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LAND USE CHANGE IMPACTS ON ECOSYSTEM SERVICES AVAILABILITY IN CZECHIA

Ph.D. Thesis

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I declare that this dissertation thesis has been composed by myself and where others' ideas or words have been included, I have adequately referenced the original sources. I agree with the thesis lending. No part of my thesis has been accepted or is currently being submitted for the same or any other academic degree.

In Prague, 9th May 2014 Jana Frélichová

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List of Abbreviations

C	Carbon
CICES	The Common International Classification of Ecosystem Services
CLES	A Consolidated Layer of Ecosystems of the Czech Republic
CO_2	Carbon dioxide
CSO	Czech Statistical Office
ELC	The European Landscape Convention
ESP	Ecosystem Services Partnership
ESVD	The Ecosystem Service Value Database
EU	The European Union
EVRI	The Environmental Valuation Reference Inventory
FAOSTAT	The Food and Agriculture Organization Corporate Statistical Database
GDP	Gross Domestic Product
H	Hypothesis
HFA	Hyogo Framework for Action
IPBES	Intergovernmental Platform on Biodiversity and Ecosystem Services
LUCC	Land Use/Cover Changes
MA	The Millenium Ecosystem Assessment
MAES	Mapping and Assessment of Ecosystems and their Services
NGO	A non-governmental organization
OECD	The Organisation for Economic Co-operation and Development
RQ	Research question
SHDI	Shannon's Diversity Index
TEEB	The Economics of Ecosystems and Biodiversity
UK NEA	The UK National Ecosystem Assessment
UN	The United Nations
NASA	The National Aeronautics and Space Administration
ESA	European Space Agency
EEA	Europen Environmental Agency
FAO	The Food and Agriculture Organization of the United Nations
GIS	Geographic Information System

Abstract

The main aim of this thesis is to analyse the impact of land use and land use changes on the provision of ecosystem services in Czechia. While land use analysis is well known approach already, the concept of ecosystem services has gained intensive scientific attention relatively recently. Consistently with the research field of geography, the concept of ecosystem services enables analysis of interactions between environment and human society from the anthropocentric point of view. The ecosystem services are in short the benefits which people obtain from natural environment and which directly or indirectly influence human well-being. Their provision is influenced by several factors, but this thesis specifically focuses on the impacts of land use change.

The thesis is delineated by six research questions, which divided its content into Theoretical Part and Analytical Part. The Theoretical Part provides background information regarding applied concepts. Except that, one chapter is dedicated to an explanation how to combine land use and ecosystem services analysis into one methodological framework. The Analytical Part of the thesis contains three case studies. All of them are situated in Czechia, however spatial scale and time scale were altered.

The first case study performs integrated assessment of ecosystem services in Czechia at national level. To estimate the total value of Czech ecosystems geographically-specific database of ecosystem service values (EKOSERV) was developed. The structure of the assessment is given by six ecosystem types and 17 ecosystem services delivered from these ecosystems. Specific literature review strategy was conducted to fill the database with biophysical and economic values of ecosystem services. Developed database consists of more than 190 values of ecosystem services, approximately half of them has been used for a benefit transfer to calculate total ecosystem values. The resulting average value of ecosystem services in Czechia represents approximately 1.5 the current national GDP (gross domestic product).

The aim of the second study was to provide spatially explicit information at a national level on land use change impacts in order to assess changes in the provision of selected ecosystem services (carbon sequestration, food production and soil erosion) in the agricultural sector of Czechia. This assessment shows that, historical land use trends (since 1948) lead to a significant decrease of arable land in the border fringes of Czechia, which is to some extent replaced by grasslands, in turn affecting the provision of ecosystem services.

The third case study studied availability of ecosystem services in the region of Cezava, Czechia since 1845 to 2000. The methodology again combines the ecosystem services analysis with an analysis of long-term land use changes. A comparison of service-provision over the centuries reveals that regulation and cultural services were significantly reduced, while provisioning services increased, due to the proliferation of arable land, land consolidation and agricultural intensification.

Despite that several uncertainties have been acknowledged during the research, the assessments provided innovative insights into the impact of long-term land use on ecosystem services in Czechia. The methodology may be used as a guideline for a long-term assessment of delivery of ecosystem services when the data for this kind of analysis are limited. As it has been shown, such an assessment clarifies the effects of land use on the environment, identifies the significance of particular services, indicates their importance for natural processes, and can potentially help in the assessments of the costs related to the loss of such services. This research also demonstrates that it is possible to analyse long-term land use trends to generate more meaningful, spatially explicit information. The LUCC Czechia UK Prague database has been a valuable resource for this analysis. Results of the spatial (and temporal) analysis of the changes can be used as a support tool for local land use management, or considered on the national scale for informing evidence-led policy decisions.

Key words: landscape, ecosystems, land use changes, ecosystem services, Cezava, Czechia

Abstrakt

Hlavním cílem této dizertační práce byla analýza dlouhodobých změn ve využití krajiny a jejich vlivu na poskytování ekosystémových služeb v Česku. Zatímco výzkum využití ploch je již dobře zavedený, koncept ekosystémových služeb začal významnější pozornost vědců získávat relativně nedávno. Podobně jako v geografii, i tento přístup se zaměřuje na vzájemné vztahy mezi společností a životním prostředím. Ekosystémové služby představují přínosy, které lidé získávají od přírody a přímo, nebo nepřímo tak ovlivňují lidský blahobyt. Dostupnost ekosystémových služeb ovlivňuje řada faktorů, avšak tato práce se zaměřuje na vliv změn využití krajiny.

Obsah a strukturu práce předurčilo šest výzkumných otázek, které byly v jejím úvodu položeny a dělí ji na část teoretickou a část analytickou. Teoretická část je souhrnem dostupných znalostí o uplatňovaných přístupech. Jedna z kapitol navíc popisuje metodologický postup kombinace sledování změn využití krajiny s analýzou ekosystémových služeb. Obsahem analytické části práce jsou tři české případové studie, lišící se zohledněnou úrovní (regionální či národní) a časovým měřítkem.

První případová studie byla zaměřena na národní integrované hodnocení ekosystémových služeb v Česku. Pro účely zhodnocení celkové hodnoty ekosystémů byla sestavena prostorově specifická databáze hodnot ekosystémových služeb (EKOSERV). Struktura databáze je dána šesti typy ekosystémů a 17 ekosystémovými službami poskytovanými těmito ekosystémy. Databáze byla naplněna biofyzikálními a ekonomickými hodnotami ekosystémových služeb na základě literární rešerše. Celkem obsahuje 190 hodnot, a přibližně polovina z nich byla využitelná pro přenos hodnot a následné vypočtení celkové hodnoty ekosystémů. Výsledná průměrná hodnota ekosystémových služeb v Česku představuje zhruba 1,5 násobek hrubého domácího produktu (HDP).

Cílem druhé studie bylo poskytnout na národní úrovni prostorově specifickou informaci o změnách ve využití zemědělské půdy a jejich dopadu na dostupnost tří vybraných ekosystémových služeb (ukládání uhlíku, produkce potravin a regulace eroze). Studie ukázala, že historické změny ve využití území (od roku 1948) vedly k významnému poklesu zastoupení orné půdy v pohraničí Česka, která byla do jisté míry nahrazena trvalými travinnými porosty, čímž byla ovlivněna dodávka zkoumaných ekosystémových služeb.

Třetí případová studie se zabývala dostupností ekosystémových služeb v zemědělsky využívaném regionu Cezava mezi lety 1845 až 2010. Srovnání dodávky ekosystémových služeb během let poukázalo na významné snížení regulačních a kulturních služeb. Produkční služby byly naproti tomu posíleny, a to zejména v důsledku nárůstu orné půdy, konsolidace půdy a intenzifikace zemědělství.

I přesto, že se výzkum potýkal s řadou nejistot, přinesl inovativní poznatky související s vlivem využití území na ekosystémové služby v Česku. Lze shrnout, že tento postup může být uplatněn i v případech omezené dostupnosti vstupních dat. Jak bylo ukázáno, tento způsob hodnocení pomáhá objasnit dopad změn ve využití krajiny na životní prostředí. V případě této práce byla velice významným zdrojem dat Databáze LUCC Czechia UK Prague. Dále výsledky ilustrují význam jednotlivých ekosystémových služeb a jejich roli pro fungování přírodních procesů, navíc mohou přispět k vyčíslení nákladů spojených se zánikem určité služby. Výzkum také potvrdil, že prostorově přesné výsledky analýzy dlouhodobých změn ve využití území je v kombinaci s poznatky o jejich vlivu na ekosystémové služby možné využít jako podklady pro místní a národní hospodaření s krajinou a s ním související rozhodovací procesy.

Klíčová slova: krajina, ekosystémy, změny ve využití ploch/území, ekosystémové služby, Cezava, Česko

1. Introduction

In global terms, increasing human pressure on landscapes often leads to degradation of environmental conditions, while people are dependent on an ever-increasing share of the biosphere's resources (Foley et al., 2005). Desired landscape multi-functionality is created by heterogeneity of landscape components along with simultaneous support for habitat, regulatory, productivity, and socio-economic functions, contrary to this, landscape functions are continually simplified (Mander et al., 2007). Therefore, it is important to seek a balance among competing interests when deciding how to best manage and allocate natural resources (Wainger et al., 2010).

One of the determining factors, affecting all ecosystem types, introducing a bundle of trade-offs and increasing human-environment systems vulnerability is the change of land cover/land use, especially land transformation for agricultural purposes (Lambin et al., 2001, Li 2007). For this reason, land use is focused in and analysed this thesis.

From a disciplinary perspective, land change has been a focus of scientific attention since the 1950s. However, a multifunctional system such as a landscape necessitates a holistic approach accounting for its integrity, connectedness and complexity (Palang et al., 2000, Verburg et al., 2009a). For this purpose, an integrated (environmental) approach, which is an interdisciplinary and policy-oriented synthesis of scientific information, is applied (Toth and Hizsnyik, 1998). Although multidisciplinary collaboration in landscape research has become more frequent recently, integrating information across disciplines continues to present challenges (Frank et al., 2011, Metzger et al., 2008, MA 2005).

This thesis therefore applies the methodological framework that combines land use analysis with the concept of ecosystem services. Ecosystem services represent the benefits of natural capital provided by landscapes (De Groot, 2006). Over the last two decades this new way of framing the relationships of biodiversity, ecosystem services and human well-being has gained more attention and has been spread through a variety of scientific disciplines and in the decision-making sphere (Lamarque et al., 2011).

The integrated framework applied in this thesis assesses the states and changes of human-environmental systems through qualitative and quantitative measures (indicators). Additionally to the employment of combined measures, the estimations are made on diverse spatial and time scales. The results describe positive outcomes for society and their changes in relation to land use. They could potentially contribute to the incorporation of scientific knowledge into political and corporate discourse.

1.1. Purpose of the Thesis

This thesis focuses on the discussion of the role of ecosystem services in respect of functioning of socio-ecological systems. The main aim of the study is to analyze the impact of land use and land use changes on the landscape by identification and quantification of ways in which ecosystems and services support and sustain the quality of human life.

In order to achieve these aims, the thesis builds on three **hypotheses** (H) that lead to the formulation of six **research questions** (RQ). The first hypothesis assumes that it is possible to quantify and value ecosystem services in terms of biophysical and economic values (H1). Also land use changes can be quantified, but the tradition of their quantification is already well established, contrary to developing science around ecosystem services. The second hypothesis considers the provision of ecosystem services to be determined by biophysical conditions and the form of land management (H2). In addition, the level of the ecosystem services provision changes over time and scale (H3).

RQ set a guideline for the analysis and can be devided into two types: theoretical and analytical. Answers to theoretical research questions should provide a concise overview of the theoretical basis for the concept of ecosystem services and its integration with the analysis of long-term land use changes. Out of the two approaches, more detailed attention is paid to the concept of ecosystem services as it still represents new and less described topic contrary to land use research. Following theoretically oriented research questions were asked:

RQ 1. What is the state of the art in land use research?

RQ 2. What is the state of the art in ecosystem services research?

RQ 3. How to combine land use analysis with the assessment of ecosystem services into a methodological framework?

RQ 1 and RQ 2, rather extensively formulated, aim to describe theories, paradigms and current knowledge in land use and ecosystem services research and their role in respect to functioning of socio-ecological systems. RQ 3 focuses on the combination of the two approaches.

The theoretical framework is applied in the analytical part of the thesis. This should examine the framework‘s applicability in practice and ability to generate orginal findings and to further develop existing traditional research practices. The main analytical research questions asked were:

RQ 4. Which ecosystem services are provided by Czech ecosystems and what is their value?

RQ 5. How have the changes in land use influenced the delivery of ecosystem services from agricultural ecosystems in Czechia in the period 1948 – 2010?

To further develop findings resulting from a national level, a regional case study is added with the aim to reflect on national trends in land use change and ecosystem services delivery:

RQ 6. How have the changes in land use influenced the delivery of ecosystem services in the Cezava region in the period 1845 – 2010?

The questions RQ 4 – RQ 6 should help to understand how the ecosystems in Czechia are qualitatively and spatially influenced by different forms of utilization. In these terms, the thesis gives three examples of application of the integrated approach in the system analysis and describes which impacts might result from changing the landscape function. Last but not least, the goal is to introduce and link the concept of ecosystem services to/with (Czech) geography and vice versa.

1.2. Outline of the Thesis

This thesis is divided into four main sections – Introduction, Theoretical Part, Analytical Part and Summarising Discussion and Conclusions. The Theoretical Part is further subdivided into three thematic blocks related to a) landscape and land use change, b) ecosystem services and c) involvement of the two concepts in this thesis.

The Analytical Part of the thesis contains three case studies. The first case study assesses ecosystem services currently (in 2012) available in Czechia. The other two case studies, at either national or regional level, provide spatially explicit information on land use change impacts in order to assess the change in the long term the provision of ecosystem services. The national case study on agricultural ecosystems changes examines three ecosystem services (carbon sequestration, soil erosion and food provision) in the period from 1948 to 2010. The regional case study in Cezava looks at availability of seven ecosystem services at regional level from a historic perspective and describes its development from 1845 to 2010. Each case study specifically follows the structure of introduction, methodology, results and discussion. The last section of the thesis, Summarising Discussion and Conclusions, provides a synthesis of general patterns and uncertainties that have been experienced based on the case studies introduced, and concluding remarks.

I. Theoretical Part of the Thesis

2. Landscape and Land Use Research

2.1. Landscape

This thesis attempts to approach the landscape from an integrated perspective based on two disciplines of which the landscape is essential subject of the research: geography and landscape ecology. This perspective is further developed by an application of the concept of ecosystem services, which helps to integrate both disciplines as it combines ecological principles with the focus on human-environmental relationships and dependencies.

2.1.1. Landscape as a Subject of the Research

Firstly, it is necessary to define the object of the subject of the research – the landscape. Of course, the definition is dependent on the expert and his field of study and on the decision context in which it is applied (compare e.g. Sauer, 1925, Forman and Godron, 1986, Daniels and Cosgrove, 1988, Lipský, 2000, Lambin et al., 2001). Apart from being the essential subject of the research in geography and landscape ecology, the landscape emerges also in the field of law, architecture, economy, history, art (Sklenička, 2003). Such a diversity of approaches, which deal with the landscape, along with the relatively long tradition of the landscape research, has led to the existence of numerous different landscape definitions. However, Balej (2012) argues that despite this variety, it is possible to recognise two main dimensions emerging among numerous definitions: material (physical, tangible) and immaterial (spiritual, perceptual).

Both these dimensions are included in the landscape definition introduced by the European Landscape Convention ("ELC"), which has been signed by 40 European countries (as of 31.12.2012). According to it, the landscape represents "*an area, as perceived by the people, whose character is the result of the action and interaction of natural and/or human factors.*" Moreover, the agreement declares the awareness that the landscape is a limited resource, natural and cultural heritage, and it influences human life quality. The Convention thus demonstrates the importance of the landscape by identification of its "[…] *important public interest role in the socio-cultural, ecological and environmental field, and* […] *economic activity* […] (ELC, 2000)." This definition of the landscape is suitable for the purposes of this thesis as it encompasses geographical and ecological perspectives as well as the perspective applied by the ecosystem services approach.

Geographers regard the landscape as a complex of relationships between the environment and human activities from local to global level, while landscape ecologists focus not merely on the human species, but more generally, on the links between communities of species and their demands on living conditions (Troll, 1939)[1]. The latter discipline builds on the European traditions of regional geography and botany. Thus, it combines the spatial (geographic) and functional (ecological) approaches. In the focus are the interactions between spatial

[1] It was Carl Troll, who introduced the term "landscape ecology" in 1939. One of the key reasons for the rapid development of landscape ecology was availability of aerial photographs in the 1930s.

arrangement and ecological processes, or more precisely, the causes and consequences of spatial heterogeneity on different levels (Turner et al., 2001).

Both disciplines share the interest in dynamic changes in the landscape dependently on time and scale. Also, both disciplines show a dichotomy in the research approaches. When considering landscape ecology, the ecosystem approach, represented by the American and Italian schools, competes with the geosystem approach favoured by Central/East European schools (Balej, 2012). The ecosystem approach is based on the central role of the biosphere and interactions of individual ecosystems spatially, while interactions between abiotic components are regarded with scant attention. Following this approach, the Millennium Ecosystem Assessment ("MA, 2005") characterizes the landscape as a place for natural and semi-natural ecosystems from which people obtain goods and services enhancing human well-being. In contrary, geosystem, polycentric approach does not favour any geosphere (atmosphere, lithosphere, pedosphere, hydrosphere, biosphere) and deals with the relations between abiotic components of the landscape (Novotná, 2001).

In the case of geography, the dichotomy rests on the traditional division of the discipline into the physical and social (human) geography. In the field of physical geography the exploration of natural ingredients of the complex prevails, while the human geography predominantly investigates activities of human society (Castree et al., 2009). When considering the landscape as a subject of the research, three dominant approaches in contemporary geography can be recognised: landscape ecology, land use research, and cultural geography (Kučera, 2009). It has been already said that landscape ecology is one of the background disciplines for this thesis. Additionally, this thesis also builds on knowledge generated by land use research. Despite the divergence in geography, a recent trend is to advocate an interdisciplinary approach and to prevent the disintegration of the physical and social geography and other sciences.

For the purposes of this thesis, the landscape is considered as a socio-ecological system, in particular a group of ecosystems (ecological systems; Nátr, 2011), of which human society is an integral and critical component. According to Zonnenveld (1990), such a comprehensive consideration of the landscape allows for holistic examination, which is greatly emphasised in the modern landscape ecology and geography (Bergandi and Balandin, 1998). This is supported also by Nátr (2011) or Hampl (1998), who conclude on the basis of Bertalanffy's general system theory that the value or importance of a complex system is greater than that acquired by its component elements individually. The added value of system-level results from the strong interactions between the components of the complex feedback loops, the space-time variability and energy flows within the system. Correspondingly, the complex adaptive system theory (Holland, 1995) provides a similar basis. Systems are composed of agents, which in a physical, biological or social context generate feedback and forward loops among each other. Aggregation of these feedback and feedforward loops influences system behaviour, not predictable from observation of any single agent in the system (Ruhl et al., 2007).

Building on this theoretical basis and the research needs, this thesis looks at the landscape from a holistic, (eco)system-level perspective and tries to overcome the dichotomy of the two above-mentioned fields **by integration of the two particular concepts: long-term land use changes and ecosystem services**. Consequently, the methodology also develops an interdisciplinary approach to the problem analysis.

2.1.2. Landscape Capital

Due to diverse (eco)system-building components and various interactions and feedbacks, resulting systems (landscapes) differ in their capacities to provide ecosystem goods and services (Burkhard et al., 2009) and in the character of the capital, respectively. The three types of landscape capital can be reflected in accordance with the recognition of the ecological, socio-cultural and economic public interest roles of the landscape by ELC: natural, social and economic.

Natural capital is given by the physical environment, by the stocks of ecosystems present in the landscape that yield a flow of valuable ecosystem goods or services into the future (MA, 2005). This provision of goods or services for human needs and well-being is what makes ecology inevitably relevant to the economy and why ecologists analogise the ecosystem structure to capital (Ruhl et al., 2007).

Generally, the benefits provided to people by ecosystems present in the landscape are at the core paradigm of the concept of ecosystem services (MA, 2005). This approach integrates social and ecological subsystems into the one system and introduces a great potential for the natural capital recognition, an explanation of linkages and feedbacks across scales, and an assessment of and societal preference for some services (Robards et al., 2011). Even though the scientific recognition of the concept, which enables complex and holistic analysis of coupled socio-ecological systems, is rapidly increasing (see Chapter 3.2., Theoretical Part), the benefits and value recognition remains research challenge, desirable for understanding the implications of land management. This thesis should contribute to the related knowledge base.

Second type of the capital, *social capital,* is an asset through which people control their access to resources and other actors (Bebbington, 1999). By shared norms, values and understandings that facilitate co-operation within or among groups, it generates benefits for society over time. Additionally to social capital, human and cultural capitals are also recognized. Cultural capital of landscape forms a local culture and identity, while human capital, residing rather in individuals contrary to other two more public related types, represents knowledge and skills of people (OECD, 2001). In this thesis, social, human and cultural capitals are grouped into one type of capital, albeit others, including Bebbington (1999) and OECD (2001), consider these capitals separate categories. However, because all of them are directly related to humans, their activities and attitudes, for the purpose of this thesis one category is sufficient.

Third type of capital, *economic capital*, enables self-sufficiency of a landscape/region (Stöglehner et al., 2013). Economic development is conditioned by natural resources, which are present, and by people who manage them. When maximising economic capital, without the control of natural and social capitals, resources depletion may occur and result in unequal distribution of goods and services (Farina, 2003).

The types of the capitals and their management, respectively, can be linked to the sustainable development concept and its three componential domains – environmental, social and economic, usually recognised as the three pillars. These three dimensions facilitate the existence of trade-offs, which must be made, even though not all efforts to achieve sustainability can result in win-win solutions (Johnston et al., 2001).

2.1.3. Sustainable Landscapes

Sustainable systems have the capacity to adapt to changing circumstances and to survive in the long-term (Valdivia et al., 2010). Otherwise they become vulnerable due to limited or lost ability to get over the shocks. The degradation of the landscape represents a loss of capital assets, which strongly influences human cultures, knowledge systems, religions, and social interactions (Renaud et al. 2013, MA, 2005). The concept of ecosystem services aims to prevent the loss of ecosystem services and vulnerability of humans by the recognition of relationships and knowledge generation.

Several projects have been recently carried out to fulfil the objectives of the sustainably used landscapes and of the ELC, respectively. For example, MOUNTLAND project aimed to provide management and policy options that support society, including policymakers and ecosystem managers, to make choices in order to promote and improve sustainable development of mountain ecosystems (http://www.cces.ethz.ch/projects/sulu/MOUNTLAND). Another example, relevant to Czechia, is the Vital Landscapes project, focused on promoting and supporting the sustainable development of cultural landscapes in Central Europe (http://www.vital-landscapes.eu).

Apart from research projects or bottom-up initiatives oriented on sustainable land use in Czechia, sustainable land utilisation is formally supported by the legislative framework and the related policies, strategies and programs. Legislation, as an important pillar, underpins the relations among diverse groups interested in the landscape and contributes to the creation of guidelines for the landscape management and development. Two periods having the greatest impact on the new acts creation were the period at the turn of 1980s and 1990s (after the Velvet Revolution) and the period associated with accession to the EU in 2004. The years at the beginning of 1990s were typical for their environmentally friendly atmosphere, which supported the socio-political tendency to improve the unpleasant state of the environment in the country. The second period, correlating with joining the EU, created the need for transposing many European directives into national law.

National laws concern the landscape and landscape planning either indirectly, through nature/environment protection laws, or directly, through the landscape components management. The fundamental framework defining the landscape as part of the environment and its sustainable use is established by Nature and Landscape Conservation Act No. 114/1992 Coll., and Act No. 17/1992 Coll., on the Environment. Acts directly related to the landscape are focused on protection, improvement and utilization of landscape components like water, land cover, and agricultural land resources. The most important laws related to the landscape or its components are: Building Code No. 183/2006 Coll., Act No. 334/1992 Coll. on the Protection of Agricultural Land Resources, Act No. 100/2001 Coll. on the Evaluation of the Construction Impact on the Environment (amended by Act No. 93/2004 Coll.), the Water Act No. 254/2001 Coll., and the Forestry Act No. 289/1995 Coll.

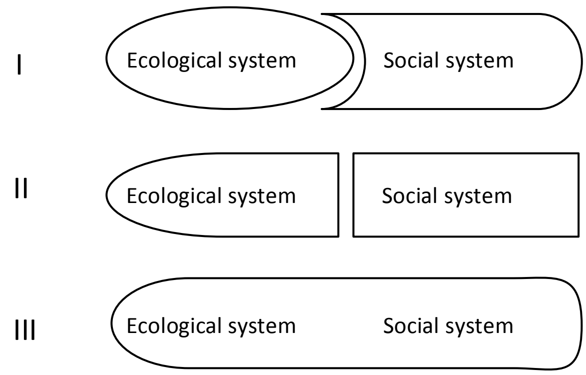
In terms of the landscape governance, several documents define the strategic framework for the implementation of the law and international conventions (CBD, ELC, Agenda 21): State Environmental Policy of the Czech Republic for the years 2012-2020 (SEP), the State Programme of Nature and Landscape (SPNL, 2009), National Biodiversity Strategy of the Czech Republic (NBS, 2005), Spatial Development Policy (2008), and Strategic Framework for the Sustainable Development of the Czech Republic (SFSD, 2010). NBS (2005) and SPNL (2009) directly require compliance with the obligations imposed by the ELC.

However, despite the legislative framework, concepts and strategies, landscape planning in Czechia is being criticised and evaluated as insufficient (Boucníková et al., 2006, STUŽ, 2014). For example, limited awareness and poor practice in information and experiences exchange resulted in the realization of the project ”Consequences and Risks of Breaking the European Landscape Convention“. The outputs of the project are going to be available by the end of 2014 (http://www.bioinstitut.cz). Some already identified fundamental problems associated with the planning of the landscape are outdated or missing land planning documents in smaller municipalities, disunity in the ecological networks planning, poor accessibility of the plans, conflicts of interest in the planning processes or lack of coordination of activities between the competent ministries and other stakeholders (SOBR, 2005).

2.2. Land Use and Land Use Change

Human society is transforming its environment. **Land use** can be seen as the human modification of natural environment or wilderness into a built environment such as settlements; or agriculture. It is deliberately connected with the attribution of new functions to the landscape by society (Bičík et al., 2010a). **Land use change** is a complex, dynamic process linking natural and human systems through their interactions. The character of the change reflects specific environmental and socio-economic conditions to a large extent (Bičík et al., 2010a). However, human systems usually change more dynamically contrary to natural systems and create more complex hierarchical organisations (Hampl, 1998).

Reciprocal linkages between the human (social) system and the rest of the ecosystem create specific types of interactions, developing in time. So far, three main periods or types of human-nature interactions have been identified (Hampl, 2002, Balej, 2007) (Figure 1).



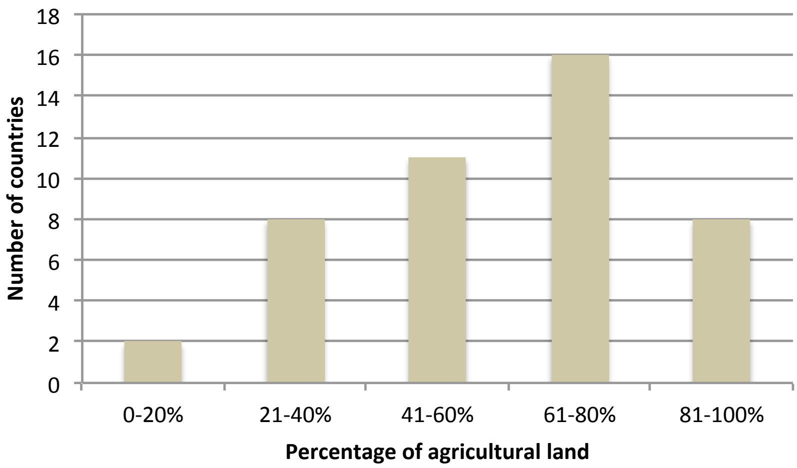
Source: Adapted from Balej et al., 2007

Figure 1: Changing interactions between ecological and social systems in periods of pre-industrial (I), industrial (II) and post-industrial (III) development.

While in the pre-industrial period natural conditions have primarily determined the way in which people utilized the environment, since the industrial revolution the competitive influence of social impacts has prevailed. Society has reshaped the character, organisation and function of the landscape (Hampl, 2002). This happened more than 200 years ago, when the development of the fossil fuels usage accelerated the scope and pace of Earth's surface changes by human activities (Geist and Lambin, 2006). Today, in the post-industrial period, an attempt to take into account the social consequences of the impacts on nature is apparent. A tendency towards a balanced relationship between the social needs and the preservation or restoration of the natural environment has emerged (Bičík et al., 2010b). This trend, however, is more evident in developed countries (Hampl, 1998).

In their reflection on human-nature interactions and consequential land use change, Lambin and Geist (2006) identified an agricultural activity of humans as the main cause of land use change, which has already lead to extensive transformation of about one third of the global surface. According to Revelle (1984), almost 852 million hectares of diverse natural ecosystem types turned into arable land between 1860 and 1978, which corresponds roughly to 6% of the Earth's land. The changes are nowadays localised mostly in developing regions of Africa, Asia and Latin America, although initially they took place in developed countries (Turner et al., 1990). This indicates the shifts of the change dependent on the spatial and temporal scale and economic development of a particular area. More recently, another project - BIOME 300, monitored changes in the shares in agricultural land in the last 300 years. The results of the project show that area of arable land increased five times between 1700 and 1990 (Lambin and Geist, 2006).

When considering solely the region of Europe, recent FAOSTAT data (2014) show that agricultural land represents dominant or considerable land use category in about 50% of European countries (out of 45), including Czechia as one of them (Figure 2). In total, agricultural land covered 21% of Europe in 2011 (FAOSTAT, 2014).



Source: Compiled based on FAOSTAT (2014)

Figure 2: Frequency distribution of agricultural land percentage for 45 European countries in 2011

Increasing demand for agricultural products (food and biofuel) triggers agricultural land expansion at the expense of other ecosystems, such as forests. On the other hand, total deforestation rate has decreased at a global level over the last decade (GEO 5, 2012). However, reforestation has regional character and occurs mainly in the zones of boreal forests (Russia, Canada, Alaska), tropical forests in some parts of Amazonia, the Congo Basin, and South East Asia, and also in the temperate zones of Europe, USA and Asia (GEO 5, 2012). The increased rate of afforestation may be caused by leaving the barren soil fallow or by the relocation of deforestation to other regions (Meyfroidt et al., 2010). This seems to be in accordance with the forest transition theory (Mathers et al., 1999): now-developed countries (including Czechia) face net reforestations, while net deforestation remains to be an issue in developing countries. Despite the increased area of global forests and overall decrease in their reduction from 16 million hectares per year in 1990 to 13 million hectares per year in 2010 the deforestation rate remains high. Also, it is necessary to consider the character of newly planted forests. Plantations or intensively managed forest represent a relatively large portion (7%), therefore some natural functions such as biodiversity conservation are significantly limited (GEO 5, 2012). Apart from forests, other types of natural ecosystems such as savannahs, steppes and wetlands decrease in size. In addition to agriculture, the key causes of their reduction is population growth, infrastructure development, urbanization and climate change (GEO 5, 2012).

2.2.1. Consequences of Land Use Change

Land conversion to human utilisation introduces the risk of undermining human well-being and long-term sustainability (Rockström et al, 2009). Particularly, it is considered to be **one of the drivers[2] of global environmental change** (Shao et al., 2005).

Transformation of ecosystems into other land use categories, primarily the conversion of various vegetation covers to agricultural land and urban areas, impacts water flows and the biogeochemical cycle, and is closely linked to climate change (Milad et al., 2011, Schulp et al., 2008). The joint effects of land use and climate change are perceived as the most important driver of biodiversity loss (Sala et al., 2000). Because biodiversity is known to represent a key prerequisite for the functioning of an ecosystem and delivery of bundles of ecosystem services (MA 2005, De Groot et al., 2010), land use change may undermine regulatory capacities of the ecosystems, e.g. in terms of the ability to avoid and minimise hazards (Rockström et al, 2009, Preston et al. 2011). A number of risks initiated by land use change or its consequences originate in diminished land productivity, land degradation, disruption of water regime, water contamination, or extra losses of biodiversity (Shao et al., 2005).

On the other hand, change in land use patterns, in terms of desirable functional changes of physical features of the landscape or componential ecosystems, can be applied as an adaptation option to most of the risks listed above. For example, disruption of water regime resulting in flood risk can be reduced by conservation and restoration of forests that would stabilise land slopes and regulate water flows or through the reinforcement of water storage capacity of upland wetlands and floodplains by sustainable management (Opdam et al., 2009, Langhammer, 2009, Sandhu and Wratten, 2013). These ecosystem approaches to disaster risk reduction and adaptation are being strongly argued recently (Uy and Shaw, 2013). Ecosystem based adaptations are considered to be cost effective, more accessible, and able to integrate biodiversity (and ecosystem services) into an overall adaptation strategy of global change (Sandhu and Wratten, 2013). By this, ecosystems can contribute to hazard mitigation and vulnerability reduction as well as to climate change adaptation (HFA, 2005).

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[2] DRIVER or *driving force*, as one of the components of the DPSIR conceptual framework for systems analysis, describes social, demographic and economic developments in societies and the corresponding changes in life styles. In relation to global environmental change, it is primarily the need for food, water or shelter which initiate land transformation, additionally it might be also the need for health, security or culture (EEA Technical Report No. 25, 1999, Jeleček, 2002).

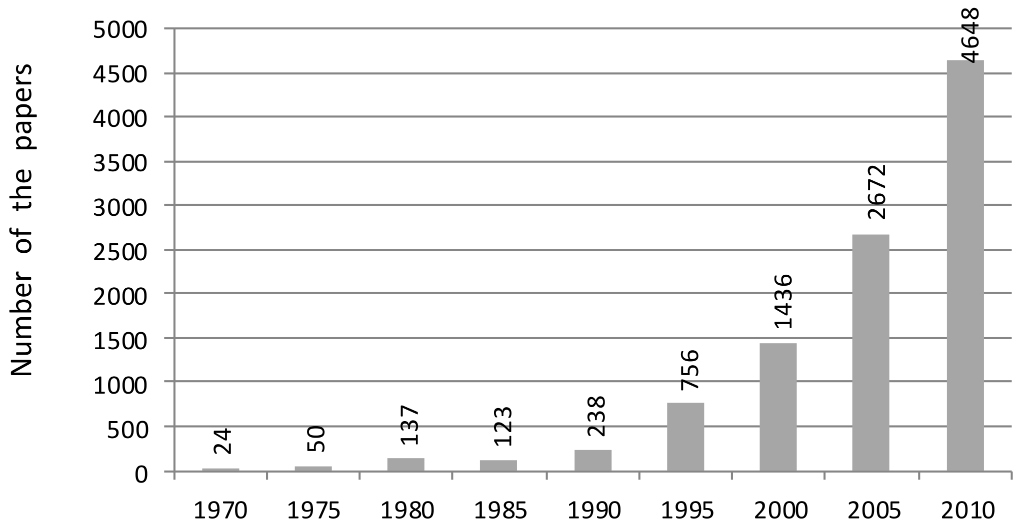
Ecosystem based adaptations however require the application of complex knowledge and integrated measures to secure well-functioning ecosystems, which might be a challenge. As Langhammer and Vilímek (2008) point out on the example of flood course, the direct impact of change in land use patterns is rather complicated and depends on many factors. Another kind of substantial constraint is that the significant role of ecosystems in hazard minimisation and vulnerability reduction has not yet been fully appreciated as a powerful tool in disaster risk management by planning authorities (Renaud et al., 2013). In this context, the concept of ecosystem services has the capacity to facilitate the process.

2.2.2. Land Use Research

Land use research was initially (by the end of the 19th century) a subject of interest in agricultural geography, which aimed to describe and explain the distribution of agricultural activities around the world. In the first half of the 20th century, under the influence of the French geographer de la Blache and the French School, the research started to be oriented towards a spatial arrangement of agricultural activities. In the post-war period, a development of typology of agriculture began to be the main aim (Kostrowicki, 1974). From the second half of the 20th century, international scientific interest started to be concerned with land use on its own. The term “land use”, meaning functional classification of an area into categories derived from the way of use, was coined by the British geographer Stamp (1945). Its application has been associated with Stamp’s presidency in the International Geographical Union (IGU) later in 1948. Since then the meaning of the term has expanded and it encompasses additionally landscape management, problems related to the change of landscape functions, regional development, and environmental conservation (Johnston et al., 2001). In the Czechia, Häufler was among the first who reflected on this issue at the beginning of the 1950s, particularly by the land use change research in mountain areas (Bičík et al., 2012). Other acknowledged experts in the field of land use science are e.g.Turner II (1994), Robinson, Douglas, Huggett eds. (1996), Atkins, Simmons, Roberts (1998), Mather et al. (1999), Gabrovec et al. (2001), Milanova et al., (2001), Himyiama et al. (2005), Lambin and Geist (2006) and Aspinall (2008).

In the context of the scope of the thesis, one of the shifts in the interest of geography towards the environmental issues in the 1970s has been of considerable importance. Development of environmental geography was a reaction to the fact that the discipline was surprisingly unprepared to respond to, and analyse, the environmental effects of industrial society (Braun, 2009). Today, environmental geography is one of the specific branches of the discipline, which describes and explains the spatial aspects of interactions between humans and their natural environment. In addition to that, it links the physical and human parts of the discipline together (Castree et al., 2009). Techniques applied to understanding the impacts of human activities on the environment/ecosystems involve some of those used in land use change research. Also, given the fact that land use change research studies human-social and ecological-change processes that are geo-referenced and coded into the frameworks of spatial analysis (Zimmerer, 2009), these two geographical branches have importantly a similar footing.

With increasing awareness of the significant impacts of human activities on the environment, the interest in tracking the land use changes and in describing their drivers grew, including their impact on biodiversity, ecosystem services, health, and climate change. Figure 3 provides an evidence of increased attention to the topic by the growing number of papers on the use of land registered in the database Web of Knowledge over the last 20 years, especially since 1995. Scientometric analysis was performed on the records based on the key words "land use OR land-use“ in the topic of the article. Also, after 2010 the trend in increasing number of publications per year on this topic continues. This confirms findings of similar analysis done earlier by Balej (2012). Additionally to the volume of the scientific papers, the number of research projects on land use supported by government agencies and international organisations (such as NASA, ESA, EEA or FAO) increases as well.



Source: Compiled by the author in February 2013

Figure 3: Number of land use related papers in Web of Knowledge database

New findings are made thanks to a number of studies conducted in diverse case-study areas and on diverse scales, from local to global. Modernization of the techniques for remote sensing (RS) and geographic information systems (GIS) for analysing changes contributed to this considerably. The knowledge base is developed also on the existing models, designing the effects of changes in land use, such as CLUE, IMAGE (modelling of global environmental change), GEOMOD (land use conversion from forested to non-forest), and others.

2.2.3. Long-term Land Use Research in Czechia

Long-term land use research commands large attention in Czechia and is respected by a number of experts. A detailed overview of schools and approaches is given in dissertations by Štych (2007) and Rašín (2010).

One of the Czech schools deliberately focused on land use research is Albertov School in the Faculty of Science of Charles University in Prague. Contributors to the discussion of theoretical and methodological frameworks of the land use research include Bičík (1995), Bičík et al., (2001a), Kupková (2001), Jeleček (2002, 2007), Kabrda (2008) and others. The resulting findings of the Albertov School are of key importance for this thesis, particularly those of LUCC Czechia UK Prague Database.

From a landscape-ecological perspective, land utilisation and its changes are studied by a team in the Department of Physical Geography of Charles University in Prague, by Lipský (Lipský et al., 1999), Romportl, and Chuman (Romportl et al., 2008). Their main research interests lies in mapping of contemporary land use changes, classification of landscapes in Czechia, and changes in diversity and heterogeneity of the Czech landscape based on a set of landscape indicators. A team at Palacký University (e.g. Balej) pursues, apart from geoecological and landscape ecological theories an interdisciplinary approach to the impact assessments of environmental stressors on the landscape (Balej et al., 2007).

Another major institute focusing on landscape studies is the Silva Tarouca Research Institute for Landscape and Ornamental Gardening. Recently, the Institute took the lead in the preparation of Landscape Atlas of the Czech Republic (Hrnčiarová, T., Mackovčin, P., Zvara, I. eds., 2010), which represents valuable and inspirational map resource. Research topics within the scope of the institute are land use changes and landscape fragmentation based on old topographic maps and aerial pictures, analyses of landscape functions, develop indicators of cultural landscape biodiversity and analyse spatial data in geographical information systems. GIS technologies and remote sensing are applied and utilised also by Štych and Kupková (e.g. Štych, 2007) in the Department of Applied Geo-information and Cartography at Charles University in Prague, Kolejka and Svatoňová (e.g. Svatoňová and Lauermann, 2010) in the Department of Geography at Masaryk University, and Létal (Létal, 2004) in the Department of Geography at Palacký University in Olomouc.

LUCC Czechia UK Prague Database

The database was created as part of the research projects supported by the Czech Scienece Foundation (GAČR) whose results have been presented since 1997 in the activities of the International Commission IGU / LUCC (International Geographical Union - Commission on Land Use and Land Cover Change).

For the purpose of the database, the total area of Czechia is divided into 8 903 standardised units (comparable territorial units). Unit area is 8.86 km^2 on average. The database provides statistical information on changes in land use in six periods from 1845 to 2010 in the territory of Czechia encompassing 13,000 cadastre units in total. Cadastral data from 1948 was acquired from the Central Land Survey and Cadastre Archive files. More recent land use data (1990, 1999) came from the computerised database of the Central Czech Land Survey Office in Prague (Bičík et al., 2001b).

Land use categories within the database fall into eight basic categories, which can be associated with three aggregate categories – agricultural land, forests, and other areas (Table 1). For further details on the database development see e.g. monograph by Bičík et al. (2010b).

Table 1: Classification of land use categories in the LUCC Czechia UK Prague Database

Aggregate category	Basic category	Specification
Agricultural land	Arable land	
	Permanent cultures	Includes orchards, gardens, vineyards and hop gardens
	Meadows	Permanent grasslands
	Pastures	
Forests	Forests	
Other areas	Water bodies	Water courses and water bodies
	Urbanized areas	
	Remaining areas	

Source: Bičík et al. (2010b)

The database provided the data for the analysis of long-term changes covering the period 1845-2010 in this thesis.

3. Ecosystem Services

3.1. The Concept of Ecosystem Services

The concept of ecosystem services enables analysis of interactions between environment and human society from the anthropocentric point of view. The ecosystem services are in short the benefits which people obtain from natural environment and which directly or indirectly influence human well-being (MA, 2005). Given the problem of orientation on human-nature/land interrelations, the approach resonates with the research field of geography.

For example, Ruhl et al. (2007) declare that the natural capital and ecosystem services have been of interest to geographers throughout modern times. He refers to *Marshal's Man and Nature: or, Physical Geography as Modified by Human Action* from 1864. The study indicated the character and the extent of the changes induced by humans and their impacts on the surrounding world, and questioned the need for (pre)caution in all operations with respect to our dependency on nature (Ruhl et al., 2007). This strongly corresponds with the rationale behind the ecosystem services approach. Similarly, Sauer in his 1925 *The Morphology of the Landscape* regards the landscape as a product of co-evolution between nature and society, which can be interpreted as an indication of substantial influence of humans on land use patterns and generation of ecosystem services (Ruhl et al., 2007).

Overall interest of some geographers in the topic, either directly or indirectly, can be also associated with consideration of environmental questions by the discipline since the 1970s. In general, much more has been produced by geographers in relation to the concept of ecosystem services, even though they did not use the related terms yet, e.g. Diamond in 1997[3]; but he already applies the term in a later work from 2005[4] (Ruhl et al., 2007). Some leading geographers dealing with the ecosystem services are Peter Verburg (VU University Amsterdam), and Benjamin Burkhard and Felix Müller (University of Kiel). However, such a complex concept cannot be developed just by a single scientific discipline. In fact, experts from other fields, such as ecology, environmental science, agriculture and forestry, and economy, also pay attention to it and apply it.

3.1.1. Basic Terms Definition

A diversity of disciplines, operating with the concept of ecosystem services, introduces terminological variety. This complicates the application of the concept as well as interdisciplinary discussion not only among scientists, but also among landscape managers, politicians or corporations. Therefore, Lamarque et al. (2011) carried out a study that aims to facilitate the common discussion by summarisation of basic terms used so far and their

[3] Diamnod, J. (1997). *Guns, Germ and Steel: The Fates of Human Societies*. New York: W.W. Norton and Company

[4] Diamnod, J. (2005). *Collapse: How Societies Choose to Succeed or Fail*. New York: Viking Books.

interpretation in a broader context. An overview of terminology related to the concept of ecosystem services based on their application is given in Table 2.

Table 2: Summary of the terms related to the concept of ecosystem services

Terms	Meanings	Main aim of studies using the term
Ecosystem services/goods	The benefits provided by ecosystems or habitats.	Biodiversity conservation
Ecological services	a) Synonym of ecosystem services. b) Benefits provided by a particular species or community.	Biodiversity conservation
Landscape services	The benefits provided by a particular region/landscape. Account for spatial patterns, landscape elements and horizontal landscape processes.	Landscape multifunctionality
Environmental services	a) Synonym of ecosystem services in PES schemes[5]. b) Human-made services, which substitute ecosystem services. c) The services provided by the abiotic environment, e.g. the wind or water regimes used for generating electricity	Landscape multifunctionality

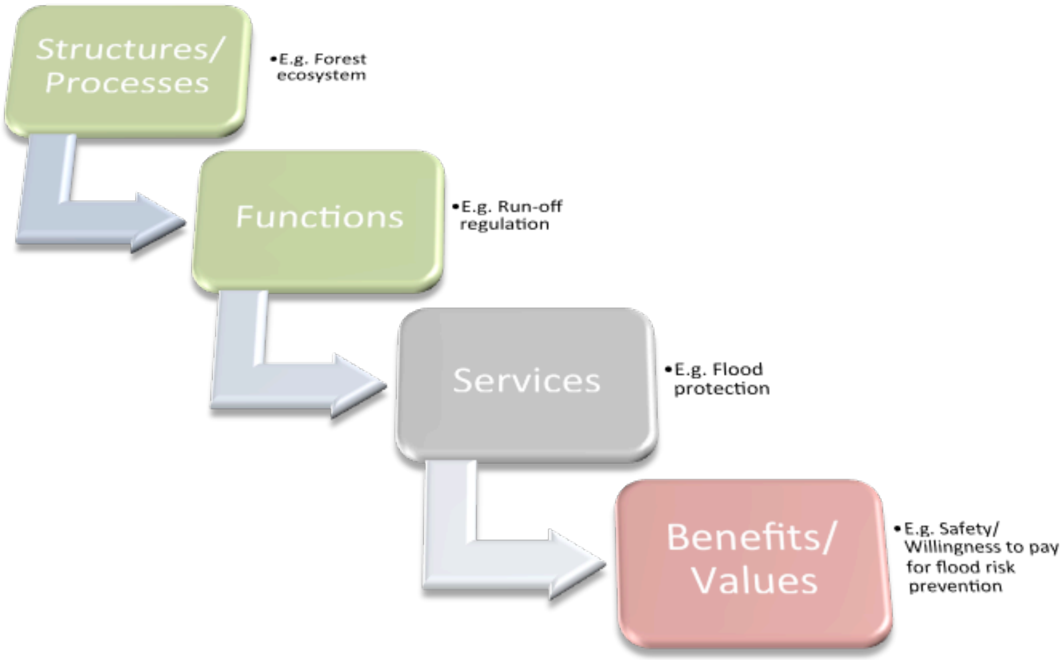
Source: Compiled based on Lamarque et al., 2011

For the purposes of this thesis, the basic and the most often used term "ecosystem services" is going to be used. In relation to other terms, ecosystem services are considered to be a synonym of "ecological/environmental services". The term "landscape services" would be applicable to the regional case study Cezava introduced in this thesis. However the term "ecosystem services" is used instead in this case to standardise the terminology used in this thesis. Another reason is that, as Lamarque et al. (2011) suggest, landscape services are more appropriate for studies directly oriented on landscape planning, which is mentioned only marginally in this study.

3.1.2. Ecosystem services cascade

The next important step for understanding of the concept of ecosystem services it the explanation of the framework, its components and their interactions. Figure 4, depicting a cascade is helpful. The cascade represents a simplified functional chain revealing the essence of logic developed around the ecosystem services paradigm. It shows interrelations between its components, from ecosystem structures and processes to the benefits supporting human well-being. Simultaneously, it declares the necessity of consideration of human society in relation to the concept, otherwise the concept would not be applicable (Haines-Young and Potschin, 2010).

5 PES = Payments for ecosystem services; stewards are paid by third party beneficiaries for an activity aimed at intentionally transforming or maintaining some useful characteristics of an ecosystem/landscape (Lamarque et al., 2011)



Source: Compiled based on Haines-Young a Potschin, 2010 and Lamarque et al., 2011

Figure 4: The ecosystem services cascade from structures/processes towards benefits/values

”Structures and processes“ are the first component of the cascade. They are given by the species composition of ecosystems and interactions between species and their environment. They create the conditions for the potential of ecosystems to provide services, particularly “functions” of the ecosystem (the second component of the cascade). De Groot (1992) defined ecosystem functions as “the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly”. Contrary to the services, the functions are ubiquitous (Ruhl et al., 2007). The third element in the cascade is “ecosystem services”. Given the diversity of ecosystem services being provided, they are grouped into four basic categories based on their functional characteristics (see next Chapter 3.1.3.). The last component of the cascade is “benefits and values”. Early studies on ecosystem services have not separated the services and the benefits (values), e.g. MA (2005), Daily (1997) or Costanza et al. (1997). It is only a recent attempt to consider services and values individually, or more precisely, benefits as a product of services. This is mainly due to the conduction of more precise economic valuations and double-counting errors prevention (Lamarque et al., 2011). Benefits and values communicate importance of the ecosystem services for human society. They can be direct benefits (e.g. drinking water) or indirect benefits (e.g. accumulation of ground water).

3.1.3. Classification of ecosystem services

Although ecosystem services are classified in several ways, three main approaches are applied most often. Out of them, classification introduced by the MA in 2005 is largely used. MA (2005) was conducted by more than 1,300 experts worldwide to assess the state of global ecosystems and consequences of ecosystem change for human well-being. The MA framework departs from the more traditional linear DPSIR framework by considering a dynamic system in which changes have a feedback effect on the acting pressures (Busch et al., 2012). It provides the scientific basis for action needed to enhance the conservation and sustainable use of ecosystems, while minimising their damage (MA, 2005). Apart from that, it systematically reflects on the implementation of the Convention of Biological Diversity (CBD), the Convention to Combat Desertification, the Convention on Wetlands, and the Convention on the Conservation of Migratory Species of Wild Animals.

MA (2005) fits ecosystem services into four categories: supporting services (e.g. primary production, nutrient cycle, water cycle), regulating services (air quality regulation, erosion regulation, disturbance regulation), provisioning services (food, fuel, medicine), and cultural services (recreation, cultural heritage, education). Supporting services are conditional for the availability of other three categories, hence their link to human well-being is indirect. Figure 5 shows the four categories, their interlinkages and the intensity of the linkages.

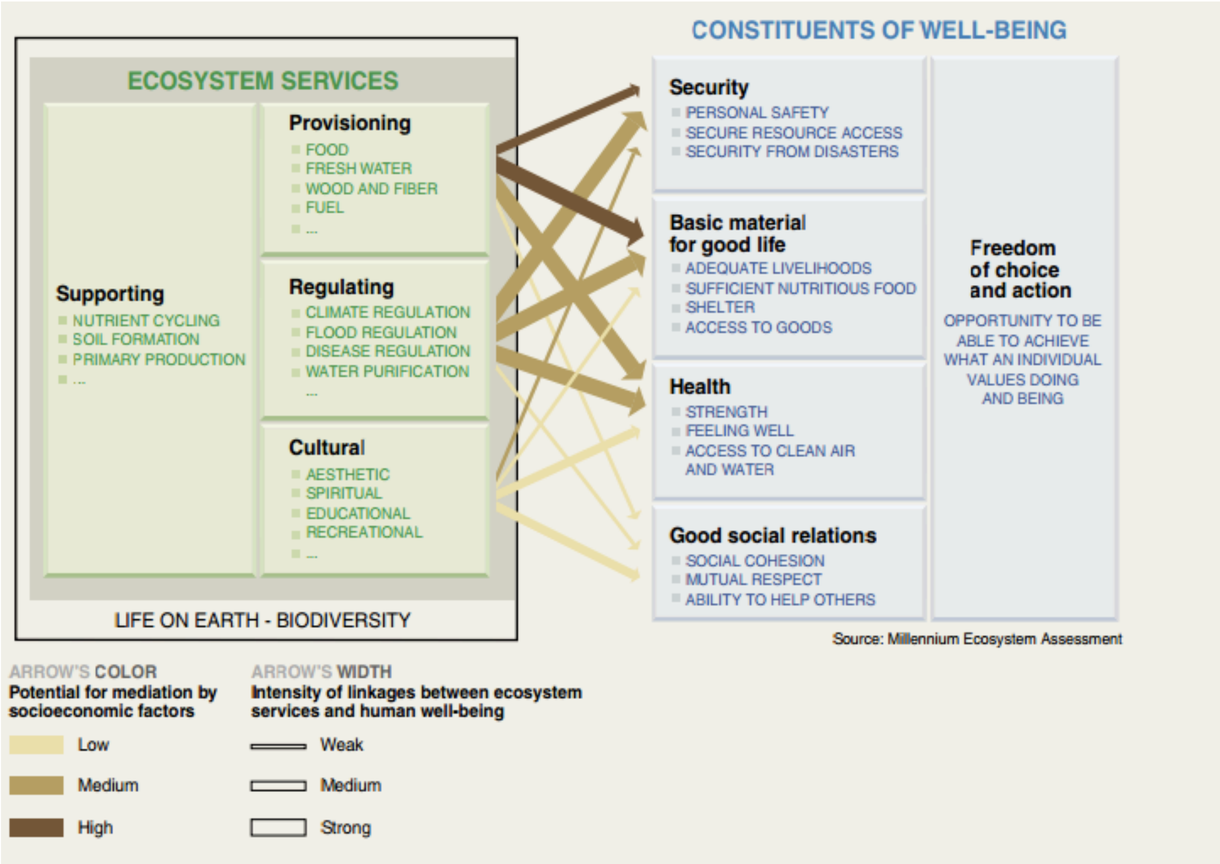


Figure 5: Classification of ecosystem services and linkages between ecosystem services and components of human well-being (taken from MA, 2005).

As Figure 5 shows, biodiversity is the key influential factor that affects the functioning of ecosystems because it increases their flexibility and resilience in the face of a change. Biodiversity (gene/species variation per hectare) secures availability of the services in term of food security, clean water and air availability, and contributes to livelihood, economic development and poverty reduction (MA, 2005, UK NEA, 2011). Reduction in biodiversity increases the possibility of reducing ecosystem services as more species are lost due to reductions in substitutability (Tilman et al., 2012). Contrary to this preconditioned role of biodiversity for ecosystem functioning and services delivery, Mace et al. (2012) consider biodiversity crucial at all levels of the ecosystem service hierarchy: as a regulator of underpinning ecosystem processes, as a final ecosystem service, and as a good that is subject to valuation. Clearly, this issue still needs additional research to better understand the role of the biodiversity for ecosystem functioning (UK NEA, 2011).

The leading methodological approach to ecosystem services assessments remains to be MA (2005), which has been applied also in follow-up studies, for example The Economics of Ecosystems and Biodiversity ("TEEB", 2010). However, the TEEB study alternates the categorisation of services. Supporting services category presented in MA (2005) has been replaced by habitat services category. Habitat services are meant to provide habitats to species and support biodiversity at the gene level (which maintains the possibility of natural selection). Another classification system has been recently introduced: The Common

International Classification of Ecosystem Services (CICES). The classification development was undertaken by the European Environment Agency and the last version available is from 2013. CICES is a multi-level classification on five hierarchical levels. Contrary to the MA (2005) and TEEB (2010), it aims to provide as general typology as possible at basic level, which further develops in much detail categories at lower levels. This should provide some flexibility to users in the application of the classification system according to the current needs and requirements (MAES, 2012). The differences between the three discussed ecosystem services classifications are shown in Table 3 using the example of provisioning services.

Table 3: Comparison between MA, TEEB and CICES classifications using the example of provisioning services

	MA category	**TEEB category**	**CICES group**	**CICES class**
Provisioning services	Food (fodder)	Food	Nutrition	Cultivated crops
				Reared animals and their outputs
				Wild plants, algae and their outputs
				Wild animals and their outputs
				Plants and algae from in-situ aquaculture
				Animals from in-situ aquaculture
	Fresh water	Water	Water	Surface water for drinking
				Ground water for drinking
	Fiber, timber	Raw material	Materials	Fibres and other materials from plants, algae and animals for direct use or processing
	Genetic resources	Genetic resources		Materials from plants, algae and animals for agricultural use
	Biochemicals	Medicinal resources		Genetic materials from all biota
				Biotic materials (Medicinal and cosmetic resources)
	Ornamental resources	Ornamental resources		Biotic materials (Ornamental resources)

Source: compiled based on MAES (2012) and CICES (2013)

Despite the differences, the three classifications are compatible (given their evolution as the later classifications are based on MA, 2005). Thus, with some effort, it is possible to convert one classification into another, if need be.

3.2 State of the Art of Ecosystem Services Research

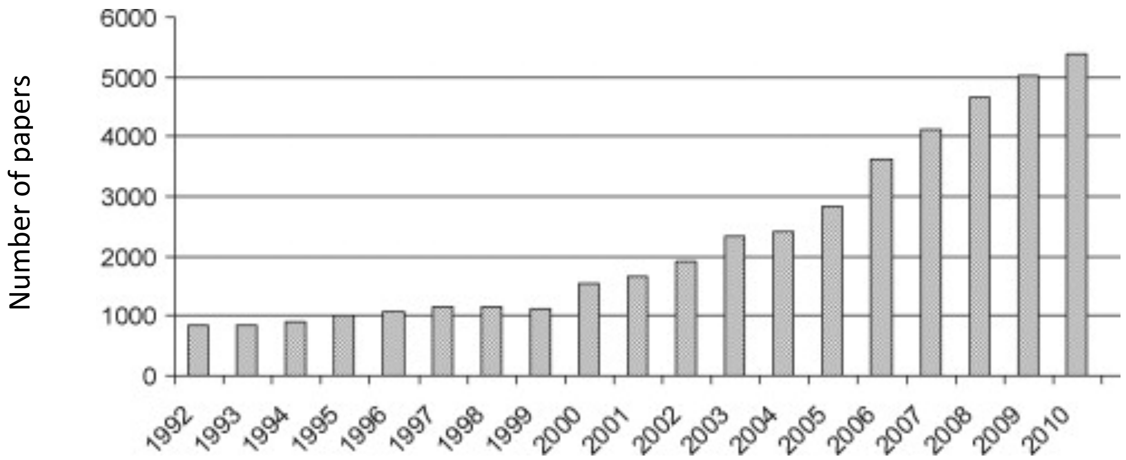
Ecosystem services started to attract scientific interest by the end of the 1960s and at the beginning of the 1970s (Mooney and Ehrlich, 1997, De Groot et al., 2010). It is no accident that the initialisation of the concern corresponds to the period of increasing global awareness about degradation of the environment and the finite nature of natural resources (Jeleček, 2007, Lambin and Geist, 2006). This concern piqued the interest in the analysis and evaluation of manifold benefits provided by the environment (Hein et al., 2006).

It was Walter Westman in 1977, who made the first attempt to assess the values of nature's services (Ruhl et al., 2007). The concept of ecosystem services in a modern sense, as a new way of framing the relationships between biodiversity, ecosystems and human well-being, was introduced during the 1990s (Lamarque et al., 2011). The first studies were those of Pearce and Turner (1990), De Groot (1992; 1994) and Pimentel et al. (1997). Two major publications about this issue were released in 1997.

First of them, Nature's Services: Societal Dependence on Natural Ecosystems by Daily (1997), viewed the ecosystem services mainly from the ecological perspective and discussed in theory economic value of the services provided by the environment. The second study, by Costanza et al. (1997), published in *Nature*, applied the theory as a practical exercise and estimated the global ecosystem services value in economic terms. The resulting value was estimated at more than USD 30 trillion. The estimate provided by this ambitious but controversial study grabbed a good deal of attention not only on the part of the scientists but the public media too, and invited appreciation as well as criticism.

The critics saw the estimate as an attempt to put a price on nature, which questions morals and ethics. The authors refuted this interpretation vigorously (Costanza et al., 1998). Other questions were raised by some mainstream economists such as Toman (1998) and Bockstael (2000). They entered into a critical discussion on the methodology applied in the study. According to them, value can be measured only in the context of a specific exchange, which is not really the case when it comes to most of the ecosystem services in the condition of an absent market. They also questioned the aggregate figure of USD 33 trillion. In response, the authors of the *Nature* paper admitted that there were many errors, underestimates and uncertainties and therefore they did not place much credibility on the figure (this was already communicated in the paper - Costanza et al., 1997, and again later in Costanza et al., 1998). Their main motivation was to synthesize the existing information in order to address a new and important question related to the value of ecosystems, and to stimulate additional research and debate (Costanza et al., 1998). Consequently, a need for closer communication between ecologists and economists was identified. This main purpose - to stimulate scientific and political debate about the value of ecosystem services being underestimated - has been considered useful even by the critics (Costanza et al, 1997, Toman, 1998, Bockstael, 2000). Since then, the value of ecosystems and their services has become an indisputable research objective, although their value and account for it still remains a challenge (Ruhl et al, 2007).

Since the late 1990s and the 2000s the development of the discipline has been remarkable. The first studies on ecosystem services had a theoretical character. It was necessary to define the paradigms, research questions and methodological frameworks and to generate the first results. The very first outcomes sparked political attention, which triggered an initialisation of assessments such as MA (2005), TEEB (2010), and the strategic documents (CBD Targets 2002, 2010). Consequently, the concept was transformed from a marginal one to a mainstream one, especially abroad. This trend supports the findings of the sociometric analysis done by Vo et al. (2012). It uses the records found in the Web of Knowledge database based on the key words ”ecosystem services“ in the topic of the article, published from 1992 to 2010. Figure 6 shows an increase in the number of publications related to ecosystem services. The same trend occurs in the case of valuation studies, even at the level of hundreds of studies.

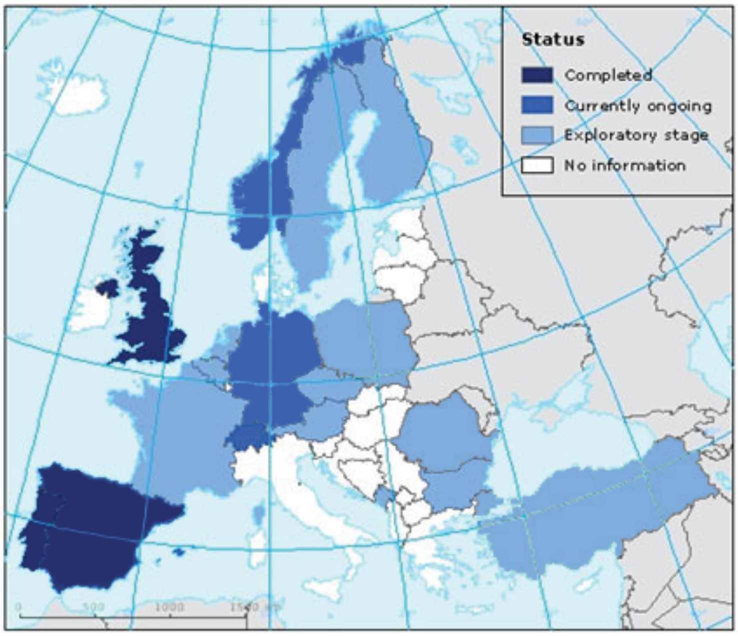


Source: taken from Vo et al., 2012

Figure 6: The total number of papers published on ecosystem services over time

After global and sub-global assessments within the framework of the Millennium Ecosystem Assessment (MA, 2005) have been made, the scientific and policy demand tends to initiate and conduct studies at a lower, national scale. Scientific interest has increased steadily since the late 1990s (Costanza et al., 1997; de Groot et al., 2002) while policy demand for specific ecosystem governance has become notable only recently (Perrings et al., 2011). Actual policy demand is driven mainly by the Aichi Targets (Strategic Goal D) and the EU Biodiversity Strategy to 2020 (Action 5) which promote consistent ecosystem assessments at a national or regional level. Both documents stress the importance of ecosystem services in maintaining human well-being and prosperity. Therefore, conducting an inventory of ecosystems and their services through mapping and assessment is one of the keystones of the EU Biodiversity Strategy. National biophysical assessments by the EU member states are expected to be delivered by 2014 and economic accounting is expected to be complete by

2020 (COM, 2011). An overview of ecosystem services assessment related activities among European countries (not only EU members) is given Figure 7.



Source: BISE, 2013 (http://biodiversity.europa.eu/ecosystem-assessments/assessments)

Figure 7: Status of national ecosystem assessments in European countries

Currently, national ecosystem services assessments have been completed in the United Kingdom, Spain, and Portugal. Switzerland, Germany and Norway are currently working on them and 12 European countries (including Czechia) are in an exploratory phase (BISE, 2013). Within the European region, an initiative on national assessments is to some extend assisted by the European Commission. Currently, a coherent analytical framework "Mapping and Assessment of Ecosystems and their Services (MAES)", which would ensure consistent approaches to national assessments, is under development. A detailed description of the integrated Assessment in Czechia is introduced in Analytical Part of the thesis, Chapter 2. Initiatives on ecosystem services assessments take part also outside Europe. The most recent, Bhutan (Kubishewski et al., 2013) and Georgia (UNEP and WWF, 2013) finished their national assessments. Assessments in a preparatory phase are under development for example in Israel (http://www.hamaarag.org.il/en/content/inner/ecosystem-services) and in Russia (http://www.ioer.de/1/projekte/aktuelle-projekte/teebi-rus/).

Recently, the concept of ecosystem services is also applied in reports summarising the state of the environment such as The European Environment: State and Outlook 2010 released by the European Environment Agency in 2010, or Global Environmental Outlook 5 released by the UN in 2012.

Furthermore, in relation to environmental security, some steps have been taken to integrate the concept of ecosystem services in one of the key documents. In 2005, Hyogo Framework for Action (HFA) aiming to build a resilience[6] of society to disasters was released. In the context of the year 2015, when the HFA expires, countries and other stakeholders confirmed their interest in and the need for the HFA2 to be a post-2015 instrument (UNISDR, 2013). Contrary to HFA, HFA2 already specifically operates with the ecosystem services framework and apart from addressing mismanagement of the environment, it calls for an enhancement of social and environmental vulnerability assessments and ecosystem services accounting (UNISDR, 2013).

An important platform, where individuals and organizations interested in ecosystem services may communicate and plan further steps and activities, is the 'Ecosystem Services Partnership' (ESP). The partnership was launched in 2008 with the primary aim of enhancing the science and practical application of ecosystem services assessments (http://www.es-partnership.org/). In 2012, another platform was established - the 'Intergovernmental Platform on Biodiversity and Ecosystem Services' (IPBES). It is an independent intergovernmental body open to the member countries of the United Nations. Its main aim is to facilitate the dialogue between the scientific community, governments, and other stakeholders on biodiversity and ecosystem services (http://www.ipbes.net).

Presented overview shows that a lot has been done to establish the theoretical and paradigm basis of the concept of ecosystem services in the last decade. Also the formation of several communication platforms, where science can meet practice and vice versa seems to contribute to the development of more grounded analytical approach. The main strength of the method can be seen in its complexity, as it enables to observe numerous proxies, bringing broader perspective contrary to e.g. traditional conservation approaches. On the other hand, despite some policy demand, results of national assessments are only slowly translated into policy documents or actions. The main aim of making the nature´s value soundly resonating in decision-making and to streamline ecosystem services into management by developing maps and indicators is still rather at the beginning.

[6] Resilience represents the ability of a system to deal with the effects of hazardous event. For this, ensuring the preservation, restoration, or improvement of basic structures and functions of the (socio-ecological) system is essential (United Nations Plan of Action on Disaster Risk Reduction for Resilience, 2013).

3.3. Research on Ecosystem Services in Czechia

Despite wide scientific interest in ecosystem services abroad and obligations imposed by EU commitments, the concept is rarely being recognized and not applied in Czech decision-making processes. The approach is still relatively new and only partially developed in our conditions. However, several researchers have already taken up the concept.

One of the institutes, which consider the approach as a key concept, is the Centre of Global Change Research (CzechGlobe), in particular the Department of Human Dimensions of the Global Change. In addition to CzechGlobe, ecosystem services are analysed at the Environment Centre of Charles University in Prague by Moldan and his team. To foster scientific and public awareness of the concept, the Centre of Global Change Research jointly with the Environment Centre of Charles University in Prague held a conference, ”Ecosystem Services, Human Values and Global Change“ in Prague in April 2012. The conference introduced the current state of the art in the field of ecosystem services, applications of the concept in environmental management and decision- making related to life quality and global changes.

Some universities have also started to pay attention to this topic, such as the Faculty of Science at Charles University in Prague, South-Bohemian University, Jan Evangelista Purkyně University, and Mendel University. A compact monograph dedicated to ecosystem services and to human well-being dependency on nature was published by Nátr from Charles University in 2011. The monograph written in Czech summarizes the basic knowledge about the concept of ecosystem services, their (economic) valuation and links to the available information about environmental issues like food security, climate change or water availability (Nátr, 2011).

So far, two studies on the assessment of ecosystems services in Czechia are available. First, a pilot study accounting for the ecosystem services provided by grasslands (Hönigová et al., 2011), second an integrated assessment of ecosystem services at the national level (Frélichová et al., 2014), which is described in detail in Analytical Part in Chapter 2. National Agency of Nature and Landscape Protection was involved in the activity and results were communicated to the Ministry of the Environment of the Czech Republic. The Ministry is currently taking a more active role in planning the consequent activities related to the assessment of ecosystem services.

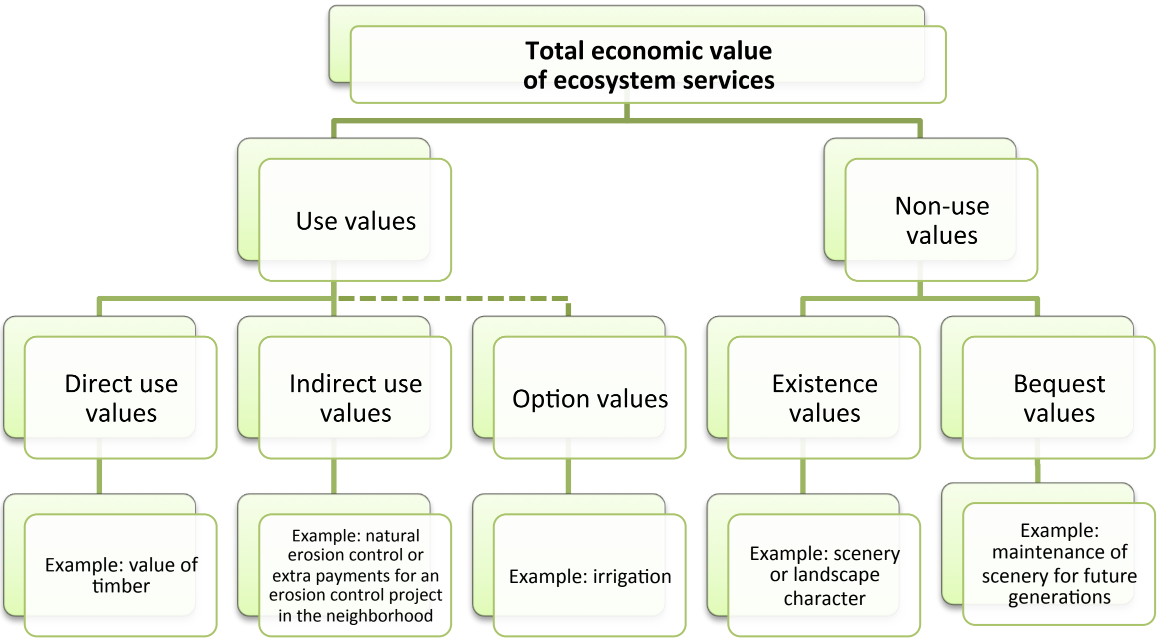
3.4. Valuation of ecosystem services

Valuation of ecosystem services, in both economic and biophysical units, aims at a quantification of the benefits provided by ecosystems to society, or alternatively losses related to damage or destruction of ecosystems. Despite the multidimensional importance of ecosystems (ecological, socio-cultural and economic), expression of the ecosystem services value in monetary terms facilitates the discussion with policy makers (De Groot et al., 2012). Because the value of ecosystem services is not fully recognised yet or it is even ignored in decision-making processes, an initiative like TEEB (The Economic Valuation of Ecosystems and Biodiversity, 2010) has been launched to stimulate public debate and policy action. Apart from that, economic valuation also provides guidance in understanding users' preferences and their appreciation of ecosystem services (De Groot et al., 2012).

Several methodologies for valuation of ecosystem services exist. The diversity of approaches basically rests in the purpose of the ecosystem services assessment, data availability, and analytical tools applied. Based on the character of data and the methodology, it is possible to distinguish two main types of the assessment: qualitative and quantitative. Qualitative methods are usually applied in case of (quantitative) data limitation. This happens quite frequently in case of ecosystem services assessments because of the complexity and integrity of the issue. Qualitative assessment investigates potential changes induced within socio-ecological systems by specific drivers of a change through rating of environmental and social changes relative to an actual state or reference condition (Busch et al., 2012). Besides, qualitative assessment is of critical importance in the assessment of cultural services. Socio-cultural value includes the importance people attach to the cultural identity and the degree to which that is related to ecosystem services (De Groot et al., 2010). Even though qualitative assessment poses a risk of subjectivity, it is often applied. Input data may be found in literature or derived from interviews, questionnaires or expert knowledge. An example of quantitative assessment provide Vihervaara et al. (2010) or Chapter 4 in Analytical Part of this thesis.

In case of quantitative assessments, biophysical (ecological) or economic indicators are used. Indicators can express actual state (*state indicator*): which ecosystem service is provided and to what extent (e.g. total biomass production), or performance (*performance indicator*): to what extent is consumption of an ecosystem service sustainable (maximal sustainable yield of biomass) (De Groot et al., 2010). Biophysical assessments yield results in biophysical units, such as tons of carbon fixed by one hectare of crop, while economic assessments operate with monetary values, such as euros or dollars per hectare.

Based on anthropogenic perspective, ecosystems are valuable in terms of use values and non-use values (De Groot et al., 2010). Use values represent values attributed to direct consumption or utilisation of ecosystem services (e.g. value of timber or recreation). Indirect use values are attributed to indirect utilisation of ecosystem services, through the positive externalities that ecosystems provide (e.g. pollination or erosion prevention). On the other hand, non-use values measure the importance attributed to an aspect of the environment in addition to, or irrespective of its use values. It represents the welfare the ecosystem may give to other people, the welfare the ecosystem may give to future generations (bequest value) and/or existence value (value of knowing that the ecosystem exists). There is one more type of value, regarded as transitional between use and non-use: option value attributed to preserving the option to utilise ecosystem services in the future (De Groot et al., 2010, Nátr, 2011). When adding use and non-use values together, the total economic value (TEV) is calculated (Figure 8).



Source: Modified based on De Groot et al. (2002) and Kaval and Baskaran (2013)

Figure 8: Components of total economic value of ecosystem services

Valuation methodologies

Even though a number of techniques to estimate the value of ecosystem services have been developed, there are nine accepted primary valuation methodologies. The technique estimating market values is **market pricing**$_{(1)}$. Prices set in the marketplace appropriately reflect the value to the "marginal buyer." The price of a good tells how much society would gain (or lose) if a little more (or less) of the good were made available. Methods for measuring non-market values fall into two general categories: **revealed preference** (travel cost method$_{(2)}$, hedonic pricing$_{(3)}$) and **stated preference** (contingent valuation method$_{(4)}$, choice experiments$_{(5)}$) (Kaval and Baskaran, 2013).

The travel cost method represents the cost of travel required to consume or enjoy ecosystem services. Travel costs can reflect the implied value of the service. For example, a recreation area attracts tourists whose value placed on that area must be at least what they were willing to pay to travel to it. *Hedonic pricing* reflects the service demand in the prices, which people pay for associated goods. For example, proximity to urban greenery tends to increase housing prices (Melichar and Kaprová, 2013).

The contingent valuation method value for service demand elicited by posing hypothetical scenarios that involve valuation of land use alternatives. It reveals how much people would be willing to pay (or willing to accept) for a change in a particular service. For example, how much would visitors pay to visit a national park. *Choice modelling* is another type of stated preference method in which people select their most preferred alternative from a choice set (choice experiment), group their preferences (contingent grouping), rate their preferences (contingent rating), or rank their preferences (contingent ranking) (Kaval and Baskaran, 2013).

Apart from these methods, the following four approaches are employed in valuing ecosystem services. *Avoided cost method*$_{(6)}$ quantifies value of costs avoided by ecosystem services that would have been incurred in the absence of those services (e.g. flood control provided in wetlands). *Replacement cost*$_{(7)}$ defines cost of replacing ecosystem services with man-made systems. For example, natural nutrient cycling waste treatment of a wetland replaced by treatment systems. *Restoration costs*$_{(8)}$ are costs associated with the restoration of an ecosystem to the natural state that existed prior to environmental damage. *Factor income*$_{(9)}$ is the value of an ecosystem service that enhances the market value of ecosystem services (Kaval and Baskaran, 2013). For example, water quality improvements increase fishermens' catches and incomes.

Applicability of methods is determined by a particular ecosystem service under observation as researchers use differrent methods to calculate values of ecosystem services. One method can estimate values of several services as well as one service can be valued by several methods. There are no standard methodologies for ecosystem valuation. Not only research objectives but also factors like avalability of data, time and resources influence the method selection. Said individuality of the studies and many issues that need to be considered will probably continue to prevent elaboration of unified or standardized valuation methodology. On the other hand, growing volume of valuation studies makes obvious which methods are commonly used to calculated ecosystem services value. An overview of ecosystem services and valuation methods can be find e.g. in de Groot et al. (2002) or Kaval and Baskaran (2013). This thesis introduces such an overview as a part of one of the case studies (Table 6, Chapter 2.3.2 in Analytical Part).

Benefit transfer

The methods listed above represent techniques that are based on primary data collection. However, when it comes to valuation of ecosystem services, researchers often use **benefit transfer** method (or value transfer) that applies secondary data (Kaval and Baskaran, 2013). In principle, the method enables the derivation of values (and other information) of the ecosystem based on data which have been previously used in order to value similar goods and services in a similar context (Liu et al., 2010). The transfer exercise derives a unit value from studies of a particular ecosystem type and service and multiplies this value by the area of the ecosystem type in the landscape under consideration (Plummer, 2009). The source of values is, in the benefit transfer terminology, the “study site” from which they are transferred to the “policy site” representing the considered landscape or ecosystem.

The strengths of this method are its time and cost effectiveness as well as the potential to substitute primary data when specific data is unavailable (Wilson and Hoehn, 2006). Additionally, it facilitates an estimation of values on scales that would be unfeasible in primary research (e.g. valuation of a number of sites across multiple countries); or provides consistent estimates of values across the sites (Brander et al., 2010).

On the other hand, the application of the method introduces a risk of fundamental errors and biases. Eigenbrod et al. (2010) defined uniformity error (the value of an ecosystem service is considered constant for a particular ecosystem type), sampling error (due to usually very limited choice of study sites) and regionalisation error (small and geographically localised study site, which might not be representative for the whole region). These are the three main components of generalisation inaccuracy. Apart from generalisation error, Brander et al. (2010) recognise measurement error, introduced by errors in primary valuation estimates because of weak methodology or unreliable data; and publication selection bias.

Commonly two basic types of benefit transfer are distinguished – unit transfer and function transfer (Plummer, 2009). Brander et al. (2010), Schägner et al. (2013) add to these two types adjusted unit values transfer and meta-analytic value function transfer. Alternatively, others recognise four levels of value transfer such as basic value transfer, expert modified value transfer, statistical value transfer and spatially explicit functional modelling

(Kubiszewski et al., 2013). These four approaches differ in accuracy, time demands and costs requirements in an upward trend from the first to the fourth level.

When applying *unit value transfer*, the values at a policy site are estimated by multiplying a mean unit value estimated at a study site by the quantity of that service at the policy site. The quantity of the service is usually related to an area unit (e.g. hectare) or in other cases to a household (Brander et al., 2010). This type of transfer is utilised in this thesis to valuate ecosystem services in all case studies; firstly applied at the national level and later scaled down to the regional level in the case study of Cezava. *Adjusted unit value transfer* develops the former method by modification of transferred unit values in a way to reflect differences in site characteristic, e.g. by adjustments for the differences in income or price levels over time between study and policy sites (Brander et al., 2010). *Value function transfer* is a more rigorous type of benefit transfer, which applies a value function to compute values for policy site using the variable in the equation. A function transfer can be an estimated preference function from a single study or a meta-analytical function of results from multiple studies (Brander et al., 2010). *Meta-analytic value function transfer* allows the value function to include explanatory variables, usually related to ecological characteristics (e.g. ecosystem type), geographical and socio-economic characteristics (e.g. abundance of lakes, GDP, population density) and valuation method (e.g. willingness to pay) (De Groot et al., 2012).

To give an example of practical application of the method, it has been used to construct the Ecosystem Service Value Database (ESVD) by De Groot et al. (2012). The database consists of more than 1,350 records on ecosystem services provided by the main types of biomes on the Earth. Except for the economic ecosystem services values, the database includes additional parameters such as geographic location of the ecosystem, type of the valuation method, and details on source of information. The ESVD is one of the largest databases of this type so far. To add to the database, the EKOSERV database has been developed under the project on the Integrated assessment of ecosystem services in the Czech Republic (grant No. TD010066 supported by Technology Agency of the Czech Republic), which is described in detail in Analytical Part, Chapter 2. Another example of the database built based on benefit transfer is The Environmental Valuation Reference Inventory (EVRI, http://www.evri.ec.gc.ca/evri/).

Supporting Valuation Tools

Another approach often applied in case of quantification of ecosystem services is an involvement of the models, which map and value the services. Three illustrative examples of frequently used models are briefly introduced.

InVEST (**In**tegrated **V**aluation of **E**nvironmental **S**ervices and **T**rade-offs) model explores how changes in ecosystems are likely to influence human well-being. Particularly, the model delineate supply of, demand of and a value of particular ecosystem service. InVEST currently includes sixteen distinct modules suited to terrestrial, freshwater, and marine ecosystems. By application of diverse scenarios of land (waters) uses, it enables quantification, visualisation and comparison of the delivery of key ecosystem services (http://www.naturalcapitalproject.org/InVEST.html). In addition, the tool identifies the trade-offs among ecosystem services. The trade-offs among ecosystems arise from management choices made by their users. They originate in changes of ecosystems and result in reduction of one ecosystem service as a consequence of increased use of another ecosystem service (Rodríguez et al., 2006). The model makes the trade-offs explicit and their recognition can facilitate decision-making or prevent unintentional changes (Rodríguez et al., 2006).

ARIES (**AR**tificial **I**ntelligence for **E**cosystem **S**ervices) a modelling platform rather than a single model or collection of models. ARIES maps the potential provision of ecosystem services (sources), their users (use), and biophysical features that can deplete service flows (sinks) using ecological process models or Bayesian models[7] (http://www.ariesonline.org/about/approach.html). The use of Bayesian approaches to meta-analysis and function transfer makes the ARIES project the first that systematically use Bayesian models to map ecosystem services provision, use, and spatial dynamics (Bagstad et al., 2011). The model is currently limited in provision of economic valuation (it provides biophysical or abstract units), but its development is planned in the near future.

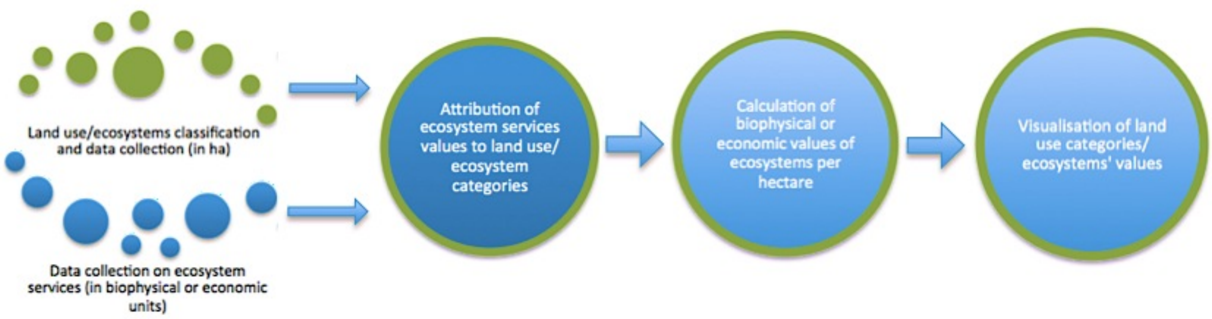
The last tool introduced here is **MIMES**, a **M**ulti-scale, **I**ntegrated set of **M**odels that assess the value of **E**cosystem **S**ervices, which provides economic arguments for land use managers to approach conservation of ecosystems as a form of economic development (http://www.afordablefutures.com/services/mimes).

With no difference, all these models can be used by various stakeholders, governments, NGOs, corporations managing natural resources or conservation organisations to qualitatively inform conservation, restoration, land use, development and other choices. Even though they are helpful tools, the uncertainties related to the models and their data inputs need to always be considered and discussed.

[7] Bayesian models are probabilistic models, which aim to describe causal relationships between variables and their associated probability measures. The causal relationships in Bayesian belief networks allow the correlation between variables to be modelled and predictions to be made, even when direct evidence or observations are unavailable (Krieg, 2001).

4. Integration of Ecosystem Services with Land Use Analysis in this Thesis

The provision of ecosystem services is considered to be dependent on biophysical conditions and land use changes (but also emissions and pollution, etc.) over time and space (Burkhard et al., 2012). Intensively utilised land is expected to expand in future to meet human needs for living space, livelihoods and food, therefore society will have to become much more strategic in its allocation of intensively managed land uses (Nelson et al., 2010). Ongoing changes in land use mentioned earlier in this thesis and increasing concerns about their impact on availability of ecosystem services is one of the main motivations for the combination of ecosystem services analysis with land use change analysis. How the approaches are combined and which steps are followed in the case studies shows Figure 9.

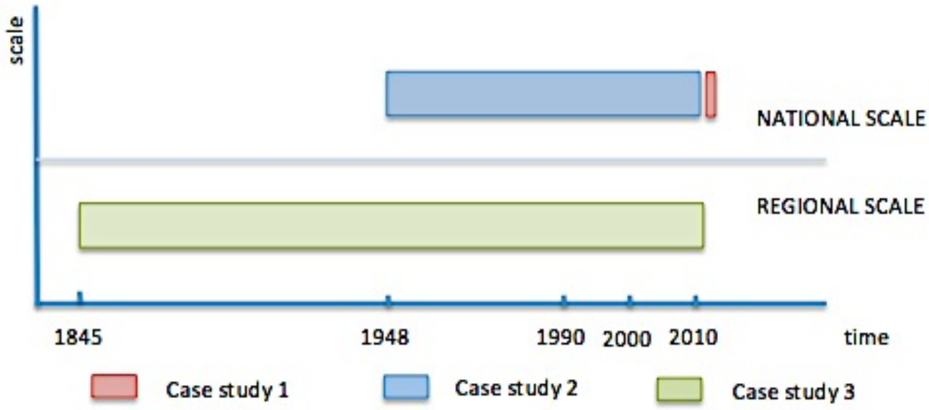


Source: Author

Figure 9: Overview of basic methodological steps followed in the case studies

The combination of these two approaches rests in the overlapping of two data sets/layers, one on land use and its changes, the other on ecosystem services and their values. Usually, to start mapping ecosystem services, spatial data on land cover/land use are classified. This thesis uses data from the Czech LUCC UK Prague Database. Then, ecosystem services are identified and their appropriate indicators are selected. In this thesis, indicators selection is determined by the content of studies taken for benefit transfer. Other options how to acquire values of indicators can be monitoring, measurements, modelling or interviews (Burkhard et al., 2012.) A similar approach applied also Maes et al. (2011) or Lautenbach et al. (2011). An integration of these data layers enables to generate the value of an ecosystem per hectare and to determine the effect of the coefficients on land conversion on values of ecosystem services. Also, an analytical approach of ecosystem service in combination with rather descriptive approach of land use change has the potential to aggregate more complex information, including visualisations. From this point of view, the concept of ecosystem services provides different perspective on land use data and interpretation of land use change and its impacts.

What needs to be accounted when it comes to the ecosystem services assessment and mapping is the scale (Hein et al., 2006). The spatial scale over which ecosystem services are provided and received is determined by the spatial scale over which an ecosystem function has effect and the spatial scale of (potential) beneficiaries (Burkhard et al., 2012). Some ecosystem services can be captured merely on a more detailed scale, as they are provided on-site (e.g. wood provision in forest), others off-site (e.g. downstream flood prevention on local scale or carbon sequestration and hence climate regulation at the global scale) (Brander et al., 2010). Time scale, especially longer time series, can also reveal some interesting patterns in changes in the delivery of ecosystem services. Figure 10 introduces the scheme of spatial and time scales covered by the case studies presented in this thesis.



Source: Author

Figure 10: Spatial and time scales covered by the case studies

Only a limited number of studies on long-term assessment of ecosystem services, as provided by the case studies 2 and 3, has been carried out so far (Kaval and Baskaran, 2013). However, the number could be to some extent enlarged by studies which analyse the issue though they do not term it as such (Kaval and Baskaran, 2013), e.g. a longitudinal ecosystem services study at a Rhone Poulenc farm started by Higgenbotham and others in 1980s (http://hgca.com/). The major difficulty in historical reconstruction of the provision of ecosystem services is the lack of data to serve as alternative to a baseline set by the current conditions. Moreover, even data for current conditions analysis are limited. Despite this data scarcity, Czech land use research and the LUCC Czechia UK Prague Database development considerably reduce data constraints and allow for interesting reflections on changes in past and their interpretation in the context of the provision of ecosystem services.

The motivation for the case studies selection, their ordering, coherence and relevancy to the research questions is discussed in the Introduction of the Analytical Part. Here, the key coefficients, which are going to be applied in each of the case studies, are introduced and their calculation is explained.

4.1. Assessment of Land Use Changes in this Thesis

After land use data acquirement from the Czech LUCC UK Prague Database, changes in the areas of particular land use subcategories are analysed. As an indicator of land use change, the index of change (I_{change}) adapted from Bičík (1995), is applied. The index assesses in percentage points the change in the area of a particular land use category during a given period.

$$I_{change} = 100 \times \frac{A_{y2}}{A_{y1}} \qquad [\%],$$

where A_{y2} is an area of a particular land use category in more recent period and A_{y1} represents an area of the same land use category in more past period. The year of more past period is the year of reference, which represents 100% state.

The index of change makes possible to compare the change in shares of land use categories between years, therefore it has been used. Lower values in more recent period symbolize a reduction of the total area of the land use category, while values higher than 100% represent an increase of the area. An identification of land use and land use change trends helps to determine national or regional provision of ecosystem services and its temporal dynamics.

4.2. Assessment of Ecosystem Services in this Thesis

To facilitate comprehensibility and comparability with other studies, ecosystem services are categorised in accordance with MA (2005) as this classification is most widely used (Fisher and Turner, 2008). Biodiversity and supporting services are considered to be a precondition for availability of regulating, provisioning and cultural services. Therefore, ecosystem services are analysed just of regulating, provisioning and cultural categories.

The desired level of ecosystem service provision is defined by environmental potential and human needs. These two phenomena can be translated into supply and demand of ecosystem services (Burkhard et al., 2012). But this thesis focuses solely on supply of ecosystem services, which is "…the capacity of a particular area to provide a specific bundle of ecosystem goods and services within a given time period" (Burkhard et al., 2012). Just to provide complete interpretation of both terms, demand for ecosystem services refers to "…the sum of all ecosystem goods and services currently consumed or used in a particular area over given time period" (Burkhard et al., 2012). Analysis of demand for ecosystem services requires stakeholders' involvement and participatory approach that is costly in terms of time, labour and resources; therefore it has not been done.

The assessment of the supply of ecosystem services employs the benefit transfer to value a service generation. After data collection, the values were converted into common metrics and, in case of monetary values, were standardized to EUR per hectares per year using 2012 as the base year. Once the values were standardized, it was possible to estimate the average values of individual ecosystem services as well as a total value per hectare of selected ecosystems. Then, to assess the long-term change in the capacity of ecosystems to provide an ecosystem service, it was calculated:

$$ES_y = ES_{indicator} \times A_y$$

where ES_y is assessed ecosystem service (in representative units dependently on ecosystem type, e.g. tons of carbon or tons of eroded soil etc.), $ES_{indicator}$ represents an indicator (unit value) for selected ecosystem service and A_y is the area (in ha) of land use type in a given year (1845, 1948, 1990, 2000 or 2010). Thus, supply of ecosystem services is a function of the area of a land use category.

In literature, the need for spatial allocation of ecosystem services and their values in relation to land use is repeatedly highlighted (e.g. Nelson et al., 2010, Burkhard et al., 2012 or Verburg et al., 2009b). In response to this, the relation between land use and the provision of ecosystem services is addressed by this thesis. Land use data are linked with statistics or other data sources to assess the provision of ecosystem services. Then they are transferred to different spatial and temporal scales.

II. Analytical Part of the Thesis

1. Introduction

This part of the thesis applies the theoretical concepts discussed in the previous section and provides practical examples and resulting insights garnered in their application. Individual chapters of the following part are based on published and accepted papers or a paper being ready for resubmission.

Common to the three case studies, Czechia is the country under consideration. The assessments of ecosystem services with regard to long-term land use/land use changes took place at two levels: national and regional. Two scales are involved, because they enable comparison to be made between different hierarchies in terms of land use trends and ecosystem services provision. As it was introduced in Chapter 4, ecosystem services are scale dependent, thus some patterns can remain hidden from only one scale perspective. The benefit of regional level study is that it has capacity to provide more detailed analysis, which would be hardly possible at a national scale in the same range. Examination at a national level on the other hand allows the integration of detailed local study into a broader geographical context.

The first study aims to answer RQ 1 and introduces an integrated assessment of Czech ecosystem services. It describes in detail the practice of such an exercise at the national level (Analytical Part: Chapter 2). The second study looks specifically on ecosystem services provided by agricultural ecosystems at the national level (RQ 2). Agricultural ecosystems were chosen for the analysis because they dominate the landscape in Czechia. The study shows changes in the ecosystem services provision, which are driven by land use changes (Analytical Part: Chapter 3). Then, a regional case study of Cezava (relevant to RQ 4) is presented. It combines the ecosystem services framework with historical land use change analysis and a more detailed assessment of the state of the environment through the use of other indicators (Analytical Part: Chapter 4). Because the study area represents an agricultural region too, it provides additional data for the results from the agricultural study at the national level.

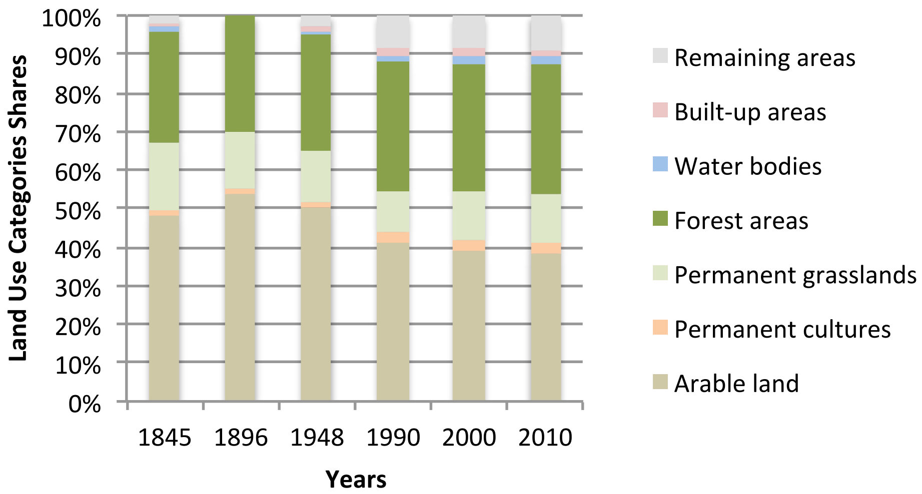
The following paragraphs outline the general background in terms of basic characteristics of Czechia and Cezava. Then, the case studies are presented. Each of the three case studies starts with an introduction to the topic and describes the methodology followed by the results. In discussion part, issues related specifically to the individual case studies are presented. Later, before the final conclusions, there follows a brief and generalising discussion based on the three case studies and frameworks given in theoretical part of the thesis.

1.1. National level – Czechia

Czechia is a landlocked country located in central Europe (between latitudes 48° and 51° N, and longitudes 12° and 19° E). Despite its medium size (compared with other European countries) of about 7,886,600 ha, the country has exceedingly varied landscape with diverse habitat types. The fauna and flora flourishing within Czechia reflect four WWF ecoregions: Western European broadleaf forests (85%), Carpathian montane conifer forests (9%), Pannonian mixed forests (4%) and Central European mixed forests (2%). The climate is temperate continental with relatively high seasonal variations as well as a great variation of temperature and precipitation dependent on the altitude. The long-term average annual precipitation is 689 mm and average annual temperature is 7.5°C. The country lies on the Main European Watershed, drained by three rivers - the Elbe River (western part), the Oder River (north-eastern part) and the Danube River (south-eastern part). Agricultural land use accounts for more than 53% of the total area of Czechia, followed by forests covering about 33%, water bodies and built-up areas (both about 2%), and other areas (9%). Protected areas (specially protected areas according to Act No. 114/1992 Coll. and its implementing regulation No. 395/1992 Coll.) cover almost 16% of the country.

The population stands at more than 10.5 million people and the population density is 134 inhabitants per km^2. Economic activity of the country as expressed by GDP (Gross Domestic Product) is CZK 3,845,926 million (EUR 152,926 million) in 2012 according to the Czech Statistical Office ("CSO"), 2014 and Eurostat, 2014. By sector, dominant contributors in the GDP budget are the services (58.6%), followed by industrial sector (39.6%) and agriculture (1.8%) (CSO, 2014).

Despite the minor contribution to GDP, agricultural land use categories dominate the Czech landscape. Land use apart from natural conditions reflects the social, economic and technical development (Bičík et al., 2010). In the Czech territory, the area of agricultural land was increasing in size until the end of the 19th century (Figure 11). Then it started to decrease due to afforestation and urbanisation (Bičík et al., 2012).



Source: Czech LUCC UK Prague Database, 2013

Figure 11: Proportions (%) of land use categories (1845 – 2010)

Agricultural intensity has been modified and production has been concentrated in the most favourable regions. These trends introduced new land use patterns and resulted in more diversified landscape functionality. Based on land use patterns, their changes and population rates, Bičík et al. (2010b) distinguish between ten typological regions. Nine of these types, determined mainly by the landscape character and land use, are introduced in Table 4.

Table 4: Overview of typological regions in Czechia

	Typological regions	Dominant LUCC categories	Dominant ecosystem types (in accordance with the categorization of the State Programme of Nature Conservation and Landscape Protection)	Dominant lsc functions
1	Urbanized areas	Built-up areas	Urban ecosystems	Residential, economic, services
2	Hinterland of main/medium-sized cities and towns	Built-up and remaining areas	Urban ecosystems	Residential, retail, transportation
3	Cultural lsc of lowlands and low-lying gently rolling regions	Arable land, permanent grasslands	Agroecosystems	Agricultural, residential
4	Uplands up to 650 m a.s.l. with average natural conditions	Arable land, built-up areas, partly forest areas and permanent grasslands	Agroecosystems	Agricultural, residential, recreational
5	Highlands and elevated regions	Permanent grasslands, forest areas	Grasslands	Environmental, recreational, residential
6	Mountainous areas	Forest areas	Mountain ecosystems	Environmental, recreational, residential
7	Military training areas	Forest areas, permanent grasslands	Forest ecosystems	Specific, environmental, recreational
8	National parks and nature protected areas	Forest areas, permanent grasslands	Protected areas	Environmental, forestry, recreational
9	Mining areas	Other areas	Urban ecosystems	Economic, residential, production

Lsc = landscape

Source: based on Bičík et al., 2010b

In this thesis, two studies focus especially on agricultural landscapes. Based on this classification, type 3 and 4 can be recognised in the agricultural case study at the national level and type 3 in the case study at the regional level.

1.2. Regional level – Cezava Region

Regional case study provides more detailed local study in order to investigate potential similarities or dissimilarities with state and trends in land use and ecosystem services at a national level. Due to examination of agriculturally utilized landscapes/ecosystems, Cezava was selected as it represents an agricultural region with very favourable natural conditions (Pannonian mixed forests), and therefore intensively managed. This make it possible to explore the impacts of intensive management and their influence on the provision of ecosystem services. Cezava covers about 0.2% of total area of Czechia and 0.4% of Czech agricultural landscape. Likewise, in other agricultural regions in Czechia, the same key processes during last decades (e.g. agricultural intensification and collectivisation, EU influence) formed current landscape functioning. In such regions, it is often the case that trade-offs take place and the environment is affected by intensive exploitation (Foley et al., 2005). Therefore, this issue provides a thought-provoking topic for the research.

The region is situated approximately 15 km south-east of Brno and covers more than 15 000 ha (Figure 12). Cezava is an alliance of 15 municipalities belonging to three districts (Brno-venkov, Hustopeče and Vyškov). The example of landscape structure and configuration illustrates Figure 13.

Source: Author

Figure 12: Location of the study area

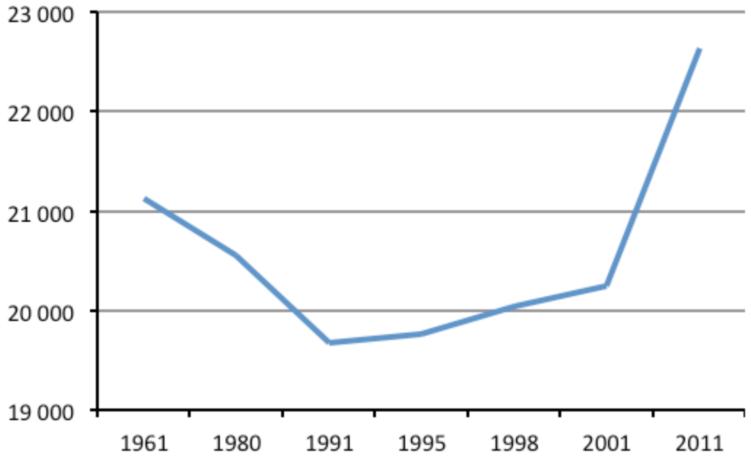
Source: GEODIS Brno (Mapy.cz, 2014)

Figure 13: Example of landscape structure in Cezava (scale 1: 95000)

The region was established in 2003 on a voluntary basis with the aim to maintain the identity of associated municipalities, conserve rural landscape, which has been affected by intensive agriculture, develop flood protection and support overall development of Brno city hinterland (Cezava, 2014).

Cezava has a concentrated settlement structure with a high number of large municipalities. However, it is difficult to identify which of these municipalities functions as a micro-regional centre, either due to comparable size of the municipalities, or the proximity to the dominant regional city of Brno. Brno is the catchment area for the Cezava region, key for labour, education and services commuting. Officially, it is Židlochovice town, which is being considered as the micro-regional centre (Cezava, 2014).

Population of Cezava stands at more than 20 200 inhabitants and population density is 129 inhabitants per km^2. Population was decreasing since 1960s until the beginning of 1990s, when it started to increase, especially after 2001 (Figure 14). This trend results from suburbanisation. However, a more extreme phase of urban sprawl (such as that around Prague) does not occur in the case of Brno and its environs (Vaishar et al., 2013). Population density in Cezava is comparable with the density at the national level.



Source: CSO, 2014

Figure 14: Population growth in Cezava region 1961 – 2011

Cezava is agriculturally exploited area since historical times (Pokorny, 2011). Management practices and the intensity of agricultural utilization have changed over time, depending on the stage of the societal development. Even though transformation of grassland in arable land continually happened also before, critical changes were introduced by the agricultural and industrial revolution, during the 19th century, and especially in the 1950s by totalitarian governmental intervention (Bičík et al., 2001). The study area was a subject to agricultural collectivization, which resulted in land consolidation and the intensification of agricultural production. Due to this uniform land use, arable land has increased, to the detriment of other uses, to include 75% (more than 11,400 ha) of land cover. Figure 15 depicts the landscape structure transformation from “strip fields” to big plots of arable land.

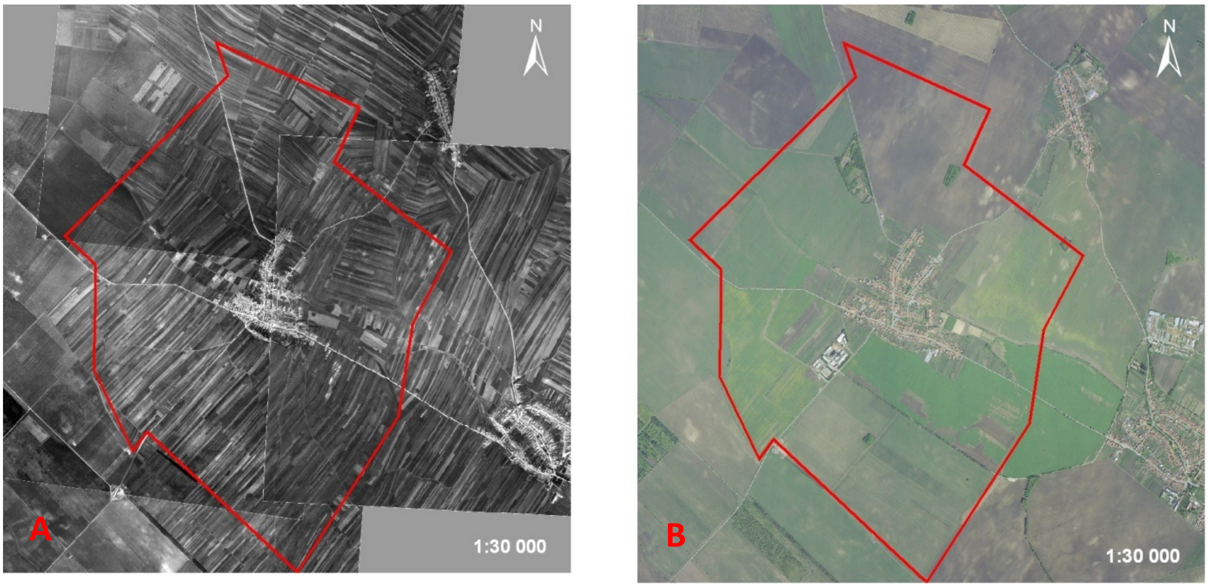


Figure 15: (A) Aerial photo of Moutnice (1949), Small scale land use (“strip fields”)
(B) Aerial photo of Moutnice (2000) Large scale land use (big plots of arable land after collectivisation); Source: VGHMÚř Dobruška, © MO ČR/HÚVG

Today, just a small portions or even the complete absence of scattered greenery is typical for the agricultural landscape of the region. Landscape elements such as permanent grasslands, gardens, vineyards and abandoned land account for less than 8% (about 1,170 ha) of the area. Forests cover about 4% (631 ha) of the region and no more than 16% of this forest cover has a natural character. The natural composition of tree species has been replaced by cultural forests, which represent 84% of the forested area (Havlíček and Navrátilová, 2005). Water bodies cover the smallest area of 1% (162 ha). Although intensively utilized, several landscape complexes with natural value can be found in the study area. One nature reserve, three nature memorials and one nature park are protected under natural conservation law (the Nature Act, No. 114/1992 Coll.). In addition to the protected nature areas, Cezava includes an ecological network and five Natura 2000 sites, proclaimed on the basis of the Habitat Directive 92/43/EEC.

Suitability of the conditions in Cezava for agriculture can be demonstrated by the official land price. The price of the agricultural land is based on the system of pedo-ecological units (PEU) and may be regarded as representative indicator of suitability of natural conditions for agriculture in given area. Every pedo-ecological unit has a unique five-digit code according to climate (first digit in the code), soil type (second and third digit in the code), relief evaluation (fourth digit in the code) and soil profile depth (fifth digit in the code). The code identifies homogenous land units and specifies their production potential. Official land price per cadastres in the study area ranges from 8.47 CZK/m^2 up to 13.61 CZK/m^2 (according to the Decree 412/2008 Coll., as updated by the Decree 427/2009 Coll. and the Decree 340/2010 Coll.). Comparing prices of Cezava with average (6.23 CZK/m^2; Ministry of Agriculture, 2012) or with minimal (0.7 CZK/m^2) and maximal (14.81 CZK/m^2) national land price values, they demonstrate high production potential of soil in the study area. Despite high soil productivity potential, the way and intensity of agricultural utilization also importantly affect the real production and its sustainability.

2. National Assessment of Ecosystem Services in Czechia

*The chapter is based on **Frélichová, J.**, Vačkář, D., Pártl A., Loučková, B., Harmáčková, Z.V., Lorencová, E. (2014) Integrated Assesment of Ecosystem Services in the Czech Republic. Ecosystem Services (in press).*

2.1. Introduction

In response to the EU Biodiversity Strategy and national TEEB assessments (TEEB, 2013), this study follows national ecosystem services assessment initiatives and introduces an integrated assessment of ecosystem services in the Czech Republic. After a pilot study accounting for the full benefits provided by grassland ecosystems in the Czech Republic (Hönigová et al., 2011), this assessment represents the first inclusive assessment of ecosystem services provided by the diverse ecosystem types in the country.

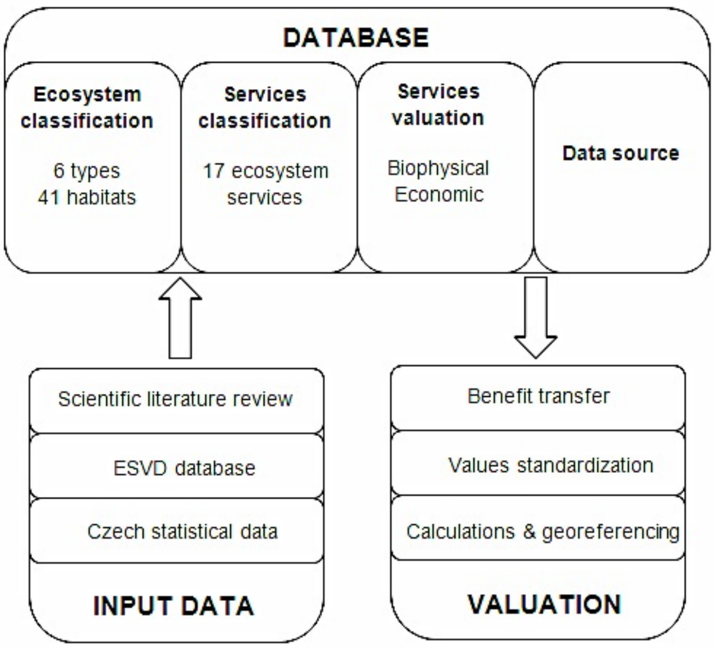
The main aims of the study are to identify ecosystems services being delivered specifically by the ecosystems in the Czech Republic and to valuate these services based on a benefit transfer. Another substantial aim of the study is to provide a methodology that would be applicable for an integrated assessment of ecosystem services in the Czech Republic at both a national and a regional scale in order to enable the development of effective policy responses to ecosystem service degradation in the future.

To fulfill these aims, the ecosystem services were quantified in terms of biophysical and economic values and mapped. The approach combines a synthesis of the state of the art, review of data resources, development of the valuation database, values geo-referencing and mapping and finally, data analysis and interpretation.

2.3. Methods

2.3.1. Methodological framework

An overall approach to an integrated assessment of national ecosystems and their services is based on the method of benefit transfer (the method has been introduced in Chapter 3.4., Theoretical Part). This method has been involved because the data available for the majority of ecosystems as well as services are limited. It allowed to transfer data from existing studies to the case of the Czech Republic. Based on the biophysical and economic values of ecosystem services present in the EKOSERV database, values were transferred according to the ecosystem accounting approach, which takes into account specific natural ecosystem units occurring on the area of the Czech Republic. The approach to a benefit transfer application as well as detailed explanation of data collection, classification of ecosystems, their services and ecosystem services values calculation is described in the following section. Figure 16 illustrates the methodological framework of the database set up of biophysical and economical values and subsequent benefit transfer valuation. The methodological framework included four basic components, namely (1) systematic review of literature, (2) the database construction, (3) benefit (value) transfer and (4) the analysis and subsequent data interpretation.



Source: Frelichova et al., 2014

Figure 16: A methodological framework

2.3.2. Data collection

To collect input data for the database on biophysical and economical values a specific search strategy was followed. The search was done in two electronic journal databases, Web of Science (WoS) and Scopus. A combination of keywords (as demonstrated on an example of grasslands) "Ecosystem service*[8] AND valuation AND grassland*" and "Ecosystem service* AND assessment AND grassland*" was applied. For other ecosystems "grassland*" were replaced by one of the relevant keywords as stated in Table 5. All document types published from 1st January 2000 to 31st December 2012 were considered. Additionally, first 50 records for each keywords combination at Google Scholar, published in the same period, were analysed to check grey literature. However, no extra contribution was found for the predefined keyword chains and therefore this data resource was not exploited more thoroughly.

Table 5: Overview of keywords used for literature search and records statistics

Ecosystem	Keywords	Total number of studies (for the period 2000-2012)				Number of relevant studies
		Scopus		WoS		
		"assessment"	"valuation"	"assessment"	"valuation"	
Agricultural	Agriculture	211	61	122	70	**5**
Forests	Forest*	432	173	282	179	**10**
Grasslands	Grassland*	78	23	59	27	**0**
Urban	Urban OR city OR cities	233	81	123	87	**3**
Aquatic	Water*	670	260	403	271	**3**
Wetlands	Wetland*	166	84	109	115	**5**

Source: Frelichova et al., 2014

Criteria for data selection were defined similarly to those applied in the case of the Ecosystem Service Valuation Database (ESVD) creation (Van der Ploeg et al., 2010). Among other reasons, this approach increases the compatibility of the findings with the database, where they can be potentially added.

In order to ensure applicability of the transferred data to Czech conditions, data sources used only were from between the latitudes of 44° – 56° N and simultaneously from European countries, or from the US and Canada. Similarly to Liu et al. (2010), the intention was to ensure similarity in socio-economic factors by an application of these conditions. Because most of the studies selected for the transfer had been conducted in Europe (90%), initial geographical zone was narrowed and focused on European studies only. As another criterion, studies needed to provide either original data or data properly referenced to the

[8] The asterisk (*) is a wildcard character, which replaces multiple characters anywhere in a word, e.g. service* finds service and services.

source. Another requirement was that studies needed to provide a biophysical or economic value of an ecosystem service with a reference to a particular ecosystem type/habitat. The value was also related to a particular surface area and the method used to derive it was discussed.

As a complementary data resource the ESVD was utilized, which has been compiled by the Ecosystem Services Partnership (ESP). The ESVD includes more than 1300 original values in monetary units organized by service and biome (Van der Ploeg et al., 2010). To select transferable records from the database with a capacity to contribute to Czech ecosystems' value determination, the same strategy as for the literature research was followed. Finally, the existing Czech studies and reports, including primary data, provided some extra data to work with.

A preliminary sorting of the findings showed that for some ecosystem types and several ecosystem services records are either scarce or missing entirely. For this reason the data search was repeated, following the same criteria but with key words adjusted, focusing on this specific data (focusing on the following specific data categories: natural forests, orchards, water, disturbance, nutrient regulation, pollination, pest control, crop and non-timber provision and aesthetic value). This additional resource mining increased the number of transferable data points. Table 6 gives an overview of ecosystems and ecosystem services that fall in the scope of the research and introduces valuation methods that have been applied to derive the value.

Table 6: An overview of ecosystem services in the scope of the research and valuation methods

Service category	Services	Ecosystems	Valuation methods	
			Biophysical	Economic
Provisioning	Crop	A	-	NP
	Biomas	A, F, G, W, WET	Modeling, productivity	DMP, NVA
	Fish	W, WET	No. of professional fisherman	MA, DMP, NVA
	Game	F	Gross animal weight	DMP
	Non-timber	F	Non-timber production	DMP
	Timber	F	Timber production	DMP, LEV
	Water	W, WET	Extraction, infiltration	AC, CV, MA, NVA
Regulating	Air quality	F	Average dry deposition of PM_{10}	AC
	Climate	A, F, G, U, WET	Carbon sequestration	AC, BT, CV, ET, MAC, DMP, SCC
	Disturbance	W, WET	-	DC, CV
	Erosion	A, F, G, WET	Model of erosion risk control, RUSLE	AC, BT, MA, RC
	Nutrient	A, G, W, WET	Review	BT
	Pest control	A, F, G, WET	-	BT, CV
	Pollination	A	-	BT, IPEV
	Water cycle	A, F, G, U, WET	Run-off, modeling	AC, BT, MA, RC
	Water quality	G, F, WET	Review	AC, BT, CV, MA, PES, RC
Cultural	Aesthetic value	A, F, W, WET	-	BT, PV, CV, MA
	Recreation	A, F, G, U, W, WET	No. of visitors/visits	BT, CPS, CV, DMP, FI, MA, MAC, NVA, TCM

Acronyms for the ecosystems: A – agricultural, F – forests, G – grasslands, U – urban, W – water, WET – wetlands
Acronyms for the valuation methods: AC – avoided cost, BT – benefit transfer, CV – contingent valuation, ET – emission trading scheme, IPEV - insect pollination economic value, LEV - land expectation value, MA – meta-analysis, MAC - marginal abatement costs, DMP – direct market pricing, NP – net production, NVA – net value added, SCC - social costs of carbon, DC – damage costs, RC – replacement costs, PES – payments for ecosystem services, PV – property value, CPS - Consumer and producer surplus, TCM – travel cost

Source: Frelichova et al., 2014

2.3.3. Ecosystem Services Classification

To increase uniformity in the application of the concept, ecosystem services classification and the comparability of the results with other studies, the categorization of the services distinguishes between provisioning, regulating and cultural services in this study, in accordance with the MA (2005), (Table 6). The services were selected based on their relevancy to environmental conditions and ecosystems present in the Czech Republic, the significance of such services for people and a preliminary assumption that it is theoretically possible to acquire data for their quantification. Supporting services are not included in the assessment as they are conditional for the availability of the other three types of services (de Groot et al., 2002, MA, 2005). Moreover, a double counting error would be introduced in by considering them (Bateman et al., 2011).

2.3.4. Spatial Data and Classification of Ecosystems

As the existing spatial data sources were not directly applicable for a national-level ecosystem assessment, a Consolidated Layer of Ecosystems of the Czech Republic (CLES) was prepared. The consolidated layer has been produced in cooperation with the Nature Conservation Agency of the Czech Republic and utilized all the major sources of land cover/land use data in the Czech Republic. The main data source was the Habitat Mapping Layer (HML) initially produced to provide the Natura 2000 site identification. The HML was combined with Corine Land Cover 2006, Urban Atlas, the Czech ZABAGED data (Fundamental Base of Geographic Data), the Czech LPIS (Land Parcel Identification System) and other specific geographic data on waters (DIBAVOD).

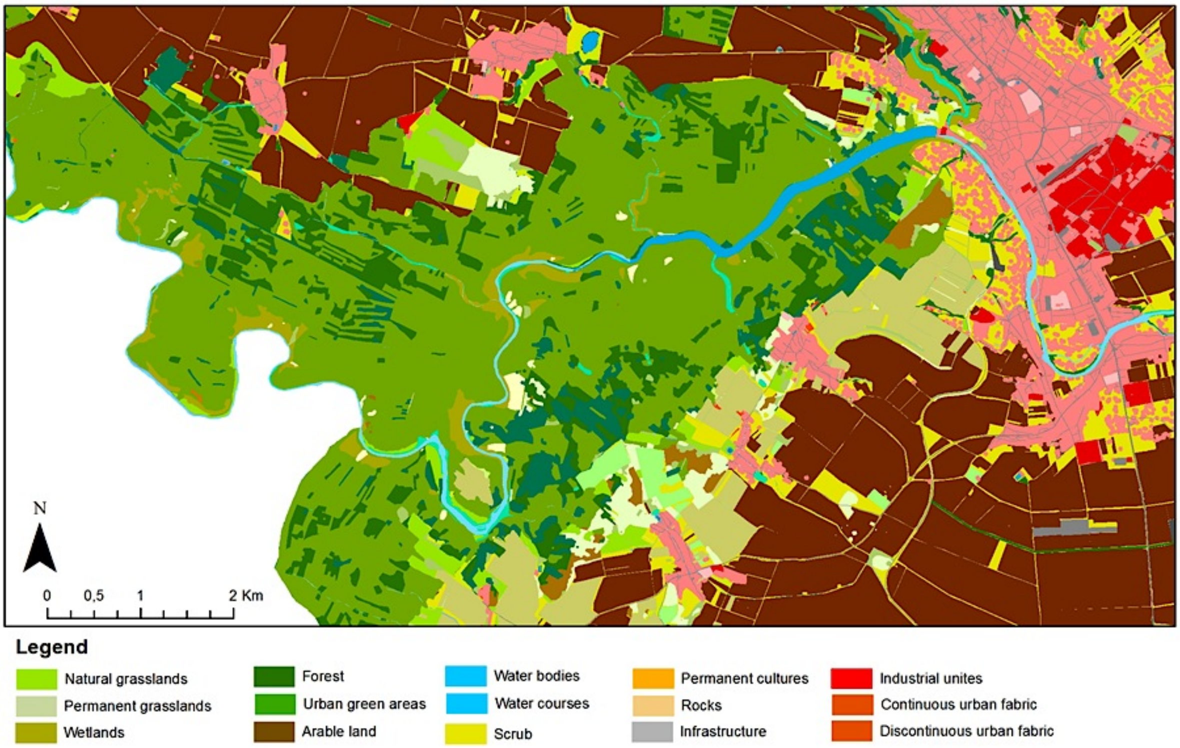
The resulting consolidated layer is comprised of 41 categories of habitats, classified at four hierarchical levels (Table 7). At the highest (first) hierarchical level agricultural land, grasslands, forests, urban areas, aquatic ecosystems and wetlands were considered.

Table 7: Hierarchical classification of the Consolidated Layer of Ecosystems

Level 1	Level 2	Level 3	Level 4
Urban areas	Continuous urban fabric	Continuous urban fabric	Continuous urban fabric
	Discontinuous urban fabric	Discontinuous urban fabric	Discontinuous urban fabric
	Industrial and commercial units	Industrial and commercial units	Industrial and commercial units
	Transport units	Transport units	Transport units
	Dump and construction units	Dump and construction units	Dump and construction units
	Green urban areas	Natural urban green areas	Urban nature
		Artificial urban green areas	Parks, gardens, cemeteries
			Recreation and sport areas
Agricultural land	Arable land	Arable land	Arable land
	Permanent cultures	Orchards and gardens	Orchards and gardens
		Hop fields	Hop fields
		Vineyards	Vineyards
	Permanent grasslands	Intensive grasslands	Intensive grasslands
Grasslands	Natural grasslands	Natural meadows	Alluvial meadows
			Dry grasslands
			Mesic meadows
			Alpine grasslands
			Heaths
Forests	Forested areas	Intensive forests	Intensive mixed forests
			Intensive broad-leaved forests
			Intensive coniferous forests
		Natural forests	Alluvial forests
			Oak and oak-hornbeam forests
			Ravine forests
			Beech forests
			Dry pine forests
			Spruce forests
			Bog forests
	Scrub	Areas with no forest cover naturally	Natural *Pinus mugo* scrub Natural shrub vegetation
		Areas with introduced no forest cover	Introduced *Pinus mugo* scrub
			Introduced shrub vegetation
Wetlands	Wetlands	Natural wetlands	Wetlands and litoral vegetation
		Natural peatbogs	Peatbogs and springs
		Anthropogenic swamps	Swamps
Aquatic ecosystems	Water bodies	Natural water bodies	Lakes
		Anthropogenic water bodies	Ponds
	Water courses	Natural water courses	Natural water courses
		Anthropogenically influenced water courses	Anthropogenically influenced water courses

Source: Frélichová et al. (2014)

Values for evaluation have been initially searched for this highest level (see the key words selection in 2.2.1.Data Collection). Due to missing data, searching based on some key words related to lower hierarchical level has been done in as well (Level 3, e.g. orchards and natural forests). Figure 17 presents an example of a segment of consolidated layer at the second hierarchical level.



Source: Frelichova et al., 2014

Figure 17: Example of consolidated layer of ecosystems for the national assessment and mapping of ecosystem services at the hierarchical level 2.

2.3.5. Calculation of Czech Ecosystem Services Values

A basic value transfer applied in this study is considered to be the best option for an initial assessment of ecosystem services values (Kubiszewski et al., 2013). Such an approach is cost-effective; however, it introduces the largest amount of measurement and generalization errors. To reduce potential inaccuracies introduced by the application of a benefit transfer, a specific search strategy was followed to ensure the maintenance of the required quality of original studies (see Chapter 2.2.1. Data Collection). Additionally, to prevent biases, a benefit transfer was not applied to calculate values of crop production. This is because of the considerable impact of specific national conditions. Apart from environmental conditions, national agricultural management and policy strategies play important role.

The literature review provided a diversified set of values in terms of economic and biophysical metrics. Therefore, the values were converted into common metrics and, in case of monetary values, were standardized to EUR per hectares per year using 2012 as the base year. For currency standardization, the official exchange rates from Eurostat (Euro/ECU exchange rates - annual data, available from http://epp.eurostat.ec.europa.eu/) and Main Economic Indicators (MEI) from OECD StatExtracts database (http://stats.oecd.org/Index.aspx?DataSetCode=MEI_PRICES) were used. Monetary values which could not be standardized based merely on the information given in the original study were excluded.

Once the values were standardized, it was possible to estimate the average values of individual ecosystem services as well as a total value per hectare of selected ecosystems. Also, a matrix of ecosystem services expected to be provided by particular ecosystem types was assembled. A total value per hectare of ecosystem was counted as a sum of the means of available services values. Afterwards, the values of Czech ecosystems were generated by attributing total values to each individual land use type based on the following formula:

$V_E = A_y * V_{ES}$,

where V_E is a value of assessed ecosystem, A_y is the area (in ha) of ecosystem/land use type and V_{ES} represents an assumed total value of given ecosystem/land use type per hectare.

2.4. Results

2.4.1. Descriptive statistics of data

In total 197 records, consisting of 55 biophysical and 142 monetary values, were found. An overview of the basic character of the database records is given by Table 8. The values were transferred from over 50 source studies.

Economic values dominate the database. Initially, more studies providing biophysical as well as economic types of values had been expected to be available, but only six papers contained both characteristics. Only 19 records that provided information in biophysical and economic units were transferred from these studies. Although this might partly result from the selection of key words, it also indicates limited coherence of data in valuation studies. Further research should take this knowledge gap into consideration and provide more coherent values, because it would contribute to increased relevance of transferred data.

Out of 142 economic values, 102 are in complete accordance with the criteria outlined for the benefit transfer, which are further referenced as strong values. Also a category of weak values was distinguished within the database. Weak values originate from studies reporting minor limitations in terms of the initial criteria, e.g. data that were published before 2000, sources introducing inter-study comparison with unspecified localization of the study area, or values based on direct market price specific for the original case study area. Values that gave a total ecological value without specifying a particular service were omitted. This study presents calculations based only on strong values.

Table 8: An overview of data character within the database

	Total no. of records	No. of standardized values (per hectare)	Character of values
Biophysical values	55	51	-
Economic values	142	121	Strong values: 102 Weak values: 19 ESVD values: 40

Source: Frelichova et al., 2014

Deriving from the literature review, several findings regarding data resources and the character of the data have been recognized. Firstly, the Scopus database provides greater body of literature on ecosystem services than Web of Science. Secondly, when the ecosystem types in terms of research frequency were considered, water bodies, forests and urban areas proved to be the most thoroughly studied land use types. However, it seems there is another triad of ecosystems being studied most frequently when it comes to valuation studies. These are forests, cropland and wetlands. On the other hand, grasslands appear to be the ecosystem to which scientific attention on ecosystem service assessment is paid most scarcely. This contrasts with the high scientific attention paid to the biodiversity of grasslands and to experiments that relate grassland plant diversity to ecosystem functioning (e.g. Loreau et al., 2001; Tilman et al., 2006; Orwin et al., 2013). However, in the case of the Czech Republic, grassland ecosystems have been chosen for a pilot assessment of value of ecosystem services (Hönigová et al., 2011). These values were included in this study.

2.4.2. Ecosystem services delivery

When comparing average values of ecosystem services with respect to classification into categories of provisioning, regulating and cultural service, the results indicate the highest value for the group of cultural services (4,081 EUR ha^{-1} yr^{-1}), followed by regulating services (2,519 EUR ha^{-1} yr^{-1}), meanwhile the group of provisioning services reaches the lowest value of 1,257 EUR ha^{-1} yr^{-1}. Average values of particular ecosystem services per hectare are introduced by Table 9.

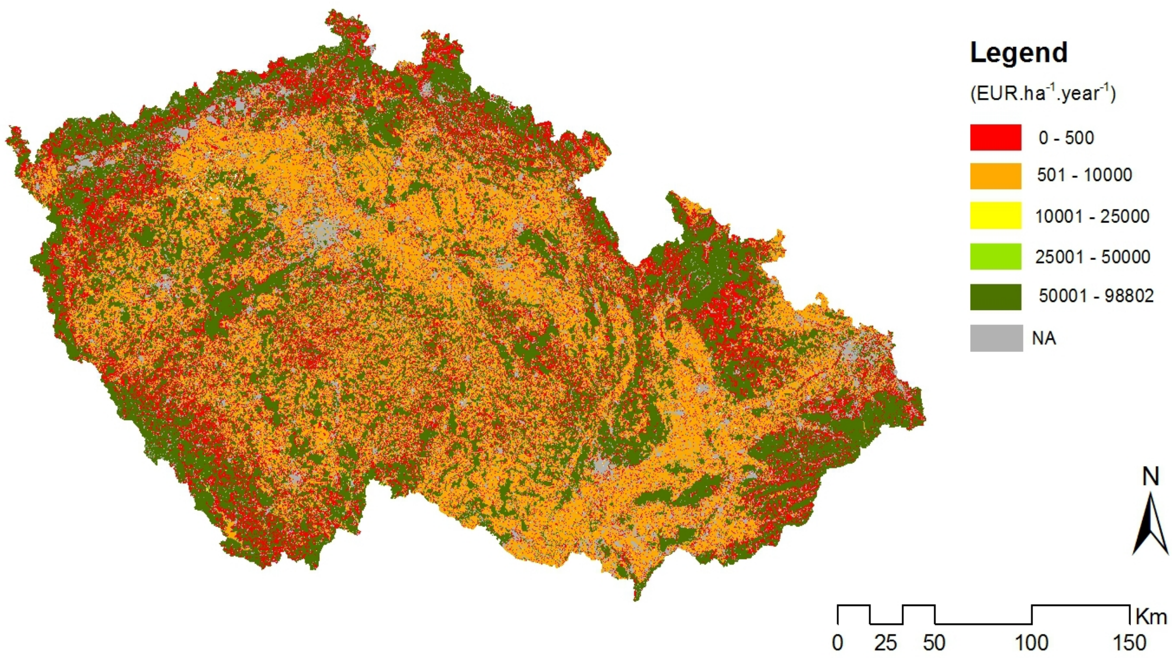
Table 9: Valuation of ecosystem services

Service category	Service	Average Value (in EUR $ha^{-1} yr^{-1}$)
Provisioning	Biomas provision	**421**
	Fish provision	**108**
	Game provision	**10**
	Non-timber provision	**57**
	Timber provision	**6,912**
	Water provision	**32**
Regulating	Air quality regulation	**266**
	Climate regulation	**4,016**
	Disturbance regulation	**8,456**
	Erosion regulation	**5,767**
	Nutrient regulation	**200**
	Pest control	**7**
	Pollination	**1,379**
	Water cycle regulation	**1,373**
	Water quality regulation	**1,211**
Cultural	Aesthetic value	**5,972**
	Recreation	**2,191**

Source: Frelichova et al., 2014

The ecosystem service of the highest average value is disturbance regulation provided by wetlands (almost 8,500 EUR $ha^{-1} yr^{-1}$). Timber provision in forests is the second most valuable service (6,912 EUR $ha^{-1} yr^{-1}$). Forests also significantly contribute to an average value of services such as aesthetic value (5,972 EUR $ha^{-1} yr^{-1}$), erosion regulation (5,767 EUR $ha^{-1} yr^{-1}$) and climate regulation (4,016 EUR $ha^{-1} yr^{-1}$). Ecosystems displaying the highest values of recreation (2,191 EUR $ha^{-1} yr^{-1}$) are forests, permanent cultures (particularly orchards) and urban parks. On the other end of the value spectrum, non-timber provision (57 EUR $ha^{-1} yr^{-1}$), water provision (32 EUR $ha^{-1} yr^{-1}$), game provision (10 EUR $ha^{-1} yr^{-1}$) and pest control (7 EUR $ha^{-1} yr^{-1}$) represent the services with the lowest values. Variability of the values can be primarily explained by a different number of input values for an average value calculation. Those ecosystem services based on an individual value record (disturbance regulation and biomas, fish, game and water provision) should be therefore interpreted cautiously. Especially provisioning services counted based on direct market price need to be taken as rough values only due to the high influence of locally specific market conditions.

For each ecosystem type, the average values of ecosystem services were summed up to come with the final value of ecosystem per hectare. As a last step, the total values per hectare of individual ecosystems were joined to the Consolidated Layer of Ecosystems. The resulting map (Figure 18) gives an illustration of spatial distribution of values of ecosystems in Czechia.



Source: Frelichova et al., 2014

Figure 18: Valuation map of ecosystems in Czechia

Economic values for unit benefit transfer used enabled to conduct the first indicative approximation of the national value of ecosystems and ecosystem services. The average annual value of the services, which represents 1.5 the current national GDP, is a good demonstration of the considerable value of ecosystems in the Czech Republic.

2.5. Discussion

2.5.1. Data

Based on the number of the records related to individual groups of services, regulating services are represented mostly (111 values), followed by cultural services (52 values) and provisioning services (27 values). Not surprisingly, the prevailing valuation method for provisioning services is direct market pricing. Both regulating and cultural services were most often valued based on benefit transfer with no more specific information on particular valuation methods. Additionally, contingent valuation and avoided costs were applied. The highest number of values comes from studies geographically attributed to the United Kingdom (39) and the Netherlands (23).

Finally, in all the studies included in the review, climate regulation service dominates the database with the total number of 43 values, followed by recreation (36 values). Although studies from European region only were reviewed, these findings are similar to the results of the global review study by Schägner et al. (2013), where recreation was found to be the most frequently mapped, followed by greenhouse gas control.

2.5.2. Verification of the Results Accuracy

The appropriateness of the resulting values has been tested by their comparison with values published in earlier ecosystem services valuation studies. Two of the studies provide global estimates of ecosystem values (Costanza et al., 1997 and de Groot et al., 2012), meanwhile the study of Liu et al. (2010) matches with observation level at a national scale. All values of ecosystems per hectare have been converted into euro per hectare per year using 2012 as the base year. Table 10 provides a summary of total values of ecosystems per hectare based on the findings (counted based on strong values only) and compares them with average values from earlier studies (Costanza et al, 1997, Liu et al. 2010, De Groot et al., 2012).

Table 10: Total values of ecosystems per hectare (in EUR 2012)

<table>
<tr><th>Ecosystem</th><th>EKOSERV total value</th><th>De Groot et al. (2012)</th><th>Liu et al. (2010)</th><th>Costanza et al. (1997)</th></tr>
<tr><td>Parks</td><td>5,813</td><td>-</td><td>5,971</td><td>-</td></tr>
<tr><td>Arable land</td><td>1,267</td><td>-</td><td>56</td><td>120</td></tr>
<tr><td>Orchards</td><td>6,601</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Pastures</td><td>452</td><td>-</td><td>-</td><td rowspan="2">302</td></tr>
<tr><td>Natural grasslands</td><td>519</td><td>157</td><td>15</td></tr>
<tr><td>Conifers</td><td>82,947</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Broadleaved f.</td><td>94,412</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Mixed forests</td><td>98,802</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Natural forests</td><td>73,264</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Forests - all types</td><td>89,886</td><td>5,488</td><td>872</td><td>393</td></tr>
<tr><td>Litoral</td><td>13,120</td><td>-</td><td>-</td><td rowspan="2">25,496</td></tr>
<tr><td>Swamps</td><td>13,862</td><td>-</td><td>-</td></tr>
<tr><td>Wetlands - all types</td><td>13,917</td><td>8,843</td><td>20,980</td><td>19,252</td></tr>
<tr><td>Ponds</td><td>1,257</td><td>-</td><td>1,847</td><td>11,065</td></tr>
</table>

"-" the value is not calculated in the study

Source: Frelichova et al., 2014

As the comparison (Table 10) implies, for parks, grasslands and ponds the values are almost identical with sooner estimates, although the source data for calculations differs (results of none of the three studies are not included in the EKOSERV database, due to filters application, therefore comparison is possible). Wetlands showed a wider range of the values; however, they could still be regarded as comparable.

Even though none of the studies shows predominant accordance compared to the others, similar patterns to EKOSERV values have been recognized. This study value estimates of urban areas and water bodies correspond most closely to findings of Liu et al. (2010). Pastures and grasslands reach similar value as Costanza et al. (1997). Another comparable figure was found for wetlands, which corresponds to the global valuation by De Groot et al. (2012). In addition, Liu et al. (2010) and Costanza et al. (1997) present almost identical estimate for this particular ecosystem type. The most noticeable difference was found in case of a forest ecosystem (in terms of total value with no specific distinction among forest types) and arable land. This might be explained by a consideration of market based values typical for Czech economy, which are usually unique and reflect local market conditions. According to findings of this study, the value of forests reaches almost 90 thousand EUR/ha, which is much higher compared to the three previous studies. The service responsible for this deviation is recreation; however, it could reflect the important role of forests in Central Europe. In general, the values produced by the Czech pilot study can be regarded as comparable with earlier findings by other scholars.

Such a comparison, besides indicating the correctness of EKOSERV values, provides additional findings regarding earlier studies. Costanza et al. (1997) is one of the most cited articles in the field of ecosystem services research, even though it thought of as presenting only preliminary and rather rough global estimates. Therefore, this approach might be criticized for an excessive degree of inaccuracy by other scholars (Loomis, 2000). Despite this fact, a number of studies performed at diverse geographical levels are based on this study and extract values of ecosystem services just and only from this resource (e.g. Porter et al., 2009). 15 years later, this study came to similar values at least in the case of some ecosystem types based on much finer data. It implies that, despite all the uncertainties included, the study of Costanza et al. (1997) provides appropriate values and their credit can be confirmed.

In addition to ecosystems, values of ecosystem services per hectare were compared as well (Table 11). For this, climate regulation and recreation were selected as they are the two ecosystem services most often valued in general (Schägner et al., 2013). The comparison implies that values for recreation (and aesthetics) resemble to values derived from earlier studies, however climate regulation values show some variability.

Table 11: Comparison of values of climate regulation and recreation services (in EUR 2012 per hectare)

	Czech value	De Groot et al. (2012)	Liu et al. (2010)	Costanza et al. (1997)
Climate regulation	4,016	2,207	991	891
Recreation	2,191	5,063	-	1,061
Recreation and aesthetics*	8,163	-	10,388	-

*Liu et al. (2010) do not provide recreation value individually

Source: Frelichova et al., 2014

The results comprised a map depicting the spatial distribution of ecosystem service values (Figure 18). The spatial perspective on ecosystem values distribution applied in this study provides innovative information about variation of ecosystem service values across Czechia, which has not been done so far. Although the development of GIS technology and models (e.g. InVEST, Aries, Mimes) provide sufficient equipment for analyses of this type, the research is still limited and in the development phase (Schägner et al., 2013, Verburg et al., 2013). Data are often the limiting factor; therefore proxies, functions and indicators are being usually applied to produce at least indicative knowledge. Despite its rather simplifying character, the combination of land cover/land use with average values of ecosystem services is one of the most frequent methodological approaches in the field of ecosystem services accounting (Schägner et al., 2013) and this study followed this approach as well. The spatial distribution of ecosystem services and differences between their values derived from the analysis, corresponded to the factual distribution of valuable natural ecosystems and biodiversity hotspots within the landscape of Czechia.

The integrated method enabled to answer which ecosystem services are provided by which ecosystems and what their value is (RQ 4). The aggregate annual value of ecosystems in Czechia reaches 237 billion EUR. This represents approximately 1.5 times Czech gross domestic product (GDP), which was in 2012 around 153 billion EUR (CSO, 2013). This is comparable to the ratio introduced by the study of Costanza et al. (1997), even though it is at a global level, which estimated average annual value of the services to be 1.8 times the current global GDP. On the other hand, the estimate of the economic value of ecosystem services is incomplete because many categories of ecosystems and ecosystem services are not included due to data limitations. As such, this number is likely an underestimation. In addition, ecosystem services assessment should also account for stakeholders' preferences, as they are the agents assigning (economic) importance to ecosystem services. In this study, calculations are based on European preferences, which might slightly differ from Czech preferences. It would be therefore desirable, as a part of future research, to also identify consumers of ecosystem services and include their choices, even though this would certainly introduce other sources of error.

3. Long-term Impacts of Land Use Change on Agricultural Ecosystem Services in Czechia

*The chapter is based on Lorencová, E., **Frélichová, J.**, Nelson, E. and Vačkář, D. (2013) Past and future impacts of land use and climate change on agricultural ecosystem services in the Czech Republic. Land Use Policy 33, 183-194.*

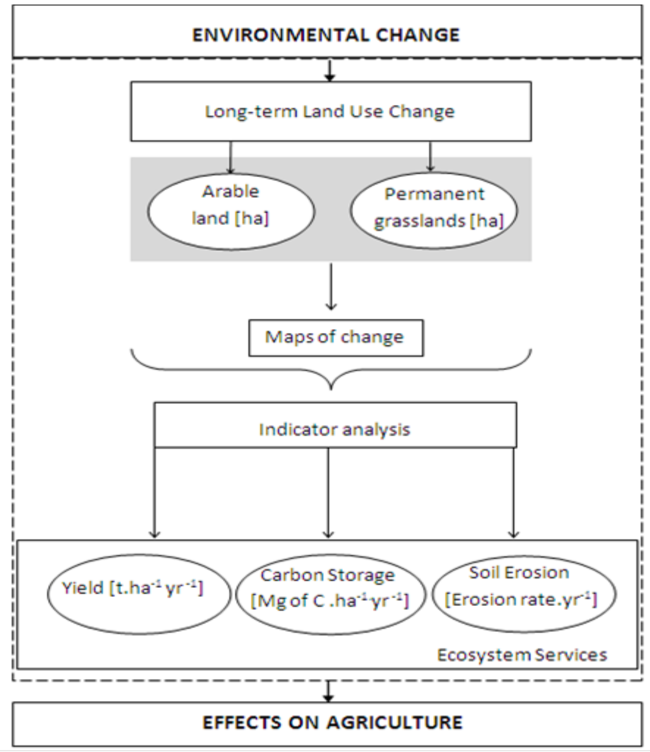
3.1. Introduction

This case study is linked with the research question focused on the delivery of ecosystem services from agricultural ecosystems in Czechia (RQ 5). It looks at the impacts land use change on selected ecosystem services specifically in the agricultural sector of Czechia, which is of strategic national importance. Whilst agriculture only contributes about 2% of GDP, agricultural land use represents more than 50% of the total area of Czech Republic (CSO, 2011). Its importance rests not only in food and other agricultural production, but it also has great significance for landscape management and landscape conservation. As such, the Czech agricultural sector represents an area of considerable economical, ecological and social value. Moreover, in Czechia, the prevailing contribution to human appropriation of aboveground net primary production (aHANPP) originates mainly from arable land (50%) and pastures (15%) (Vačkář and Orlitová, 2011).

Contrary to the previous case study, this one aims at valuation of selected ecosystem services solely in biophysical (non monetary) units. Thus it introduces alternative approach to the assessment of ecosystem services. Provision of selected agricultural ecosystem services – carbon sequestration, food production and erosion regulation (per hectare) was set against long-term land use changes data from Czech LUCC UK Prague Database. The integration of these two data sets should provide a feedback on applicability of the introduced methodological framework and enable more complex perpsective on environmental changes and their impacts. Some influential underlying socio-political transitions in Czechia were also considered. Last but not least, results of this study are meant to be a national reference for the following study at a regional level (Chapter 4).

3.2. Methods

Firstly, changes in agricultural land use from 1948 to 2010, utilising data from Czech LUCC UK Prague Database were analysed. Maps resulting from this exercise represented a basis for the comparison of the trends in land use changes over time period defined. Then potential impacts of environmental change on the three indicators of selected ecosystem services were described (Figure 19).



Source: Adapted from Lorencová et al. (2013)

Figure 19: A methodological framework

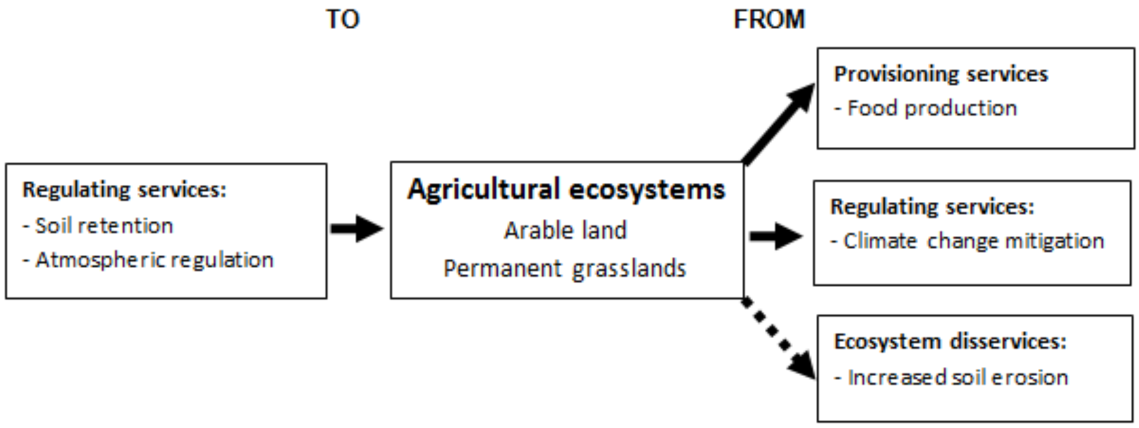
3.2.1. Land use change assessment

The land use category under observation is agricultural land that consists of arable land, grasslands (pastures and meadows) and permanent cultures (orchards, gardens vineyards and hop-fields). Because the category of permanent cultures covers a relatively low proportion of Czech land (3%), and represents a very heterogeneous category, the study focuses merely on the dominant categories of arable land and grasslands.

TO describe the changes in the areas of particular land use subcategories, an index of change (Bičík, 1995) was applied.

3.2.2. Ecosystem Services in Agricultural Landscapes

The provision of ecosystem services occur not just as one way benefits from natural to social systems, but also a flow to and from managed ecosystems, in the form of services (Figure 20) (Zhang et al., 2007). Dale and Polasky (2007) additionally identified that agricultural practices may affect the quality and quantity of ecosystem services provided by other non-agricultural systems (e.g. pollinators increasing agricultural crop yield).



Source: Modified from Zhang et al. (2007)

Figure 20: Benefits to and from agricultural ecosystems

Agricultural systems are managed by society to obtain a certain set of field and landscape characteristics and functions (ES) that serve key objectives, such as maximising the provision of yield, fiber and fuel outputs. Besides, these services agricultural systems provide a number of other benefits in terms of supporting services (e.g. soil fertility), regulating services (regulation of soil loss, water cycle, carbon sequestration or biodiversity by a capacity of agricultural landscapes to regulate population dynamics of species), or cultural services providing aesthetics and recreation possibilities (Swinton et. al., 2007). The existence of **disservices** (e.g. increased soil erosion or pest damage) is also important issue in a case of agricultural ecosystems (Zhang et al., 2007).

The selected ecosystem services aim to qualitatively and quantitatively assess the impact of the studied changes on agriculture. Given the spatial and temporal range of the study, three ecosystem services are designated and focused. The ecosystem services considered under the scope of this research are *provisioning services*, represented by yield production; and *regulating services*, represented by carbon storage and erosion regulation. The three services are essential for agricultural ecosystems functioning and were chosen as highly relevant to describe the trends in historical land use. The ecosystem services (yield production, carbon storage and erosion regulation) were translated into three specific indicators, which are applied.

Carbon storage is an indicator directly linked to climate change, and has of course a large role in climate mitigation. $ES_{indicator}$ of carbon storage is estimated based on carbon sequestration rate for arable land and for pastures and managed grasslands.

Yield and erosion regulation suitably reflect agricultural management and adaptation measures effectiveness. $ES_{indicator}$ of food production was quantified in terms of yield in tonnes per hectare and year based on statistical data provided by Czech Statistical Office. The selected crops were wheat, barley and maize, which are three commonly produced cereals in Czechia.

$ES_{indicator}$ of erosion level was estimated for arable land and permanent grasslands with the use of Universal Soil Loss Equation (USLE). Data was obtained from the Department of Irrigation, Drainage and Landscape Engineering of the Czech Technical University in Prague (Krása, 2010). As previously stated, the level of erosion is calculated based on average erosion values for arable land and grasslands.

To assess the long term change in the capacity of arable land and grasslands to sequester carbon, and to estimate the level of soil erosion from 1948 to 2010, it was calculated:

$$ES_y = ES_{indicator} \times A_y$$

where ES_y is assessed ecosystem service, $ES_{indicator}$ represents an indicator for selected ecosystem service and A_y is the area (in ha) of land use type (arable land or grasslands) in a given year (1948, 1990, 2000 or 2010).

3.2.3. Data

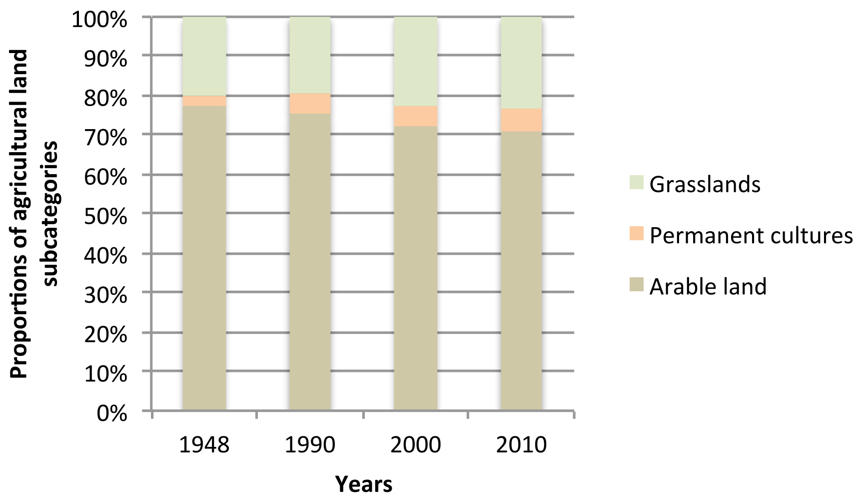
The evaluation of changes in the proportions of land use categories from 1948 to 2010 is based on statistical data from the Czech LUCC UK Prague Database. The data inputs for ecosystem services analysis were selected based on the literature review (*benefit transfer*) and data of the Czech Statistical Office. In Czechia, the carbon sequestration rate of 0.5 Mg C ha^{-1} yr^{-1} for pastures and managed grasslands is used for calculations (Hönigová et al., 2011), meanwhile arable land represents a carbon source by the release of 0.358 Mg C ha^{-1} yr^{-1} (Janssens et al., 2005). Relating to data on long-term food production in Czechia, it was looked at yields of wheat, barley and maize in 1948, 1990, 2000 and 2010 (Czech Statistical Office). National annual loss of soil from arable land (including vineyards, orchards and hop fields) reaches, in average, 3.32 tonnes per hectare (Krása, 2010). Grasslands contribute to soil erosion at 0.18 tonnes of released soil per hectare per year (Krása, 2010). The limitation is however consideration of these rates to be constant.

3.3. Results

3.3.1. Agricultural land use change

This section provides the analysis of land use changes within agricultural land from 1948 to 2010 and discusses socio-political causes. Figure 21 presents changes within the category of agricultural land. National share of arable land dropped since 1948 by approximately 6%. The area of permanent cultures increased by about 3%, and area of grasslands increased also by 3% compared to 1948. Despite arable land reduction, nationalization and socialist industrialization caused an enormous increase in the exploitation of natural resources over this period (Bičík et al., 2001). The land consolidation and agricultural production intensification triggered landscape degradation and landscape structure simplification. Large fields of arable land under the supervision of co-operative and state farms started to dominate the agricultural landscape (Bičík et al., 2001). The exceptional socio-political changes within this period, were an important contributory factor for land use change.

After 1989, new political and economic conditions led to changes in key land use characteristics. In this new era, sustainable utilization and landscape management emphasizing the agricultural land protection were expected. Unfortunately, the privatization of agriculture neither reduced the size of fields and intensity of farming, nor enriched the diversity on the fields (Janeček, 2007). One of the reasons is highly fragmented ownership patterns. Moreover, Czech agriculture typically has a high share of leased land (about 90%), which may affect farmers behaviour and their attitude towards landscape. Ownership fragmentation rate is important especially for grassland fragmentation whereas arable land fragmentation, is driven mainly by soil conditions (Sklenicka and Salek, 2008). Another factor positively influencing not only provisioning and regulating services, but cultural services is landscape heterogeneity (Fahrig et al., 2011). The study Sklenička and Pixová (2004) indicates the reduction of landscape heterogeneity and change towards a simpler land use pattern in Czech Republic over last 150 years (1845 – 2000). Similar trends, especially in agricultural landscapes, are recognized by Romportl at al. (2010) in the period 1990 – 2000.

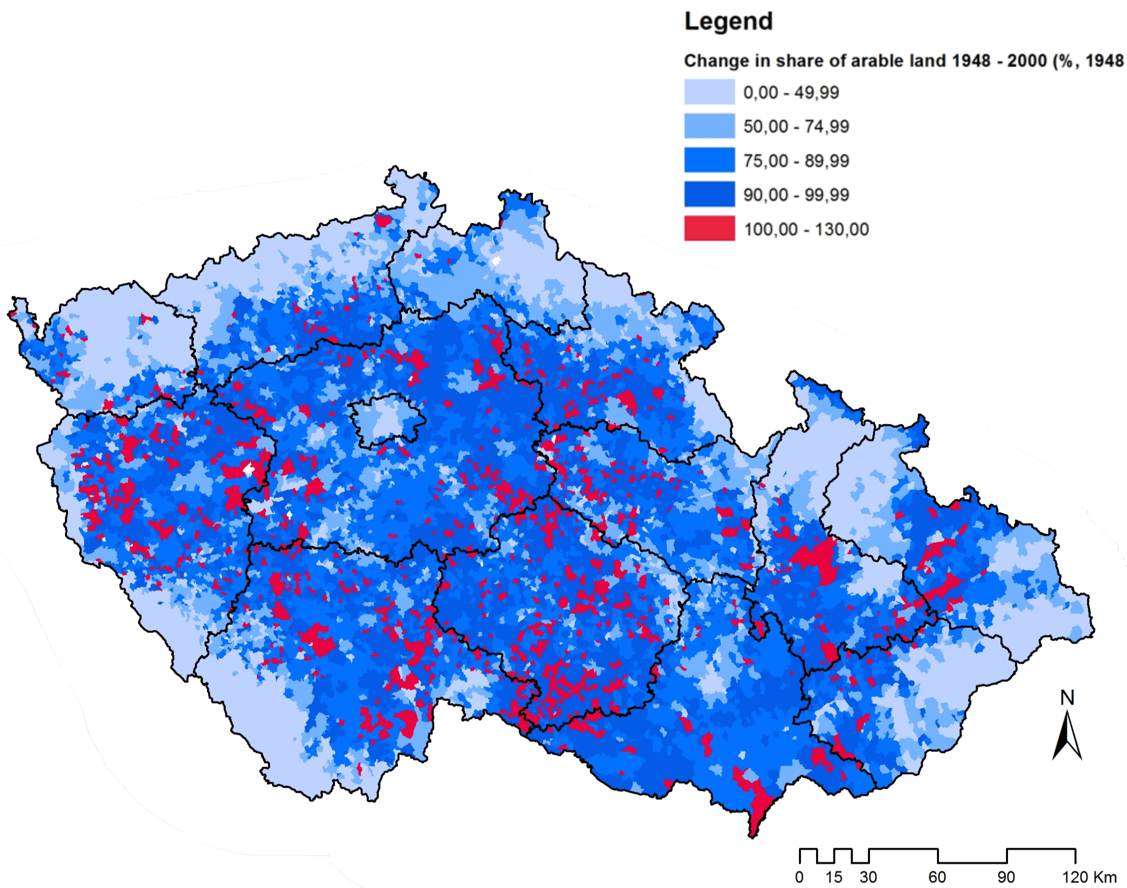


Source: Czech LUCC UK Prague Database, 2014

Figure 21: Area (%) of agricultural land subcategories (1948 – 2010)

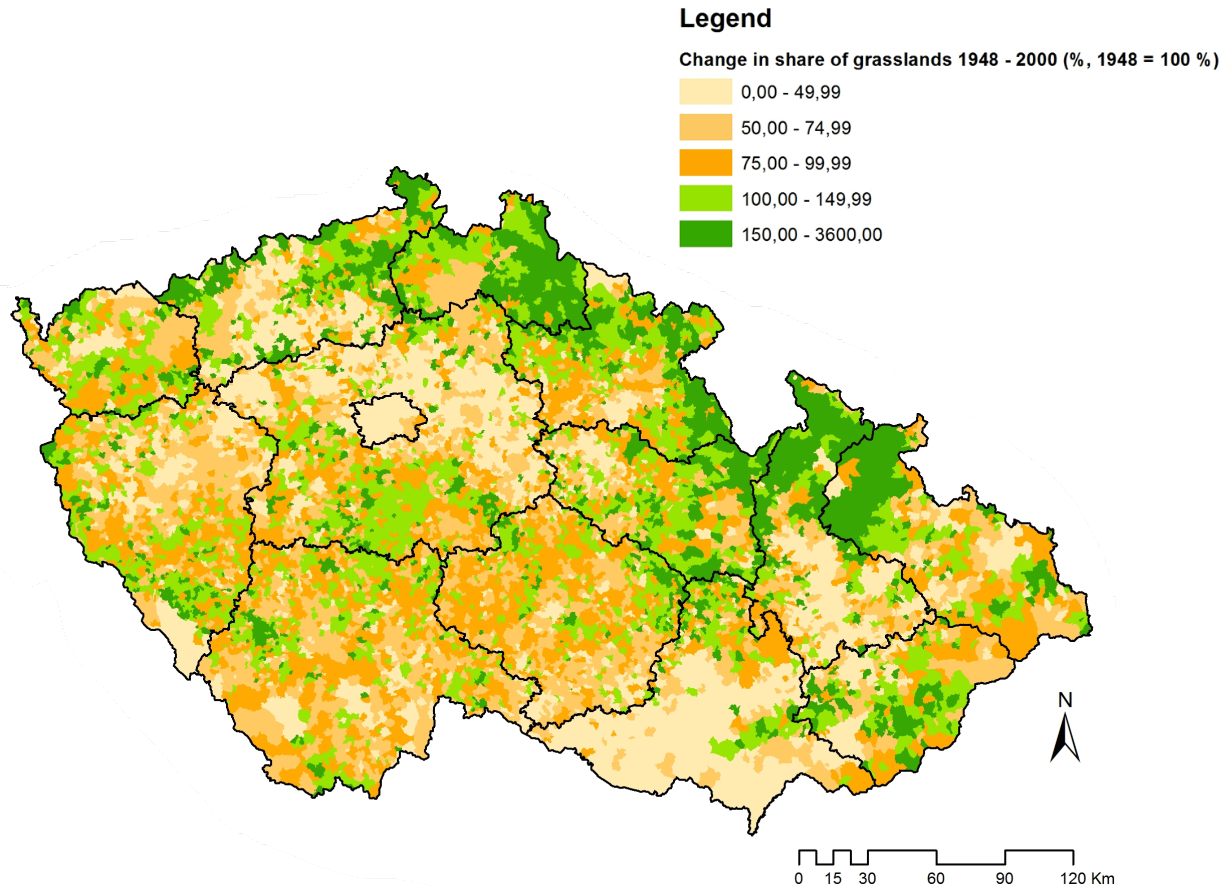
In summary, after 1989 Czech agriculture turned to intensive use of fertile lands, and the conversion of those less fertile into permanent grasslands or forests. The conversion of arable land into permanent grasslands or forests or its abandonment is notable trend in 1990. The difference of land use change at a national level between 2000 and 2010 is negligible as it makes 1% of share only in the case of arable land category and less then 1% in the other two land use categories.

The most important milestone in recent Czech historical socio-political development is its accession to the European Union in 2004, opening up the country to European regulations and law. Consequently, several subsidisation programmes supporting agro-environmental management were applied (e.g. Rural Development Programme). However, existing programmes such as EU agri-environment schemes instead of encouraging landscape level coordination usually favour farm scale approach that leads to individual, disconnected actions (Prager et al., 2012). According to the latest data, the area of arable agricultural land follows a downward trend. Grasslands, in contrary, increased its area in size by about 20,000 ha from 2000 to 2008 (CSO, 2011). The main motivation for grassing is agricultural extensification, agricultural land maintenance, soil conservation, water erosion prevention and subsidies.



Source: LUCC Czechia UK Prague Database

Figure 22: Relative change in share of arable land from 1948 to 2000, Czechia



Source: LUCC Czechia UK Prague Database

Figure 23: Relative change in share of grasslands from 1948 to 2000, Czechia

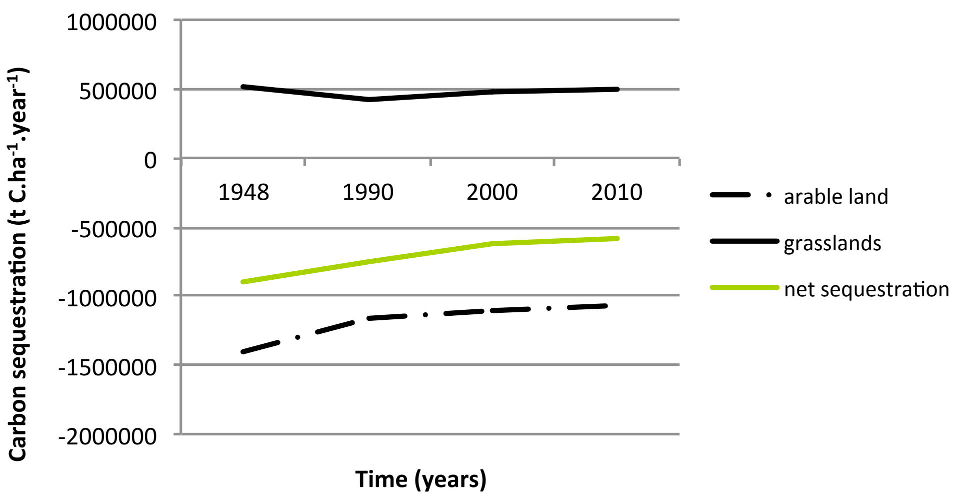
The relative change in share of arable land between years 1948 to 2000 is illustrated by Figure 22. It identifies areas with an increase or a decrease of arable land area, when comparing 1948 and 2000. The areas with increased share of arable land (the category of more than 100%) are in the minority and correspond to the zones with the richest soils and suitable climate conditions. In contrary, the area of arable land decreased in the borderland the most (by more than 50%). These regions of higher altitudes and cold climate are naturally not so suitable for agricultural production and therefore have been grassed (or forested). Another determinig factor was also the expulsion of Czech Germans and consequential landscape extensification.

Figure 23 shows the relative change in share of grassland between years 1948 to 2000 (increase of grassland area in green, decrease in orange colour). As Figure 23 indicates, grassland increased in area mainly on the northern and partly on the eastern borders of the country. The central area in the western part of the country, and the central and southern area in eastern part of the country, represents the regions with greatest decrease in grassed area. These areas are agriculturally utilized or underwent development and became urbanized. The trend corresponds with the changes of arable land and also reflect natural conditions.

3.3.2. Availability of selected ecosystem services

Carbon sequestration

Land use categories differ in the amount of carbon stored in soil and vegetation. In general, soil organic carbon stocks under cropland are lower than the stocks under pastures (Schulp et al., 2008). Figure 24 shows the trend of the change in carbon sequestration service provision from 1948 to 2010.



Source: Lorencova et al. (2013)

Figure 24: Carbon sequestration by arable land and grasslands in Czechia (1948-2010)

Net sequestration communicates the capacity of arable and grasslands to fix carbon. Being the source of carbon, arable land, which continues to dominate the category of agricultural land, pushes net sequestration into negative values as conversion of pastures to cropland, always reduces the C stocks by 50% (Guo and Gifford, 2002). However, from a long term point of view, net sequestration is increasing and the conversion of cropland to grassland can lead to increases in soil C of up to 30% (Guo and Gifford, 2002). Acceleration of the increase is visible since 1990. This correlates also with the discussed socio-political changes in 1989. Net carbon seqestration by arable land and grasslands at a nationl level increased by approximately 35% between 1948 and 2010 (when 1948 = 100%).

Food production

Food production yielded in particular years of the time period observed is introduced by Table 12. At the beginning of the observed period in 1948, yields were significantly lower because of post-war conditions, the expulsion of Gremans and adverse weather conditions. From around the 1950's to 1990, the data (CSO) on food production reflect the intensive agricultural production introduced by the Communist regime, targeting the agricultural self-sufficiency of the country. In the period 1990 to 2000, the production of wheat and barley radically dropped by approximately 12% and 48%. Production of these two crops has remained more or less stable since 2000. Maize for grain started to increase its share in total production, approximately trebling (to 200,000 t) from 1990 to 2000, and still rises. One of the reasons for this is its utility as a biofuel.

Table 12: Production of selected crops from 1948 to 2010 (in tonnes)

	1948	1990	2000	2010
Wheat	925,887	4,624,190	4,084,107	4,161,553
Barley	537,872	3,157,299	1,629,372	1,584,456
Maize for grain	35,176	98,381	303,957	692,589

Source: CSO, 2012

Erosion regulation

In nature, soil erosion is an essential natural process, reflecting the translocation of soil particles by factors related to climate, soil, topography, and vegetation. However, human activities often significantly influence this natural process (Renschler and Harbor, 2002). Soil erosion introduced by intensive agriculture, limits soil functioning as a habitat and gene pool of soil organisms and contributes to soil degradation. Soil degradation further reduces productive potential and other services such as regulation of water quality, and nutrient cycling, platforms for human activity and a functional element of landscape and cultural heritage (Elgersma et al., 2008). About 50% of arable land in Czechia is under water erosion risk and about 9% of arable land is affected by wind erosion (CENIA, 2011). For the purpose of this study, wind erosion is considered as marginal and dominating water erosion only is taken into account.

According to the results, soil erosion rate on arable land reached their highest level in 1948 and have decreased since (from approximately 13 Mt to 10 Mt, which makes about 23% less). In the case of grasslands, erosion rates are considerably lower. Here the highest erosion level was reached in 1948 (184 thousand tons), with the lowest in 1990 (150 thousand tons). Since 1990 soil erosion started to rise, almost approaching the 1948 rate (177 thousand tons in 2010). This trend, however, results from an increase in share of grasslands in the country.

In addition to soil erosion rates being driven by changes in the relative composition of land use categories, other drivers should be considered. Accelerated erosion and sediment processes on arable land may be explained by collectivization and mass production in the 1950s. Land degradation was also speed up by the introduction of crops not suitable for local conditions (e.g. maize on unfertile parcels with steep slopes). Also animal production contributed to soil degradation. Intensive animal production in stables replaced grazing and required the growing of modified feeding mixtures (Van Rompaey et al., 2003). Since 1989, Czechia has undergone some reorganization of the landscape structure once again. Nevertheless, the (re)introduction of sustainable systems that take into account actual and possible future landscape functions, and services remains challenging (Van Rompaey et al., 2003).

3.3. Discussion

Historical land use change shows a significant decrease of arable land in the border region, which is to some extent replaced by grasslands. The trend is explainable by environmental conditions given by higher altitude and less fertile soils (but it is not limited to border areas only) and socio-political drivers, mainly depopulation of the region.

Considering the ecosystem service assessment, net carbon sequestration indicates negative results. Looking at the trend in the period 1948 – 2010, grasslands showed limited ability to compensate negative carbon balance introduced by arable land. In total, selected ecosystems represent a source of carbon. Acapacity to sequester carbon in current times is (2010: -584 Gg $C.yr^{-1}$). Similarly to these findings, Müller et al. (2007) assessed that effects of land use change will cause terrestrial carbon losses of up to 445 GtC by 2100.

Food production has been estimated for the period from 1948 to 2010, for three selected crops. While the production of wheat and barley dropped, other less traditional crops like maize follow increasing pattern. Changes in agricultural production also took place in other European countries in last 50 years. Despite a number of drivers for this change, technological progress is commonly considered as the central factor (Busch, 2006). Similar patterns in the production of the three selected crops can be found at a European level (EU-15) in the last decade: lower yields of wheat and barley and increased yields of maize (Eurostat, 2007). Lobell and Field (2007) found negative response of global yields to increased temperatures for wheat, maize, and barley. Moreover, results of the study of Busch (2006) show that the structure of agricultural production and spatial patterns of agricultural land use in Europe are expected to face major changes over the next decades due to changes in global trade, technology, demography, biofuel production and policies.

From a long-term perspective, total national erosion on arable land and grasslands seems to follow an improving trend so far as the area of the most vulnerable ecosystems decreases. However, under a changed climate, soil erosion might cause changes in productivity and sustainability of agro-ecosystems (Lee et al., 1999). It is important to mention that the results are interpreted just based on changes in shares of arable land versus grasslands. Other erosion relevant factors, e.g. like changes in crop preference, which with high probability play a relevant role were not accounted.

An assessment of the impact of land use changes on ecosystem services is challenging, due to the complexity and multifunctionality of natural and managed environmental systems (Li et al., 2007). The simplification in the qualitative or quantitative evaluation of such impacts, through indicators, is inevitable and justifiable, but can introduce bias.

Dealing with agro-ecosystems, this approach displays an underlying societal preference between the characteristics of one ecosystem regime over another. Here, as in resilience theory (e.g. Walker et al., 2004), system management and intervention seeks to maintain essential services and functions for society, even through disturbance (e.g. climate or land use change). Three selected ecosystem services were considered as “umbrella services” (meaning that a support of these services indirectly influence a support of many others) and are thus sufficient for the purpose of this study. A more significant limitation rests in the consideration of ecosystem size as the only forcing variable influencing long-term availability of ecosystem services. Although size of an ecosystem crucially influences availability of services, other factors may significantly contribute. Carbon sequestration, production and soil erosion are for example influenced also through crop type, age of the community, soil conditions or management practices.

Regarding soil erosion in particular, some additional limitations were introduced. Firstly, estimation of soil erosion is based on a number of variable factors such as rain fall, soil erodibility, slope, vegetation type and management (see Universal Soil Erosion Equation). Results may importantly differ dependently on variables substituted, methodological modifications or model selection (Krása, 2010). Secondly, Krása (2010) counted average soil erosion based on data from the LPIS database (Land Parcel Identification System), which is still under development and does not quite cover the real total area of agricultural land in Czechia (Krása, 2010). When compared to the LUCC Czechia UK Prague Database, the difference in the agricultural land area is about 10%. Consequently, this divergence is another source of slight inaccuracy in the soil erosion estimation. Last but not least, the ambiguity is introduced by a back-casting of the soil erosion level based on the area of land use categories only. However, no suitable historical data for soil erosion level estimation are available.

This study described impacts of land use changes on the delivery of ecosystem services from agricultural ecosystems in Czechia (RQ 5). Added value of this study rests in its potential to provide an estimation of long term trends in a development of arable land and grasslands area; with changes conveyed spatially and with reference to ecosystem services availability. This has not been done previously for Czechia. Moreover, research generating spatially distinct outcomes describing environmental change with respect to ecosystem services is scarce in the literature in general. It is felt that this research, and where its methods are applied elsewhere, especially in countries that have undergone distinct rural land use change driven by socio-economic transition, could stimulate valuable policy discussions regarding the implications of future agricultural demand, productivity and resulting impacts on rural areas.

4. Analysis of Ecosystem Services Availability in a View of Long-term Land Use Changes at a Regional Level

The chapter is based on two studies: ***Frélichová, J.*** *(2012) Integrated Landscape Assessment of Cezava Region In HIMIYAMA, Y., BICIK, I., FERANEC, J. eds.: Land Use/Cover Changes in Selected Regions in the World. Volume VII., Charles University in Prague, Faculty of Science and Institute of Geography, Hokkaido University of Education Asahikawa and IGU/LUCC, and* ***Frélichová, J.*** *and Fanta, J. (ready for resubmission) Multi-temporal Analysis of Ecosystem Services Availability in a View of Land Use Changes: a Regional Case Study in Czechia.*

4.1. Introduction

This case study introduces integrated assessment of ecosystem services at a regional level. It combines both, biophysical and economic valuation. Here, the case study applies and further develops metodology applied in the case studies at a national level.

A South Moravian region in Czechia is a case study area, where the ecosystems, their change and the delivery of multiple ecosystem services over more than 160 years were studied. Even though few regional studies of this type exist so far, they usually analyze much shorter time period or report on either only a single service or a specific driver of the change (e.g. Lautenbach et al., 2011). The aim of this study is to introduce a conceptual approach, which combines the assessment of ecosystem services with long-term land use change and landscape functioning.

To fulfill the aim to capture fluctuations in ecosystem services availability from 1845 to 2010, the changes in economic values are analyzed. However, looking back to more distant past is a challenge, because reliable information resources are lacking. Bičík et al. (2010a). To deal with such a limitation, a simple scoring method defining functional features of the ecosystems was applied to evaluate the change of qualitative characteristics of the observed ecosystems. Finally, the findings of the assessments were compared with the analysis performed by landscape metrics.

4.2. Methods

4.2.1. Land Use Change Analysis

The evaluation of changes in the proportions of land use categories was based on statistical data from the LUCC Czechia UK Prague Database (see Chapter 2.2.3. in Theoretical Part of this thesis). The land use categories under observation were classified so as to be consistent with the categories in the database. Five land use categories were analysed – agricultural land, forests, water bodies, urbanized areas and remaining areas. The proportion (in ha) of particular categories was compared for five time periods: 1845, 1945, 1990, 2000 and 2010. To increase accuracy of statistical data interpretation, additional resources concerning land use change were consulted, such as historical maps (the 1st, 2nd and 3rd Military Surveys dating from the 18th and 19th centuries), contemporary maps, aerial photos and municipality chronicles.

4.2.2. Ecosystem Services Analysis

To capture the landscape multifunctionality and to indicate environmental quality of the study area, seven services in parallel provided by arable land, forests and water bodies were studied (Table 13). Although ecosystem services are also generated by ecosystems within the urban and remaining areas (e.g. Bolund and Hunhammar, 1999), this study is limited just to these three ecosystem types, which dominate the study area and cover almost 90%. Main reason for the exclution of remaining areas, despite the category includes areas of significant ecological function, is excessive heterogenity of the category.

Likewise in the previous chapters and assessments, this analysis follows MA (2005) classification of ecosystem services. The quantification of ecosystem services is based primarily on values (benefit) transfer from existing literature and further on chronicle reviews and map analysis. Table 13 introduces indicators for service quantification in biophysical terms. The relevancy of the selected services arises out of the agricultural character of the study area along with its land use patterns and spatial context as introduced by Costanza (2008). Here, atmosphere related services are represented by *air quality regulation* and *carbon sequestration*, both relevant at the local level, but with an outreach to the global scale. Another two services - *erosion control* and *flood protection* - indicate regulatory processes as well. Due to the agricultural character of the region, *food* and *raw material provision* are considered. Finally, *recreation and tourism* show flows of people to natural features (cultural services).

Table 13: Classification of ecosystem services and selected indicators

Ecosystem services	**Arable land**	**Forests**	**Water bodies**
Regulating ***(Regulation of biogeochemical cycles and biospheric processes)***			
Carbon sequestration	Annual amount of CO_2 stored in the biomass per 1 ha of ecosystem		-
	Annual amount of CO_2 stored in the soil per 1 ha of ecosystem		-
Air quality regulation	Amount of pollutants in the air (PM_{10} – daily and annual air pollution)		
Erosion control	Amount of eroded soil per 1 ha of ecosystem per year		
Flood protection	Buffering the extreme water flows/run-offs of water (unitless)		
Provisioning ***(Supporting humans with material benefits)***			
Food production	Average crop output (t/ha/year)	Meat of hunted deer (t/ha/year)	Fish production (t/ha/year) Mineral water collection (m^3/year)
Material production	Average yield of lucerne and silage maize (t/ha/year)	Annual growing stock of timber (m^3/ha/year)	
Cultural ***(Supporting humans with knowledge, information, cultural values etc.)***			
Recreation and tourism	Cycling trails: length in km	Number of visitors/ha/year	Number of members of the fishing clubs/year

Source: modified based on Pešoutová (2007) and MA (2005)

Added to biophysical valuation of ecosystem services, total value of ecosystems and their services in economic terms is presented. Ecosystem values per hectare were adopted from the EKOSERV database (see Chapter 2, Analytical Part). Following the same methodology, values of Czech ecosystems were generated by an attribution of the total values of ecosystems to a land use type based on the following formula:

$V_E = A_y * V_{ES}$,

where V_E is a value of assessed ecosystem, A_y is the area (in ha) of ecosystem/land use type and V_{ES} represents an assumed total value of given ecosystem/land use type per hectare. Similar approach has been applied by Li et al. (2007) for the quantification of land use change impacts on ecosystem services in Chinese case study.

4.2.3. Changes in the Delivery of Ecosystem Services

In addition to economic valuation of ecosystems and their services of today, the same methodology was applied for a calculation of economic value of ecosystems in 1948. Then, 1948 and 2010 time horizons were compared.

Also, the long term change in the study area capacity to sequester carbon and changes in soil erosion rates and yields from 1845 to 2010 were estimated and compared with national data. Additionally, for the purpose of complex description of the change a simple scoring method was developed to track the variability of the ecosystem services delivery. Scoring methods in general are helpful and widely used in assessments of complex systems (e.g. MA, 2005, Lovell, 2010, UK NEA, 2011). A crude numerical scale describing the range of changes in the provision of ecosystem services from 1845 is introduced. The methodology applied was adopted from the study published by Lovell et al. (2010).

The three groups of ecosystem services (regulating, provisioning and cultural) were attributed to three categories of landscape functions (ecological, production and cultural). For each category of landscape function, four features were identified as indicators (listed in Figure 29). Based on the literature review (e.g. Flynn et al., 2009, Frank et al., 2011, Meixler and Bain, 2010, Wallenius et al., 2010 and Zhang et al., 2007), results from previous case studies in this thesis and expert knowledge, every feature was assigned a score ranging from –2 to 2, where 2 = strongly improves functional feature, 1= slightly improves functional feature, 0 = neutral impact, –1 = slightly negative impact on functional feature and –2 = strong negative functional feature. Each category of landscape functions could receive eight points in maximum (either positive or negative). In total, maximum score for all functional categories was 24. To show the changes, the assessment was conducted for Cezava's ecosystems appearance in 1845 and in 2010. The results of the assessment indicate the contributions of each feature to the overall multifunctionality of selected ecosystems.

4.2.4. Implications of Environmental Problems on the Ecosystem Services Availability

An overview of environmental problems is considered as a straightforward way to indicate disturbed ecosystem functions, because limited ecosystem functioning undermines the provision of ecosystem services that support human well-being (Flynn et al., 2009). It has been recognized that intensive agricultural utilization modifies the quality of the environmental conditions and markedly influences the provision of bundles of ecosystem services (Metzger et al., 2008). Optimization of agricultural ecosystems for the provision of food, fibre and fuel usually requires simplification of their structure and management intensification. Consequently, landscape characteristics (e.g. heterogeneity) are changed. This triggers a chain of causal effects further influencing biological diversity, ecological functioning and finally, the delivery of regulating and supporting services on which the harvest significantly depends (Flynn at al., 2009, Frank et al., 2012, Zhang et al., 2007). Due to this causality, attention should be paid to changes in landscape heterogeneity as one of the symptoms of the environmental change.

Changes in landscape heterogeneity can be described by **Shannon's Diversity Index** (SHDI). SHDI indicates diversity of land based on information about landscape composition (the number of land use categories present) and the relative abundances of different categories (Frank et al., 2012). The SHDI increases as the number of different patch types (i.e., patch richness) increases and/or the proportional distribution of area among patch types becomes more equal. The SHDI was counted based on the following formula:

$$\text{SHDI} = -\sum_{i=1}^{m} (\text{Pi} * \ln Pi)$$

m = number of land use categories

Pi = proportion of area covered by land use category i.

Except information regarding the landscape diversity, resp. ecological functionality, the index reflects on the aesthetic value of the landscape. It is because landscapes naturalness, land cover diversity and heterogeneity contribute to the aesthetic perception by people (Herbst et al., 2009). SHDI was counted for the case study area and for agricultural land subcategories in the period 1845 – 2000 (2010 was excluded, because data on shares of meadows and pastures separately were not available). Additionally to SHDI, **size of fields** during observed time period is compared to indicate changes in arable land structure.

To further analyze environmental conditions and ecological functioning, the study focuses on **soil erosion and water pollution**, which point to harmed energy flows within ecosystems. Soil erosion is one of the indicators of soil degradation (European Commission 2002), which threatens many ecosystem services, e.g. soil productivity, the habitat and gene pool of soil organisms, water and nutrient regulation, the soil as a functional element of landscape or as a platform for human activity, etc. (Elgersma et al., 2008). In Czech conditions, water is the main cause of erosion, meanwhile wind erosion is a minor contributor to Czech erosion rate (less than 20%). Soil erosion level was adopted from the study of Havlíček and Navrátilová (2005), in which erosion data were derived from the vulnerability of landscapes to water erosion according to Wischmeier and Smith’s Empirical Soil Loss Model as the mean potential annual soil loss (t/ha/year).

To estimate water quality, basic chemical parameters (O_2, NH_4^+, NO_3^- and overall P in mg/l) were checked in seven main watercourses, flowing through the study area (Dunávka River, Hranečnický Brook, Moutnický Brook, Litava River, Otnický Brook, Říčka Brook and Svratka River). These data are available from databases of The Agricultural Water Management Authority and The Czech Hydro-Meteorological Institute (Pešoutová, 2007). Chemical parameters were referenced to the Czech norm of classification of surface water quality (ČSN 75 7221), thus water quality was determined.

4.3. Results

4.3.1. Land Use Changes

Changes in the proportions of land use categories based on data from the LUCC Czechia UK Prague Database are shown in Table 14. The proportion of agricultural land and water areas dropped by 7% and 2% in between 1845 and 2010, while the area of forests, built-up and remaining areas increased by 2%, 1% and 7%, respectively.

Table 14: Changes in land use of the region Cezava from 1845–2010

Land use category	*Years*									
	1845		*1948*		*1990*		*2000*		*2010*	
	Area (ha)	*%*	*Area (ha)*	*%*	*Area (ha)*	*%*	*Area (ha)*	*%*	*Area (ha)*	*%*
Agricultural land	**13,540**	**91**	**13,464**	**92**	**12,431**	**85**	**12,390**	**85**	**12,306**	**84**
Arable land	*10,957*	*74*	*12,740*	*87*	*11,240*	*77*	*11,240*	*77*	*11,134*	*76*
Permanent cultures	*363*	*2*	*374*	*2*	*1,012*	*7*	*1,010*	*7*	*962*	*7*
Meadows	*822*	*5*	*106*	*1*	*19*	*0*	*17*	*0*	*210*[9]	*1*
Pastures	*1,398*	*9*	*244*	*2*	*161*	*1*	*123*	*1*		
Forests	**363**	**2**	**431**	**3**	**624**	**4**	**628**	**4**	**631**	**4**
Water areas	**473**	**3**	**73**	**1**	**136**	**1**	**149**	**1**	**162**	**1**
Built up a.	**167**	**1**	**190**	**1**	**298**	**2**	**309**	**2**	**313**	**2**
Remaining areas	**285**	**2**	**412**	**3**	**1180**	**8**	**1180**	**8**	**1245**	**9**
Total area[10]	14,830		14,570		14,670		14,656		14,657	

Source: adapted from http://www.lucc.ic.cz

Among the five categories, the dominant change trends were conversion of agricultural land and water bodies to forests, built up areas or remaining areas. Agricultural land shows only a slight downward trend in total area during the years examined (Table 14). However, this category encompasses diverse types of land – arable land, permanent cultures, meadows and pastures. When considering these subcategories, the heterogeneity of agricultural lands has notably decreased over time (see also SHDI in Chapter 4.3.4.). Landscape elements such as meadows or pastures practically disappeared, despite they enhance the landscape functioning (Fahrig et al., 2011). This phenomenon can be explained primarily as a result of ongoing agricultural intensification in the mid-20th century, strengthen by the introduction of new socio-political drivers that arose out of changes in socio-political regime. The socialist government suppressed private ownership and small-scale land use (typically strip fields), providing livelihood to individual farmers, was transformed into large-scale land use under the supervision of co-operative and state farms (Sklenička, 2005). The results reveal a

[9] Shares of meadows and pastures individually are not available in 2010 due to aggregation of the categories into "grasslands".
[10] Total area of Cezava slightly changes in time because of differenciations in borders of the cadastres.

distinct simplification of the agricultural land microstructure after the 1950s. Due to favourable natural condition, the region has been attributed agricultural function and remains agricultural up today as the efficiency of intensive management is higher contrary to less favourable areas, e.g. in higher altitudes.

Forested area has been increasing since the 19th century. The most intensive afforestation period took place in the second half of the 20th century (Table 14). Until 1948, the forests exhibited a natural alluvial character with typical species composition (e.g. *Alnus* sp., *Salix* sp, *Populus* sp., *Quercus* sp. etc). Apart from the continuous forest cover, scattered islands of trees and a number of solitary trees were present in the landscape. According to maps of Military Surveys, most roads were lined with alleys. The forests of today are managed at odds with natural species composition and the tree species planted (*Picea* sp.) have mainly production function.

In contrast to forest cover, the area of water bodies and streams has significantly decreased over the examined period. The first significant reduction in the number of water bodies, ponds particularly, is evident already in maps from the 18th and 19th centuries (Maps from the 1st and 2nd Military Surveys, Figure 25). The ponds were drained and replaced by fields. According to the chronicles, these fields were rich in nutrients and very fertile. For example the chronices mention that great sugar beet and lucerne yields were obtained for 20 years, without any fertilization. Stream regulation and drainage, in response to regular spring floods and summer droughts, caused another reduction in water area (e.g. Moutnice municipality in the 1920s). The total area of water bodies dropped to a minimum in the mid-20th century. Since then, the area of water bodies has increased again, although many flows were dredged or regulated. Such a management causes the groundwater level decrease and degradation of aquatic ecosystem functions (Meixler and Bain, 2010).

Due to industrial and agricultural development, the number of inhabitants in the region increased. This common trend resulted in the enlargement of total urbanized area. Since the 1950s, the total extent of remaining areas has developed in a fashion similar to built-up areas. Today, the landscape of Cezava, which had primarily agricultural function, transformed into the landscape with agricultural and residential functions.

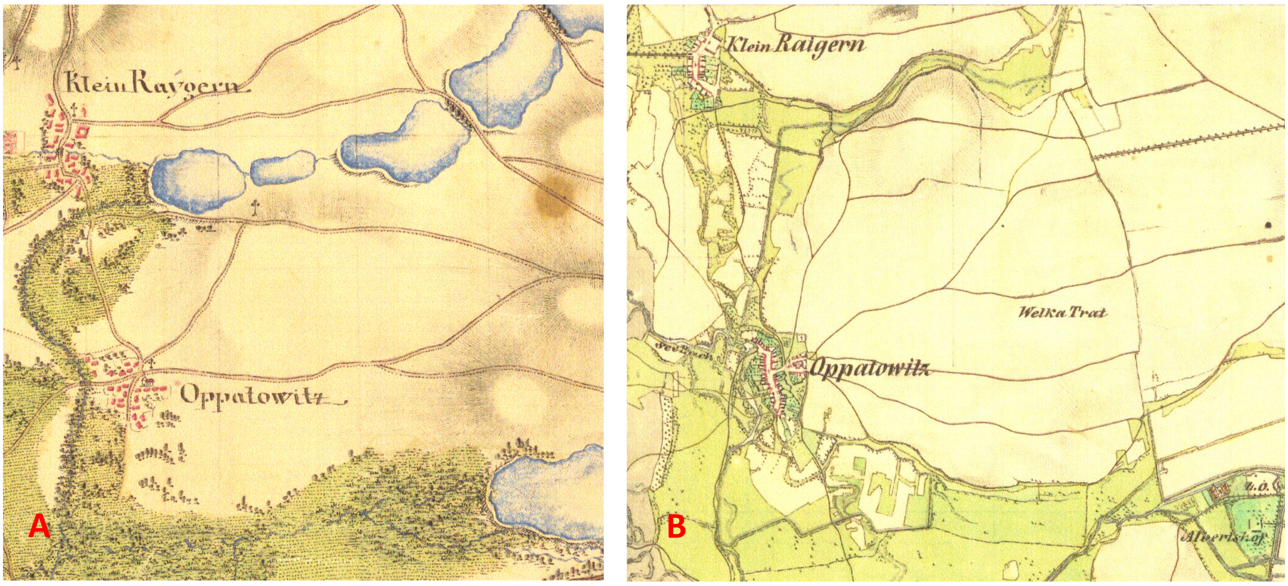


Figure 25: Reduction of water bodies in Municipality Rajhradice: (A) 1st Military Survey, 1764 – 1768, 1:28 800, (B) 2nd Military Survey, 1836 – 1852, 1:28 800

An overview of land use development in the study area indicates that socio-political development has been the major influential factor acting as the driving force behind land use changes over the past two centuries. The same factor is among the most important for land use changes throughout Czechia, resulting, typically, in the reduction of agricultural land and in the increase of forest and remaining areas (including built-up areas) (Bičík et al., 2000 and Chapter 3.3.1. in this thesis). The only exception is the period from 1845 to 1948, when the area of agricultural land in the study area increased. When considering changes within the agricultural land category, the study area is again consistent with primary types of change occurring at the national level. The most common type of change in the 1845–1948 period was the increase of arable land and permanent cultures, accompanied by the reduction of meadow and pasture lands. Later, from 1948 to 1990, the dominant type of change was the significant decrease of arable land, meadows and pastures. Permanent cultures, on the other hand, increased (Bičík et al., 2000).

More recently, remarkable socio-political development to significantly impact land use in Czechia was the Velvet Revolution (1989) and the subsequent transformation period (1990–2000). Since the 1990s, agricultural utilization of land has concentrated more significantly in the lowlands, whereas extensive agricultural production in hilly and mountainous regions has been greatly reduced to increase the economic effectiveness of agricultural production (Bičík and Jančák 2005). Due to favourable natural conditions, the study area continues to be utilized agriculturally up to these days, eventhough the area of arable land dropped by 1% in the decade after 2000. Also permanent cultures were slightly reduced, but all other categories show increase in area.

4.3.2. Ecosystem Services Provided in the Present

Three dominant ecosystem types were analyzed in terms of the ecosystem services provision – arable land, forests and water bodies. They cover abput 80% of the study area. Regulation, provisioning and cultural services of the ecosystems were identified and, where possible, quantified in biophysical units (Table 15).

Table 15: Ecosystem services recently provided by ecosystems of the region Cezava

	Ecosystems		
	Arable land	Forests	Water bodies
Regulating Services			
Carbon sequestration	Amount of C stored by arable land (-0.358 t/ha/y)*	Amount of C stored in forests (0,494 t/ha/y)*	Fixation of C by water vegetation C dissolved in water No quantitative data available
Air quality regulation	Disservice – air pollution limits for PM_{10} daily exceeded Negative moisture regulation	Dust catching Positive moisture regulation	Dust catching Positive moisture regulation
Erosion control	Disservices – soil loss due to water erosion (2.7 t/ha/year of arable land) Soil loss due to wind erosion Siltation of the streams and water reservoirs due to erosion	Prevention of soil erosion	Soil loss reduced– regulated water streams
Flood protection	Disservices – diminished infiltration and retention of water on the fields due to the low humus content Increased risk of floods	Regulation of hydrological flows Retention of rainwater Runoff reduction Infiltration of water to the ground	Storage of rainwater (flood prevention)
Provisioning Services			
Food production	Total average yield: 9.4 t/ha/y (94% of the sown area)	0.06 t of meet/ha/y from hunting	Fish production – 0.14 kg/ha/y Collection of mineral water is about 1 800 m^3/y
Material production	Lucerne average yield: 24 t/ha/y Silage maize average yield: 28 t/ha/y (6% of the sown area)	Total growing stock of timber – 7.8 m^3/ha/y	-
Cultural Services			
Recreation and tourism	Biking trails: 140 km Wine tourism Disservice – monotonous landscape	87 visitors/ha/y	Fishing 101 members of the fishing clubs/y

* taken from Janssens et al., 2010. Source: Adapted from Pešoutová (2007)

In addition to biophysical assessment of the ecosystem, services, economic valuation of ecosystems has been conducted. Based on data from EKOSERV database it was possible to generate a map of the study area, indicating spatially explicit values of ecosystems (Figure 26). The explicit values for the key ecosystems, which were considered for the detailed analysis of the change in time, are shown in Table 16 (in EUR per hectare).

Table 16: Total values of ecosystems per hectare (in EUR 2012)

Ecosystem	Value (EUR/ha)
Arable land	1,267
Meadows	486
Pastures	486
Forests	89,886
Water (ponds)	1,257

Source: Frelichova et al., 2014

Forests are the ecosystem, which generates the highest value out of the group of ecosystems under observation. It is due to high values of particular services relevant for this ecosystem type, in particular timber provision, recreation and erosion and climate regulation. In contrary, grasslands (meadows and pastures) are the ecosystem type of the lowest value (out of natural or close to nature ecosystems). It is usual that relatively low values are assigned to agricultural ecosystem or their subcategories, partly because of lack of data (Poter et al., 2009). On the other hand, agricultural ecosystems offer the best chance to increase global ecosystem services by definition of appropriate goals for agriculture and land management regimes that favor the provision of ecosystem services (Porter et al., 2009).

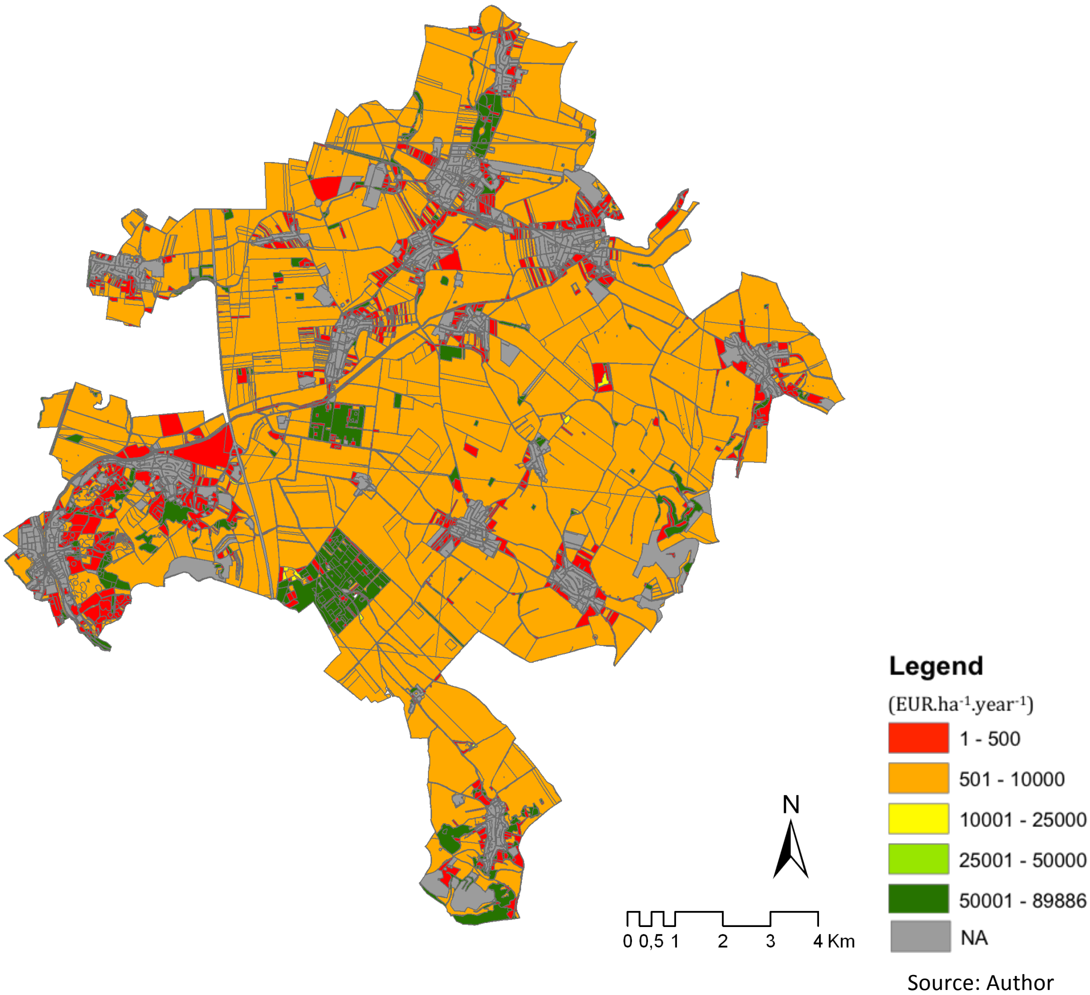


Figure 26: Valuation map of ecosystems in Cezava (2012)

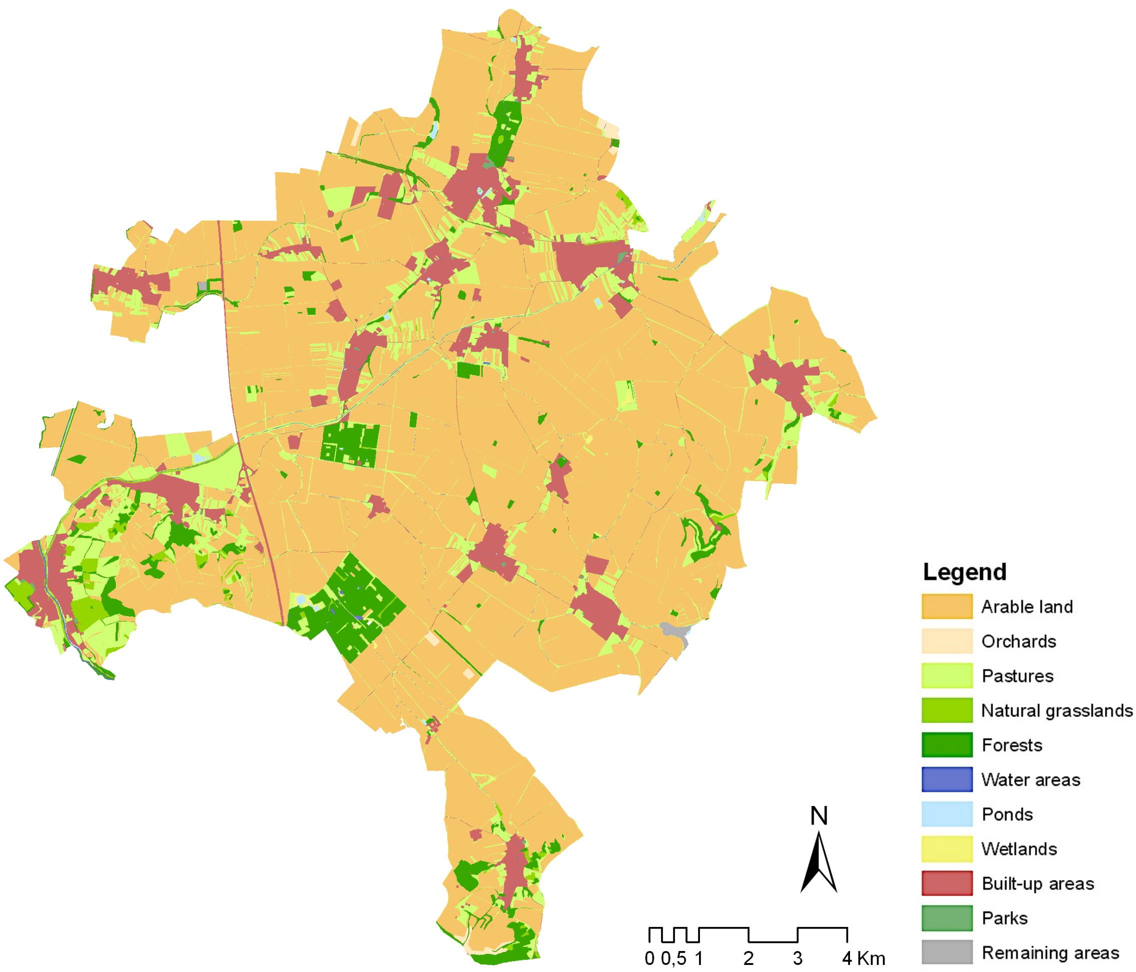


Figure 27: Contemporary land use map of Cezava (2012)

The map (Figure 26) gives an illustration of spatial distribution of values of ecosystems in Cezava. Similarly to the monetary valuation of ecosystems at a national level, the highest values are attributed to natural and close to natural ecosystems (compare with Figure 27).

4.3.3. Changes in Ecosystem Services Provision

The changes in the provision of ecosystem services are based on the two assessments. Firstly, the modifications in economic values between 1948 and 2010 were analysed. Secondly, a qualitative assessment of the landscape performance is introduced.

Changes in economic values

Total economic value of each ecosystem/land use category and its change between 1948 and 2010 was estimated. The year 1845 has been left out as the time horizon is too distant to be included in the analysis under consideration of the same socio-economic conditions as in the later periods. The results of the analysis demonstrates Table 17.

Table 17: Impact of land use changes on ecosystems' value (in EUR 2012 x 10^6, respectively in%)

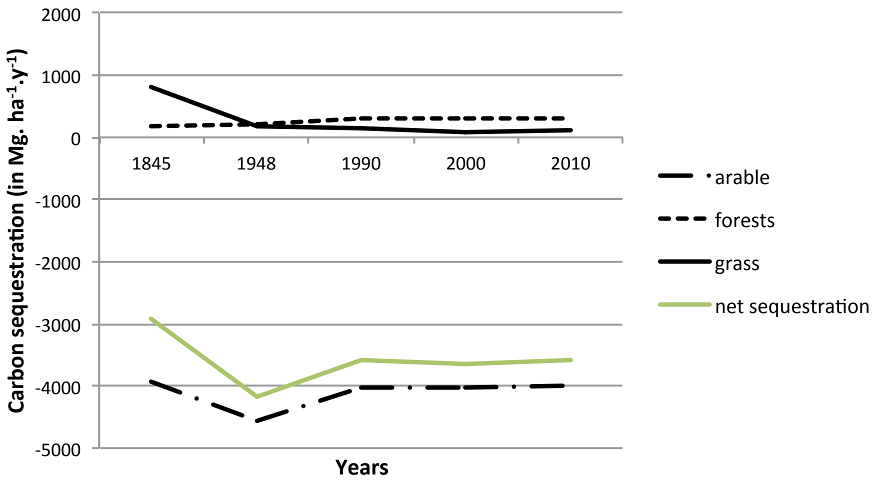
Land use category	Total Ecosystem Value (mil.EUR/year)				1948-1990		1990-2010		1948-2010	
	1948	1990	2000	2010	Mil. EUR	%	Mil. EUR	%	Mil. EUR	%
Arable land	16.1	14.2	14.2	14.1	-1.9	-12	-0.1	-1	-2.0	-13
Forests	38.7	56.1	56.4	56.7	17.3	45	0.6	1	17.9	46
Water	0.1	0.2	0.2	0.2	0.1	86	0	17	0.1	122
Meadows	0.1	0	0	?	0	-82	?	?	?	?
Pastures	0.1	0.1	0.1	?	0	-34	?	?	?	?
Grasslands	0.2	0.1	0.1	0.2	-0.1	-49	0.1	74	0	-11
Built-up area	0	0	0	0	0	0	0	0	0	0
TOTAL	**55.1**	**70.6**	**70.9**	**71.2**	**15.4**	**28**	**0.6**	**1**	**23.3**	**29**

Source: Author's calculations

During the observed period, the total ecosystem services value increased by EUR 23 million. This increment was caused by almost doubling of the forested area and water bodies. Another category, which contributed considerably to total ecosystem value (despite decreasing in time), is arable land. The contribution of arable land is not more significant as energy inputs by humans were not accounted for, particularly crop production (and similarly, fish farming in ponds). In general, agricultural land use categories were altered and their value declined. This is the case of arable land, meadows and pastures. A decline in values of these ecosystems was compensated mainly by forest ecosystems, which are the largest contributors to the total ecosystem services in the study area (by EUR 17.3 million or 75%). Interestingly similar findings in terms of the highest total ecosystem value of forests have been provided by Li et al. (2007). The results suggest that forests have the capacity to compensate for economic losses caused by land use changes. However, alternative (non-economic) indicators should be also accounted for. In the following chapter the analysis is developed further with respect to landscape metrics/environmental indicators to view this phenomenon.

Changes in carbon sequestration capacity

Similarly to the case study on agricultural land at national level, here the trend of the change in carbon sequestration service provision in Cezava from 1845 to 2010 is estimated and compared with the national situation (Figure 28).



Source: Author

Figure 28: Change in net carbon sequestration rate by ecosystems in Cezava

Net sequestration is the capacity of arable lands, grasslands and forests to fix carbon. In the case of Cezava the negative values of carbon sequestration of arable land pushes net sequestration into negative values. Even though forests and grasslands capture carbon, their representation in agricultural landscape does not outweigh the negative carbon values from dominant croplands. From a long-term point of view, net sequestration is increasing since 1948, however initial capacity of the study area to sequester carbon has not been reached again. The difference between initial carbon sequestration in 1845 and in 2010 is about 22% less carbon fixed (when 1845 = 100%).

When comparing trends in carbon sequestration by arable land and grasslannds at regional and national level (between 1948 and 2010), they show similarity in terms of increasing capacity to sequestre carbon since 1948. But meanwhile this trend is continual at the national level, carbon fixation by agricultural land in Cezava after increase slightly dropped in 2000 and then started to rise again. The difference between initial carbon sequestration in 1948 and in 2010 is approximately 35% more carbon fixed at national level and approximately 14% more carbon fixed in case of Cezava (when 1948 = 100%).

Changes in Proportions of Functional Groups of Ecosystem Services

The results of the landscape performance assessment illustrate Figure 29. It introduces the worksheets for the ratings based on ecosystem features and land use in 1845 and 2010, respectively. Three categories of ecosystem functions (ecological, production, cultural) contain four features, in total 12 features were evaluated for each ecosystem type. Based on the literature review and available knowledge, the ecosystems' features scores for arable land, grasslands, forests and water bodies were determined.

Particularly, in 1845 **arable land** was managed less intensively than today, and therefore a less negative impact on the erosion rate and water quality was indicated by the score. Also, the economic value was slightly lower (according to the results of economic analysis). Another feature, visual quality was importantly influenced by the transformation of small-scale land use to large-scale land use (Figure 25).

As stated in Chapter 4.3.1., reservoir desiccation and water stream regulation occurred after 1845. Hence, a higher score for wild life habitat (due to more natural character) and water quality (due to higher self-cleaning capacity) was attributed to **aquatic ecosystems** in 1845. Contrary to this, erosion was regarded as higher due to unregulated river banks around 1845. As the presence of water in the landscape is an important feature influencing landscape aesthetics (Frank et al., 2012), the aesthetic value of the study area in 19th century was considered to be higher than today.

Similarly, the character of **forests** has changed over the centuries. Today, spruce monocultures, which do not match the natural forest species composition, prevail (Chapter 1.2.). According to a growing number of studies, conifer-dominated plantations have lower capacity to provide potential benefits than mixed-species forests (Felton et al, 2010). Spruce monocultures show lower habitats quality for biodiversity, increased vulnerability to pests, pathogens and invasions, worsen soil conditions, and increase the risk of damage by wind and fire, (Felton et al., 2010, Nasi et al., 2002, Main-Knorn et al., 2009). The change of natural species composition was reflected in the lower score for wild life habitat in 2010 compared with 1845.

Even though the character of **grasslands** (meadows and pastures) and their management has been modified over the last 160 years, no specific information in this regard is available. Therefore, the same scores were attributed to meadows and pastures for both time periods.

Based on the scores, bar charts have been produced (Figure 30 and Figure 31). They indicate the relevant functional groups of features for each ecosystem type in two time periods. Consequently, categories of ecosystem services can be interpreted. Ecological features are presented by regulating services and, similarly, cultural features by cultural services. In case of production features, two of them represent provisioning ecosystem services - food and material production. In addition to the illustration of the (multi)functionality of ecosystems, the bar charts (widths of the columns) reflect the shares of ecosystem type in the study region.

1845

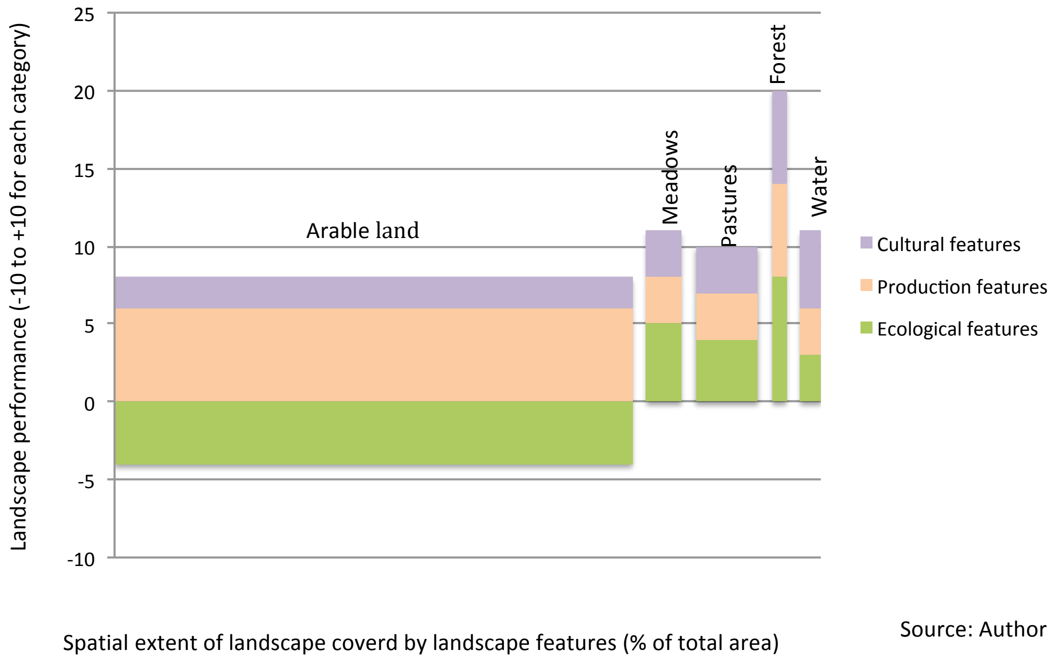
Functional Attributes (-2 to +2)	Arable land	Meadows	Pastures	Forest	Water	*TOTAL*
Ecological features						
Carbon seqvestration	-1	1	1	2	0	
Wild life habitat	-1	1	1	2	2	
Erosion control	-1	2	1	2	-1	
Water quality	-1	1	1	2	2	
TOTAL	*-4*	*5*	*4*	*8*	*3*	*16*
Production features						
Food production	2	0	0	1	1	
Material production	2	1	1	2	0	
Efficiency of input	1	1	1	1	1	
Economic value	1	1	1	2	1	
TOTAL	*6*	*3*	*3*	*6*	*3*	*21*
Cultural features						
Recreation	0	1	1	2	2	
Visual quality/Aesthetics	1	1	1	2	2	
Education/Research	1	1	1	2	1	
Living place	0	0	0	0	0	
TOTAL	*2*	*3*	*3*	*6*	*5*	*19*
Performance Sum	**4**	**11**	**10**	**20**	**11**	**56**

2010

Functional Attributes (-2 to +2)	Arable land	Meadows	Pastures	Forest	Water	*TOTAL*
Ecological features						
Carbon seqvestration	-2	1	1	2	0	
Wild life habitat	-1	1	1	1	1	
Erosion control	-2	2	1	2	0	
Water quality	-2	1	1	2	1	
TOTAL	*-7*	*5*	*4*	*7*	*2*	*11*
Production features						
Food production	2	0	0	1	1	
Material production	2	1	1	2	0	
Efficiency of input	1	1	1	1	1	
Economic value	2	1	1	2	1	
TOTAL	*7*	*3*	*3*	*6*	*3*	*22*
Cultural features						
Recreation	0	1	1	2	2	
Visual quality/Aesthetics	-1	1	1	2	2	
Education/Research	1	1	1	2	1	
Living place	0	0	0	0	0	
TOTAL	*0*	*3*	*3*	*6*	*5*	*17*
Performance Sum	**0**	**11**	**10**	**19**	**10**	**50**

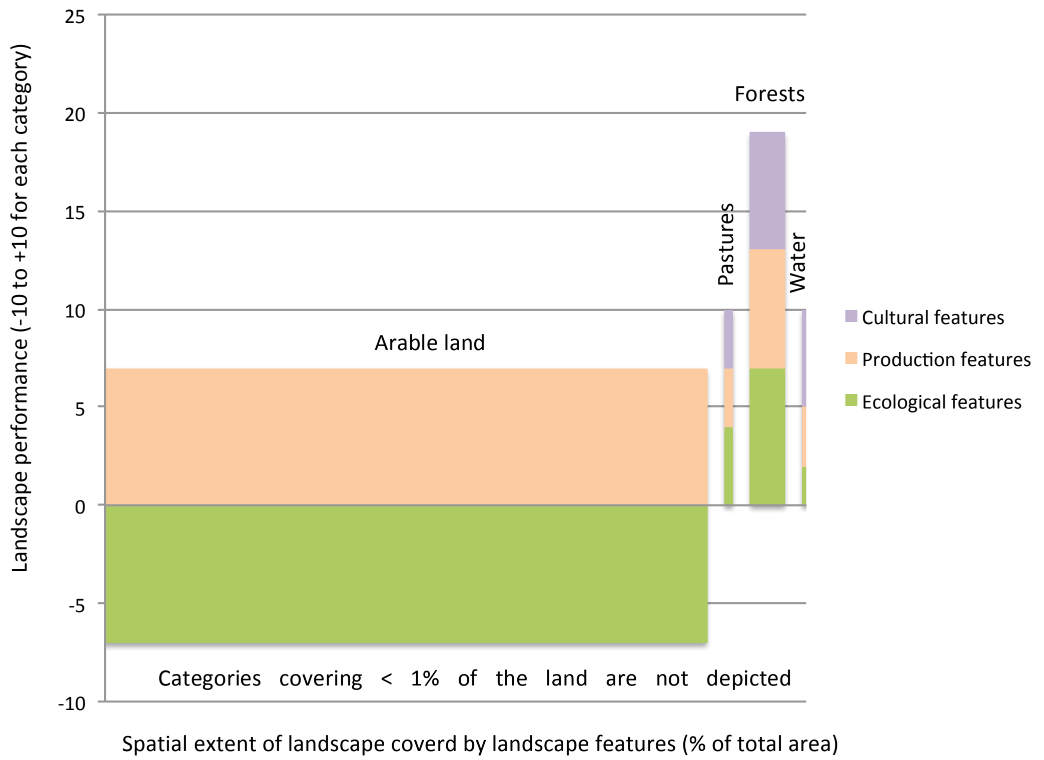
Compiled by the author based on Lovell et al., 2010

Figure 29: The scoring worksheets for functional attributes in 1845 and 2010



Source: Author

Figure 30: Performance of landscape featrues in 1845



Source: Author

Figure 31: Performance of landscape features in 2010

The results show that in general the highest production was performed by arable land, especially in 2010. Forests, followed by meadows, had the highest scores for the ecological functions. For cultural features, forests and water obtained the highest score. The most highly rated ecosystem performance sum was determined in case of forests in 1845 (20). Later, in 2010, it was one point less, but still the highest score. This indicates forests to be more multifunctional ecosystem compared to others. Negative scores were assigned only to ecological functional features of arable land (-4 in 1845 and -7 in 2010).

The findings also show that the level of the landscape multi-functionality has been reduced since 1845. Meanwhile all ecosystems performed all three functional features in 1845, cultural function of arable land was considerably reduced later in the observed period. In contrary, production on arable land is the only feature, which increased with time. Other features either remained at the same level or were reduced. Besides, the chart clearly illustrates trade-offs (conflicts), which exist between particular groups of functions or services. The most notable trade-off occurs between ecological and production function (regulating and provisioning services). This supports also the findings from the biophysical assessment, particularly the identification of several disservices such as reduced air quality and soil retention capacity or increased soil losses (Table 15). Another trade-off can be identified between provisioning and cultural services. While the production function remained the same in time, the cultural function of the ecosystems has been reduced.

4.3.4. The Environmental Problems Analysis

The findings on the agricultural land heterogeneity modifications are provided here, followed by a reflection on intensive management and its impact on the study area (in terms of levels of biodiversity loss, soil erosion and water pollution).

Results of landscape heterogeneity assessment indicate that the study area was the most uniform around 1948, when the value of SHDI reached lowest level (0.29). Since 1990 SHDI keeps on values around 0.5, which is even higher than in 1845 (SHDI 0.33). This is caused by reduction of the agricultural land area and increase of shares of other land use categories. However, another pattern is demonstarted by the agricultural land subcategory and related levels of SHDI. In 1845 the heterogeneity level was the highest (0.67). In 1948 SHDI of agricultural land was 0.26, later 0.36 (1990) and 0.35 (2000). Time horizon 2010 was not analysed because of changes in the classification of grasslands. Until 2000 all agricultural land use subcategories represented at the beginning of the period remain present (also meadows in terms of 0.14% in 2000) and the results can be interpreted as an slightly increasing evenness of the structure of agricultural land despite the dominant share of arable land (by about 76 – 77%). A proportion of arable land reaches almost the same share like in 1845, but the average field size has considerably increased, correspondingly to the national trend. According to data of the Ministry of Agriculture (2009), average size of fields reached about 0.2 ha in 1948 (before collectivisation) contrary to 20 ha average size of fields today. High land use intensity may also be demonstrated by minimal size of the area of natural land cover types (Frank et al., 2012). In Cezava, landscape segments under nature protection cover less than 1% of the study area.

Intensive management of the study area introduces several considerable environmental issues typical for agricultural regions in general. Firstly, the conversion of land from complex natural systems to simplified agricultural ecosystems is a major cause of the biodiversity loss (Flynn et al., 2009). Disappearance of many of the semi-natural and natural habitats is accompanied by animal species diversity reduction (Sundseth, 2009). Cezava is a region of major importance for birds. A decline in bird populations in the study area has already been observed and described. Like in other places of Czechia (and Europe), the number of farmland birds decreased by 50% approximately since 1980s (Voříšek et al., 2008 or Voříšek et al., 2009). Examples of the threatened bird species of today, but historically abundant in the study area are the Grey Partridge (*Perdix perdix*) and the Common Quail (*Coturnix coturnix*). Another species of interest, the Great Bustard (*Otis tarda*), occurred (even sporadically) in the study area since the 1900s. This species indicates diversified, species-rich habitats and heterogeneous and sustainable landscapes (Škorpíková, 2008). Its disappearance from the South Bohemia region is dated 1990s, as a consequence of further changes in crop selection and substantial reduction of areas sown by alfalfa (*Medicago sativa*) (Škorpíková, 2008). The decline in bird species populations or their absence even despite bird conservation indicates inappropriate landscape management (Sundseth, 2009). Except for biodiversity, reduced landscape heterogeneity means a monotonous landscape and limits the provision of cultural ecosystem services (Frank et al., 2012).

Soil erosion and water pollution are other environmental issues in the Cezava region (Havlíček and Navrátilová, 2005, Janeček et al., 2007). Most of the main water streams of the study area show chemical concentrations corresponding to polluted, heavily polluted or very heavily polluted water. The only exception with clean water is the Říčka brook (adapted from Pešoutová, 2007). Regarding soil erosion, an area of about 42% is threatened by soil erosion (municipalities Blučina, Nikolčice, Otnice, Těšany and Židlochovice) (Havlíček and Navrátilová, 2005). Arable land erosion rate is average aggregate of erosion levels on arable land in individual municipalities (2.7 $t.ha^{1}.y^{-1}$), while for grasslands national average rate by Krása (2010) was accounted (0.18 $t.ha^{1}.y^{-1}$).

Erosion on arable land and grasslands reached the lowest level in 1845 (29.8 thousand tons). The time of highest level of erosion rate was around 1948 (34.5 thousand tons), which correlates with the national case. After 1990, erosion level reached lower rates again, around 30.4 thousand tons. In 2010, annual soil loss was about 30.1 thousand tons (1% more than in 1845). Besides, soil quality has also changed. The dynamic properties of soil, soil structure and soil biodiversity were affected and resulted in soil degradation. Several areas in the Cezava region already miss a nutrient rich layer in a soil horizon. The main reason for soil degradation is that a large part of land reclaimed for agriculture is not suitable for intensification practices and heavy machinery use (Elgersma et al., 2008).

4.4. Discussion

In this study, an assessment of landscape dynamic processes by the use of multi-temporal land use data is presented. In combination with the concept of ecosystem services, this methodological approach helped to develop a framework reflecting changes in the landscape multi-functionality over about 160 years. Being the subject of the research, the Cezava region is a case study area of great agricultural importance, which shows symptoms of monofunctional over-use as demonstrated by environmental problems identification. From this perspective, its agricultural potential might be at risk in the future. Despite the Czech example presented as a case study, the applied approach and the results will hopefully be supportive for other studies dedicated to environmental impacts of long-term land use change in different regions. The following chapter discusses the methods, findings, and the related opportunities and limitations of this approach.

4.4.1. Land Use Change Analysis

The statistical data acquired from the LUCC Czechia database made it possible to analyse long-term land use changes in the period from 1845 to 2010. The categorisation of land use changes data into time intervals brings up several points for discussion. Firstly, the four highpoints within the 1845–2010 period do not divide the period into equally long intervals (but rather into 103, 42 and two 10-year long periods[11]). Such differences affect the comparability of the observed periods. On the other hand, despite being a shorter period of time, more intensive changes occurred between 1948 and 2010.

Secondly, the intensity of a change within a given period is difficult to predict – a change could have been quite sudden (e.g. within one year) or gradual (developing over several years or even decades). Another limitation lies in the fact that the statistical data are not spatially specific and do not provide qualitative information on ecosystems. This "black box" character of the gathered data limits their subsequent interpretation. For that reason, the statistical data were supplemented with historical maps from three Military Surveys and with aerial photos. The First Military Survey originates in times even beyond the period in the scope (from 1763-1785), the Second and the Third Military Survey maps originate in 1836-1852 and 1874-1880 and photos are from 1949 and 2000. Old maps and aerial photos provide relevant source data that enable assessments of landscape history (Engstová and Skaloš, 2009), although they (particularly aerial photos) show only arbitrary snapshots in time (Hietel et al., 2004). It is also important to consider the different explanatory values of these records. Old maps show particular parcels of land and their use, but they do not express information on landscape microstructure – if or how the parcel is further divided into smaller fields. The microstructure is observable from aerial photos, which provide a portrait of contemporary land cover, with a mosaic of different crops on small land parcels (Engstová and Skaloš, 2009). As such, pictorial data proved to be useful supplement to the statistical data. The last type of specific historical data resource, the chronicles, partly aided

[11] Year 1896 has also been recently added to the LUCC Czechia database, however it was not considered in this thesis.

in clarification of causes behind changes and helped shed light on ways that socio-political events have contributed to land use development.

Drawing upon available literature, major reasons for the change in land use patterns are population growth, collectivisation and production intensification followed by introduction of diverse agro-environmental schemes after 1990 and accession to the EU. In the national context, Cezava is one of the regions where an emphasis is placed on agricultural production. This prevailing function has been determined based on the distribution of land use categories in combination with ecosystems functional features assessment. Apart from favourable conditions, this trend is strengthened by reduction of the arable land (and agricultural land in general) in areas at higher altitudes and/or with less fertile soils. Drawing upon the classification of typological regions introduced by Bičík et al. (2010b), Cezava fits into the category of cultural landscape of lowlands and low-lying gently rolling regions. The key feature of such a region except agricultural production function is residential function.

4.4.2. Ecosystem Service Provision over Time

According to the results, provisioning services from arable land declined, but in fact current provision of food and raw material is higher (CSO, 2012). Statistical data on production are available from the beginning of the 20th century. Until 1920 crop yields at a national level did not exceed 3 million tons (contrary to 7 million tons produced today). This is due to modernisation of agriculture and additional energy inputs to agroecosystems. This analysis was primarily oriented on natural capacity of ecosystems to provide services, therefore this inconsistency appeared. Delivery of other services remains lower than it was at the beginning of the observed period in 1845. However, an improving tendency in ecosystem services delivery has been introduced after 1990. This can be explained by the new socio-political regime. Lautenbach et al. (2011) drew similar conclusions in a case study from East Germany and this trend might be typical for some more post-communist countries in Central Europe.

The method provides for simple identification of dominant trade-offs among ecosystem services. One weakness, however, is that data for the historical analysis were limited and one should be therefore careful with their interpretation. For example, details on forest species composition in the 19th century were not available, despite a complex search in several types of historical sources. The chronicles were expected to have the potential to contain the desired information, but they include primarily demographic or economic data, while ecological data are limited to extreme weather conditions. Therefore, the chronicles did not prove to be very relevant sources for this type of analysis, at least in this case study area. Because data restrictions are often a problem when analysing past or current ecosystem services (e.g. Costanza et al., 1997 or Gaodi et al., 2006, Bičík et al, 2010a), past ecosystem services were assessed based on the land use development and expert knowledge of political events and social life.

Current ecosystem services were discussed in more detail – qualitatively and when possible quantitatively – based on existing studies. Despite the availability of more consistent data for the assessment of present services, several uncertainties emerged (see Summarising Discussion, Chapter 5). One type of ambiguity is introduced by the data transfer from literature. For example, data on actual rates of carbon storage and sequestration are based on measurements taken from only a few forests, from which amounts of carbon for a given forest are then estimated. In reality, the actual amounts of carbon for a given forest type vary greatly depending on plant community composition, age structure, soil fertility and other local environmental conditions (Kremen et al., 2004). Another weakness could arise out of the assumption that the availability of ecosystem services is linear. In fact, the availability of services follows more sophisticated patterns than simple linear trends as the ecosystems are complex and dynamic (De Fries, 2004, Koch et al., 2009).

Changes in the provision of ecosystem services were discussed based on the development of economic value in time. However, a change is demonstrated by current values only relative to the acreage of an ecosystem. It is likely that the values developed in time differently as they are attributed by humans. On the other hand, such an analysis is impossible due to missing data sources. Therefore, even though the values are expressed with contemporary perspective, they allow a unified comparison throughout a long time period. But still, they should be viewed as special indicators rather than strict numbers.

Although the results of ecosystem services analysis revealed increasing value in economic terms, in terms of the multifunctional performance of the ecosystems some showed a decline. This might be seen as a contradiction. The forests importantly increase total value of ecosystem services. When the forests are not accounted for, a reduction in total value would appear (from EUR 16.4 million in 1845 to 14.5 EUR million in 2010). This confirms intensive land use on the most of the areas of Cezava. To broaden the context of the findings, a reflection on environmental problems of Cezava is provided and their potential impact on the services availability is discussed.

4.4.3. Implication of environmental problems on ecosystem services delivery

Shannon's Diversity Index on agricultural land heterogeneity and a development of average field size indicated intensive agriculture in the study area. The estimation of SHDI represents a formalized assessment enabling fast interpretation of particular land pattern development (Frank et al., 2012). In accordance with Frank et al. (2012), this exercise confirm usefulness of the combination of landscape metrics and ecosystem services even though landscape metrics are not able to evaluate all aspects of the service provision capacity of landscape.

In agroecosystems, many of the natural ecosystem functions are substituted by human labour or fertilisers (Swift et al., 2004). Reduction of biodiversity in these systems implies from the production intensification due to specific purpose to maximise production under minimal additional energy inputs. Effects of the impacts of agricultural intensification on biodiversity are still poorly understood, but it almost always results in fewer species with lower genetic variation and less functional groups (Swift et al., 2004). However, biodiversity is conditional for delivery of ecosystem services (MA, 2005) not only from agricultural ecosystems but also from all other ecosystem types which are not in isolation. Risks associated with lowered biodiversity may be possible production losses through reduced diversity of pollinators or increased soil erosion.

Increased levels of soil losses in turn threaten agricultural production and food safety, affect water quality (Otero et al., 2012) and contribute to soil degradation, which is considered as one of the key environmental problems (Robinson et al., 2009, Adhikari and Nadella, 2011, UK NEA, 2011). Changes in a quality of soil result in limited availability of other ecosystem services such as regulation of greenhouse gasses (Robinson et al., 2009). Because soil acts as a major sink for carbon and source of greenhouse gases (Elgersma et al., 2008, UK NEA, 2011), increased soil erosion potentially contributes to climate change and, consequently, to global changes (UK NEA, 2011). As demonstrated on the example of soil erosion, relationship between the provision of ecosystem services and environmental problems discussed in this study can be characterised as a downward spiral. The environmental problems negatively affect services provision and in turn, the limited capacities of ecosystems to deliver services aggravate the existing problems. For that reason, the complex interactions and trade-offs should be taken into account in landscape planning.

This case study has shown how land use and ecosystem services have changed in the period from 1845 to 2010 in Cezava, thus it answered RQ 6. Apart from partial biophysical and economic valuation, the scoring method evaluating functional features of the landscape and their changes was applied. Consideration of presented indicators of environmental problems (water quality, erosion, biodiversity loss) helped to assess the consequences of land use in a broader context. Therefore, their integration into the research framework makes the analysis more robust, reduces data unavailability limitations and clearly demonstrates side effects of intensive agricultural land use. The results of this case study also provided a feedback to the previous studies conducted at a national level.

5. Summarising Discussion

Topics specifically related to the case studies have already been discussed. Here, more general issues, commonalities and repeating patterns are reflected in the discussion.

5.1. Commonalities of Land Use and Ecosystem Services Concepts

From the theoretical framing, some consistencies between both concepts have emerged. First commonality is predetermined by the ultimate interest of both approaches that is human-nature interactions and related issues. Both disciplines are anthropocentric, where humans represent key influential agents. On the other hand, their approach to human–environment relations can be considered as fully symmetrical in sense of equally detailed attention paid to both people and other species as they interact (Castree et al., 2009). Secondly, the variety in terminology and in definitions is typical for both fields, however this can be regarded as common for many scientific disciplines, especially for those from the social branch. Thirdly, land use as well as ecosystem services represent interdisciplinary approaches and the concepts related provide methodologies and tools with a potential for integrated research.

Another resemblance can be traced based on scientometric analysis on land use and ecosystem services. Publication activities (indicating scientific interest or concern) in both fields have flourished recently. In the case of land use it was just after 1995, while in a case of ecosystem services a little later, after 2000, or 2005 respectively. Total number of annually issued studies in each of the fields was also comparable. Only about 550 studies made the difference in behalf of ecosystem services. Actually, the number of papers on ecosystem services was bigger in the observed period. This might be seen as an indicator of stronger scientific interest and more rapid development of the ecosystem services science. It is probably given by the fact that ecosystem services research is younger, and even though much has been done in the last 20 years, the concept and its paradigms is still being developed. In this sense, land use science can be considered as more firmly established.

The Combination of the Concepts

Requirements on combination of land use and ecosystem services researches have already been formulated in a number of papers (Frank et al., 2012, Lautenbach et al., 2012). The need also became tangible due to the development of activities, such as the establishment of a working group on mapping ecosystem services under the Ecosystem Services Partnership. The complementarity of the approaches is facilitated by said commonalities, especially due to common problem orientation and interdisciplinary nature of both methods. However, identifying a closer link between geography in general and ecosystem services science remains to be a challenge to this day. Kozak et al. (2011) claim that the application of geographical concepts to the analysis of ecosystem services and their values to society has lagged, meanwhile Ruhl et al. (2007) sees the limitation in the fact, that geography has not embraced the ecological economic concept of natural capital and ecosystem services. Evidently, there is untapped potential in both approaches.

Therefore, a combined approach based on ecosystem services analysis and land use analysis at diverse spatial and time scales has been made operational by the methodological framework applied in this thesis. It has shown that the combination of the methods enables original interpretation of human impacts on the environment. While the land use analysis is rather descriptive and presents the state as it exists (or existed), the ecosystem services assessment adds more analytical perspective. Resulting from the experience introduced in the three case studies, the combination of land use and ecosystem services analysis proved to be beneficial for several reasons. It implied that land use analysis provides meaningful data for the ecosystem services research. Moreover, the integration of methods further develops state data on land use by adding variables to be considered. Particularly the combination proved to be useful in description of land use impacts on the environment and in identification of the specific services significance and their importance for natural processes.

Another important benefit of the methods combination is the opportunity for spatial analysis of the provision of ecosystem services. Thanks to spatially explicit data, resulting maps illustrate distribution of the services or their values. The visualizations are generally assumed to be valuable tools for the interaction with stakeholders and decision-makers and can potentially help in the assessments of the costs related to the loss of such services (Lautenbach et al., 2012, De Groot, 2006). This would need to be verified for the Czech decision making process related to ecosystem services as it has not been done before. The thesis might provide relevant input data for this, even tough their adaptation according to the stakeholders preferences would be desired.

Despite the benefits implied by the integrated methodology, these go hand in hand with uncertainties and limitations. These will be discussed in the following chapters.

5.2. Land use analysis

The identification of modifications in land use over time, from 1845 until 2010, provided important background information for an assessment of the services provision. Here, several points for discussion related to land use analysis is introduced.

The LUCC Czechia Database was an important data resource enabled original interpretation of multiple impacts of land use changes during a long period. Although the database represents a substantial resource for landscape change analysis, some limitations were identified. The real situation in the field may differ from an assessment based on data from the database. One of the reasons for remaining data ambiguity may be a delay, introduced by landowners, in the registration of changes within the categories, mainly in the category of agricultural land (Bičík and Kupková, 2007). To reduce such data ambiguities, some additional resources were employed where available (e.g. historical maps, aerial photos). Some other constrains related to database utilization, e.g. irregular distribution of milestones along the time scale or black box character of statistic data, were pointed out and discussed in Chapter 4.4.1. (Analytical Part).

Land use change analysis was elaborated in two of the case studies on two scales. Hence it was possible to compare how specific trends in land use change at the regional level correspond with or differ from national trends. Taking into consideration general findings of Bičík et al. (2010b), compliance showed up for years from 1845 – 1948 – 1990. In the period from 1845 – 1948 transformation of grasslands into arable land and increase of built-up areas dominated at both levels. Later, from 1948 – 1990, the similarity with national trend rested in further development of built-up areas and remaining areas and decrease of agricultural areas. National data after 1990 are seen by Bičík et al. (2010b) as less reliable. Despite that, they revealed the tendency to conversion of arable land into grasslands. Suburbanisation has been partially captured in the data from LUCC Czechia Database and has been confirmed also from other sources (e.g. Sýkora and Mulíček, 2012). It occurred in Cezava as well. In accordance with Bičík et al. (2010b), an overview of land use development in the study areas indicated that socio-political development and natural disposition have been the major influential factors acting as the driving force behind land use changes over the past two centuries on both levels.

5.3. Ecosystem services analysis

This part of the discussion is going to reflect on the concept of ecosystem services, its application, benefits and limitations, and suggestions regarding the concept development for its use in future.

5.3.1. Critique of the concept

This thesis touched on several characteristics of the concept, which are being repeatedly questioned. They are specifically related to anthropocentric focus of the concept, its terminological diversity, ambiguous role of the biodiversity, and questionable purpose of economic valuation (Chapter 3 in Analytical Part). These plus some extra arguments have recently been introduced by a paper, which provided summary of the main critiques on the concept of ecosystem services. Schröter et al. (2014) have grouped the arguments into three types of consideration: ethical, political and scientific. Table 18 gives a synthesis of particular types of criticism and counter-arguments. The overview provides crucial essence of the criticism, which is very constuctive and useful for further improvements of the concept.

Counter-arguments can also be seen as explanatory statements how the concept of ecosystem services (even with its limitations) can enhance sustainable management of (natural) resources. Yet there are some other points not specifically discussed by Schröter et al. (2014), but worth to be considered. It is for example that economic valuation in addition enables conversion of diverse units to one common metric (e.g. carbon sequestration and erosion reates), which consequently facilitates weightening of decision options. Or, introduces common language for the debate with economists or politicians. What remains a challenge is to tackle a high portion of subjectivity, which can be introduced in the ecosystem services assessments. It relates to relative vagueness of the concept. However some steps can be taken to limit or at least reduce subjectivity involvement, e.g. by solid data and methods as in the case of application of filters in benefit transfer process or by additional consultations with experts from the field, etc. Because the critique of the concept closely relates to uncertainties, some of these points are going to be touched once again in the following chapters.

Table 18: Overview of the critique and counter-argumnets on the concept of ecosystem services

	Critique	Counter-arguments
Ethical consideration	Antropogenic focus.	The concept do not replace biocentric view, it rather provides antropocentric arguments for protection and sustainable human use of ecosystems.
	Exclusion of the intrinsic value of different entities in nature.	Intrinsic value is not exluded, it is encounted by cultural services (e.g. existance value).
	ES promote exploitative human-nature relationship and may turn people into consumers separated from nature.	Society has already become isolated from nature (mainly Western world). The concept provide holistic perspective, which could reconceptualize human-nature relationship. Besides, the concept communicates human dependance on nature.
Policy related consideration	Empirical proof of relationship between ES provision and biodiversity is weak.	Overlaps between biodiversity and ES have been acknowledged by empirical evidence, especiall on how biodiversity underpins provision of ES.
	ES are used as a conservational goal at the expense of biodiversity.	Several ES-based initiatives aim to broaden biodiversity conservation practices (e.g. REDD+ or CBD's Biodiversity 2020 targets).
	Valuation of ecosystem services comprises economic framing.	Monetary valuation helps to raise awareness about the importance of ES and highlights the undervaluation of externalities. It does not replace ethical, ecological, or other non-monetary arguments.
	Economic valuation commodifies nature.	Economic valuation is not necessarily connected to marketizaion, it rather can e.g. help to assess efficiency of policy instruments.
Scientific consideration	ES is vague and inconsistent concept ("catch-all" phrase).	The definition of ES is intentionally vague, thus appropriate for diverse ES assessments. ES definitions and classifications depend on the aim of the assessment. Moreover, the flexible boundaries of the object allows creativity and facilitates interdisciplinary cooperation and co-development of the research.
	The concept implies that all ES are good or desirable despite some ecosystems provide "disseriveces" (e.g. increased risks of diseases).	The "services" (or "goods" or "benefits") evoke positive association. They have been chosen intentionally, as the "services" are the reaserch interest.

Source: Compiled by author based on Schröter et al. (2014)

5.3.2. Ecosystem services assessment

Several uncertainties have repeatedly emerged within the assessments of ecosystem services in all three case studies. One of the constraints is introduced to assessments by inevitable originality of each and every ecosystem. Ecosystems are individual and they are likely to vary in their characteristics, such as area, integrity, type and age of ecosystem, species composition (Brander et al. 2010). Besides natural variability, management practices importantly affect the potential to deliver the services (Lorencová et al, 2013). Additionally, also beneficiaries of the services differ (number, income, preferences), as well as the context (availability of substitute and complementary sites and services) (Brander et al. 2010). All of these variables should be taken into account in a site-specific value estimation, but this is not always possible, most often because of data limitation. Another uncertainty is introduced by the assumption that ecosystem services are available linearly, whereas in reality they are delivered as non-linear as they are conditioned by highly dynamic processes in nature (Chapter 4.4.2., Analytical Part). From the methodological point of view, crucial constraints rest in relatively high portion of subjectivity of the author of the ecosystem service assessment. This is partly given by the vague character of the method as it was introduced by scientific considerations in the previous chapter, partly by data limitation.

This thesis has identified an availability of ecosystem services based on land use changes. Given the ever-increasing demands on ecosystems and the related reduction, fragmentation or degradation of said ecosystems (MA, 2005, Metzger et al., 2008, Opdam, 2006), it was presumed that the quantity of service provision is proportional to the area of the ecosystems in question. However, the area of the question ecosystem also represents an influential factor. As Brander et al. (2010) discussed, adding an additional unit of area to a large ecosystem increases the total value of ecosystem services less than an additional unit of area to a smaller ecosystem. Besides the area, other factors control the provision of services, ecological and social characteristics were included in the scoring (in a case of agricultural case study at the national level and Cezava study as well). However, unavailability of quantitative measures for the factors did not allowed their full consideration in weighing the overall impact. This would need to be developed by additional research.

Economic Valuation of Ecosystem Services

Apart from the general critique of the concept as a whole, its componential part, economic accounting is most often debated. It is due to its economic terminology, anthropocentric orientation and underestimation of biological principles (Burkhard et al., 2009). As Table 18 shows, such a critique is argued by solid counter-arguments in terms of its contribution to awareness raising or policy efficiency insurance. Also this type of valuation has a potential to present values of diverse ecosystem services in one common quantitative unit, which not possible in the case of biophysical assessments, and facilitates their comparability. Here again, ambiguity may be introduced. Even though the units are usually converted and standardized, it is not always possible or their modification might be another source of considerable errors. Therefore, valuation of ecosystem services should be regarded mainly as a communication tool introducing common language for different partners in the dialogue. Based on the valuation experience from this research supported by some other resources (e.g. De Groot et al., 2012), reported values should not be used as a basis for setting prices and they should not be treated as private commodities that can be traded in private markets.

Once the economic values are estimated, their following interpretation can often be complicated. As De Groot et al. (2012) show, values ranges are often large, may depend on the size or remoteness/accessibility of a given ecosystem and related utilisation rate by humans. To enhance correctness of values interpretation, additional statistical characteristics such as median values, minimal and maximal values and standard deviation errors are helpful. At any rate, any transfer or extrapolation of values between different sites or generalisation to a larger scale must be done with care (De Groot et al., 2012). Despite these limitations and constraints, economic valuation applied in this thesis proved to be a powerful tool for declaration of importance of ecosystems for human well-being and for consideration of externalities.

Apart from generally complicated adoption and interpretation of the resulting valuations by a wider audience, the valuation methods themselves are burdened by uncertainties. Again, data availability plays the key role. Because benefit transfer technique was the methodology employed for the value calculations in this thesis, detailed attention is paid to this method.

Benefit transfer

Despite the fact that the method of values transferring from studies already completed has been applied for more than 20 years, limitations in contemporary benefit transfer practice remain (Eigenbrod et al., 2010). However, developing research in the field of ecosystem services valuation increases the reliability of transfers across populations and sites (Brouwer and Spaninks, 1999). In the last decade, and especially after the MA 2005 publication, a number of studies have applied benefit transfer methodology (e.g. Porter et al., 2009, Liu et al., 2010, De Groot et al., 2012, Kubiszewski et al., 2013). Increasing the use of this method is also motivated by the development of the Ecosystem Services Development Database (ESVD) and the ECOSERV database.

Currently, there is a tendency in this area to unify the benefit transfer methodology (e.g. Wilson and Hoehn, 2006, Bergstrom and Taylor, 2006) and to conduct benefit transfer using GIS (Liu et al., 2010). Following these trends, two case studies in this thesis applied the method of benefit transfer similarly as related studies (e.g. Liu et al., 2010, De Groot et al., 2012). Also the use of GIS was included, as it enabled overlay analysis, geo-processing of input data and added spatial dimension of economic valuation. Such an approach is considered to be beneficial especially for the decision-making process and landscape planning (Troy and Wilson, 2006).

The accuracy of benefit transfer very much depends on factors such as the quality of original studies, the extent of measurements, and generalization error occurrence (Plummer, 2009). An assumption of the constant value of ecosystem service, neglecting spatial differences of habitat types or lack of measurements and potentially a poorly representative size of study sites used for extrapolations, are seen as key problematic issues (Eigenbrod et al., 2010, Nelson et al., 2010). Indeed, the case studies considered these limitations, particularly the inaccuracy introduced by the assumption of constant values. However, it is unfortunately not possible to eliminate these errors in the conditions of the current sources. This might be resolved in follow-up research, for example by development of a value function, which would enable consideration of additional variables representing context characteristics and specifics.

Another issue often mentioned in connection with the limitations of this method is that original studies may not contain all the information desirable for benefit transfer. If so, biases rest in unlike biophysical and socio-economic conditions that are not identical when comparing the original site to the study area (policy site) in the scope of physical research (Wilson and Hoehn, 2006). Such an ambiguity has been minimised by introducing specific filters, search strategies, and selections of suitable values only to be considered (strong values, Chapter 2.4.1., AnalyticalPart).

Valuation of ecosystem services in the past

Data unavailability represented a major limitation in analyses of ecosystem services in the past. Most often, many past or present assessments collect data on ecosystem services for the first time (Pereira et al., 2005). This is especially true for Czechia, where the research in terms of services assessment and mapping is in its infancy. Except for the case studies presented in this thesis, some other areas received attention in this term, fox example the Šumava National Park or protected landscape area Třeboňsko. In this regard, there is currently a growing national and international body of data with the potential for such assessments in the future. This thesis might provide a background and methodological guidelines for such a research.

Also, the values themselves change in time. It is because of transformation of economic conditions (e.g. inflation or other respective changes at the market), due to alterations in social attitudes and preferences, or because of modifications in availability of natural resources. Economic instruments enable technical standardisation of values in terms of common units or comparability of main economic indicators (as presented by the case studies). But other factors, especially social and natural resources related, are difficult to be accounted for. Brander et al. (2010) for example reports that when an ecosystem service becomes scarcer, its marginal and average values tend to increase. This opens a window of opportunity for further research, which might not only address the factor of scarcity (or abundance), but could also enable future projections of ecosystem services availability trends.

Given the linkage of services and the preferences and consumption by beneficiaries, unavailability of former consumers of the services represents another limitation in assessments of ecosystem services in the distant past. Thus it is rather coarse application of current experts knowledge on historical conditions. Alternative sources such as historical annual reports may shed light on the attitudes of our ancestors towards natural capital, but this is not always possible (information is not captured at all in a source or is not available on the desired geographic level). This knowledge gap might be closed or at least narrowed by experts from the field of ethnography or cultural anthropology. This confirms once again the interdisciplinary character of the ecosystem services concept.

The variability of conditions and preferences makes economic valuation of ecosystem services problematic and means that simply multiplying a constant per-unit value by the total quantity of ecosystem service provision is likely to underestimate the total value of a change (Brander et al., 2010). This is true for economic as well as for biophysical valuation, because of e.g. ecological succession or human interventions. Another dimension of underestimation is introduced by incomplete total economic value because many categories of ecosystems and ecosystem services are not included due to data limitations. Adding to this, a problem of underestimation is not relevant only for valuation of ecosystem services in the past, but is pressing in current or future assessments as well.

Need for combined valuations

Building on the difficulties related to economic valuation, De Groot et al. (2012) stress that importance of the ecosystems rests not only on their economic value, but no less importantly, on their ecological and socio-cultural value. Similar outcomes are in the case study focused on the Cezava region, which combined quantitative and qualitative assessment. The scale enabled more detailed analysis of additional characteristics as presented in landscape features assessment and consideration of environmental and cultural issues. Based solely on the economic assessment results, total value of the area increased in time, which might be seen as a positive trend. However, the results of the qualitative assessment showed that a higher economic value do not necessarily mean a higher ecological or social value. Such an aggregated assessment brings different results by showing more complex information. This confirms that values (and functions) of ecosystems are multidimensional and trade-offs needs to be considered. The case studies of this thesis showed that the main trade-offs occurred between provisioning versus regulating and cultural services.

5.3.3. Implication of land use changes on the provision of ecosystem services

The research presented in this thesis brings a multilevel perspective on ecosystem services assessment with respect to landscape/ecosystem type and impact of related land use change. Resembling patterns occurred in changes of long-term availability of ecosystem services at the regional and national level. The long-term analysis revealed noticeable dependency of the services provision on land use management. Based on this and some other studies (e.g. Rounsevell et al., 2012; Müller and Burkhard, 2012), ecosystem services confirmed to be useful as measurable indicators of the functioning and change of the land system, and therefore provide tools for management-relevant communication concerning recent, past or potential future states of human-environmental systems.

Moreover, understanding previous land use changes and related drivers behind them can be useful for extrapolations into future and more precise scenario development, although an isolation of conditions shaping past decisions still remains challenging (Nelson et al., 2010). Lorencová et al. (2013) combined past analysis with future projections that have been found valuable as the approach enabled simplification and translation of complex processes into a common robust framework.

6. Conclusions

This chapter synthetises findings of this thesis. Firstly, main aim, research questions, and the structure of the thesis are recapitulated. Then results of the case studies are presented and validity of three hypotheses formulated at the beginning of this thesis is reflected.

The thesis provided a theoretical basis for the landscape research and the concept of ecosystem services. The main aim of the thesis was to analyse the impacts of land use and land use changes on the provision of ecosystems services. The listed approaches have been combined into an integrated framework, which has been applied to the assessments of ecosystem services delivered by Czech ecosystems and the analysis of land use and land use change impacts on the ecosystem services provision.

The thesis was divided into two main parts, theoretical and analytical, and so can be the outputs. As the entity under observation was the landscape, the Theoretical Part reflected on the conception of landscape research in geography and landscape ecology. Geographical approach has been involved primarily in land use change analysis, while ecological, respectively ecosystem approach was engaged mainly during the ecosystem services analysis. The Theoretical Part was divided into three main chapters focusing the landscape and land use research, ecosystem services research and the integration of these two approaches. The description of the state of the art of land use and ecosystem services science provided an overview of current knowledge and methods being applied both internationally and in Czechia (RQ 1 and RQ 2). The concept of ecosystem services still represents less known approach in Czech scientific environment, therefore more detailed attention has been paid to it. Its introduction was more needed and brings some new insights. The last chapter in the Theoretical Part explains how land use analysis has been combined with ecosystem services analysis (RQ 3). The principal integration rests in the overlapping of the data sets to generate the (biophysical or economic) value of an ecosystem per hectare and to determine the effect of land use changes on these values.

The integrated methodological framework was made operational in the case studies presented in the Analytical Part of the thesis. Three case studies were conducted on diverse spatial and time scales. The ecosystem services assessment was based on the methodology as introduced in the Millennium Ecosystem Assessment (2005) and some previous studies such as Liu et al. (2010), or De Groot et al. (2012), while land use changes were analysed based on changes in proportions of individual land use categories in the way followed by applied by Bičík et al. (2010b). Crucial data sources were existing databases on ecosystem services (ESVD and EKOSERV databases) and the LUCC Czechia UK Prague database. Using the benefit transfer technique, ecosystems services being delivered specifically by the ecosystems in Czechia have been identified and valued.

The first case study answered RQ 4 and represented the first attempt at national-level ecosystem service assessment in Czechia and estimated the aggregate value of Czech ecosystems at around EUR 237 billion, which is approximately 1.5 times the Czech GDP (Frélichová et al., 2014). Further detailed attention was paid to agricultural ecosystems/region (RQ 5 and RQ 6). At the national level, availability of key ecosystem services provided by Czech arable land and grasslands has been set against wider land use trends. The regional analysis was done in a similar way. The national study looked specifically at three ecosystem services: carbon sequestration, erosion regulation and food provision. In the same context the services were checked at the regional level and the resulting trends were compared.

The change in capacity of agricultural systems to sequester carbon has shown to have similar trend on both levels in the period from 1948 to 2010. The capacity to sequester carbon was the lowest in 1948 and has been increasing since then. Agricultural land represents a source of carbon. Although the area of grasslands has increased at the national level and forests were added to the carbon balance at the regional level, they do not compensate for the negative carbon balance introduced by arable land. Erosion rate on arable land and grasslands in Czechia and Cezava seems to have been declining since 1948 as the area of arable land has been decreasing. Food provision, no matter on the observation level scale, can be more easily intensified today due to modernisation of agriculture and availability of extra energy inputs. Therefore, the changes in food production are related more to alterations in crop preferences, which are most often influenced by policies, subsidies, or global trade. Said conflicts between e.g. negative carbon sequestration balance and food provision or generally regulating and cultural services contrary to provisioning services are the trade-offs, which would need to be accounted by the decision-making. Except that this type of research is able to identify and quantify them if data allow, stakeholders should be addressed too to express their preferences.

The regional ecosystem assessment in Cezava showed that results of increased total economic value of the observed region do not necessarily mean an improvement of the ecological and/or cultural values. The total economic value increased primarily due to enlargement of forested area, which might be simultaneously regarded as being in favour of other functions. Despite that, the ecological and cultural functions have been reduced. Ecosystem services provision and landscape heterogeneity today show lower levels than in 1845. Thus, this case study highlighted the importance of combined assessment through qualitative and quantitative measures and biophysical and economic values, when it comes to the analysis of states and changes in human-environmental systems.

After the case studies, Summarising Discussion providing a reflexion on general issues implying from the research has been included. For the same reason as in the case of the concept introduction, more comprehensive critique of the concept of ecosystem services has been conducted. The case studies dealt with several limitations and uncertainties. However, it is commonplace for the interdisciplinary and integrated approach taken. Despite that the assessments provided innovative insights into the impact of long-term land use on ecosystem services in Czechia. The methodology may be used as a guideline for a long-term assessment of delivery of ecosystem services when the data for this kind of analysis are limited. As it has been shown by the case studies in this thesis (case study 2 and 3), such an assessment clarifies the effects of land use on the environment, identifies the significance of particular services, indicates their importance for natural processes, and can potentially help in the assessments of the costs related to the loss of such services. This research also demonstrates that it is possible to analyse long term land use trends to generate more meaningful, spatially explicit information, which can form the basis for landscape planning. The method additionally enables consideration of landscape structure, which is one of the aspects influencing ecosystem services delivery, though it is often ignored in many studies on ecosystem services. Even here, landscape structure was touched only marginally (in case study 3), therefore further research would be desirable to broaden the findings of this thesis.

The results allow to reflect on the validity of three hypotheses introduced by the thesis. The first hypothesis assumed that it is possible to quantify the value of ecosystem services in terms of biophysical and/or economic values. This hypothesis was found to be valid as it implies from the theoretical overview presented in the Theoretical Part by the chapters focused on the research of ecosystem services as well as from every case study presented in the Analytical part of the thesis. It is possible to conclude that while biophysical quantification is accepted, economic accounting is often debated.

Also the second hypothesis (the dependence of ecosystem services provision on biophysical conditions and land management) can be confirmed based on the findings of this research. The theoretical confirmation implied from literature review and was introduced e.g. in Chapter 4 of the Theoretical Part. Practical confirmation, especially of the impact of land management or land use, was provided by the case studies. Recognition of the interrelations between land use and ecosystem services holds the promise of more effective management interventions and the extension of the method's utilisation as an analytical tool.

Even the last hypothesis (the level of the provision of ecosystem services changes over time and space) has been confirmed in both, theoretical and analytical parts, of the thesis. The experience from this research showed that analysis of the changes in the provision of ecosystem services is easier along the spatial scale, while in the case of the time scale data scarcity often limits the investigation. Therefore only few studies on long-term changes in availability of ecosystem services have been done so far. From this point of view, the Czech LUCC UK Prague Database represents exceptional data source, which allowed somewhat original analysis based on land use changes in the given range of time. Understanding previous land use changes can be useful for extrapolations into the future and more precise scenario development, although isolation of the conditions shaping past decisions remains to be challenging (Nelson et al., 2010). Results of the spatial (and temporal) analysis of the changes can be used as a support tool for local land use management, or considered on the national scale for informing evidence-led policy decisions. For example, considerable value of ecosystems in Czechia should be taken into account in decision-making processes and management practices related to the natural environment.

Whilst this work has taken the example of the agricultural sector and agricultural region in Czechia, entailing in itself justifiable assumptions and limitations, its approach of applying spatial trend data could be applied to many other socio-economic sectors, such as urban areas, forestry or aquatic environments, and of course in other countries, especially those that have undergone socio-economic changes in landownership and management. Additional involvement of temporal scale in terms of long-term changes in land use can bring some interesting insights and introduce broader context of their impacts.

To conclude, this thesis showed that the use of ecosystem services highlights the significance of ecological processes beneficial as services for society. Besides the knowledge applicability in sustainable environmental management, further research can develop the understanding of the role of ecosystems in safety of social and natural systems with respect to increasing risks related to the regional, national or even global environmental change.

7. References

ASPINALL, R.J., HILL, M.J. (eds) (2008): Land Use Change. Science, Policy and Management. Boca Raton, FL, USA, xxvii + 185 s.

ADHIKARI, B. and NADELLA, K. (2011): Ecological economics of soil erosion: a review of the current state of knowledge in “Ecological Economics Reviews.” Robert Costanza, Karin Limburg & Ida Kubiszewski, Eds. Ann. N.Y. Acad. Sci. 1219: 134 – 152.

ATKINS, P., SIMMONS, I., BRIAN ROBERTS, B. (1998): People, Land and Time. Taylor Francis Ltd, United Kingdom.

BAGSTAD, K.J., VILLA, F., JOHNSON, G.W., AND VOIGT, B. (2011): ARIES – Artificial Intelligence for Ecosystem Services: A guide to models and data, version 1.0. ARIES report series n.1. (http://www.ariesonline.org/docs/ARIESModelingGuide1.0.pdf).

BALEJ, M. ANDĚL, J., ORŠULÁK, T. (2007): Environmental Sterssors and Sterss as a Natural/Humna Interdisciplinary Issue: A Case Study from Northern Part of the Czech Republic. Acta Universitatis Carolinae. Geographica 1-2: 53–74.

BALEJ, M. (2012): Using of Landscape Metrics from the view of Central-European Landscape Ecological Approach. Prešov University in Prešov (Prešov, Slovakia). Faculty of Humanities and Sciences. Department of Geography and Regional Development. 161 pp.

BATEMAN, I.J., MACE, G.M., FEZZI, C., ATKINSON, G., TURNER, K. (2011): Economic Analysis for Ecosystem Service Assessments. Environmental and Resource Economics 48:177–218.

BEBBINGTON, A. (1999): Capitals and Capabilities: A Framework for Analyzing Peasant Viability, Rural Livelihoods and Poverty. World Development 27(12): 2021–2044.

BERGANDI, B., BALANDIN, P. (1998): Holism vs. Reductionism: Do Ecosystem Ecology and Landscape Ecology Clarify the Debate? Acta Biotheoretica 46 (3): 185–206.

BERGSTROM, J.C., TAYLOR, L. O. (2006): Using Meta-analysis for Benefit Transfer: Theory and Practice. Ecol. Econ. 60: 351–360.

BIČÍK, I. (1995): Possibilities of Long-Term Human – Nature Interaction Analysis: The Case of Land-Use Changes in the Czech Republic. In: Simmons, I. G., Mannion, A. (Eds.): The Changing Nature of the People-Environment Relationship: Evidence from a Variety of Archives. Charles University of Prague, Faculty of Science, Department of Social Geography and Regional Development, Prague.

BIČÍK, I., JELEČEK, L. AND ŠTĚPÁNEK, V. (2000): Major Types of land Use Changes in the Czech Republic 1845–1848–1990. In: Hwang, M. and Himiyama, Y. (ed): Land Use and Land Cover Change: The Contribution of Geography. Journal of Geography Education, 43. Special Issue for the IGU Study Group on Land Use and Land Cover Change. 29th International Geographical Congress, Seoul, Korea, 14–18 August 2000.

BIČÍK, I., CHROMÝ, P., JANČÁK, V., JELEČEK, L., KUPKOVÁ, L., ŠTĚPÁNEK, V., WINKLEROVÁ, J. (2001a): Land Use/Land Cover Changes in Czechia over the past 150 Years – An Overview. In: Himiyama, Y., Mather, A., Bičík, I., Milanova, E.V. (eds): Land Use/Cover Change in Selected Regions in the World. Vol. I, Part IV, Issued by IGU SG LUCC. IGU-LUCC Research Reports IL-2001-01, Institute of Geography, Hokkaido Univ. of Education, Asahikawa, s. 29–39.

BIČÍK, I., JELEČEK L., ŠTĚPÁNEK, V. (2001b): Land-use Changes and Their Social Driving Forces in Czechia in the 19th and 20th Centuries. Land Use Policy 18, 65–73.

BIČÍK, I., KUPKOVÁ, L. (2007): Land Use Development in the Czech Republic and Possibilities of Generalization and Modelling. In: Dostál, P., Langhammer, J. (ed): Modelling Natural Environment and Society, P3K, Praha.

BIČÍK, I., ANDĚL, J. AND BALEJ, M. (2010a): Landscape Function Transformations with Relation to Land Use Changes. In: Anděl, J. et al. (ed): Landscape Modelling. Urban and Landscape Perspectives 8. Springer Science + Business Media B.V.

BIČÍK I., JELEČEK, L., KABRDA, J., KUPKOVÁ L., LIPSKÝ, Z., MAREŠ, P, ŠEFRNA, L., ŠTYCH, P., WINKLEROVÁ, J. (2010b): Vývoj využití ploch v Česku. Česká geograická společnost. Edice Geographica, Praha, 250 s.

Bičík, I., Kupková, L., Štych, P. et al., 2012. Změny funkcí České krajiny a jejich dopad na využití ploch. In: Chodějovská, E., Šimůnek, R. (eds.) Krajina jako historické jeviště. K poctě Evy Semotanové. Historický ústav Akademi věd ČR, v.v.i., Praha.

BOCKSTAEL, N. E., FREEMAN, A.M., KOPP, R.J., PORTNEY, P.R., SMITH, V.K. (2000): On Measuring Economic Values for Nature. Environmental Science and Technology 34:1384-1389.

BOLUND, P. and HUNHAMMAR, S. (1999): Ecosystem Services in Urban Areas. Ecological Economics 29 (6): 293–301.

BOUCNÍKOVÁ, E., FANTA, J., LÍŠKOVÁ, Z. (2006): Úloha krajinného plánování v systémovém řízení vývoje a využívání krajiny. In: Dreslerová, J., Packová, P. (eds.) Ekologie krajiny a krajinné plánování. Sborník ekologie krajiny 2. Sborník příspěvků z konference CZ-IALE, 14. – 16. září 2006 v Lednici. 250 s.

BRANDER, L. GHERMANDI, A., KUIK, O., MARKANDYA, A., NUNES, P.A.L.D., SCHAAFSMA, M., WAGTENDONK, A. (2010): Scaling up Ecosystem Values: Methodology, Applicability and a Case Study. Nota di Lavoro 41.2010. FondazioneEni Enrico Mattei. Milano.

BRAUN, B. (2009): Nature. In Castree, N. (ed.). A Companion to Environmental Geography. Wiley-Blackwell, London.

BROUWER, R. AND SPANINKS, F.A. (1999): The Validity of Environmental Benefits Transfer: Further Empirical Testing. Environ. Resour. Econ. 14: 95–117.

BURKHARD, B., KROLL, F., MÜLLER, F., AND WINDHORST, W. (2009): Landscapes‘ Capacities to Provide Ecosystem Services – a Concept for Land-Cover Based Assessments. Landscape Online 15, 1–22.

BURKHARD, B., KROLL, F., NEDKOV, S., MÜLLER, F. (2012): Mapping supply, demand and budgets of ecosystem services. Ecol. Indicat. 21: 17–29.

BUSCH, G. (2006): Future European Agricultural Landscapes—What Can We Learn from Existing Quantitative Land Use Scenario Studies? Agriculture, Ecosystems and Environment 114, 121–140.

BUSCH, M., LA NOTTE, A., LAPORTE, V., ERHARD, M. (2012): Potentials of Quantitative and Qualitative Approaches to Assessing Ecosystem Services. Ecological Indicators 21: 89–103.

CASTREE, N. DEMERITT, D., LIVERMAN, D. AND RHOADS, B. (2009): A Companion to Environmental Geography. Wiley-Blackwell, London.

COM (2011): Communication from the Commission to the European Parliament, the Council, the Economic and Social Committee and the Committee of the Regions: Our life insurance, our natural capital: an EU biodiversity strategy to 2020.

COSTANZA, R., D’ARGE, R., DE GROOT, D., FARBER, S., GRASSO, M., HANNON, B., LIMBURG, K., NAEEM, S., O’NEILL, R., PARUELO, J., RASKIN, R.G., SUTTON, P. AND VAN DEN BELT, M. (1997): The Value of the World's Ecosystem Services and Natural Capital. Nature 387: 253–260.

COSTANZA, R., D’ARGE, R., DE GROOT, D., FARBER, S., GRASSO, M., HANNON, B., LIMBURG, K., NAEEM, S., O’NEILL, R., PARUELO, J., RASKIN, R.G., SUTTON, P. AND VAN DEN BELT, M. (1998): Special Section: Forum on Valuation of Ecosystem Services: The Value Of Ecosystem Services: Putting The Issues In Perspective. Ecological Economics 25: 67–72.

COSTANZA, R. (2008): Ecosystem Services: Multiple classification systems are needed. Biological Conservation. 141: 350–352.

DAILY, G. (ed.) (1997): Nature’s Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, DC.

DALE, V.H., POLASKY, S. (2007): Measures of the effect of agricultural practices on ecosystem services. Ecological Economics 64, 286–296.

DANIELS S. AND COSGROVE D. (1988): Introduction: Iconography and Landscape, in Cosgrove, D. and S. Daniels,. The Iconography of Landscape: essays on the symbolic representation, design, and use of past environments. Cambridge Studies in Historical Geography, Cambridge University Press, Cambridge.

DE FRIES, R., ASNER, G.P. AND HOUGHTON, R. (2004): Trade-Offs in Land-Use Decision: Towards a Framework for Assessing Mulitple Ecosystem Responces to Land-Use Change. In: De Fries, R., Asner, G.P. and Houghton, R. (eds.): Ecosystems and Land Use Change. Americal Goephysical Union. Washington, DC.

DE GROOT, R. (1992): Function of nature: evaluation of nature in environmental planning, management and decision making. Amsterdam: Wolters-Noordhoff. pp. xviii + 315 pp.

DE GROOT, R. (1994): Environmental functions and the economic value of natural ecosystems. Investing in Natural Capital: The Ecological Economics Approach to Sustainability. Island Press, International Society for Ecological Economics. 151 –168 pp.

DE GROOT, R.S. (2006): Function-analysis and Valuation as a Tool to Assess Land Use Conflicts in Planning for Sustainable, Multi-functional Landscapes. Landscape Urban Plan 75: 175–186.

DE GROOT, R.S., WILSON M.A. AND BOUMANS R.M.J. (2002): A Typology for the Classification, Description and Valuation of Ecosystem Services, Goods and Services. Ecol. Econ. 41: 393–408.

DE GROOT, R.S., ALKEMADE, R., BRAAT, L., HEIN, L., WILLEMEN, L. (2010): Challenges in Integrating the Concept of Ecosystem Services And Values in Landscape Planning, Management and Decision Making. Ecological Complexity 7: 260 –272.

DE GROOT, R.S., BRANDER, L., VAN DER PLOEG, S., COSTANZA, R. ET AL. (2012): Global Estimates of the Value of Ecosystems and their Services in Monetary Units. Ecosyst. Serv. 1: 50–61.

DOUGLAS, I., HUGGETT, R.J., ROBINSON, M.E. (1996): Companion encyclopedia of geography: The environment and humankind. ROUTLEDGE.

EEA Technical Report No. 25, (1999): European Environment Agency. http://www.eea.europa.eu/publications/TEC25.

EIGENBROD, F., ARMSWORTH, P.R., ANDERSON, B.J., HEINEMEYER, A., GILLINGS, S., ROY. D.B., THOMAS, C. D., GASTON, K.J. (2010): Error Propagation Associated with Benefits Transfer-Based Mapping of Ecosystem Services. Biol. Conserv. 143: 2487–2493.

ELGERSMA, A.M., DHILLON, S.S., ARNOLDUSSEN, A., FANTA, J. AND BOUCNÍKOVÁ, E. (2008): Understanding the Quality of Land in Agricultural Land Use Systems. Sustainable Land Management: Strategies to Cope with the Marginalisation of Agriculture. Edward Elgar Publishing, Cheltenham, UK, Northampton, MA, USA.

ENGSTOVÁ, B. AND SKALOŠ, J. (2009): Methodology for mapping non-forest wood elements using historic cadastral maps and aerial photographs as a basis for management. Journal of Environmental Management 91: 831–843.

EUROPEAN COMISSION (2002): Europe’s Environment: The Third Assessment Report from EEA: Europe's environmental progress at risk from unsustainable economic activities.

EUROSTAT, 2007. Agricultural Statistics: Data 1995–2005. Office for Official Publications of the European Communities, Luxembourg. ISBN 92-79-02955-X.

FAHRIG, L., BAUDRY, J., BROTONS, L., BUREL, F., CRIST, T.O., FULLER, R.J., SIRAMI, C., SIRIWARDENA, G.M., MARTIN, J. (2011): Functional Landscape Heterogeneity and Animal Biodiversity in Agricultural Landscapes. Ecology Letters 14: 101–112.

FALLOON, P. AND BETTS, R. (2010): Climate Impacts on European Agriculture and Water Management in the Context of Adaptation and Mitigation - The Importance of an Integrated Approach. Science of the Total Environment 408, 5667–5687.

FARINA, A., JOHNSON, A.R., TURNER, S.J., BELGRANO, A. (2003): 'Full' world versus 'empty' world paradigm at the time of globalisation. Ecological Economics 45 (1): 11–18.

FELTON, A., LINDBLADH, M., BRUNET, J. AND FRITZ Ö. (2010): Replacing coniferous monocultures with mixed-species production stands: An assessment of the Potential Benefits for Forest Biodiversity in Northern Europe. Forest Ecology and Management 6: 939–947.

FISHER, B. AND TURNER, E.K. (2008): Ecosystem Services: Classification for Valuation. Biological Conservation 141, 1167–1169.

FLYNN, D.F.B, GOGOL-PROKURAT, M., NOGEIRE, T., MOLINARI, N., TRAUTMAN RICHERS, B., LIN, B.B., NICHOLAS, S., MAYFIELD, M.M. AND DECLERC, F. (2009): Loss of Functional Diversity under Land Use Intensification Across Multiple Taxa. Ecology Letters 12: 22–33.

FOLEY, J.A., DEFRIES, R., ASNER, G.P., BARFORD, C., BONAN, G., CARPENTER, S.R., CHAPIN, F.S., COE, M.T., DAILY, G.C., GIBBS, H.K., HELKOWSKI, J.H., HOLLOWAY, T., HOWARD, E.A., KUCHARIK, J., MONFREDA, C., PATZ, J.A., PRENTICE, I.C., RAMANKUTTY, N., SNYDER, P.K. (2005): Global Consequences of Landuse. Science 309: 570–574.

FORMAN, T. T.J. AND GODRON, M. (1986): Landscape Ecology. John Wiley and Sons, Inc. Canada.

FRANK, S., FÜRST CH., KOSCHKE, L. AND MAKESCHIN, F. (2012): A Contribution Towards a Transfer of the Ecosystem Service Concept to Landscape Planning Using Landscape Metrics. Ecol. Indic. 21: 30–38.

FRÉLICHOVÁ, J., VAČKÁŘ, D., PÁRTL A., LOUČKOVÁ, B., HARMÁČKOVÁ, Z.V., LORENCOVÁ, E. (2014): Integrated Assesment of Ecosystem Services in the Czech Republic. Ecosystem Services (in press).

GABROVEC, M., PETEK, F., KLADNIK, D. (2001): Land Use Changes in the 20th Century in Slovenia. In: Himiyama, Y. a kol. (eds): Land Use/Cover Changes in Selected Regions in the World – Volume I. IGU-LUCC Research Reports IL-2001-01, Japonsko, p. 41–52.

GAODI, X., YU, X., CHUNXIA, L. (2006): Study on Ecosystem Services: Progress, Limitation and Basic Paradigm. Journal of Plant Ecology 30: 191–199.

GEO 5 (Global Environmental Outlook 5) 2012. United Nations Environment Programme.

GUO, L.B. AND GIFFORD, R.M. (2002): Soil Carbon Stocks and Land Use Change: a Meta-Analysis. Global Change Biology 8: 345–360.

HAINES-YOUNG, R. AND POTSCHIN, M. (2010): The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D.G. and Frid, C.I.J. (eds.): Ecosystem Ecology: a new synthesis. Cambridge University Press, BES, p.110–139.

HAMPL, M. (1998): Realita, společnost a geografická organizace: hledání integrálního řádu. Přírodovědecká fakultak Univerzity Karlovy, Praha. 110 p.

HAMPL, M. (2002): Regioníální organizace společnosti: Principy a problémy studia. Geografie-sborník české geografické společnosti 107 (4).: 333–348.

HAVLÍČEK T. AND NAVRÁTILOVÁ M. (2005): Nová krajina regionu Cezava – Koncepce revitalizace krajiny na katastrech členských obcí. Atelier Fontes, Brno.

HAYGARTH, P.M. AND RITZ, K. (2009): The Future of Soils and Land Use in the UK: Soil Systems for the Provision of Land-based Ecosystem Services. Land Use Policy 26S: S187–S197.

HEIN, L., VAN KOOPEN, K., DE GROOT, D., VAN IERLAND E.C. (2006): Spatial Scales, Stakeholders and the Valuation of Ecosystem Services. Ecological Economics 57, 209–228.

HERBST, H., FÖRSTER, M., KLEINSCHMIT B. (2009): Contribution of Landscape Metrics to the Assessment of Scenic Quality – the Example of the Landscape Structure Plan Havelland/Germany. Landscape Online 10: 1-17.

HIETEL, E., WALDHARDT, R., OTTE, A. (2004): Analysing Land-cover Changes in Relation to Environmental Variables in Hesse, Germany. Landscape Ecology 19: 473–489.

HIMYIAMA, Y., MATHER, A., BIČÍK, I., MILANOVA, E. V. (eds.), (2005): Land use/cover changes in selected regions of the world. IGU Study Group on LandUse/Cover Change, Hokkaido University of Education.

HÖNIGOVÁ, I., VAČKÁŘ, D., LORENCOVÁ, E., MELICHAR, J., GÖTZL, M., SONDEREGGER, G., OUŠKOVÁ, V., CHOBOT, K., HOŠEK, M. (2011): Survey on grassland ecosystem services. Nature Conservation Agency of the Czech Republic, Prague, 78 pp., ISBN 978-80-87457-25-2.

HRNČIAROVÁ T., MACKOVČIN, P., ZVARA, I. (eds.), (2010): Atlas krajiny České republiky, MŽP Průhonice, VÚKOZ, 332 s.

JANEČEK, M. (2007): Agricultural Land Erosion Protection – Methodology (In Czech: Ochrana zemědělské půdy před erozí – Metodika). VÚMOP, v.v.i., Praha.

JANSSENS, I.A., FREIBAUER, A., SCHLAMADINGER, B., CEULEMANS, R., CIAIS, P., DOLMAN, A.J., HEIMANN, M., NABUURS, G.J., SMITH, P., VALENTINI, R., SCHULZE, E.D. (2005): The carbon budget of terrestrial ecosystems at country-scale - a European case study. Biogeosciences 2 (1): 15–26.

JELEČEK, L. (2002): Historical Development of Society and LUCC in Czechia 1800-2000: Major Societal Driving Forces of Land Use Changes. In: Bičík, I. a kol. (eds): Land Use/Land Cover Changes in the Period of Globalization. Proceedings of the IGU-LUCC International Conference, Prague, 2001. KSGRR PřF UK, Praha, s. 44-57.

JELEČEK, L. (2007): Environmentalizace vědy, geografie a historické geografie: environmentální dějiny a výzkum změn land use Česka v 19. a 20. století. Klaudyán 4 (1): 20–28.

JOHNSTON, R.J., GREGORY, D., PRATT, G., WATTS, M. (2001): The Dictionary of Human Geography. 4th Edition, 958 pp., Blackwell, Oxford.

KABRDA J. (2008). The changing spatial structure of agricultural land use in Czechia since the mid-19th century. Geografický časopis 60 (3): 255-278.

KAVAL, P. AND BASKARAN, R. (2013): Key Ideas and Concepts from Economics for Understanding the Roles and Value of Ecosystem Services. In: Wratten, S., Sandhu, H., Cullen, R. And Costanza, R. (eds.) Ecosystem Services in Agricultura and Urban Landscapes, First Edition. John Willey and Sons Ltd.

KOCH, E.W., BARBIER, E.B., SILLIMAN, B.R., REED, D.J., PERILLO, G.M.E., HACKER, S.D., GRANEK, E.F., PRIMAVERA, J.H., MUTHIGA, N., POLASKY, S., HALPERN, B.S., KENNEDY, C.J., KAPPEL, C.V., WOLANSKI, E. (2009): Non-linearity in Ecosystem Services: Temporal and Spatial Variability in Coastal Protection. Frontiers in Ecology and the Environment 7(1): 29–37.

KOSTROWICKI, J. (1974): The typology of world agriculture. Principles, methods and model types. International Geographical Union, Commission on Agricultural Typology, Warsaw.

KOZAK, J., LANT, CH., SHAIKH, S., WANG, G. (2011): The geography of ecosystem service value: The case of the Des Plaines and Cache River Wetlands, Illinois. Applied Geography 31(1): 303–311.

KRÁSA, J. (2010): Empirical Models for Water Erosion in Czech Republic (In Czech: Empirické modely vodní eroze v ČR). Habilitation Thesis. Department of Irrigation, Drainage and Landscape Engineering, Czech Technical University in Prague.

KREMEN, C.W., WILLIAMS, N.M., BUGG, R.L., FAY, J.P., THORP, R.W. (2004): Total Area Requirements of an Ecosystem Service: Crop Pollination by Native Bee Communities in California. Ecology Letters 7: 1109–1119.

KRIEG M. L. (2001): A Tutorial on Bayesian Belief Networks. DSTO Electronics and Surveillance Research Laboratory, Australia. (http://dspace.dsto.defence.gov.au/dspace/bitstream/1947/3537/1/DSTO-TN-0403.pdf)

KUBISZEWSKI, I., COSTANZA, R., DORJI, L., THOENNES, P., TSHERING, K. (2013): An Initial Estimate of the Value of Ecosystem Services in Bhutan. Ecosystem Services 3: e11–e21.

KUČERA, Z. (2009): Krajina v české geografii a otázka relevance přístupů anglo-americké humánní geografie. Geografie –Sborník české geografické společnosti 2: 145–154.

KUPKOVÁ, L. (2001): Analýza vývoje české kulturní krajiny v období 1845-2000. Doktorská dizertační práce, PřF UK, Praha, 218 s.

LAMARQUE, P., QUÉRTIER, F., LAVOREL, S. (2011): The Diversity of the Ecosystem Services Concept and its Implications for their Assessment and Management. Comptes Rendus Biologies 334(5-6):441–449

LAMBIN, E.F. ET AL. (2001): The Cause of Land-use and Land-cover Change: Moving Beyond the Myths. Global Environmental Change 11: 261–269.

LAMBIN, E.F. AND GEIST, H.J. (eds.), (2006): Land-Use and Land-Cover Change: Local Processes and Global Imapcts. Springer-Verlag Berlin Heidelberg. Germany.

LANGHAMMER, J. AND VILÍMEK, V. (2008): Landscape changes as a factor affecting the course and consequences of extreme floods in the Otava river basin, Czech Republic. Environmental monitoring and assessment 144: 53–66.

LANGHAMMER, J. (2009): Applicability of hydromorphological monitoring data to locate flood risk reduction measures: Blanice River basin, Czech Republic. Environmental monitoring and assessment 152: 379–392.

LAUTENBACH, S., KUGEL, C., LAUSCH, A., SEPPELT, R. (2011): Analysis of Historic Changes in Regional Ecosystem Services Provisioning Using Land Use Data. Ecological Indicators 11: 676–687.

LAUTENBACH, S., VOLK, M., GRUBER, B., DORMANN, C.F., STRAUCH, M., SEPPELT, R. (2012): Quantifying Ecosystem Service Trade-offs. International Environmental Modelling and Software Society (iEMSs) 2012 International Congress on Environmental Modelling and Software Managing Resources of a Limited Planet, Sixth Biennial Meeting, Leipzig, Germany. R. Seppelt, A.A. Voinov, S. Lange, D. Bankamp (Eds.)

LEE, J.J., PHILLIPS, D.L., BENSON, V.W. (1999): Soil erosion and climate change: Assessing potential impacts and adaptation practices. Journal of Soil and Water Conservation 54 (3): 529-536.

LI R.Q., DONG M., CUI J.Y., ZHANG L.L., CUI Q.G., HE W.M. (2007): Quantification of the Impact of Land Use Changes on Ecosystem Services: A Case Study in Pingbian County, China. Environmental Monitoring and Assessment 128: 503–510.

LIPSKÝ, Z., KOPECKÝ, M., KVAPIL, D. (1999): Present land use changes in the Czech cultural landscape. Ekológia (Bratislava) 18: 31–38.

LIPSKÝ Z. (2000): Observation of Changes in the Cultural Landscape (in Czech: Sledování změn v kulturní krajině). Textbook for seminars in Landscape ecology. Czech University of Life Sciences in Prague, Department of Applied Ecology Kostelec nad Černými lesy.

LIU, S., COSTANZA, R., TROY, A., D'AAGOSTINO, J.D., MATES, W. (2010): Valuing New Jersey's Ecosystem Services and Natural Capital: A Spatially Explicit Benefit Transfer Approach. Environmental Management 45: 1271–1285.

LOBELL, D.B. AND FIELD, C.B. (2007): Global scale climate–crop yield relationships and the impacts of recent warming. Environmental Research Letters 2 (1). DOI:10.1088/1748-9326/2/1/014002.

LOOMIS, J., KENT, P., STRANGE, L., FAUSCH, K., COVICH, A. (2000): Measuring the Total Economic Value of Restoring Ecosystem Services in an Impaired River Basin: Results from a Contingent Valuation Survey. Ecol Econ 33: 103–117.

LOREAU, M., NAEEM, S., INCHAUSTI, P., BENGTSSON J.,. GRIME, J.P., HECTOR, A., HOOPER, D.U., HUSTON, M.A., RAFFAELLI, D., SCHMID, B., TILMAN, D.,. WARDLE, D.A. (2001): Biodiversity and Ecosystem Functioning: Current Knowledge and Future Challenges. Science 294 (5543): 804-808.

LORENCOVÁ, E., FRÉLICHOVÁ, J., NELSON, E., VAČKÁŘ, D. (2013): Past and future impacts of land use and climate change on agricultural ecosystem services in the Czech Republic. Land Use Policy 33, 183-194.LUCC Czechia UK Prague Database (2008) Database of the project of the Grant Agency of the Czech Republic GAČR 205/09/0995 "Regional differentiation and possible risks of land use as a reflection of functional changes of landscape in Czechia 1990 – 2010". http://lucc.ic.cz/.

MA (Millennium Ecosystem Assessment), (2005): Ecosystems and Human Well-Being: A Framework for Assessment. Report of the Conceptual Framework Working Group of the Millennium Ecosystem Assessment. Island Press, Washington, DC.

MACE, G.M., NORRIS, K., FITTER A.H. (2012): Biodiversity and ecosystem services: a multilayered relationship. Trends in Ecology and Evolution 27 (1): 19 – 26.

MAES (2012): An Analytical Framework for Ecosystem Services Assessments Under Action 5 of the EU Biodiversity Strategy to 2020. Discussion Paper – version 9.6.

MAES J., PARACCHINI, M.L. AND ZULIAN G. (2011): A European Assessment of the Provision of Ecosystem Services - Towards an Atlas of Ecosystem Services. Publications Office of the European Union, Luxembourg. 81 p.

MAIN-KNORN, M., HOSTERT, P., KOZAK, J. AND KUEMMERLE, T. (2009): How Pollution Legacies and Land Use Histories Shape Post-communist Forest Cover Trends in the Western Carpathians. Forest Ecology and Management 2: 60-70.

MANDER, Ü., HELMING, K., WIGGERING, H. (2007): Multifunctional Land Use: Meeting Future Demands for Landscape Goods and Services. Springer Berlin/Heidelberg.

MATHER, A.S., FAIRBAIRN, C., NEEDLE L. (1999): The Course and Drivers of the Forest Transition: The Case of France. Journal of Rural Studies 15: 65–90.

MEIXLER, M.S. AND BAIN, M.B. (2010): Landscape Scale Assessment of Stream Channel and Riparian Habitat Restoration Needs. Landscape and Ecological Engineering 6: 235–245.

MELICHAR, J. AND KAPROVÁ, K. (2013): Revealing preferences of Prague's homebuyers towards greenery amenities: the empirical evidence of distance-size effect. Landscape and Urban Planning 109 (1): 56–66.

METZGER, M.J., SCHRÖTER, D., LEEMANS, R., CRAMER, W. (2008): A spatially explicit and quantitative vulnerability assessment of ecosystem service change in Europe. Regional Environmental Change 8: 91–107.

MEYFROIDT, P., RUDEL, T.K., LAMBIN, E.F (2010): Forest transitions, trade, and the global displacement of land use. PNAS 107 (49) 20917–20922.

MILAD, M., SCHAICHA, H., BÜRGIB, M., KONOLDA, W. (2011): Climate change and nature conservation in Central European forests: A review of consequences, concepts and challenges. Forest Ecology and Management 261: 829–843.

MILANOVA, E.V. et al. (2001): Land Use / Cover Change in Russia since the end of the 19th century. In: Himiyama, Y. a kol. (eds): Land Use/Cover Changes in Selected Regions in the World – Volume I. IGU-LUCC Research Reports IL-2001-01, Japonsko, p. 61–72.

MOONEY, H.A. AND EHRLICH, P. R. (1997): Ecosystem Services: A Fragmentary History. In Daily, D. G., 1997. Nature`s Services: Societal Dependance in Natural Systems, 11-19. Washington, DC: Island Press.

MÜLLER, C., EICKHOUT, B., ZAEHLE, S., BONDEAU, A., CRAMER, W., LUCHT, W. (2007): Effects of changes in CO2, climate, and land use on the carbon balance of the land biosphere during the 21st century. Journal of Geophysical Research G: Biogeosciences 122 (2).

MÜLLER, F. AND BURKHARD, B. (2012): The indicator side of ecosystem services. Ecosystem Services 1, 26–-30.

NASI, R., WUNDER, S. AND CAMPOS A.J.J. (2002): Forest Ecosystem Services: Can They Pay Our Way Out Of Deforestation? A discussion paper prepared for the GEF, for the Forestry Roundtable held in conjunction with the UNFF II, New York on March 11, 2002.

NÁTR, L.. (2011): Příroda nebo člověk? Služby ekosystémů. Nakladatelství Karolinum, Univerzita Karlova v Praze. 1. vydání.MA 2005

NELSON, E., SANDER, H., HAWTHORNE, P., CONTE, M., ENNAANAY, D. WOLNY, S., MANSON, S. AND POLASKY, S. (2010): Projecting Global Land-Use Change and Its Effects on Ecosystem Services Provision and Biodiversity with Simple Models. PLoS ONE (5) 12. Available at http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0014327.

NOVOTNÁ, D. (ed.), (2001): Úvod do pojmosloví v ekologii krajiny. Praha: MŽP+Enigma, 399 s.

OECD (2001):The Well-Being of Nations. The Role of Human and Social Capital. Education and Skills.

OPDAM P., STEINGROVER E., VAN ROOIJS S. (2006): Ecological Networks: A Spatial Concept for Multi-actor Planning of Sustainable Landscapes. Landscape and Urban Planning 75: 322–332.

OPDAM, P., LUQUE, S., JONES, K.B. (2009): Changing Landscapes to Accommodate for Climate Change impacts: A Call for Landscape Ecology. Landscape Ecology 24: 715 – 72.

ORWIN, K.H., OSTLE, N., WILBY, A., BARDGETT, R.D. (2013): Effects of species evenness and dominant species identity on multiple ecosystem functions in model grassland communities. Oecologia, DOI 10.1007/s00442-013-2814-5. Perrings, Ch., Duraiappah, A., Larigauderie, A. and Mooney, H., 2011. The Biodiversity and Ecosystem Services Science-Policy Interface. Science 331 (6021): 1139–1140.

OTERO, J.D., FIGUEROA, A., MUÑOZ, F.A., PEÑA, M.R. (2012): Loss of soil and nutrients by surface runoff in two agro-ecosystems within an Andean paramo area. Ecological Engineering 37 (12): 2035 – 2043.

PALANG, H., MANDER, Ü., NAVEH, Z. (2000): Holistic Landscape Ecology in Action. Landscape and Urban Planning 50: 1–6.

PEARCE, D.W. AND TURNER, K.R. (1990): Economics of Natural Resources and the Environment. BBCC Wheatson Ltd., Exeter, UK. 378 pp.

PEREIRA, H.M., REYERS, B., WATANABE, M., BOHENSKY, E., FOALE, S., LEE, M., PALM, C., AND PATWARDHAN, A. (2005): Condition and Trends of Ecosystem Services and Biodiversity. In Ecosystems and Human Well-Being: Multi-scale Assessments (eds C. Samper, D. Capristano, M. Lee and C. Raudsepp-Hearne), Vol. 4, pp. 171-203. Island Press, Washington, D.C.

PERRINGS, CH., DURAIAPPAH, A., LARIGAUDERIE, A., MOONEY, H. (2011): The Biodiversity and Ecosystem Services Science-Policy Interface. Science 331 (6021): 1139–1140.

PEŠOUTOVÁ, R. (2007): Integrated Assessment of the Trade-offs of a Landscape Restoration Plan in the Cezava Region, the Czech Republic. Master Thesis, Wageningen University, Wageningen.

PIMENTEL, D., WILSON, C., MCCULLUM, C., HUANG, R., DWEN, P., FLACK, J., TRAN, Q., SALTMAN, T., CLIFF, B. (1997): Economic and Environmental Benefits of Biodiversity. BioScience 47 (11): 747–757.

PLUMMER, M.L. (2009): Assessing Benefit Transfer for the Valuation of Ecosystem Services. Frontiers in Ecology and the Environment 7(1): 38–45.

POKORNY, P. (2011): Neklidné časy: Kapitoly ze společných dějin přírody a lidí. Dokořán. 370 s.

PORTER, J., COSTANZA, R., SANDHU, H., SIGSGAARD, L., WRATTEN, S. (2009): The Value of Producing Food, Energy, and Ecosystem Services within an Agro-Ecosystem. Ambio 38 (4).

PRAGER, K., REED, M., SCOTT, A. (2012): Encouraging collaboration for the provision of ecosystem services at a landscape scale – Rethinking agri-environmental payments. Land Use Policy 29 (1), 244–249.

PRESTON, B.L., YUEN, E.J., WESTAWAY, R.M. (2011): Putting vulnerability to climate change on the map: a review of approaches, benefits, and risks. Sustainability Science 6 (2), 177-202.

RAMANKUTTY, N., EVAN A.T., MONFREDA C., FOLEY J.A. (2008): Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000, Global Biogeochem. Cycles: 22, GB1003, doi:10.1029/2007GB002952.

RAŠÍN, R. (2010): Krajina česko-rakouského pohraničí: vývoj a dědictví. Disertační práce. UK v Praze, PřF, KSGRR, Praha, 187 s. + příl.

RENAUD, F., SUDMEIER-RIEUX, K., ESTRELLA, M. (2013): The Role of Ecosystems in Disaster Risk Reduction. United Nations University.

RENSCHLER, C.S. AND HARBOR, J. (2002): Soil erosion assessment tools from point to regional scales-the role of geomorphologists in land management research and implementation. Geomorphology 47 (2-4): 189–209.

REVELLE, R. (1984): The Effects of Population Growth on Renewable Resources. Population Resources, Environment and Development. Population Studies, 223–240. UN, New York.

ROBARDS, M.D., SCHOON, M.L., MEEK, C.L., ENGLE. N.L. (2011): The Importance of Social Drivers in the Resilient Provision of Ecosystem Services. Global Environmental Change 21: 522-529.

ROBINSON, D.A., LEBRON, I., VEREECKEN, H. (2009): On the Definition of the Natural Capital of Soils: A Framework for Description, Evaluation, and Monitoring. Soil Science Issues 73:1904 – 1911.

ROCKSTRÖM, J., W. STEFFEN, K. NOONE, Å. PERSSON, F. S. CHAPIN, III, E. LAMBIN, T. M. LENTON, M. SCHEFFER, C. FOLKE, H. SCHELLNHUBER, B. NYKVIST, C. A. DE WIT, T. HUGHES, S. VAN DER LEEUW, H. RODHE, S. SÖRLIN, P. K. SNYDER, R. COSTANZA, U. SVEDIN, M. FALKENMARK, L. KARLBERG, R. W. CORELL, V. J. FABRY, J. HANSEN, B. WALKER, D. LIVERMAN, K. RICHARDSON, P. CRUTZEN, J. FOLEY (2009): Planetary Boundaries: Exploring the Safe Operating Space for Humanity. Ecology and Society 14 (2): 32

RODRÍGUEZ, J. P., T. D. BEARD, JR., E. M. BENNETT, G. S. CUMMING, S. CORK, J. AGARD, A. P. DOBSON, AND G. D. PETERSON (2006): Trade-offs across space, time, and ecosystem services. Ecology and Society 11(1): 28. [online] URL: http://www.ecologyandsociety.org/vol11/iss1/art28/

ROMPORTL, D., CHUMAN, T., LIPSKÝ, Z. (2008): New method of landscape typology in the Czech Republic. Landscape Classification - Theory and Practice, The Problems of Landscape Ecology. Faculty of Geography and Regional Studies of the University of Warsaw, Vol. 20, p. 315 – 320. ISSN 1899 - 3850

ROMPORTL, D., CHUMAN, T., LIPSKÝ, Z., (2010): Landscape heterogenity changes and their driving forces in the Czech Republic after 1990. In: Bičík, I., Himiyama, Y., Feranec, J.: Land Use/Cover Changes in Selected Regions in the World. Volume V, IGU-LUCC Research Report, Charles University in Prague, Faculty of Science, ISBN 978-4-907651-05-9, 41–50.

ROUNSEVELL, M.D.A., PEDROLI, B., ERB, K. H., GRAMBERGER, M., BUSCK, A.G., HABERL, H., KRISTENSEN, S., KUEMMERLE, T., LAVOREL, S., LINDNER, M., LOTZE-CAMPEN, H., METZGER, M.J., MURRAY-RUST, D., POPP, A., PÉREZ-SOBA, M., REENBERG, A., VADINEANU, A., VERBURG, P.H., WOLFSLEHNER, B. (2012): Challenges for land system science. Land Use Policy 29 (4), 899–910.

RUHL, J.B., KRAFT, S.E., LANT, C.L. (2007): The Law and Policy of Ecosystem Services. Island Press, Washington DC. 345 p .

SALA, O. E., CHAPIN III, F., ARMESTO, J.J., BERLOW, E., BLOOMFIELD, J., DIRZO, R., HUBER-SANWALD, E., HUENNEKE, L., JACKSON, R., KINZIG, A., LEEMANS, R., LODGE, D., MOONEY, H., OESTERHELD, M., POFF, N., SYKES, M., WALKER, B., WALKER, M., WALL, D.H. (2000): Global Biodiversity Scenarios for the Year 2100. Science 287(5459): 1770.

SANDU, H. AND WRATTEN S. (2013): Ecosystem Services in Farmland and Cities. In: Wratten, S., Sandhu, H., Cullen, R. And Costanza, R. (eds.) Ecosystem Services in Agricultura and Urban Landscapes, First Edition. John Willey and Sons Ltd.

SAUER C. O. (1925): The Morphology of Landscape. University of California Publications in Geography 2 (2): 19-53.

SCHÄGNER, J. P., BRANDER, L., MAES, J., HARTJE, V. (2013): Mapping Ecosystem Services' Values: Current Practice and Future Prospects. Ecosystem Services 4: 33–46.

SCHRÖTER, M., VAN DER ZANDEN, E.H., VAN OUDENHOVEN A.P.E., REMME, R.P., SERNA-CHAVEZ, H.M., DE GROOT, R.S. AND OPDAM, P. (2014): Ecosystem services as a contested concept: a synthesis of critique and counter-arguments. Conservation Letters. doi: 10.1111/conl.12091.

SCHULP, C.J.E., NABUURS, G.-J., VERBURG, P.H. (2008): Future Carbon Sequestration in Europe – Effects of Land Use Change. Agriculture, Ecosystems and Environment, 127: 251 – 264.

SHAO,J. NI, J., WEI CH., XIE D. (2005): Land Use Change and Its Corresponding Ecological Responses: A Review. Journal of Geographical Sciences 15(3): 305–328.

SKLENIČKA, P, PIXOVÁ, K. (2004): Importance of Spatial Heterogeneity to Landscape Planning and Mannagement. Ekológia Bratislava 23 (1), 310–319.

SKLENIČKA, P. (2003): Základy krajinného plánování. N. Skleničková. Říčany. 321 s.

SKLENIČKA, P. (2005): Applying Evaluation Criteria for the Land Consolidation Effect to Three Contrasting Study Areas in the Czech Republic. Land Use Policy 23: 502–510.

SKLENICKA, P., SALEK, M. (2008): Ownership and soil quality as sources of agricultural land fragmentation in highly fragmented ownership patterns. Landscape Ecology 23, 299–311.

ŠKORPÍKOVÁ, V. (2008): Drop velký: letitý problém české ochrany přírody. Ochrana přírody 5.

SOBR (Strategie ochrany biologické rozmanitosti v ČR) (2005): Ministerstvo životního prostředí, Praha.

STAMP, L.D. (1945): The Land of Britain: The Report of Land Utilization Survey in Britain. London.

STÖGLEHNER, G., NEUGEBAUER, G., LÖSCHNER, L. (2013): The Concept of Vital Landscapes. In: Těšitel, J, Kolbmüller and Stöglehner, G. (eds.) Vital Landscapes: Valorisation and Sustainable Development of Cultural Landscapes Using Inovative Participation and Visualisation Techniques.

ŠTYCH, P. (2007): Územní diferenciace dlouhodobých změn využití krajiny Česka (dizertační práce). Univerzita Karlova v Praze, Přírodovědecká fakulta, katedra sociální geografie a regionálního rozvoje.

SUNDSETH, K. (2009): Natura 2000 in the Pannonian Region. European Commission, Environment Directorate General.

SVATOŇOVÁ, H., LAUERMANN, L. (2010): Dálkový průzkum Země – aktuální zdroj geografických informací. Brno: Masarykova univerzita, 96 s.

SWIFT, M.J., IZAC A.-M.N., VAN NOORDWIJK, M. (2004): Biodiversity and Ecosystem Services in Agricultural Landscapes – Are We Asking the Right Questions? Agriculture, Ecosystems and Environment 104: 113 – 134.

SWINTON, S.M., LUPI, F., ROBERTSON, G.P., HAMILTON, S.K. (2007): Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits. Ecological Economics, 64(2),245–252.

SÝKORA, L. AND MULÍČEK, O. (2012): Urbanizace a suburbanizace v Česku na počátku 21. století. Urbanismus a územní rozvoj XV (5): 27-38.

TEEB (2010): The Economics of Ecosystems and Biodiversity for National and International Policy Making, www.teebweb.org.

TEEB (2013): Guidance Manual for TEEB Country Studies. Version 1.0.

TILMAN, D., REICH, P.B., ISABELL, F. (2012): Biodiversity impacts ecosystem productivity as much as resources, disturbance, or herbivory. PNAS 109 (26): 10394–10397.

TILMAN, D., REICH, P.B., KNOPS, J.M.H. (2006): Biodiversity and ecosystem stability in a decade-long grassland experiment. Nature 441: 629-632.

TOMAN, M. (1998): Why Not to Calculate the Value of the World's Ecosystem Services and Natural Capital. Ecological Economics 25: 57–60.

TOTH, F.L., HIZSNYIK, E. (1998): Integrated Environmental Assessment Methods: Evolution and Applications. Environmental Modeling and Assessment 3: 193–207.

TROLL C. (1939): Luftbildplan and okologische bodenforschung. Zeitschraft der Gesellschaft fur Erdkunde Zu Berlin: 241–298.

TROY, A. AND WILSON, M.A. (2006): Mapping Ecosystem Services: Practical Challenges and Opportunities in Linking GIS and Value Transfer. Ecological Economics 60: 435–449.

TURNER, M.G., GARDNER, R.H., O'NEILL, R.V. (2001): Landscape Ecology in Theory and Practice. Pattern and Process. Springer Science+Business Media, Inc. 406 p.

TURNER II B.L. (ed.), (1990): The Earth As Transformed by Human Action: Global and Regional Changes in the Biosphere over the Past 300 Years. Press Syndicate of the University of Cambridge, USA.

TURNER, B. L. II, MEYER W. B., SKOLE, D. L. (1994): Global Land-Use/Land-Cover Change: Towards an Integrated Study. Ambio 23 (1), pp. 91-95.

UK NEA (UK National Ecosystem Assessment) (2011): The UK National Ecosystem Assessment: Synthesis of the Key Findings. UNEP-WCMC, Cambridge.

UNEP and WWF (2013): TEEB Scoping Study for Georgia. United Nations Environment Programme (UNEP), Geneva, Switzerland.

UNISDR (2013): Synthesis Report: Consultations on a Post-2015 Framework on Disaster Risk Reduction (HFA2).

UY, N., SHAW, R., (2013): Ecosystem resilience and community values: Implications to ecosystem-based adaptation. Journal of disaster research 8 (1): 201–202.

VAČKÁŘ, D., ORLITOVÁ, E. (2011): Human appropriation of aboveground photosynthetic production in the Czech Republic. Regional Environmental Change 11, 519–529.

VAISHAR, A. ET AL. (2013): Změny krajiny na okraji velkých měst. Je suburbanizovaný venkov ještě venkovem? Mendelova univerzita v Brně, Agronomická fakulta. MENDELU, Brno. 98p.

VALDIVIA, C., SETH, A., GILLES, L. J. ET AL. (2010): Adapting to Climate Change in Andean Ecosystems: Landscapes, Capitals, and Perceptions Shaping Rural Livelihood Strategies and Linking Knowledge Systems. Annals of the Association of American Geographers 100(4): 818–834.

VAN DER PLOEG, S., DE GROOT, R.S., WANG Y. (2010): The TEEB Valuation Database: overview of structure, data and results. Foundation for Sustainable Development, Wageningen, the Netherlands

VAN ROMPAEY, A., KRASA, J., DOSTAL, T., GOVERS, G. (2003): Modelling Sediment Supply to Rivers and Water Reservoirs in Eastern Europe during and after Collectivization Period. Hydrobiologia 494, 169–176.

VERBURG P.H. et al. (2009a): From Land Cover Change to Land Function Dynamics: A Major Challenge to Improve Land Characterization. Journal of Environmental Management 90: 1327–1335.

VERBURG, P.H., VAN DE STEEG, J., VELDKAMP, A. AND WILLEMEN, L. (2009b): From Land Cover Change to Land Function Dynamics: A Major Challenge to Improve Land Characterization. Journal of Environmental Management 90: 1327–1335.

VERBURG, P.H., ASSELEN, S. VAN, ZANDEN, E.H. VAN DER AND STEHFEST, E. (2013): The representation of landscapes in global scale assessments of environmental change. Landscape Ecology, 28(6): 1067–1080.

VIHERVAARA, P., KUMPULA, T., TANSKANEN, A., BURKHARD, B. (2010): Ecosystem Services-A Tool for Sustainable Management of Human-environment Systems. Case Study Finnish Forest Lapland. Ecological Complexity 7 (3): 410–420.

VO, Q.T., KUENZER, C., VO, Q.M., MODER, F., OPPELT, N. (2012): Review of valuation methods for mangrove ecosystem services, Ecological Indicators 23: 431-446.

VOŘÍŠEK P. ET AL. (2008): Agricultural Intensification Still Threatens European Field Birds. http://www.cso.cz/index.php?ID=1756

VOŘÍŠEK, P., KLVAŇOVÁ, A., BRINKE, T., CEPÁK, J., FLOUSEK, J., HORA, J., REIF, J., ŠŤASTNÝ, K. AND VERMOUZEK Z. (2009): State of the birds of the Czech Republic. Sylvia 45: 1 – 38.

WAINGER, L.A., KING, D.M., MACK, R.N., PRICE, E.W. AND MASLIN, T. (2010): Can the Concept of Ecosystem Services Be Practically Applied to Improve Natural Resource Management Decisions? Ecological Economics 69: 978–987.

WALKER, B., HOLLING, C.S., CARPENTER, S.R., KINZIG, A. (2004): Resilience, Adaptability and Transformability in Social–ecological Systems. Ecology and Society, 9(2), p.5 [online] URL: http://www.ecologyandsociety.org/vol9/iss2/art5/.

WALLENIUS, T., NISKANEN, L., VIRTANEN, T., HOTTOLA, J., BRUMELIS, G., ANGERVUORI, A., JULKUNEN J., PIHLSTRÖM M. (2010): Loss of habitats, naturalness and species diversity in Eurasian forest landscapes. Ecological Indicators 10 (6): 1093–1101.

WILSON, M.A. AND HOEHN, J.P. (2006): Valuing Environmental Goods and Services Using Benefit Transfer: The State-of-the art and Science. Ecol. Econ. 60: 335–342.

ZHANG, W., RICKETTS, T.H., KREMEN, C., CARNEY, K., SWINTON, S.M. (2007): Ecosystem Services and Dis-services to Agriculture. Ecological Economics, 64(2), 253–260.

ZIMMERER K.S. (2009) Biodiversity. In Castree, N. Demeritt, D., Liverman, D. and Rhoads, B.: A Companion to Environmental Geography. Wiley-Blackwell, London. p. 50–65.

ZONNENVELD, I.S. (1990): Scope and Concepts of Landscape Ecology as an Emerging Science. In: Zonnenveld, I.S. and Forman, R.T.T.: Changing Landscapes: An Ecological Perspective. Springer-Verlag New York, Inc.

Websites:

BISE (2013): http://biodiversity.europa.eu/ecosystem-assessments/assessments

CENIA (2011): http://issar.cenia.cz/issar/page.php?id=1887

Cezava (2014): http://www.region-cezava.cz

CICES (2014): http://cices.eu/

CSO (Czech statistical office) (2011): http://www.czso.cz

CSO (2014): http://www.czso.cz/csu/redakce.nsf/i/hdp_narodni_ucty

EUROSTAT (2014):
http://epp.eurostat.ec.europa.eu/portal/page/portal/national_accounts/data/main_tables

FAOSTAT (2014): http://faostat3.fao.org/faostat-gateway/go/to/download/E/EL/E

Ministry of Agriculture (2012): Situační a výhledová zpráva: Půda. Praha.
http://eagri.cz/public/web/file/181775/Zprava_Puda_kniha_web__1_.pdf

Ministry of Agriculture, (2009): Zemědělství 2008.
http://eagri.cz/public/eagri/file/17422/Zemedelstvi2008.pdf

STUŽ (2014): http://www.bioinstitut.cz