70% to 67% and rising from 2% to 8%, respectively. High rates of aHANPP are rather typical for agricultural land and can be explained as follows. In the case of crop production, usually all parts of the plant are killed during harvest. As such, the whole plant biomass is accounted for as appropriated, even those parts that are not recovered and left on the field (e.g. stubble). In this study, 88% of aNPP-act on harvested cropland and 66% of aNPP-act on grassland are accounted for as aHANPP-harv, the rest being attributed to pre-harvest losses and the maximal harvestable fraction on grassland (see chapter 3.4.3) and as such accounted to aNPP-eco. While especially in industrialized production systems only very little biomass is left behind (Gingrich et al., 2015; Haberl et al., 2007), those backflows to nature that do exist can be calculated within the HANPP framework as unused residues (i.e. the part of residues which is not recovered). In the case of above-ground cropland, these residues are usually ploughed-in stubble and straw, and are only available to detritivores.

A decomposition of aHANPP-harv and aNPP-eco can be seen in

Figure 20. Development of yields, with a rising share of crop yields over grass yields, generally reflects LULMC. Accumulated yields reach 69.5 t C $ha^{-1}$ with an additional 34.8 t C $ha^{-1}$ of residues. Backflows to nature do exist, but form a considerably small fraction of aHANPP-harv. Remaining annual biomass (RAB) corresponds to total aNPP-eco in the AGR scenario. Even though the share of fallow land, which is the largest fraction of aNPP-eco here, remains constant, a slight decrease in RAB over time (from 1.4 to 1.2 t C $ha^{-1}$ $yr^{-1}$) is also caused by the shrinking share of grassland.

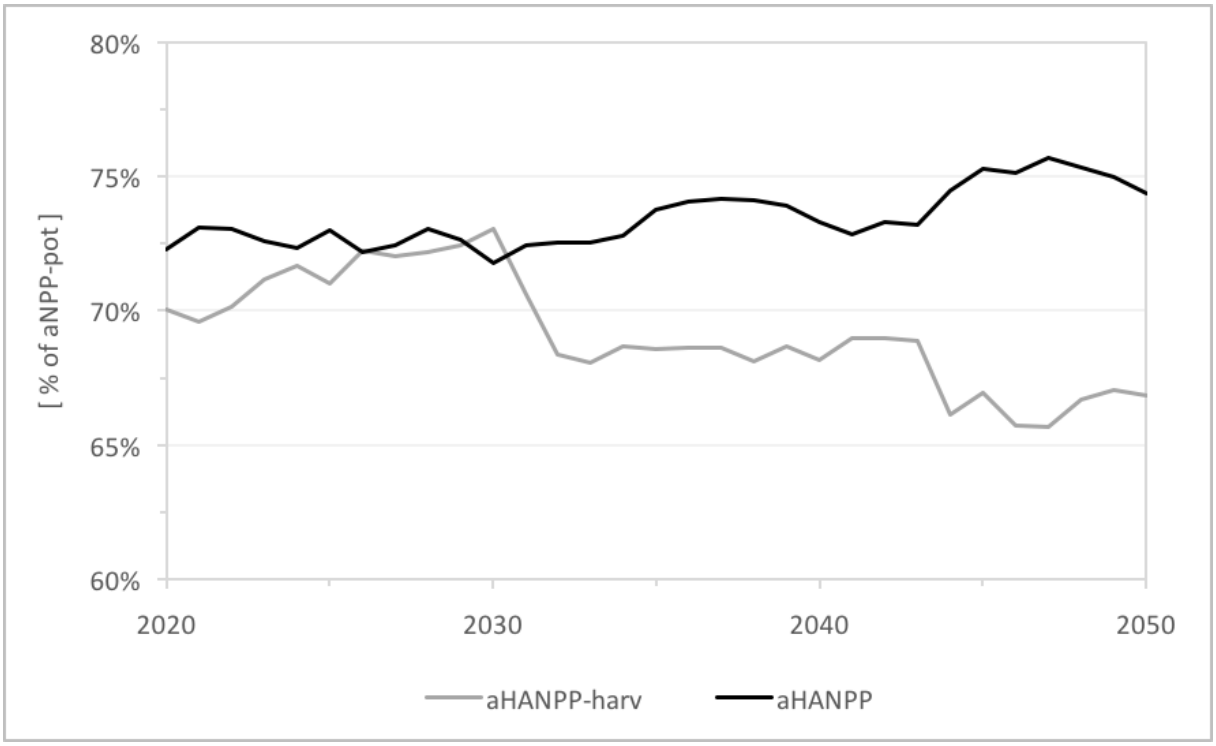



**Figure 19: aHANPP and aHANPP-harv in the AGR scenario, 2020–2050, in percent of aNPP-pot.**

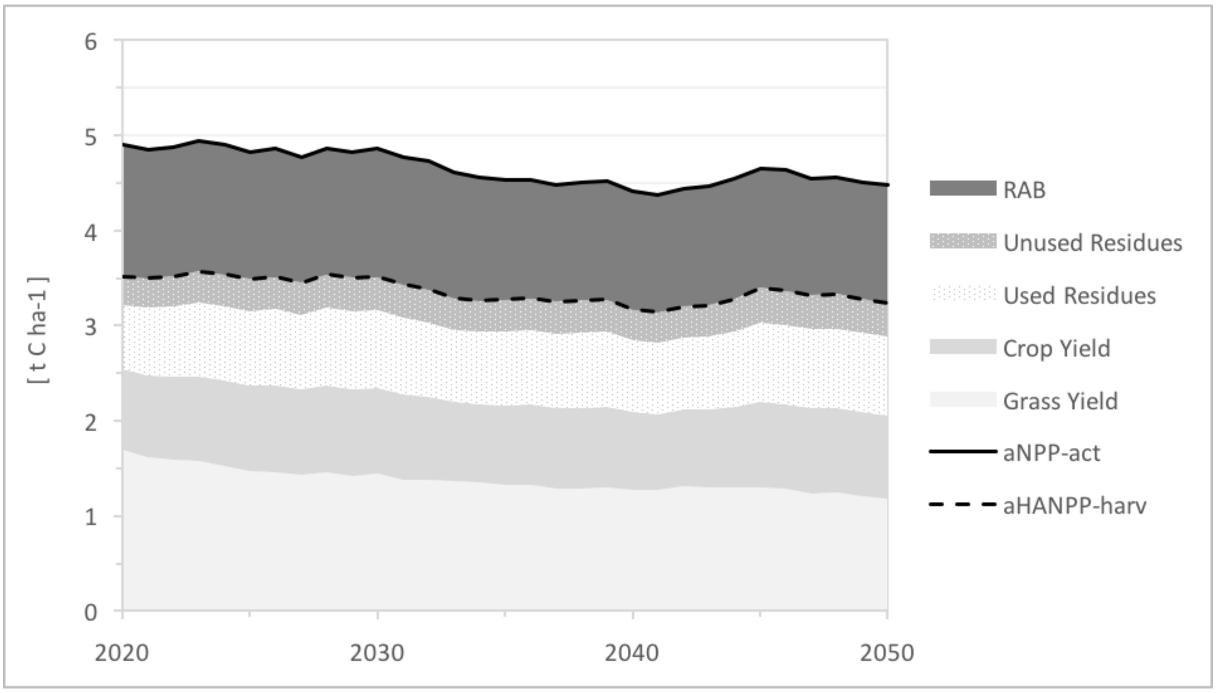



**Figure 20: Development of area-weighted yields, residues and remaining annual biomass (RAB) in the AGR scenario between 2020–2050, in t C ha$^{-1}$ yr$^{-1}$. The sum of yields and residues equals aHANPP-harv; RAB corresponds to aNPP-eco; and the sum of aHANPP-harv and aNPP-eco equals aNPP-act.**

## 4.4 Maximum agroforestry scenario (AFS-MAX)

The AFS-MAX scenario depicts the integration of trees in agricultural lands. It is based on the same land use dataset as the AGR scenario, thus equally being subject to LULMC. AFS-MAX is the maximum scenario as the transition to agroforestry takes place abruptly in 2020 on 100% of agricultural land. As such it shows the maximal effects of agroforestry in the study region.
The development of aNPP-pot, aNPP-eco and aHANPP-harv is shown in Figure 21. aNPP-pot naturally follows the same trajectory as it does in the AGR scenario. aNPP-act (depicted as the sum of the stacked areas), on the other hand, starts off at a lower level than in the AGR scenario. This is directly caused by the fact that 16% of the area is exclusively populated with trees and consequently only 84% of the area remains for crops. The trends of increasing aNPP-eco and decreasing aHANPP-harv start almost instantaneously after implementation of the AFS-MAX scenario. Gains in aNPP-eco quickly counteract losses in aHANPP-harv, resulting in aNPP-act surpassing aNPP-pot by the year 2030. aNPP-act then peaks around 2040, after which it starts to decline again. The development of aNPP-act is composed of a rather drastic decline of aHANPP-harv in the first half of the study period (from 3 t C ha$^{-1}$ yr$^{-1}$ in 2020 to 1.8 t C ha$^{-1}$ yr$^{-1}$ in 2037) offset by a sharp increase of aNPP-eco (from 1.2 t C ha$^{-1}$ yr$^{-1}$ in 2020 to 4.1 t C ha$^{-1}$ yr$^{-1}$ in 2042). After reaching their respective high and low peaks, aNPP-eco shows an attenuated decline and aHANPP-harv stagnate at this low level thereafter.

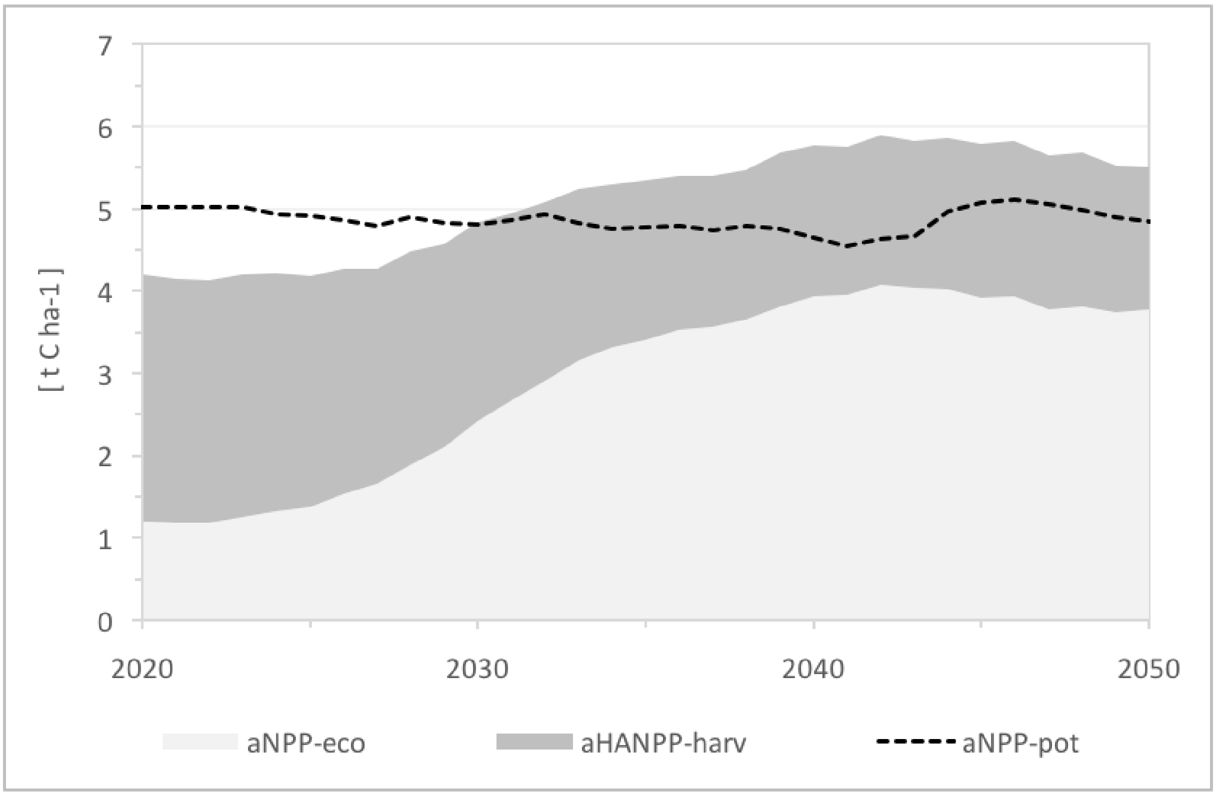



**Figure 21: Development of area-weighted aNPP-eco, aHANPP-harv and aNPP-pot in the AFS-MAX scenario, 2020–2050, in t C ha$^{-1}$ yr$^{-1}$. The sum of aNPP-eco and aHANPP-harv equals aNPP-act, whereas the difference between aNPP-act and aNPP-pot equals aHANPP-luc.**

Development of aHANPP components in percent of aNPP-pot is shown in Figure 22. Considering combined developments of tree and crop growth, the AFS-MAX scenario's overall productivity clearly surpasses that of the AGR scenario, also depicted in the relation of aNPP-act and aNPP-pot. Accordingly, aHANPP-luc declines from almost 20% of aNPP-pot at the start of the study period to a low of -27% by 2042, and then stabilizes at around -14% by 2050. aHANPP follows this trajectory, although it proportionally declines even stronger (from 76% to 22%) due to the above-mentioned growth in aNPP-eco.

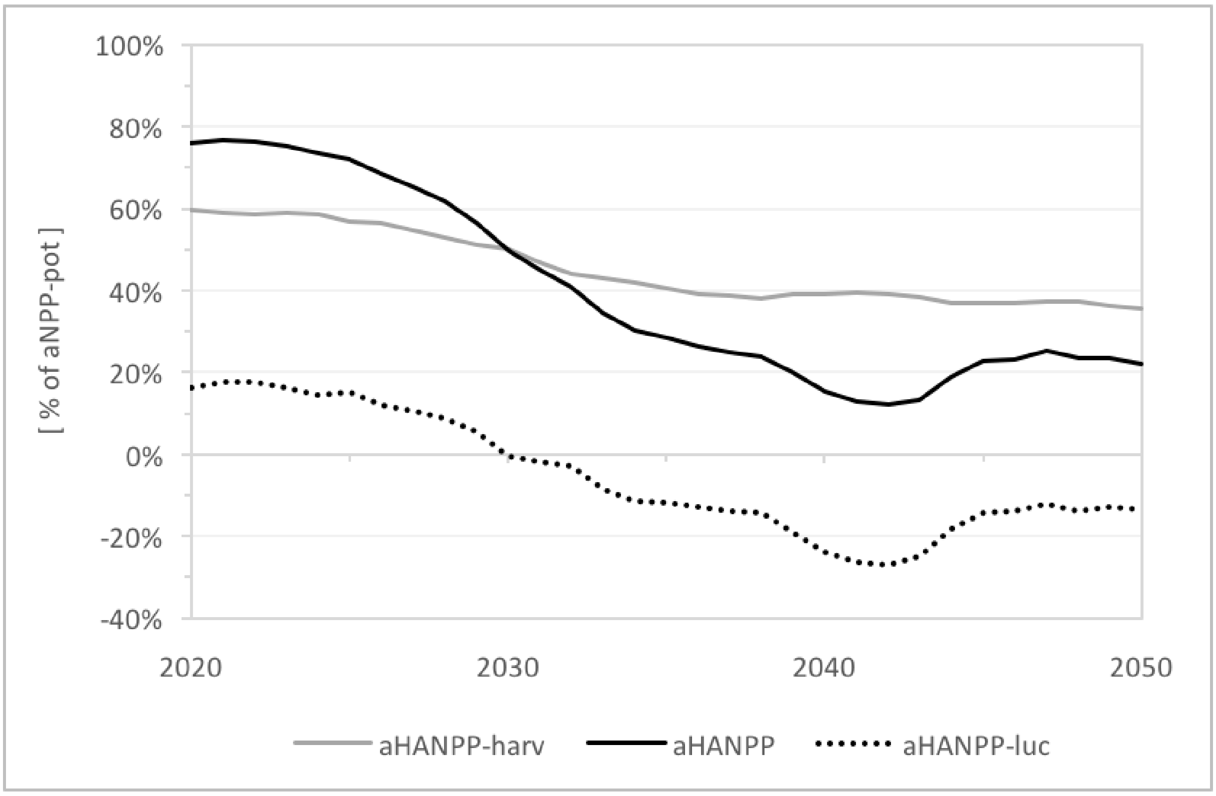



**Figure 22: Development of aHANPP, aHANPP-harv and aHANPP-luc in the AFS-MAX scenario between 2020–2050, in percent of aNPP-pot.**

The development of aNPP-act is clearly driven by the co-existence of trees and crops. Cherry trees feature a strong growth in the first 20 years, shaping the curve of aNPP-eco. Yearly tree growth peaks around 2040 with an increment of nearly 6 t C $ha^{-1}$ $yr^{-1}$. Thereafter, the growth rate starts to decline but significant annual growth is still sustained at least until 2080. Simultaneously, the productivity of crops declines sharply due to the increasing competition for water – and more importantly – solar radiation[14]. This reduction in crop productivity is slightly attenuated when looking at aHANPP-harv, because cherry fruit production makes a considerable impact starting around 2030 (Figure 23).

Figure 23 shows the composition of aHANPP-harv and aNPP-eco. Combined crop and grass yields reach an accumulated total of 38 t C $ha^{-1}$ with an additional 18.9 t C $ha^{-1}$ of residues. Accumulated crop and grass yields here thus reach a value just above accumulated residues in the AGR scenario, which shows another angle of the drastic reductions in harvest. Simultaneously, cherry production provides an additional accumulated fruit yield of 11.4 t C $ha^{-1}$. At the end of the study period (2049), fruit yields increase up to the point at which the yearly amount of harvested fruit (0.7 t C $ha^{-1}$ $yr^{-1}$) is larger than the yearly amount of harvested crops and grass (0.6 t C $ha^{-1}$ $yr^{-1}$). Backflows to nature diminish as crop yields decrease.

[14] Nutrient limitations are not considered in Yield-SAFE, as discussed in chapter 3.2.3.

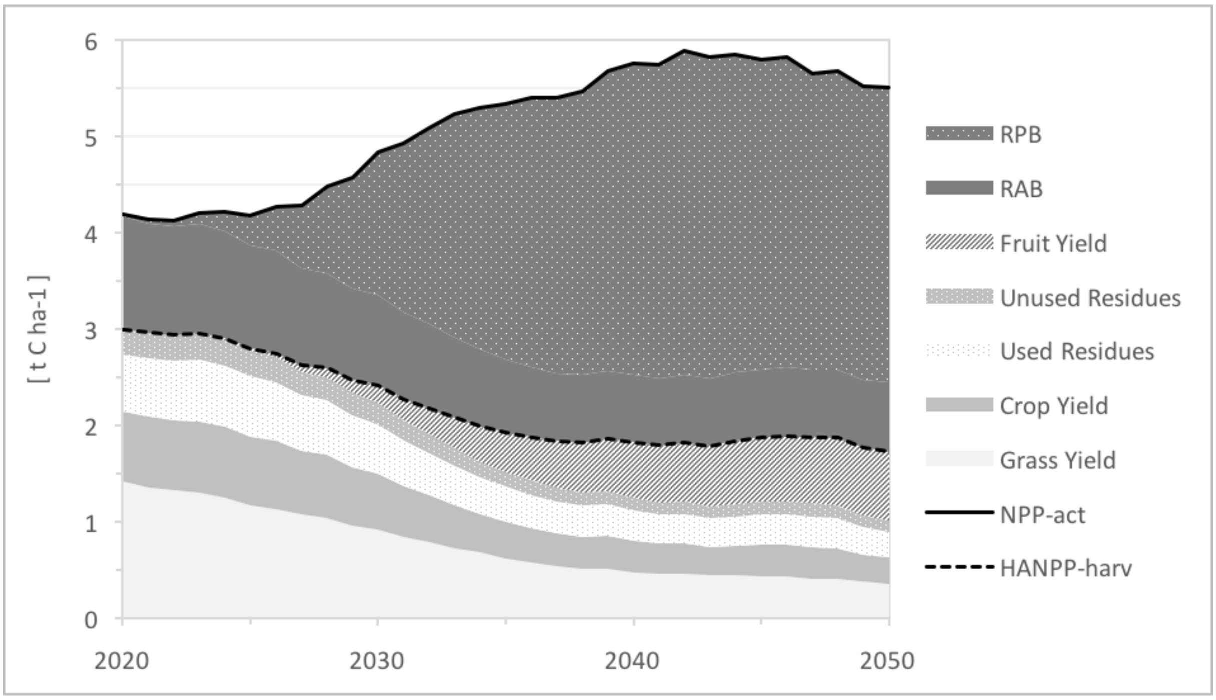



**Figure 23: Development of area-weighted yields and residues as well as remaining perennial biomass (RPB) and remaining annual biomass (RAB) in the AFS-MAX scenario, 2020–2050, in kt C ha$^{-1}$ yr$^{-1}$. The sum of yields and residues equals aHANPP-harv; the sum of RPB and RAB equals aNPP-eco; and the sum of aHANPP-harv and aNPP-eco equals aNPP-act.**

In AFS-MAX, aNPP-eco is composed of remaining annual biomass (RAB) and remaining perennial biomass (RPB). The latter, which corresponds to the amount of C sequestered in long-lived woody biomass, shows a strong increase in relation to yields and residues. RPB thereby reaches a total of 64 t C ha$^{-1}$ over the whole study period, with a maximum yearly increment of 3.4 t C ha$^{-1}$ yr$^{-1}$ in 2042. This indicator represents the climate change mitigation potential and is thus an important measure for this study.

## 4.5 Gradual agroforestry scenario (AFS-GRAD)

The AFS-GRAD scenario depicts a gradual implementation of the agroforestry system in a 5-year time step. Contrary to the AFS-MAX scenario, it thus incorporates a time-lag in the growth dynamics of the trees on the landscape-scale. This results in a more realistic – if nevertheless purely hypothetical – estimate. It further offers the possibility to better assess the CS potential over time, as trees exhibit a more widely distributed age structure by the end of the study period.

The gradual implementation causes a delayed and less pronounced development of aHANPP components (Figure 24 and Figure 25) than in AFS-MAX. Thereby, aHANPP-harv decreases (from 3.4 to 2.2 t C ha$^{-1}$ yr$^{-1}$) and aNPP-eco increases (from 1.4 to 3 t C ha$^{-1}$ yr$^{-1}$) at much slower rates, such as that aNPP-act surpasses aNPP-pot in 2037 (which is ten years later than in the AFS-MAX scenario). aHANPP-luc subsequently falls to -8% of aNPP-pot by 2050, and aHANPP decreases from 73% of aNPP-pot in 2020 to 36% in 2050.

The same general effect can be observed in the decomposition of aHANPP-harv and aNPP-eco (Figure 26). Accumulated RPB reaches 28.1 t C ha$^{-1}$ (with a yearly maximum of 2.3 t C ha$^{-1}$ yr$^{-1}$ in 2050 and rising), whereas combined crop and grass yields reach 55.2 t C ha$^{-1}$ with an additional 27 t C ha$^{-1}$ of residues and 5.6 t C ha$^{-1}$ of fruit yield.

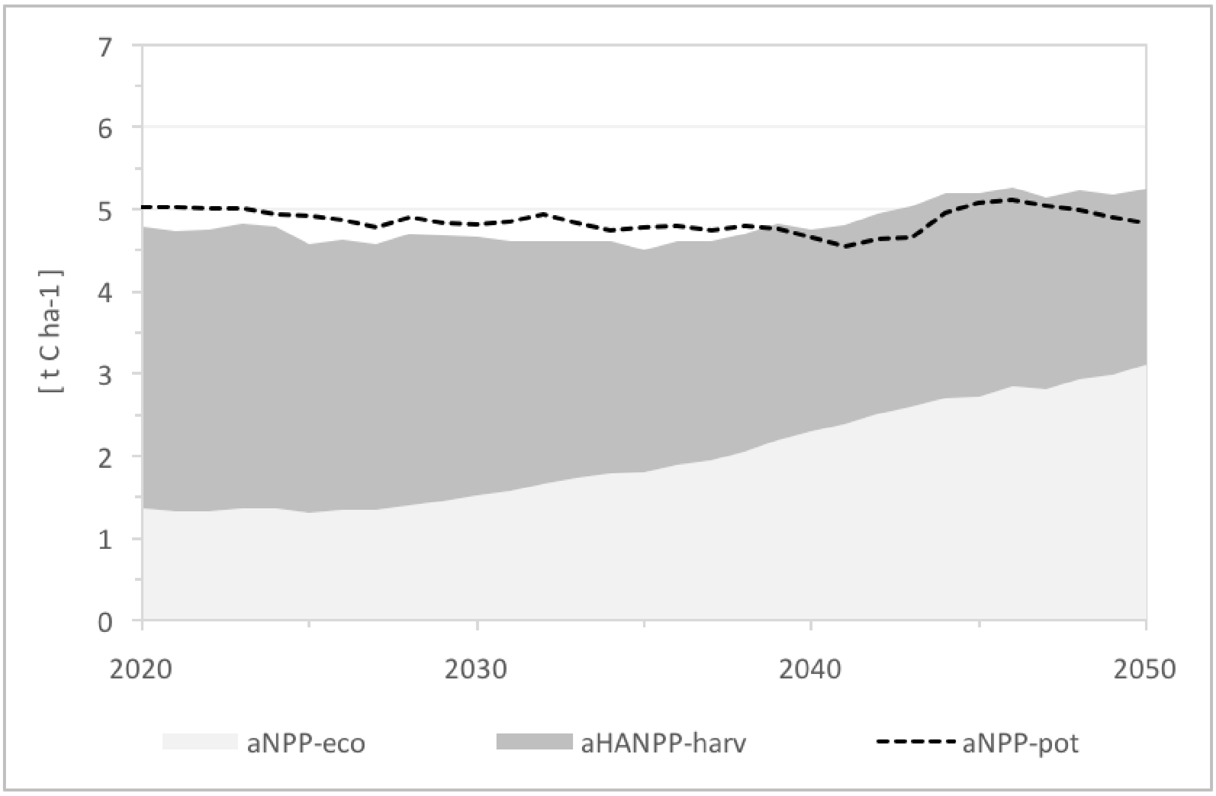



**Figure 24: Development of area-weighted aNPP-eco, aHANPP-harv and aNPP-pot in the AFS-GRAD scenario, 2020–2050, in t C ha$^{-1}$ yr$^{-1}$. The sum of aNPP-eco and aHANPP-harv equals aNPP-act, whereas the difference between aNPP-act and aNPP-pot equals aHANPP-luc.**

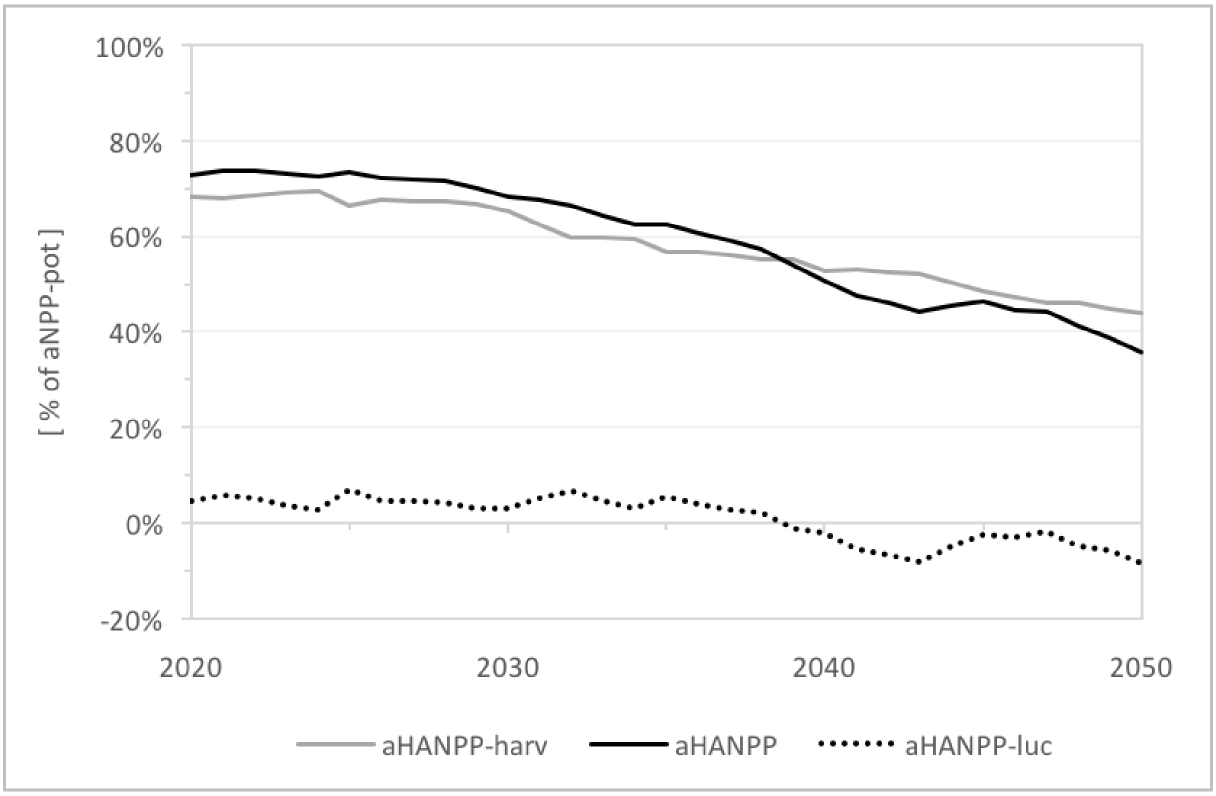



**Figure 25: Development of aHANPP, aHANPP-harv and aHANPP-luc in the AFS-MAX scenario between 2020–2050, in percent of aNPP-pot.**

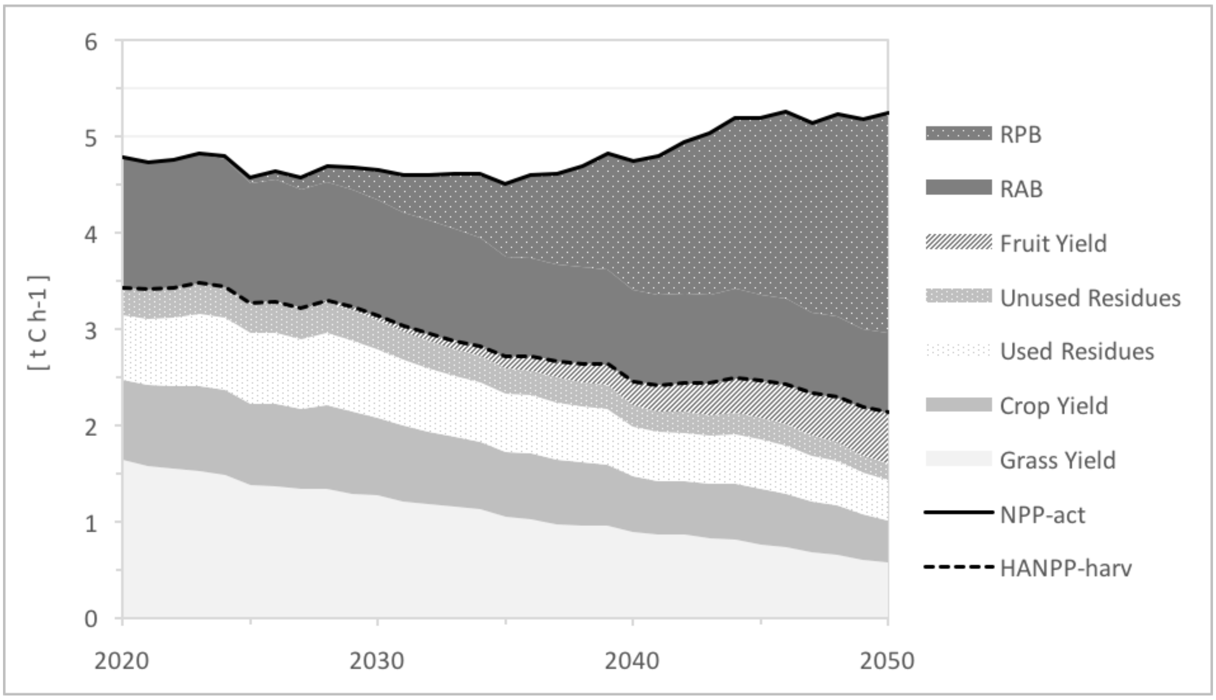



**Figure 26: Development of area-weighted yields and residues as well as remaining perennial biomass (RPB) and remaining annual biomass (RAB) in the AFS-GRAD scenario between 2020–2050, in kt C ha$^{-1}$ yr$^{-1}$. The sum of yields and residues equals aHANPP-harv; the sum of RPB and RAB equals aNPP-eco; and the sum of aHANPP-harv and aNPP-eco equals aNPP-act.**

## 4.6 Scenario comparison

The following section gives an overview of the three scenarios' results in relation to each other by comparison of total values as well as temporal trends. The comparison also includes an extended view of the two AFSs until the year 2080 (with the land use distribution of 2050 held constant). Table 11 provides the cumulative and average C flows and stocks of each HANPP indicator calculated over the whole study period. All numbers in the following section refer to this table and are thus given in accumulated kt C or area-weighted average t C ha$^{-1}$ between 2020–2050, if not stated differently. Figures 27–Figure 30 additionally visualize the developments of relevant indicators. Comparison of HANPP indicators between the different scenarios provides the basis for the discussion of research questions in the next chapter.

Direct comparison of AGR and AFS-MAX elucidates the vastly different C dynamics of the two scenarios, with aHANPP (Figure 27) in AFS-MAX almost cut in half by a reduction of -1.6 t C ha$^{-1}$ (-1,185 kt C). Analysis of aNPP-act reveals that both AFS scenarios have a higher overall productivity than the AGR scenario, with a plus of 0.5 t C ha$^{-1}$ (306 kt C) in AFS-MAX and of 0.2 t C ha$^{-1}$ (106 kt C) in AFS-GRAD.

While the upward trend of aNPP-act (Figure 28) starts around 2027 in AFS-MAX, that is seven years after the planting of the trees, AFS-GRAD has the curve continuously lying below AGR and aNPP-pot until 2037. By this time, agroforestry systems in AFS-GRAD are implemented on 50% of the area with an average tree age of 10 years. After this point, tree growth in AFS-GRAD starts to rise significantly and starts to offset reductions in biomass harvest. In relation to aNPP-pot, the increased overall productivity of the two AFSs results in a slightly negative and quasi-neutral aHANPP-luc of -0.2 t C ha$^{-1}$ (-151 kt C) in AFS-MAX and 0.05 t C ha$^{-1}$ (48 kt C) in AFS-GRAD, whereas AGR does not – if also only slightly – reach the potential productivity with aHANPP-luc of 0.2 t C ha$^{-1}$ (154 kt C). Although these numbers lie quite close together, differences in aHANPP-luc become clearer when extending the time frame. While aNPP-act in AFS-MAX continues its steady decline until almost converging with aNPP-pot by 2070, it is still notably above aNPP-pot between 2050–2070. AFS-GRAD's aNPP-act, on the other hand, keeps rising continually to 6 t C ha$^{-1}$ yr$^{-1}$ after 2060 and finally peaks in 2074 (at 6.2 t C ha$^{-1}$ yr$^{-1}$) with an average tree age of 41.5 years.

**Table 11: Comparison of aHANPP components of the three land use scenarios. Values represent accumulated carbon flows/stocks and area-weighted average carbon flows/stocks for the period of 2020–2050, in kt C and t C ha$^{-1}$, respectively. Indents of HANPP indicators represent sub-categories.**

| | *Accumulated C flow/stock, kt C* | | | *Average C flow/stock, t C ha$^{-1}$* | | |
|---|---|---|---|---|---|---|
| *HANPP indicator* | *AGR* | *AFS-MAX* | *AFS-GRAD* | *AGR* | *AFS-MAX* | *AFS-GRAD* |
| *aNPP-pot* | *3,779* | *3,779* | *3,779* | *4.9* | *4.9* | *4.9* |
| *aNPP-act* | *3,624* | *3,930* | *3,730* | *4.7* | *5.1* | *4.8* |
| *aNPP-eco* | *1,006* | *2,192* | *1,505* | *1.3* | *2.9* | *2.0* |
| *RPB* | *-* | *1,518* | *654* | *-* | *2.1* | *0.9* |
| *RAB* | *1,006* | *674* | *851* | *1.3* | *0.9* | *1.1* |
| *aHANPP* | *2,773* | *1,587* | *2,273* | *3.6* | *2.0* | *2.9* |
| *aHANPP-luc* | *154* | *-151* | *48* | *0.2* | *-0.2* | *0.05* |
| *aHANPP-harv* | *2,618* | *1,738* | *2,225* | *3.4* | *2.2* | *2.8* |
| *Yield, Crops* | *674* | *377* | *549* | *0.9* | *0.5* | *0.7* |
| *Yield, Grass* | *1,075* | *608* | *862* | *1.4* | *0.8* | *1.1* |
| *Yield, Fruit* | *-* | *267* | *128* | *-* | *0.4* | *0.2* |
| *Used residues* | *609* | *340* | *480* | *0.8* | *0.4* | *0.6* |
| *Unused residues* | *261* | *146* | *206* | *0.3* | *0.2* | *0.3* |

Combined crop and grass yields in AFS-MAX and AFS-GRAD are reduced considerably as compared to AGR, by -1 t C ha$^{-1}$ (-764 kt C) and -0.4 t C ha$^{-1}$ (-338 kt C), respectively. Additional total fruit yields of 0.4 t C ha$^{-1}$ (267 kt C) in AFS-MAX and 0.2 t C ha$^{-1}$ (128 kt C) in AFS-GRAD are able to offset a substantial part of this reduction, resulting in a moderated but nevertheless grave difference in aHANPP-harv of -1.2 t C ha$^{-1}$ (880 kt C) in AFS-MAX and -0.5 t C ha$^{-1}$ (393 kt C) in AFS-GRAD. When looking at the curves beyond the year 2050 (Figure 29), we see that average yields per hectare in AFS-MAX and AFS-GRAD do not decrease any further after 2050 and 2065, respectively. On the one hand, this suggests that trees have matured after 20–30 years to a level where no further significant increase in the interception of solar radiation and competition for water occurs. On the other hand, this interpretation is contradicted by the fact that yearly increase in woody biomass (RPB in Figure 30) is significant in both scenarios until the year 2080, even though strong early growth of RPB stagnates and shrinks after 20 years in AFS-MAX and after 45 years in AFS-GRAD.

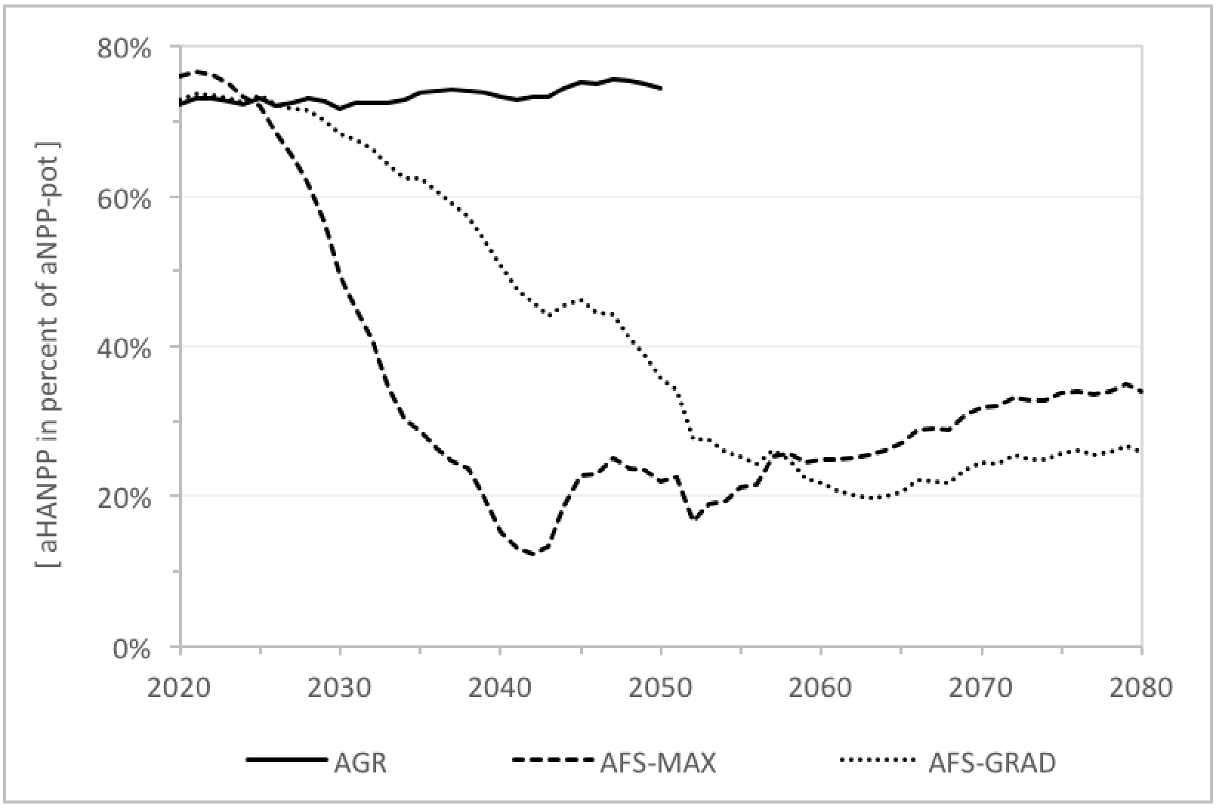



**Figure 27: aHANPP of the three modelling scenarios, 2020–2080, in percent of aNPP-pot.**

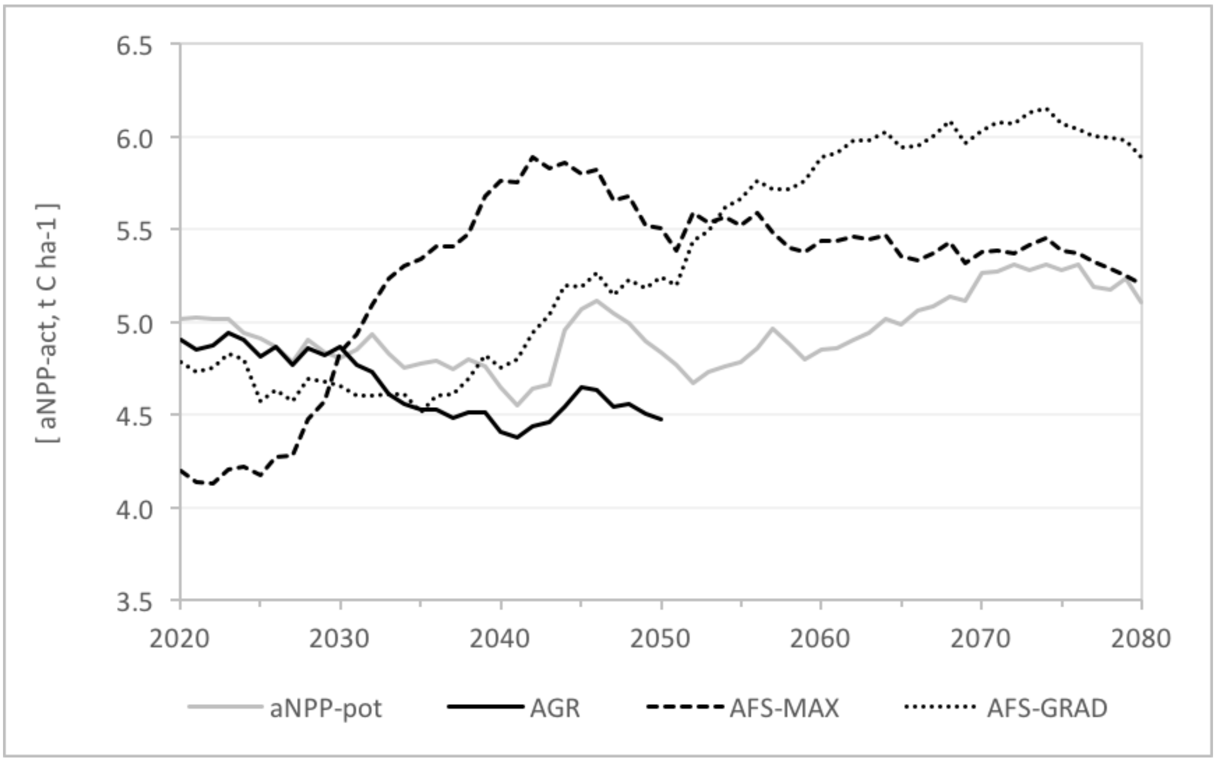



**Figure 28: Area-weighted average aNPP-act and aNPP-pot of the three modelling scenarios, 2020–2080, in t C $ha^{-1}$ $yr^{-1}$.**

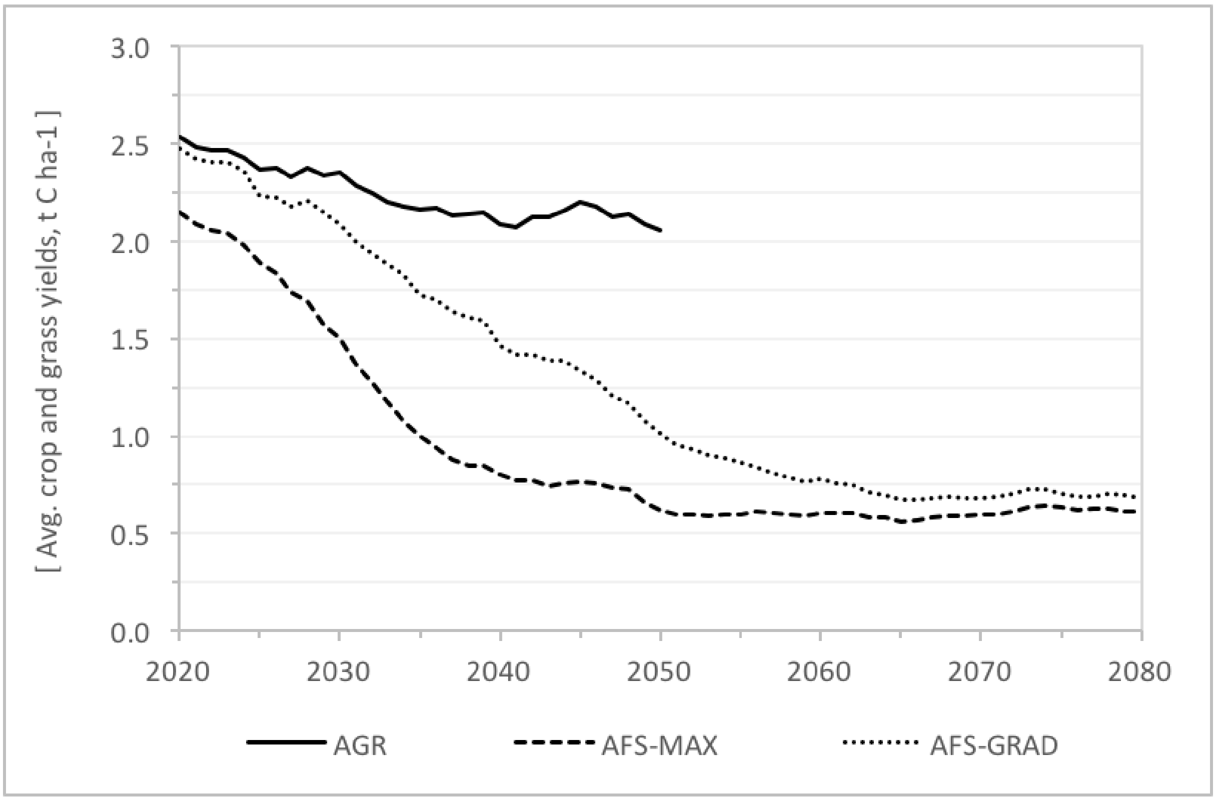



**Figure 29: Area-weighted average combined yields of crops and grass, 2020–2080, in t C ha$^{-1}$ yr$^{-1}$.**

The negative development of yields is opposed by a positive development of aNPP-eco (Figure 30). Compared to AGR, accumulated aNPP-eco from 2020–2050 more than doubles in AFS-MAX and increases by 50% in AFS-GRAD. While RAB remains almost constant in AGR, it decreases in the AFSs simply due to less area being available to crops, and especially, to crops on fallow land (see chapter 0). Total RPB, on the other hand, is inexistent in AGR but grows strongly in AFS-MAX by an average between 2020–2050 of 2.1 t C ha$^{-1}$ (1,518 kt C) and, again less pronounced but nevertheless considerably, in AFS-GRAD by 0.9 t C ha$^{-1}$ (654 kt C).

In other words, tree growth in both AFS scenarios more than offsets the reduction in crop and grass yields. This leads to higher aNPP-act than in AGR and subsequently negative aHANPP-luc by 2050. But, as yearly tree growth stagnates and decreases with advancing tree age, actual and potential vegetation in tree stands of uniform age converge again (as best observed in the curve of AFS-MAX in Figure 28 and AFS-MAX: RPB in Figure 30). A wider distribution of the age structure of the trees has a moderating effect on the development of aNPP-act, thus leading to a more constant rate of negative aHANPP-luc over time.

Figure 31 shows the accumulated C stock (ACS) and carbon carrying capacity (CCC) in the two AFSs. ACS relates to the accumulation of RPB from 2020–2080, whereas CCC depicts the maximum C sink in a hypothetical study region, where tree age is evenly distributed throughout the assumed harvest cycle of 61 years, i.e. with a mean tree age of 30.5 years.

ACS reaches 156 and 120 t C ha$^{-1}$ (or 3.37 and 2.6 Mt C in the study region) in the AFS-MAX and AFS-GRAD scenarios, respectively. ACS curves do not show a significant saturation of yearly increment, as could already be inferred from developments of RPB. This means that trees keep growing steadily, even with an age of 60 years at the end of AFS-MAX scenario. CCC reaches an average of 67.5 t C ha$^{-1}$ (or 1.46 Mt C in the study region). It is illustrated in the figure where the solid line intersects with the dashed and dotted lines, that is between 2049–2050 in AFS-MAX and 2063 in AFS-GRAD.

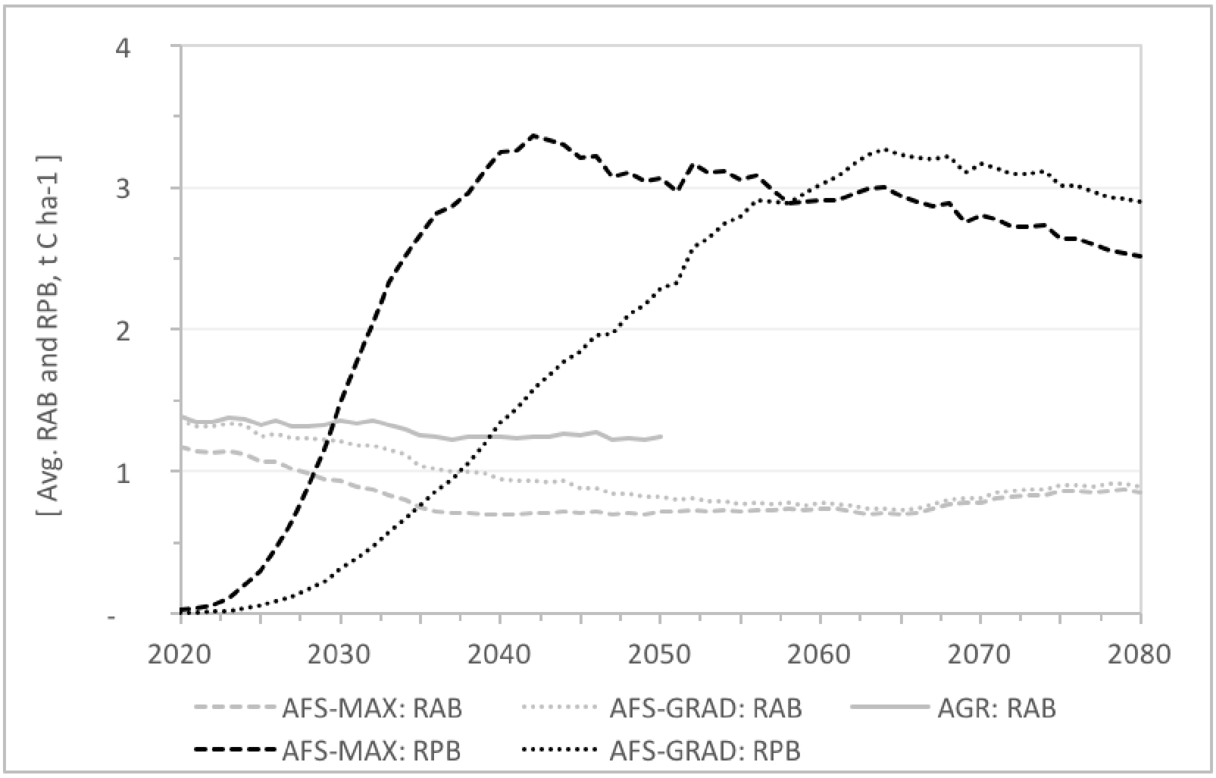



**Figure 30: Area-weighted average remaining annual biomass (RAB) and remaining perennial biomass (RPB), 2020–2080, in t C ha$^{-1}$yr$^{-1}$. The sum of RAB and RPB equal aNPP-eco.**

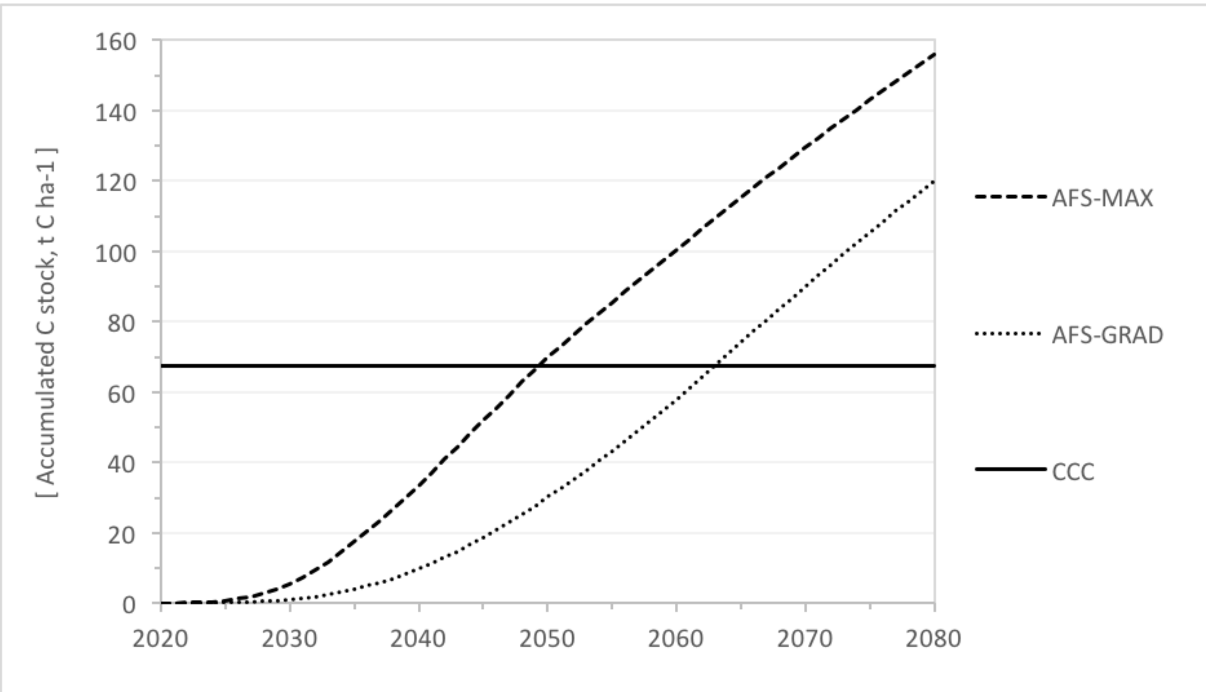



**Figure 31: Development of the area-weighted accumulated carbon stock (ACS) in the AFS-MAX and AFS-GRAD scenarios from 2020–2080 as well as the carbon carrying capacity (CCC) in the study region, in t C ha$^{-1}$. The intersection of the solid line with the dashed and dotted lines depicts the year in which the region's CCC, at a mean tree age of 30.5 years, is reached in the respective scenarios.**

# 5 Discussion

The results of this study demonstrate that the implementation of agroforestry systems on available agricultural lands in the Eisenwurzen study region severely impacts the C dynamics of the agro-ecological landscape over time. Through the application of the HANPP framework, analysis of the C flows thereby enables the assessment of the trade-offs between CS and biomass harvest.

While the yearly increment in woody biomass grows rapidly during the first three decades of the newly established agroforestry stands, average crop and grass yields diminish strongly. Yields then stabilize at a relatively low level, whereas tree growth continues until the assumed point of harvest, although, with increasing age at attenuated rates. The total amount of C stored in woody biomass and contained in the yield of cherries thereby surmounts the total amount of C lost through decreasing crop and grass yields. This not only leads to an overall higher net primary productivity of the AFSs compared to the AGR, but also compared to the potential net primary production. Despite this net increase in productivity, the trade-off between food production and CS is substantial.

These dynamics have several implications connected to a society's specific needs as well as to the normative functions it prescribes to an agroecosystem. The following four sections discuss the study's validity and relevance beyond the examined case. (1) Limitations to this study and aspects of data and model integration are summarized. (2) Results are evaluated along the lines of LULMC to disentangle effects described in chapter 4.2. (3) The agroforestry system's CS potential in the context of Austria's climate change mitigation strategies and goals are discussed. (4) The reduction in biomass harvest and yields is assessed against the backdrop of socio-economic considerations, among others relating to food security and climate change adaptation.

## 5.1 Methodological limitations, data and model integration

Working with model predictions and scenarios necessarily requires assumptions to be made about the future, inherently containing a range of uncertainties. In this case, four main sets of assumptions had to be made about (1) future land use in the study region based on the sustainability pathway SSP1, itself based on assumptions about socio-economic developments and system dynamics inherent to the SSPs in general and SECLAND in particular (chapter 3.2.2); (2) net primary productivity determined as the minimum of the annual temperature and precipitation functions without any interactions between these two variables inherent to the MIAMI model (chapter 3.4.1); (3) crop and tree interactions and their biomass productivity, based on a mechanistic understanding of plant behavior and a dynamic but simplified representation of the interaction between the trees, crops, atmosphere and soil environment inherent to Yield-SAFE (chapter 3.2.3); (4) future climatic development based on data from CMIP5 and inherent assumptions about the development of GHG concentrations and their radiative forcing potentials (in this case RCP4.5). All of these assumptions and uncertainties contained within, necessarily add-up within the methodological framework applied in this study.

The level of detail in this study, operating on a landscape-scale, was confined by data availability and working time and is mostly related to the modelling done in Yield-SAFE. These factors converge for a variety of issues: the number of modelling sites to represent climate and soil variables at a finer spatial resolution; the number of species that were incorporated into the AFS; the lack of specific parameterization of Yield-SAFE for the individual tree and crop species in the study region. These issues were already discussed in chapter 3.2.3 and are accepted within the scope of this study. They might, however, contribute to inaccuracies of the results, for example causing the deviation of predictions of crop biomass production from reported statistical data. Further work in this regard should comprise the inclusion of other species (in particular other tree species) as well as modelling sites. This extension of agroforestry model runs should also include different system designs, i.e. a variable number of trees per hectare in order to assess the trade-off between CS and food production according to different normative goals. Another important work package of future research is undoubtedly the collection of appropriate datasets of plant and soil characteristics and the site- and species-specific parameterization of Yield-SAFE.

To assess model integration, it is primarily important to look at the performances of Yield-SAFE and MIAMI models, because these two models provide the simulations of biomass productivity central to the HANPP calculations in this study. Although the land use datasets provided by SECLAND were necessary for the calculations too, they constitute input data used for the scaling-up of results to the landscape-level, which is relatively independent from the other two bio-physical models.

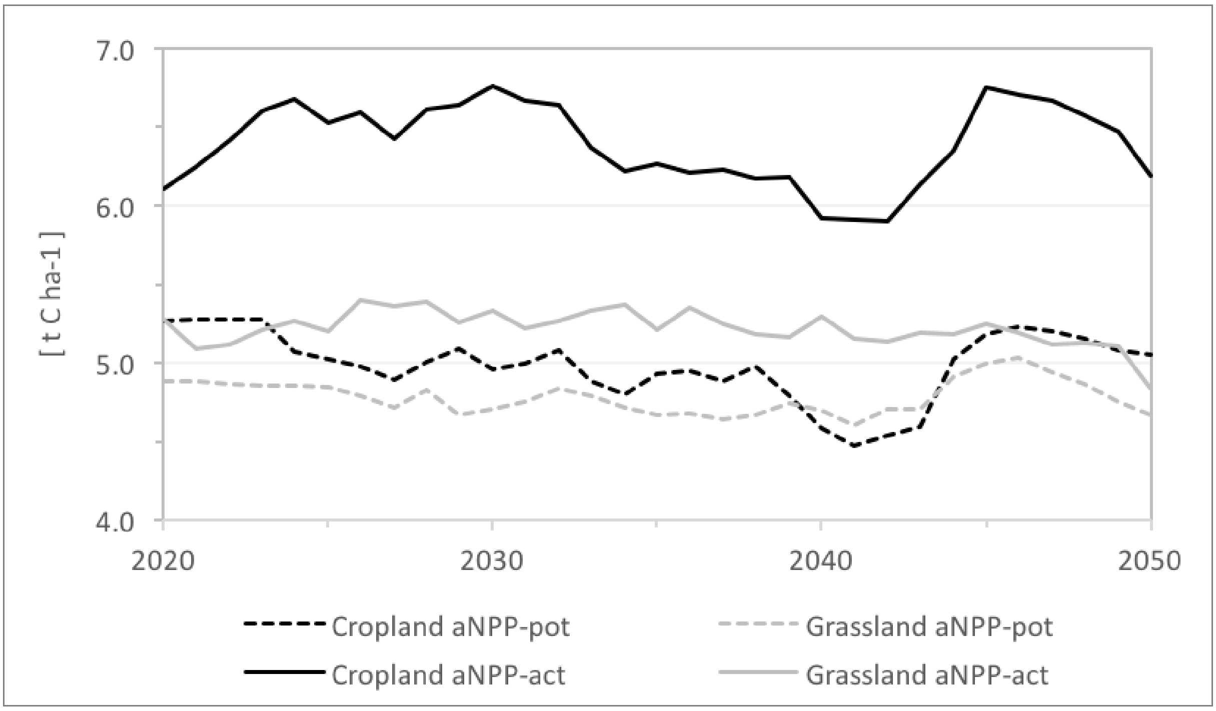



**Figure 32: aNPP-pot ha$^{-1}$ and area-unweighted average aNPP-act ha$^{-1}$ of the cropland and grassland modelling sites in t C ha$^{-1}$ yr$^{-1}$.**

To assess the fit of Yield-SAFE and MIAMI models and ascertain similar behavior of both modelling approaches, comparison of aNPP-pot (MIAMI) and aNPP-act (Yield-SAFE) in conjunction with the underlying climate data provides insight into this question. Figure 32 shows aNPP-pot and area-unweighted average aNPP-act[15] of both land use categories.

In general, values of aNPP-act (Yield-SAFE) and aNPP-pot (MIAMI) follow similar trends within the two modelling sites. Small deviations, such as the increase of cropland aNPP-act between 2020–2024, which cannot be explained by annual climate data, are ascribed to daily and

[15] All aNPP-act values in this section refer to results from the AGR scenario, used as a baseline. The unweighted average includes extensive grass, maize, oilseed, winter wheat and barley in the cropland category, as well as extensive grass and intensive grass in the grassland category.

monthly weather variations during sensitive growth stages of specific crops. This difference comes from the fact that Yield-SAFE operates on a daily time-step whereas MIAMI operates on a yearly time-step. Results nevertheless indicate, that both models supply sufficiently homogenous results despite operating with completely different algorithms and temporal resolutions. This further validates the methodological approach to combine the respective datasets in the HANPP calculation.
Significant saturation effects in tree growth did not occur during the study period and not even until the end of the extended study period, as could be seen in Figure 31 (chapter 4.6). A very slight flattening of the curve around 2040 indicates that the yearly growth rate stagnates and RPB results show that from then on RPB continuously but slowly decreases until 2080. Nevertheless, trees are still accumulating more than 2.5 t C $ha^{-1}$ $yr^{-1}$ between 2050–2080.
The extended study period of 61 years was chosen primarily because of the assumption that this time frame corresponds to one full harvest cycle of wild cherry, but also to assess the saturation in tree growth. An additional model run (Figure 33) was thus performed with matching parameters of the cropland modelling site, except for the modelling period of 150 years and the climate setting of "current climate" (because RCP4.5 data is only available until the year 2100), to see when and to what extent a saturation in tree growth would set in.

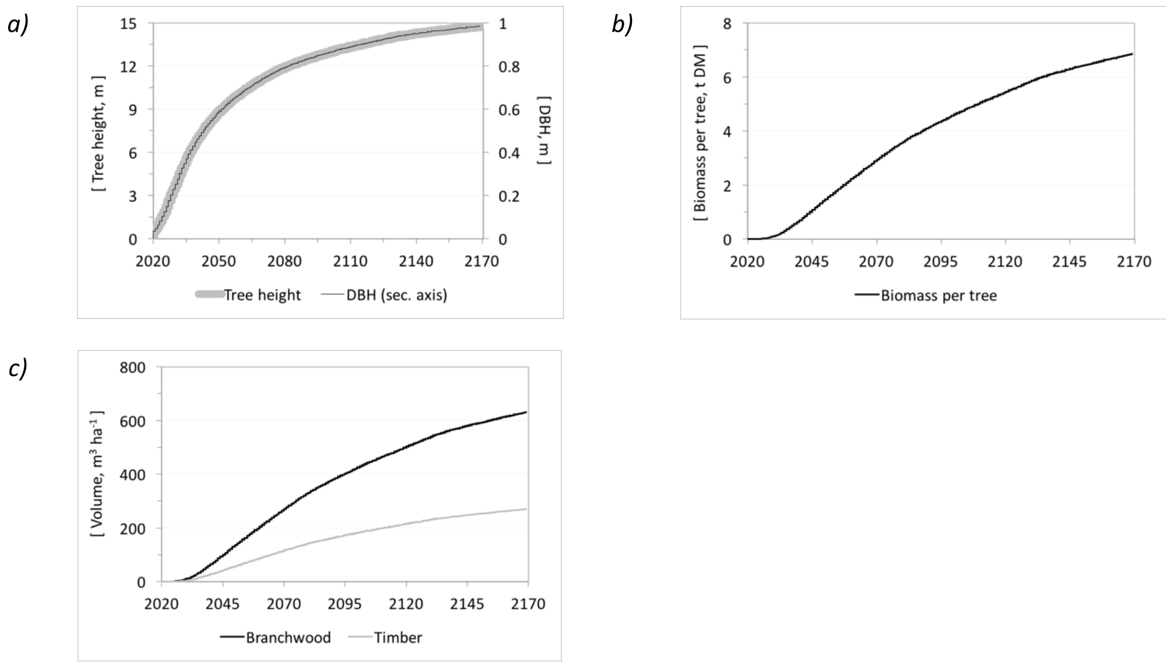



**Figure 33: Development of wild cherry on the cropland modelling site with corresponding input parameters showing [a] tree height and diameter at breast height (DBH) in m, [b] biomass per tree in t DM $yr^{-1}$ and [c] volume of timber and branch wood in $m^3$ $ha^{-1}$ $yr^{-1}$, 2020–2170.**

Figure 33-a shows the development of tree height and DBH, which are obviously exactly synchronized within the logic of Yield-SAFE. Annual increase of height and DBH clearly attenuates with growing tree age, starting around the year 2050. Despite a slighter saturation, biomass per tree (Figure 33-b) increases at least until the tree reaches 150 years of age. This might be explained by a stronger development of the tree crown compared to the stem with increasing age, also shown by the relationship between growth of timber and branch wood (Figure 33-c). As described in chapter 3.3.2, tree structures of wild cherry for fruit and timber production vary strongly. Accordingly, model predictions are a good fit with tree development associated with cherry fruit production.

## 5.2 Differentiating land use and land management change

As HANPP enables the quantification of the effects of land conversion and biomass harvest it can be used to explain typical land use change processes, for example distinguishing between changes in the extent of land use types such as agriculture and forestry, changes in production intensities and changes in ecological productivity, for example due to climatic variability or climate change (Gingrich et al., 2015). These processes are characterized by specific developments of individual HANPP components. Figure 34 illustrates two of those processes relevant to this study on a hypothetical land area. As such, forest expansion (Figure 34-a) is characterized by an increase of NPP-eco while HANPP-harv, HANPP-luc and subsequently HANPP decline, depicting additional forest growth at the expense of agricultural biomass extraction. Agricultural intensification (Figure 34-b), on the contrary, is characterized by rising HANPP-harv and to a smaller extent NPP-eco, resulting in negative HANPP-luc and decreasing HANPP, depicting higher biomass extraction on a smaller area of land.

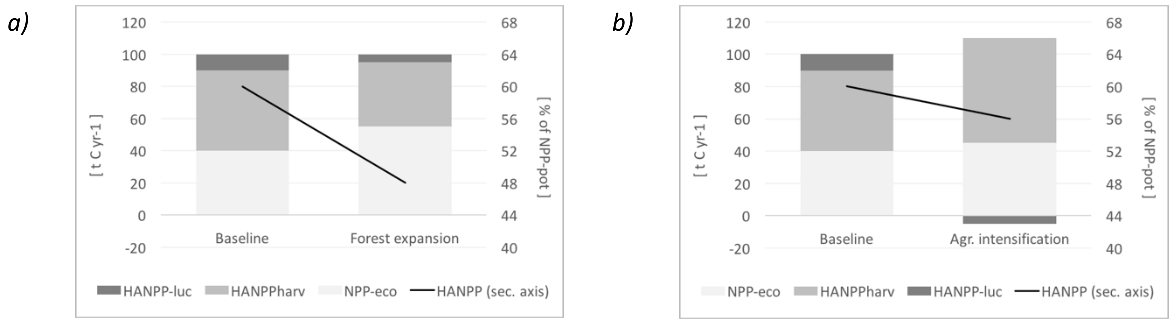



**Figure 34: Hypothetical examples of land use change illustrating changes to HANPP components in scenarios of [a] forest expansion and [b] agricultural intensification, in t C yr$^{-1}$ (primary axis) and percent of aNPP-pot (secondary axis). Own illustration based on Gingrich et al. (2015), Fig. 3.**

The above-mentioned examples are based on land use datasets that include the two land use types agriculture and forest, whereas in this study analysis is based on agricultural land only. Nevertheless, parallels exist when interpreting results from this perspective. In the AGR scenario (Figure 35-a) developments occur in reverse of the agricultural intensification hypothesis described above: the decline of aHANPP-harv and aNPP-eco is paired with an increase of aHANPP-luc and aHANPP, depicting the process of agricultural extensification (which in this case is based upon the land use change inherent to the SECLAND datasets). The AFS-MAX scenario (Figure 35-b), on the other hand, presents a completely different picture: aNPP-eco increases and aHANPP-harv decreases strongly, insofar as that aHANPP-luc becomes negative and aHANPP decreases drastically.

From this perspective, it seems that the implementation of agroforestry acts as if it were a change in the extent of land use types, i.e. agricultural land split into agricultural land and forest. In a way, it could be argued that this is actually the case, just in a very small-scale structural form (despite contradicting the very essentials of agroforestry). Nevertheless, increasing aNPP-act per area unit of land – even lying significantly above aNPP-pot – is clearly a sign of land use intensification. The higher biomass production is, in this case, just not to the benefit of extraction but sequestration. In other words, integrated and multi-functional land systems have the ability to harness the benefit of higher productivity in combining CS and biomass harvest. The challenge then lies in designing and implementing the systems in such a way, that CS and biomass harvest are balanced within a farm's or a region's socio-ecological context.

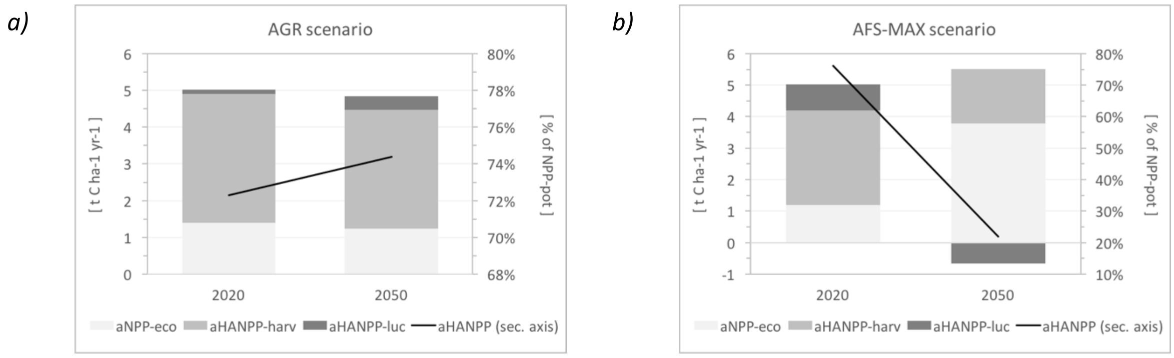



**Figure 35: aHANPP components in the [a] AGR and [b] AFS-MAX scenarios, 2020 and 2050, in t C ha$^{-1}$ yr$^{-1}$ (primary axis) and percent of aNPP-pot (secondary axis).**

Increased productivity is generally possible due to the ecological theory of niche differentiation, i.e. different species obtaining resources from different parts of the environment (J. Smith et al., 2013). The production of more than one product – or of different components of an agroforestry system – is thereby achieved by the complementary use of solar radiation, water and nutrients (Cannell et al., 1996). This can lead to a yield advantage, referring to a situation where a mixture of species produces more yield from an area unit of land than the combined yields if that area were divided into sole stands. A method named Land Equivalent Ratio (LER) to evaluate yields of intercropping situations as compared to their sole crop yields was introduced by Mead and Willey (1980) and was later adopted to the concept of agroforestry by Ong et al. (1996). On the basis of this method, higher LERs in agroforestry systems were predicted by models as well as measured on experimental sites.
Yield-SAFE already predicted LERs between 1–1.7 (LER > 1 depicts higher biomass production in the intercropping situation) in different studies at its inception (Graves et al., 2007; Keesman et al., 2007; van der Werf et al., 2007). In the study by Graves et al. (2007), which combined field experiments and modelling, the highest LERs of 1.6 were obtained by integrating deciduous trees with autumn-planted crops, because of their complementarity of light-use. This combination was also applied in the present study, with the deciduous tree species wild cherry and the winter crop species winter wheat, oilseed and the two grass species (whereas maize and barley are spring-planted). A different study by Khasanah et al. (2020), although in a completely different context and setting, found a higher LER for an oil palm and cocoa agroforestry system compared to respective monocultures in Indonesia, by using the process-based agroforestry model WaNuLCAS (Water, Nutrient and Light Capture in Agroforestry System). The tendency of increasing productivity in agroforestry systems was also corroborated in a study using experimental data by Seserman et al. (2018), which found LERs of above their specific thresholds for two agroforestry systems of short rotation coppice in combination with different crop species in Germany, as well as in a study by Sharrow and Ismail (2004) of douglas fir and ryegrass systems in north-west USA. A meta study by Torralba et al. (2016) found neutral or negative effects of agroforestry on provisioning services – although it is important to note that they only considered studies which compared individual elements such as wood or grass, and not the full amount of biomass produced – such as a meta-study (Rivest et al., 2013), finding that scattered trees on pastures do not compromise pasture yield, or a study using data from site-specific experiments and demonstration trials from temperate agroforestry systems, which found that shading, however manageable, has been shown to decrease yields of associated forage species in silvopastoral systems (Jose et al., 2004).

In conclusion, these findings suggest that increased biomass production is generally possible due to niche differentiation and complementary use of resources, but effects are very much dependent from the specific species integrated in an agroforestry system as well as climate, soil and management variables.

## 5.3 Agroforestry's carbon sequestration potential

The carbon carrying capacity (CCC) defines the maximum net C sink in the land use system when tree age is equally distributed throughout one harvest cycle, i.e. a mature system established over time in which trees are continuously harvested and replanted. This enables the assessment of the maximum climate change mitigation potential. CCC in the AFSs was calculated with an area-weighted average of 67.5 t C $ha^{-1}$ at a mean tree age of 30.5 years, amounting to a potential net C sink of 1.46 Mt C in the study region.

It was pointed out (Aertsens et al., 2013; Dixon, 1995) that agroforestry systems can also be significant sources of GHG emissions through diverse management practices ranging from shifting cultivation to chemical fertilization to tillage or the use of fossil fuel driven machinery, determining the net-flux (source or sink) of GHGs. Some of those practices would be assumed to happen in the study region, regardless of the land use scenario applied therein. Such emissions are nevertheless not considered in this study.

C storage in agroforestry systems was calculated and measured in a variety of studies, covering different regions, scales, timeframes and types of systems. Despite most data coming from the tropics, temperate agroforestry is gaining in importance in recent years. Estimates of total (above- and below-ground) CS potential in temperate agroforestry systems (with tree age ranging from 6–41 years) vary between 1–12 t C $ha^{-1}$ $yr^{-1}$, depending on species, climate, soil, management, and rotation (Lawson et al., 2019). Aertsens et al. (2013) calculated the total CS potential of agroforestry in Europe (without stating a reference timeframe) with an average of 2.75 t C $ha^{-1}$ $yr^{-1}$. IPCC default coefficients for above-ground woody biomass in cropping systems containing perennial species in temperate climate regions are 63 t C $ha^{-1}$ over a harvest cycle of 30 years, i.e. an accumulation rate of 2.1 t C $ha^{-1}$ $yr^{-1}$ (IPCC, 2006). A study including a wild cherry and rye grass/fescue system with 200 trees $ha^{-1}$ aged 26 years in France reported an above-ground C stock of 36.7 t C $ha^{-1}$, corresponding to 1.4 t C $ha^{-1}$ $yr^{-1}$ (Cardinael et al., 2017). Dixon (1995) calculated a global median above-ground C storage in the vegetation of agroforestry systems of 70 t C $ha^{-1}$ over the period of 50 years, corresponding to 1.4 t C $ha^{-1}$ $yr^{-1}$, too.

CCC results of the present study, with an average accumulation rate of 1.1 t C $ha^{-1}$ $yr^{-1}$ over the period of 61 years (with a mean tree age of 30.5 years), coincide well with the lower range of reported values. A recent meta-study by Feliciano et al. (2018) concluded, that transition to agroforestry always results in positive above-ground CS, whereby agrisilvicultural and silvopastoral systems are at the lower end of the sequestration spectrum and results differ strongly for climates, soils and management.

The actual and potential C content of mixed forests in Austria was calculated at 96 and 154 t C $ha^{-1}$, respectively, and the total reduction of the potential aboveground standing crop in Austria at 630 Mt C (Erb, 2004). In comparison, the potential CCC of the AFS of 67.5 t C $ha^{-1}$ (with a corresponding total of 1.46 Mt C in the study region) corresponds to approximately two thirds of the actual C content of mixed forests in Austria.

***Temporal dynamics of carbon sequestration***

The C sequestered in agroforestry biomass is eventually bound to be harvested, and, in the case of wild cherry can be turned into long-lived wood products, thus storing it far beyond the point of harvest. But, as pointed out before, wild cherry growth depends strongly on the tree's purpose, be it fruit or timber. Fruit trees thereby yield relatively little veneer-grade timber, and vice versa (Sheppard and Spiecker, 2015), as is the case in this study. The majority of wood here comes from branches and is not well suitable for veneer production, as previously discussed in chapter 3.3.2 and shown in Figure 33-c (see chapter 5.1). This part of the trees' biomass can, nevertheless, be used as a source for bioenergy, thereby substituting fossil fuel use. Biomass, however, is a hydrocarbon and releases (primarily) $CO_2$ when converted to energy, having a similar effect on global warming than $CO_2$ emissions from fossil fuels. The C used for biomass accumulation has, of course, been sequestered from the atmosphere by photosynthesis in the first place, which is true for fossil fuels as well as biofuels, the difference being the timing of sequestration and release. Röder and Thornley (2016, p. 3) point out:

> *"In the case of annual, these sequestration and release events may be very closely spaced […]. The lifetime of $CO_2$ in this case is short and hardly contributes to any net long-term increase in GHG concentrations. […] In case of forest-based biomass, the picture changes as at the point of harvest a large amount of $CO_2$ is released into the atmosphere and its sequestration takes a much longer time."*

Climate neutrality of forest based bioenergy is, in the short term, not a given or at least subject to a complex scientific and political debate (Cherubini et al., 2012; Gaudreault and Miner, 2015; Marland and Schlamadinger, 1995; McKechnie et al., 2011; Searchinger et al., 2018). In the case of CCC estimated in this study, harvested biomass used for bioenergy can, however, more readily be viewed as a climate-neutral source because the calculation considered an equally balanced age structure of trees between 0–60 years (corresponding to a full harvest cycle). In any case, CS in biomass takes time. In the AFS-GRAD scenario, the CCC is reached in the year 2063.

***Austria's climate strategy, GHG emissions and climate change mitigation potential***

These numbers are also interesting in the context auf Austria's climate strategy. Under current EU legislation, Austria is obliged to reduce GHG emissions from emission sources outside the emission trading system until 2030 by 36% (from 2005 levels) and is further committed to becoming climate-neutral by 2050 (BMNT, 2019a). By then still existing but unavoidable emissions (e.g. from agriculture or industry) will need to be compensated by C storage in natural or technical sinks. The so-called *transition scenario* (Umweltbundesamt, 2017a) bases emission reductions on changes in the energy, transport, industry, building, land use and consumption sectors. Within this scenario, different pathways assume natural C sinks to sequester between 3.9–17 Mt $CO_2$-e from 2020–2050 to offset excess emissions (BMNT, 2019a).

While Austria's forests are a C sink at present (Anderl et al., 2018), they cannot act as a C sink forever because of restrictions to the area extent as well as the density of forests. After centuries of gradual deforestation for the benefit of agricultural expansion as well as domestic and industrial wood consumption, forests were allowed to regrow in terms of area and density at least since 1830 (Gingrich et al., 2007). This was made possible by the transition from an agrarian to an industrial socioecological regime (Krausmann, 2001; Krausmann et al., 2016), among others characterized by a nearly fivefold increase of domestic C consumption and fossil fuels becoming its primary source (Erb et al., 2008). While these land use legacies promote future C uptake in Austria much beyond 2050, legacy effects of forest management (primarily the promotion of homogenized Norway spruce stands) put forests at particular risk to increasing disturbances under climate change, reducing future C uptake potential (Thom et al., 2018). The increased demand for bio-energy could put forests under additional intensive management. While in 2018 almost two thirds of the bioenergy used in Austria (approximately 150 PJ) came from forestry (Statistik Austria, 2020), the use of forestry biomass could, according to the Austrian Biomass Association (Österreichischer Biomasse-Verband, 2015), be increased to more than 200 PJ based on the notion of increasing forest stocks as well as increasing amounts of fallen timber due to extreme weather and pests. These developments pose uncertainties when determining future C uptake of Austrian forests and LULUCF in general (Umweltbundesamt, 2017a). Furthermore, technical solutions such as carbon capture, utilization and storage, which are part of Austria's climate strategy (BMNT, 2019a), are also prone to a range of socio-economic and technical uncertainties and discussed critically (Markusson et al., 2012; Smith et al., 2016; Stephens et al., 2011).

It follows from these considerations, that agroforestry could be a valuable contribution to Austria's land-based C sink. As a thought experiment, let us consider the implementation of agroforestry systems on the total of Austria's arable land and grassland, amounting to roughly 2.5 Mha in 2018 (BMNT, 2019b). Using data from the AFS-GRAD scenario the CCC of 67.5 t C $ha^{-1}$ would be reached by the year 2063. This would amount to a total of C sink of roughly 170 Mt C or 3.9 Mt C $yr^{-1}$ over the period from 2020–2063.

According to Austria's Annual Greenhouse Gas Inventory 1990–2016 (Anderl et al., 2018), Austria's net C sink[16] from forest land amounted to 4.3 Mt $CO_2$-e in 2016, which corresponds to roughly 1.15 Mt C[17]. Compared to 2.95 Mt C from 1990 this is, however, a reduction of 65%, mainly caused by weather conditions, wind throw as well as high timber demand and price developments. LULUCF is projected to stabilize and remain a net sink at least until 2035 (Umweltbundesamt, 2017b). According to this comparison, the potential average yearly C sink of agroforestry on all of Austria's available agricultural land (from 2020–2063) would be 3.5-times the net C sink from forest land (in 2016). Additional sinks might emerge through the long-term accumulation of C in harvested wood products, indeed having implications for considering the mitigation potential of agroforestry systems involving high-value timber tree species. According to this, the harvest of timber from agroforestry systems would be accounted as an additional C sink.

Emissions from agriculture, made up of $CH_4$ from enteric fermentation (57%) and $N_2O$ from agricultural soils (29%), amounted to 1.99 Mt C in 2016 (corresponding to 9.1% of the total national emissions). Between 2015 and 2035 an increase in emissions by 5.2% is predicted because underlying livestock projections indicate growing cattle and pig numbers for this period (Umweltbundesamt, 2017b). Agroforestry's potential C sink would thus surmount emissions from agriculture by a factor of more than 2.

[16] This number relates to the net C removals on forest land, without consideration of removals from harvested wood products as well as LULUCF emissions.

[17] To convert C into CO2 the ratio of the molecular weight of carbon dioxide to that of carbon was calculated at 3.664.

These considerations, of course, neglect all other socio-economic and ecological factors, barriers and adverse effects, in particular the reduction of yields, which is covered in the next section.

## 5.4 Conflicting objectives: the role of diminishing yields

Quantification of C dynamics with the HANPP framework also enables analysis of the developments of biomass harvest. Biomass harvest includes yields as well as used and unused residues. Yields comprise crops and grass in the AGR scenario and additional cherry fruit in the AFS scenarios. This chapter will put the modelled reduction in yields through the implementation of agroforestry into the context of socio-economic considerations as well as food security and self-sufficiency.

Between 2020–2050 biomass harvest and yields in AGR declined by 19 and 7.9% (to 3.2 and 2.1 t C $ha^{-1}$ in 2050), respectively, due to changes in crop species and production intensity. Without LULMC, biomass harvest and yields remained almost constant. During the same period in the AFS-MAX scenario, combined crop and grass yields declined by 71% (to 0.6 t C $ha^{-1}$ in 2050). This was offset by additional cherry yields of 0.7 t C $ha^{-1}$ $yr^{-1}$ in 2050, resulting in total yields declining 37.8% and biomass harvest declining by 42.4% (to 1.7 t C $ha^{-1}$ in 2050). Without LULMC, biomass harvest and total yields (including fruit) declined by 49.3 and 39.4%, respectively, combined crop and grass yields (without fruit) by 67%.

There is relatively little experimental data on the performance of arable crops in temperate agroforestry systems, especially with mature tree components. In one study (Pardon et al., 2018), the influence of tree rows on yield and quality of key western European arable crops (maize, potato, winter wheat and winter barley) in Belgium was assessed, with trees of young, moderate and older age (2–48 years). Tree species were mixed, including Prunus avium, Populus sp. Juglans regia and Sorbus terminalis. Effects on crop yields were mainly determined by the age of the trees and the distance to the tree rows, as well as by the type of crop. Winter crops (wheat, barley) were much less influenced than summer crops (maize, potato). Largest yield reductions were found between 2.5–12m distance from long-standing tree rows for forage maize (-65%), grain maize (-45%), potato (-45.5%) and winter wheat (-39.1%). Winter barley was much less affected (-13.5%). While maize and potatoes were affected strongly by middle-aged trees too (between -30.6 and -43.5%), winter wheat was affected much less in this category (-8.2%). Conclusions that can be drawn from the study are that winter crops are much better suited for silvoarable agroforestry than summer crops. Harvest of trees should furthermore not be prolonged any more than necessary, although several regulating and supporting ecosystem services such as CS and biodiversity conservation may be positively related to the size (and/or age) of the trees (Pardon et al., 2017). Yield reductions determined in the present study are within the range of the above-mentioned findings.

***Socio-economic dimensions***

Negative impacts on crop yields are not taken lightly by farmers (Graves et al., 2017; Rois-Díaz et al., 2018). Farmers often have preconceptions against agroforestry, and risk-averse attitudes in combination with high initial investments inhibit establishment of modern agroforestry systems (Nerlich et al., 2013). High investments and diminishing yields can also result in economically lean transition periods and the need for long-term planning, and additional problems can arise from decreased labor productivity, development of niche markets and the lack of institutional support and social learning networks (Borremans et al., 2018; Santiago-Freijanes et al., 2018). Rois-Díaz et al. (2018, p. 15) come to a similar insight:

*"Many farmers would be willing to implement agroforestry if they would have more knowledge on those [practices] available, their profitability, benefits and practical know-how. Undecided farmers would like to apply or expand agroforestry in their farm if the systems would be rewarding from an economic point of view."*

Graves et al. (2008) reported that 86% of 264 interviewed farmers were willing to use silvoarable systems, but only under particular conditions, the most important of which was confidence in their profitability. They conclude, that there is clearly a need for "policy, research, demonstration sites, and extension services, if silvoarable agroforestry is to become a significant feature of the European landscape" (Graves et al., 2008). These insights show that it is not merely a question of food security on a national or regional level, but primarily – and maybe even more importantly – a question of profitability and know-how for the individual farmer.
Considering the fact that cherries are not a great substitute for staple crops from the perspective of food security, fruit yields might, nevertheless, provide farmers with an additional diversified form of income. While cherry prices in the European Union (EU) are increasingly volatile throughout the season (April–August), the price trend since 2004 is rising with a mean price in 2019 in the EU of 326 € per 100 kg net weight (European Commission, 2020a). Modelled cherry yields in this study increased until 2080, if only slightly beyond 2050. The average cherry yield between 2020–2050 was 0.9 t fresh weight $ha^{-1}$ $yr^{-1}$ and between 2051–2080 it was 2 t fresh weight $ha^{-1}$ $yr^{-1}$. This would result in a hypothetical gross production value of cherries of 2,960 € $ha^{-1}$ $yr^{-1}$ between 2020–2050 and 6,431 € $ha^{-1}$ $yr^{-1}$ between 2051–2080 (calculated with average 2019 prices in the EU). In comparison, prices for cereals are much more stable throughout the year and, after several spikes between 2007–2013, relatively stable interannually since then. The mean price for wheat, barley and maize in 2019 in the EU was 173 € per ton (European Commission, 2020b). Modelled average grain yields between 2020–2050 for these three cultivars was 8.6 and 4.5 t $ha^{-1}$ $yr^{-1}$ in the AGR and AFS-MAX scenarios, respectively. Between 2051–2080 yields dropped to 2.1 t grain $ha^{-1}$ $yr^{-1}$ in the AFS-MAX scenario. This would result in a hypothetical gross production value between 2020–2050 of 1,486 € $ha^{-1}$ in AGR and 3,712 € $ha^{-1}$ in AFS-MAX as well as 6,883 € $ha^{-1}$ in AFS-MAX between 2051–2080 (calculated with average 2019 prices in the EU).

**Table 12: Estimated production value of grains and cherry in the two land use scenarios for the periods of 2020–2050 and 2051–2080. Grain and cherry prices refer to EU averages in 2019 per ton of product. Yields are given in t $ha^{-1}$ $yr^{-1}$.**

| | | | *2020–2050* | | | *2051–2080* | | |
|---|---|---|---|---|---|---|---|---|
| *Scenario* | *Grain price, $t^{-1}$* | *Cherry price, $t^{-1}$* | *Grain yield* | *Cherry yield* | *Est. total value* | *Grain yield* | *Cherry yield* | *Est. total value, €* |
| *AGR* | *173 €* | *3,260 €* | *8.6 t* | *–* | *1,486 €* | *n.d.* | *n.d.* | *n.d.* |
| *AFS-MAX* | | | *4.5 t* | *0.9 t* | *3,712 €* | *2.1 t* | *2 t* | *6,883 €* |

This comparison shows, that from an economical perspective the transition from conventional agriculture to agroforestry potentially results in a much higher production value due to the high prices for cherry. Nevertheless, it is to bear in mind that these numbers refer to simple gross value estimations that don't take initial setup and running management costs into

account. Necessary investments for the setup of an agroforestry system as well as tree management and harvesting costs for the fruit are expected to lie well above the management and harvesting costs for arable land. On the other hand, the value of wood for veneer or biofuel production that can be generated after the assumed life span of the trees, as well as generation of possible subsidies, are not accounted for either. Additional income could potentially be generated by the valuation of ecosystem services (Lusardi et al., 2020).
There exist various economic evaluations of agroforestry systems that take very different aspects of evaluation into account, ranging from the monetarization of individual aspects such as carbon (Aertsens et al., 2013) to the inclusion of a variety of additional factors such as investment and management costs, subsidy programs, return to labor or price uncertainties and market stability (Benjamin et al., 2000; Kaeser et al., 2010; Khasanah et al., 2020; Martinelli et al., 2019; Sereke et al., 2015) to the accounting of a larger suit of environmental benefits and ecosystem services (Kay et al., 2019; Ovando et al., 2016; J. Palma et al., 2007; Paul et al., 2017). Results are, again, very much dependent on the specific systems under study, but many examples identify higher overall economic gains and resilience.

***Self-sufficiency, food security and climate change***
A healthy human nutrition generally requires the supply of macronutrients (energy, protein and fat) and micronutrients (vitamins and minerals). Globally, over 820 million people are undernourished and over 2 billion people suffer from moderate levels of food insecurity including micronutrient deficiencies (FAO et al., 2019). While the Green Revolution, which led to the concentration on few crop species (such as rice, wheat and maize), achieved to reduce malnourishment globally, it was much less successful in reducing micronutrient deficiencies (Gómez et al., 2013).
The positive effects of agroforestry practices on strategies to strengthen the resilience of rural livelihoods was shown in many studies, in particular under conditions of subsistence farming in developing regions (Garrity et al., 2010; Mbow et al., 2014). In such contexts, food security is also linked to the concept of agrodiversity, as diverse food systems can increase the availability of macro- and micronutrients as well as the resilience to climate change impacts and environmental pressures (Frison et al., 2011). Rosenstock et al. (2019) further show that agroforestry in Sub-Saharan Africa is likely to improve a whole range of important health issues including food and nutrition security, the spread of infectious disease, the prevalence of non-communicable diseases, and human migration.
In the context of a welfare state such as Austria, embedded in a globally interconnected industrialized food system, impacts on food security must, nevertheless, be assessed differently. The degree of self-sufficiency in Austria varies strongly by category (BMNT, 2019b). Animal production shows overall high sufficiency, with production of milk reaching 164%, beef 149% and pork 102%, while poultry reached 72%. On the other hand, cereals reached 89%, vegetables 56%, fruits 40% and vegetable oils only 27%. While the supply with food products from Austrian agriculture shows an increasing trend between 2000 and 2017 (BMNT, 2019b), food security will be under increasing pressure in the future, among other things due to land utilization and climate change.
Land utilization, as defined by the Austrian Conference on Spatial Planning (ÖROK), is the permanent loss of biologically productive soil through the development of land for construction and transport purposes, recreation or excavation (Gruber et al., 2018). Usable space is very limited in Austria due to its alpine topography, with a decreasing gradient from east to west with >50–12% share of usable land (Gruber et al., 2018). At the same time, land utilization is disproportionately high compared to other European countries and concerns in

particular high-quality agricultural land for commercial activities and industry, settlement, transport and energy infrastructure (Umweltbundesamt, 2019a). Although decreasing since 2009, daily land utilization was 14.7 ha $day^{-1}$ throughout Austria for the period 2013 to 2016, and the three-years average was at 44 $km^2$ in 2019 (Umweltbundesamt, 2019b). According to the government program 2020–2024, land utilization is to be kept as low as possible and annual growth is to be reduced to 9 $km^2$ per year by 2030 (Gruber et al., 2018); a goal that appears far-fetched.

Projections of crop yields under various environmental and socio-economic scenarios differ strongly between different climatic regions, but also between different studies. While crops can benefit from temperature and $CO_2$ increase (Ewert et al., 2005), many other environmental conditions, such as extreme weather events or increased pressure from weeds and pests may influence yields negatively (Cogato et al., 2019; Hay, 2007; Juroszek and von Tiedemann, 2013). A meta-analyses of crop yields under climate change (Challinor et al., 2014) showed, that global production losses are expected for wheat, rice and maize in both temperate and tropical regions (with 2°C warming). The study further showed that yield losses are greater for the second half of the century as well as in tropical regions, but even moderate warming may reduce temperate crop yields in many locations. With crop-level adaptation measures, however, these losses could be partially off-set in temperate regions or attenuated in tropical regions (Challinor et al., 2014). The uncertainty about the impact of various effects, for example $CO_2$ fertilization, is nevertheless high, as well as uncertainty about developments of various drivers such as climate change, management or technological change (Müller et al., 2010). Additionally, different crop types may react differently to environmental change. In Europe, winter and spring crops may initially benefit if rainfall is sufficient, but increasing temperatures can reduce these positive effects after 2050 (although yield increase in cooler regions is possible); whereas late spring and summer crop yields decline almost everywhere by the end of the century (Supit et al., 2012). In a study by the Austrian Agency for Health and Food Safety (AGES), climate change is expected to have a direct impact on the production potential of the soils (Haslmayr et al., 2018). Simulation results in this study show that yield potentials decrease slightly under moderate climate change (ALADIN) and significantly under extreme climate change (CMIP5). This second scenario would severely impact Austria's predicted degree of self-sufficiency of many important cultivars including wheat, grain maize, potatoes and sugar beet.

One powerful lever to the issues of decreasing high-quality soils and agricultural productivity is human diet. There are many reasons why a dietary shift towards a higher share of plant-based products would be advantageous, ranging from less food-competing feedstock to environmental benefits to human health (Alexander et al., 2016; Schader et al., 2015; Tilman and Clark, 2014). A study by Erb et al. (2016), exploring the biophysical option-space of feeding the world, showed that "the world population can be fed healthily even with low cropland yields and little cropland expansion when diets with a reduced fraction of livestock products are adopted." According to their study, human diets have more impact on the biophysical option space than yields or cropland availability.

In one way or another it becomes clear, that Austrian agriculture and society as a whole will have to adapt to climate change if food security and self-sufficiency of agricultural production are to be ensured in the coming decades. Against this background, losing 40–70% of arable yields does not appear like a feasible option, even when the benefits for climate change mitigation are high. But, as argued for example by Matocha et al. (2012), the integration of climate change mitigation and adaptation measures can also create synergies. From this perspective, agroforestry holds potential to attenuate negative climate change impacts on agriculture by increasing habitat, structural and functional diversity (Hernández-Morcillo et al., 2018). Thereby, agroforestry has the ability for climate change risk abatement (for example by reducing the susceptibility to extreme weather events, improving soil fertility and decreasing erosion and flooding) as well as for enhancing resilience by diversifying sources of income and providing buffers against yield fluctuations (Hernández-Morcillo et al., 2018; Matocha et al., 2012). Subsequently, even if results suggest that a radical transition to agroforestry would have too large an impact on food production in the study region, this broader perspective still encourages a transition to agroforestry, if a more moderated one. By harnessing synergies between climate change mitigation and adaptation and due to the large variety of management options and site-specific flexibility, agroforestry systems have the potential to balance the trade-offs between carbon sequestration and food production on a regional scale.

# 6 Conclusion and final remarks

Anthropogenic land use is a major driver of global environmental change. While ensuring provision of food, feed, fiber and fuel, land use affects global biogeochemical cycles and biodiversity, contributing to climate change and jeopardizing vital ecosystem services. With population growth and rising per capita biomass demand, the extent and intensity of land use are anticipated to rise further. Additionally, agriculture and forestry are expected to contribute to climate change mitigation by simultaneously reducing GHG emissions and providing bioenergy to substitute fossil fuels, while creating additional carbon sinks in biomass and soils. Land use strategies are hence increasingly challenged to foster multi-functional agroecosystems. The question of how to simultaneously maximize carbon sequestration and biomass harvest is therefore paramount. Agroforestry is often proposed as a possible solution by simultaneously addressing climate change, food insecurity and environmental degradation. But the assessment of trade-offs and synergies is complex and losses in crop yields often seem outweighed by the impetus of environmental benefits when evaluating options for temperate industrialized regions.

The objective of this study was to evaluate the trade-offs between carbon sequestration and biomass harvest in a hypothetical transition from conventional agriculture to agroforestry in the Eisenwurzen region in Austria. This was achieved by quantifying the carbon dynamics of one agriculture and one agroforestry scenario on a landscape-level using the Human Appropriation of Net Primary Production (HANPP) framework. Implementation of agroforestry was thereby assumed on the total of available agricultural land, thus representing a maximum potential scenario. Results contribute to the broader discourse on sustainable land use policy by providing a reference frame for the assessment of agroforestry impacts on the socio-ecological system. The methodology was based on a landscape-level modelling approach integrating output data from two distinct land use models. Thereby, plot-scale productivity simulations (Yield-SAFE model) were aggregated to the landscape-scale according to simulated land use distributions in the study region (SECLAND model). The accounting framework HANPP was then used to quantify the carbon flows and stocks of the two scenarios.

Results show that the transition to agroforestry profoundly alters the carbon dynamics in the agroecosystem. Due to tree growth, a large fraction of the actual net primary productivity (NPP) remains in the ecosystem, representing a high climate change mitigation potential. The carbon carrying capacity (CCC) in the agroforestry scenario can reach 67.5 t C $ha^{-1}$, with an average accumulation rate of 1.1 t C $ha^{-1}$ $yr^{-1}$ over the period of 61 years (and a mean tree age of 30.5 years). The CCC corresponds to roughly two thirds of the actual carbon stock per hectare in an Austrian mixed forest. This is a substantial amount which leads to a total potential carbon pool of 1.46 Mt C in the study region. Biomass harvest, however, suffers a steep decline. Combined annual crop and grass yields decrease by almost 70% between 2020–2050. Although increasing fruit yields offset almost half of this loss and even surmount crop and grass yields by 2050, a 40% decrease of total biomass harvest is still severe. Total biomass production (measured as the actual NPP), however, is higher in agroforestry than in agriculture, and even higher than the potential NPP in the study region. All of these effects together result in a drastic reduction of HANPP from above 70% to 20–30% of potential NPP. This development denotes a strong reduction of anthropogenic pressure on the ecosystem in the study region, although the system-level land use intensity is higher when considering the increase in actual NPP.

Although agroforestry systems have the ability to ensure and enhance the supply of various other ecosystem services besides climate change mitigation and food provision, the trade-offs especially between these two compartments need to be carefully evaluated. Considering the current degree of self-sufficiency and land utilization in Austria as well as projected climate change impacts on agricultural productivity, a drastic reduction of arable yields does not appear advisable and could potentially compromise food security (in particular under special circumstances such as the Covid-19 pandemic). Austria's dependency on global biomass imports would subsequently grow, too. An increase of domestic fruit production, on the other hand, would be beneficial to Austrian self-sufficiency in this food category. Furthermore, Austria's land-based climate strategy in part relies on increasing carbon stocks in vegetation and soils while simultaneously enhancing biofuel production from forests and arable land. This should furthermore be achieved while reducing greenhouse gas emissions from the agriculture and forestry sectors. These competing requirements demand high performance of all individual components of a sustainable land system.

The discussion of results tends to argue in favor of integrated multi-functional agroecosystems because they harness the benefits of complementarity of resource use and provide a much wider range of ecosystem services. As such, the agroforestry scenario achieves a higher overall productivity by combining carbon sequestration and biomass harvest, while at the same time potentially restoring or enhancing additional ecosystem services and resilience. Future research should concentrate on designing and implementing agroforestry systems that manage to balance provisioning of various different ecosystem services within a region's larger socio-ecological dynamics.

From a farmer's perspective, implementation of agroforestry is subject to a variety of socio-economic barriers such as high initial investments, increased labor requirements and complex knowledge concerning design and management of the systems; barriers that could well be reduced by adequate policy support and learning networks. An estimation of the gross production value of crop and fruit yields even suggests that agroforestry has the potential to economically outperform agriculture, while new challenges for the farmers might arise from the development of niche markets and appropriate economical structures. Additional monetary value could potentially be created by introducing land-based carbon stocks into the European carbon trading scheme.

Limitations of this study primarily pertain to the case-specific calibration of the agroforestry model, leading to deviations of simulated productivity from statistical and experimental data. Apart from custom model calibration, future work could expand the model established in this study to include different tree species and modelling sites, a larger variety of system designs and management practices as well as additional environmental impact categories to better balance the trade-offs involved. This could, for example, lead to the identification of a *productivity gradient* from low tree densities on highly productive lands to high tree densities on low quality and marginal lands. In addition, forest land should also be included in the model to enable a comprehensive assessment of the trade-offs between multi-functional agroforestry and segregated agriculture and forestry systems.

Despite inaccuracies arising from above-mentioned limitations, this study can be viewed as a reference point for the magnitude of change to be expected from a transition to agroforestry practice on a landscape-scale. The potential carbon stock per hectare in the agroforestry scenario thereby corresponds to roughly two thirds of the actual carbon stock per hectare in an Austrian mixed forest, while biomass harvest decreases by 40–70% according to individual yield categories. In conclusion, this study constitutes a valuable contribution to the ongoing discussion on sustainable land use strategies and a socio-ecological transition.

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