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Integrating the Concept of Ecosystem Services into
Landscape Ecology

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TABLE OF CONTENTS

KURZFASSUNG.....	3
ABSTRACT	4
EXTENDED SUMMARY	5
INTRODUCTION	10
I. Background.....	10
II. Landscape ecology and the human value	13
III. Conceptual evolvement – ecosystem versus landscape services	14
IV. Putting the concept into practice – limitations and challenges.....	17
OBJECTIVES AND OUTLINE OF THE THESIS	21
PROJECT FRAMES OF THE ARTICLES.....	24
REFERENCES	26
CHAPTER 1 - The concept of ecosystem services regarding landscape research – a review ..	37
CHAPTER 2 - Landscape structure – service relations.....	77
CHAPTER 3 - Providing a spatially explicit assessment framework	103
DANKSAGUNG	149
CURRICULUM VITAE	150

KURZFASSUNG

Trotz zahlreicher Naturschutzaktivitäten ist der Verlust an Biodiversität und die Zerstörung natürlicher Ökosysteme in Europa nicht aufzuhalten. Gravierende Veränderungen sowohl in der Landschaft als auch in der Landnutzung werden dafür verantwortlich gemacht. Diese Entwicklung ist unter anderem darauf zurückzuführen, dass der vielfältige Nutzen von Ökosystemen in Landschaftsentwicklungsplänen nicht berücksichtigt wird. Das Konzept der Ökosystemleistungen (ÖKL) ermöglicht der Landschaftsökologie ihren wissenschaftlichen Ansatz: „Prozessanalysen räumlicher Landschaftsmuster“ in Richtung Landschaftsbewertung zu erweitern und somit zu einer nachhaltigen Landschaftsentwicklung beizutragen. Das Ziel dieser Doktorarbeit ist es neue Erkenntnisse für die Bewertung von ÖKL auf Landschaftsebene zu gewinnen. Es werden sowohl grundlegende Zusammenhänge zwischen Landschaftsstruktur und ÖKL erforscht, als auch ein räumlich expliziter Bewertungsrahmen für unterschiedliche ÖKL präsentiert und angewendet.

Zuerst informiert eine ausführliche Literaturübersicht über den aktuellen Stand der Wissenschaft. Mehr Information sowohl über den Zusammenhang zwischen Landbedeckungsklassen und ÖKL als auch über geeignete Kartierung,- und Quantifizierungsmethoden diverser ÖKL wird benötigt. Die Verfügbarkeit von Daten ist eine der größten Herausforderungen. Im zweiten Kapitel beleuchtet eine österreichische Studie den Zusammenhang zwischen verschiedenen Landschaftsstrukturvariablen und der ÖKL „Schädlingsbekämpfung“ am Beispiel des Drahtwurms. Im Umfeld von 14 mit Drahtwürmern befallenen Kartoffelfeldern wurden Landschaftsstrukturanalysen durchgeführt. Bei Grünland mit erhöhtem Sandgehalt im Boden ist besondere Vorsicht geboten, da diese Kombination mit hohem Drahtwurmbefall einhergeht. In diesem Fall wird ein Wiesenmanagement empfohlen, das den Schädling in seiner Entwicklung einschränkt. Da Biodiversität die Grundlage vieler ÖKL darstellt, informiert eine Literaturübersicht über neue Wege in der Erhebung von Biodiversitätsindikatoren auf Landschaftsebene. Entgegen herkömmlichen, terrestrischen Vermessungstechnologien kann die flugzeuggestützte Erfassungstechnik (FWF-ALS, Full waveform airborne laser scanning) hochauflösende Daten liefern, die als biodiversitäts-relevante Indikatoren verwendet werden können, wie z.B.: Landschaftsfragmentierung oder Struktur der Schichtung des Waldes. Im dritten Kapitel werden, eingebettet in einem räumlichen Bezugssystem, zahlreiche ÖKL in europäischen Grenzregionen erhoben und in Karten dargestellt. Die Umsetzung des Bewertungsrahmens ermöglicht eine schnelle ÖKL-Evaluierung auf regionaler Ebene. Die resultierenden ÖKL-Karten können somit in zukünftige Landschaftsplanungsprozesse miteinbezogen werden.

Mit besonderem Fokus auf den Zusammenhang zwischen Landschaftsstruktur und ÖKL liefert die vorliegende Arbeit einen wichtigen Beitrag zur ÖKL-Forschung sowohl in der Theorie als auch in der Praxis. Eine räumliche Bewertung von ÖKL ist unverzichtbar und benötigt lokal explizite Daten. Um sich diesen Herausforderungen ausreichend stellen zu können, ist weitere Forschungsarbeit notwendig, die sich mit diesen zugrundeliegenden Zusammenhängen beschäftigt.

ABSTRACT

Despite a variety of conservation activities, the decline of European biodiversity and the degradation of natural ecosystems are still going processes due to serious land use and land cover changes. One reason for this development could be the absence of ecosystem service values in landscape planning concepts. Therefore, landscape ecology is encouraged to extend its pattern-process approach by including the concept of ecosystem services (ESS) into their scientific scope for a sustainable landscape development. The objective of this thesis is to provide new knowledge about the assessment of ESS at landscape scale by exploring landscape structure and service relations as well as operationalising a spatially explicit ESS assessment framework.

The first chapter presents a literature analysis reviewing the current state-of-the-art. It demonstrates that a better understanding of the interactions between land cover and service on the one hand and methods to map and quantify services on the other hand is needed. Data availability is still one of the major limitations of ESS assessment. In the second chapter the relationship between landscape structure and the ESS 'pest control' with special regard to wireworms was investigated within an Austrian case study. Different landscape variables were explored around 14 randomly selected potato field plots. The study revealed that grassland habitats with high sand content are significantly correlated with wireworm infestation level. Therefore, appropriate management regimes for grassy landscape elements suppressing wireworm populations in crop fields are strongly recommended. As biodiversity has multiple roles in relation to the delivery of ecosystem services, this chapter also presents a literature review of new procedures to measure biodiversity at landscape scale. In contrast to traditional, terrestrial surveying technologies, the full waveform airborne laser scanning (FWF-ALS) can provide high resolution data and is therefore relevant for biodiversity indicators such as landscape fragmentation and forest layer structure. The third chapter describes a spatially explicit methodology for the assessment and mapping of various ESS embedded in a spatial reference framework within three European cross-border regions. The implementation of the assessment framework provided enables rapid regional service evaluations. The resulting maps could be used as a knowledge base in landscape planning procedures.

The thesis as a whole constitutes an important contribution towards the application of the ESS approach in theory and practice with special regard to landscape structure-service relationships. Spatial ESS assessment is indispensable and requires locally explicit data. However, in order to meet these challenges and to better understand the underlying relationships, more research needs to be conducted.

EXTENDED SUMMARY

Introduction

Human beings benefit from processes or structures within ecosystems that give rise to a range of goods and services called ‘ecosystem services’ (ESS) (Daily, 1997; MEA, 2005). According to the MEA (2005) they can be classified into four main categories: *provisioning* (e.g. food, fresh water), *supporting* (e.g. nutrient cycling, soil retention), *cultural* (e.g. aesthetic, spiritual experiences), and *regulating* (e.g. climate regulation, water purification) services. Cultural landscapes in particular are of great interest for human activities due to their high potentials to provide different ESS. In such areas different sectors, such as agriculture, forestry, urban sprawl and nature protection are competing for limited resources and space. Due to the fragmentation and degradation of natural ecosystems the provision of ESS is affected. One reason for the loss of ESS might be the absence of ESS values in environmental planning processes. In particular the quantification and implementation of ESS in environmental planning have been the biggest challenges of current ecosystem science (Wallace, 2007). Managing cultural landscapes in an adequate way, in order to receive sufficient *supporting, regulating, provisioning and cultural* services (MEA, 2005) will require research that is policy-relevant, multidisciplinary and spatially explicit. Therefore, landscape scientific disciplines, especially landscape ecology are encouraged to extend their pattern-process approach (Turner et al., 2001) by including perceptions of value in their scientific scope for a sustainable landscape development. They are able to provide useful information on how the spatial distribution of human activities affects important landscape processes and structures from which services are derived (Jones et al., 2008). Since landscape structure has been linked to biodiversity (Zechmeister and Moser, 2001; Zechmeister et al., 2003; Perterseil et al., 2004) and to other important landscape functions and services, such as pest control or limited erosion risk (Forman, 1995; MEA, 2005; de Groot, 2006; Willemen et al., 2008), it seems worthwhile to further explore landscape structure as a proxy for different landscape functions and services. As most indicators available are inadequate in characterizing the diversity and complexity of the services provided by ecosystems and landscapes (Egoh et al., 2008), there is a need for new techniques to collect accurate data at the landscape scale. In other words, a deeper understanding of the various scales at which services occur is needed in order to put the concept of ESS into practice. Synergies and trade-offs between services have to be identified at different scales so multifunctional goals can be negotiated (Tippett et al., 2007; Opdam et al., 2008). However, a conceptual and feasible assessment framework for various services at different scales is still missing.

The first part of this thesis therefore aims at providing a coherent knowledge base about the concept of ESS and presenting opportunities for implementing the concept into landscape research. The second chapter emphasises the relationship between landscape structure and

ecosystem services and presents new procedures to measure ESS at the landscape scale with special regard to biodiversity. The final chapter intends to provide and operationalise an assessment framework for a wide range of services at different scales in transnational areas.

Methods and Results

(1) The concept of ecosystem services regarding landscape research – a review

Based on a literature review, the first section presents different key definitions used within the concept of ESS as well as different ESS classification systems. The subsequent parts address various approaches and challenges regarding the quantifying and mapping of ESS, different aspects of valuation methods and conclude with a discussion; the debate about key definitions and a coherent classification system is still going on. This paper demonstrates that a better understanding of the interactions between land cover, use and function on the one hand and methods to map and quantify land use and landscape function on the other hand is needed, if the ESS concept is to be fully integrated into landscape planning issues (e.g. Verburg et al., 2009). By searching for appropriate ESS indicators and proxies several issues have to be faced, especially the relationship between services and scales. Furthermore, data availability and disaggregation of spatial data are still one of the major limitations to the assessment of ESS.

(2) Landscape structure – service relations

In order to gain more knowledge on the relationship between landscapes structure and service provision, an Austrian case study addressing the service '*pest control*' was carried out at landscape scale. We investigated the relationship between different landscape variables, such as landscape composition, landscape complexity, ecological variables and the wireworm infestation level - represented by the proportion of wireworm damaged potato tubers along the gradient from the lower warm-dry "Weinviertel" to the cool-humid "Waldviertel" in the north-east of Austria. The different landscape variables were characterised around 14 randomly selected potato field plots at three different spatial scales (1, 9 and 25 ha) based on aerial photographs and field surveys. In addition to that, climate characteristics (annual temperature sum, annual precipitation sum) and soil type as well as management data (number of years with grassy crops for 5 years prior to potato growing and potato cultivar) were included in the analysis. The results revealed a higher infestation level in cool-humid than in warm-dry regions. Grassy field margins with higher sand content in particular were significantly positively correlated with wireworm damage at the 25 ha scale.

Since nation-, and/or region-wide indicators for ESS at landscape scale are rare, we used a literature review to present how data collected with Airborne laser scanning (ALS) - especially with full-waveform (FWF)-ALS - can be used to derive quantities relevant for biodiversity assessments. The biggest advantage of ALS over traditional photogrammetric

measurements methods, including airborne and satellite based imagery in the visible and infra-red part of the spectrum, is its ability to “see” through small gaps in the vegetation cover, e.g. forest canopy (Naesset, 2007). Parts of the laser beam penetrate the foliage and are reflected by vegetation underneath, such as understory trees, bushes, herbaceous vegetation and also the forest floor. Vegetation, due to its penetration capacity of the foliage, usually features larger height distributions than flat terrain and consequently larger echo widths. This information can be used for the filtering process in order to increase the digital terrain model (DTM) quality (Kraus and Pfeifer, 1998; Axelsson, 2000; Vosselman, 2000; Sithole and Vosselman, 2005). Knowledge of the topography and the spatial distribution of the plant cover are considered an invaluable proxy for the estimation of biodiversity indicators. As ALS typically provides high point densities and good penetration rates, various biodiversity indicators can be estimated directly from these measurements. These can be terrain surface features like roughness, as well as vegetation parameters like height, predominant tree type, location of fallen trees or forest layer structure.

(3) Providing a spatially explicit assessment framework

To receive support for adequate choices in sustainable landscape planning, an integrative assessment framework was developed. This framework was applied within an ecologically sensitive cross-border region in Austria and Hungary. The objective was to evaluate various landscape services at different spatial scales. Five main landscape services - namely *regulation, habitat, provision, information and carrier* (adapted from de Groot, 2006) were mapped and assessed embedded in a spatial reference framework. Dependant on the service providing units, three different levels of service assessment were distinguished: (1) The *Landform Approach*; all services could be observed directly either by the use of Corine land cover or by clearly identifiable spatial indicators within seven different Landform Types defined by Konkoly-Gyuró et al. (2010). (2) The *Broader Habitat Approach*; it emphasised the assessment of services at the landscape element scale within 41 randomly selected landscape sample sites. The assessment was based on the use of an expert driven capacity matrix (Burkhard et al., 2009), where Broader Habitat Types were linked with relevant services. The resulting values were revised with the help of semi-quantitative data gained from field work. (3) The *Socio-cultural Approach*; this approach was used for the assessment of the *information* services that were generated at a broader scale, namely the Landscape Character Type scale (Konkoly-Gyuró et al., 2010; after Swanwick, 2002). Additional indicators that were mainly based on geo-data were defined. Finally, all service values were extrapolated to the seven Landform Types revealing the actual landscape service provision within the study area. The results presented hot and cold spots of service provision at different spatial scales as well as the trade-offs between the different services.

To enhance connectivity within European transnational ecological networks the status quo concerning the provision of ESS has to be evaluated. Therefore, the allocation of *regulation, habitat, provision, information and carrier* services was assessed within the following project regions (TEN, www.transeconet.com): *North* included the ‘Ore Mountains (Erzgebirge/ Krušné hory) and Elbe Sandstone Mountains border region’ (Germany/Czech Republic),

Central North contained the 'Czech Republic' and *Central South* the cross border region 'Neusiedlersee/Fertö' (Austria/Hungary). As data availability was limited within these project regions, we used a capacity matrix (Burkhard et al., 2009), in which Broader Habitat Types were linked to specific landscape services by expert knowledge. In order to be spatially explicit, these capacity matrix values were adapted to the different regional conditions. Whereas the case studies in the project regions *North* and *Central North* based their assessments only on the values of the regional capacity matrices, those in the project region *Central South* extended their evaluations by semi-quantitative data gained from field work. While the main service distributions were visualized as spider-web diagrams within each investigation area, the individual services delivery, exemplified by soil formation, was displayed in a map format.

Conclusions and outlook

Several main conclusions can be drawn from the presented thesis. Assessing and mapping ecosystem services with special regard to the landscape structure and service delivery relation are important contributions towards the applications of the ecosystem service approach in theory and in practice. Despite the fact that a great deal of work has already been done regarding the implementation of the ecosystem services concept into landscape planning, it still remains an innovative research field with many challenges to be faced - regarding the assessment and visualization of ESS in particular.

Concerning the biodiversity assessment, full waveform airborne laser scanning (FWF-ALS) has such great promise because a number of biodiversity indicators can be estimated directly from the measurements in an accurate and reliable way, especially if efficient data acquisition over large areas is needed. Up to now, a lot of research has been carried out in forested areas, and it would be important to apply this method in open cultural landscapes within future projects. The results of this thesis also reveal the necessity of considering landscape structure as a crucial factor in ESS assessment, especially for pest control. As semi-natural habitats may act both as sources and pest pressure depending on their quality, further research regarding appropriate management regimes for grassy landscape elements suppressing wireworm populations in crop fields is strongly recommended.

Furthermore, a good overview of the distribution of different landscape services within transnational study areas could be provided embedded in a spatial reference framework. The proposed method makes a rapid service assessment possible and illustrates first trends for landscape service provision. Applying a spatially explicit methodology of this kind provides useful information on how spatial patterns contribute to different service provision levels. A visualisation of the actual service provision could help local and regional spatial planners to make informed choices, in particular for areas with high pressure on land resources.

The implementation of our assessment framework into nature conservation and sustainable landscape development processes has already started within our Austrian investigation area

‘Neusiedlersee’ and has revealed that the concept of ecosystem (landscape) services has great potential to evaluate ecologically sensitive regions. However, when evaluating ESS, it became obvious that social constructs always have to be taken into account. In practice, different people have different viewpoints and needs, and - to give an example - not everything that is potentially nutritious is accepted everywhere as food (Daniel et al., 2012). Therefore, for the identification of ESS demands participative approaches, such as the integration of different stakeholders are considered indispensable, but still remain a challenge (de Groot et al., 2010; Schops, 2011). By including stakeholders of different disciplines into the evaluation process both conflicting and compatible views about ESS demands may arise. However, these insights are important for managing ESS in a stakeholder-related sense and for recognising and reducing conflicts of interest (see Backhaus et al., 2007; Soliva et al., 2008). The inclusion process represents an opportunity to meet this challenge and offers new potential for a sustainable ESS management in the future.

The presented work may contribute to the international research on ‘mapping and assessing ecosystem services’ (MAES), which is one of the keystones of the EU Biodiversity Strategy 2020 (EC, 2011). In the short-term, the essential challenge of Action 5 under the EU Biodiversity Strategy is to make the best use of and to operationalise the information and scientific knowledge currently available on ecosystems and their services across Europe. A MAES working group has been set up, whose task it is to overview the drafting of methodological guidelines on mapping ecosystem services. This should include a flexible set of indicators for mapping ESS as well as mapping tools, methodologies and training options. As all member states should provide specific case studies (ground-truth) from on-going or recently completed national or sub-national assessments and research projects, my thesis results could contribute to the overall EU picture.

INTRODUCTION

I. Background

a) Problem identification

“Halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss”, is the headline target of the EU Biodiversity Strategy for 2020 (EC, 2011). Despite a multiplicity of conservation activities, the decline of European biodiversity is still on-going. Over the last few decades the demand for natural resources has grown considerably due to increasing human population size, exponential economic growth and global consumption resulting in an enormous pressure on Earth’s ecosystems (Vitousek, 1997). As a consequence, we are faced with negative effects on the environment, such as habitat loss or change, fragmentation and degradation, climate change, biological invasions, overexploitation and pollution at global, national and regional level. The relative importance of these pressures varies from place to place and they are frequently occurring together (CEC, 2006). Recent data indicate that while areas of extensive agriculture, grasslands and wetlands continue to decline across Europe, artificial surfaces are increasing (EEA, 2011). Large parts of Europe are therefore highly fragmented by transportation infrastructure and urban development. According to an IUCN analysis (EEA, 2011) almost a quarter of wild species are threatened by extinction and 60% of habitats are in unfavourable conditions. For instance, 27 % of mammals, 10 % of reptiles and 8 % of amphibians whose natural habitat is the forest are endangered in the EU region (IUCN, 2009).

Due to the loss of biodiversity and the degradation of habitats, the provisioning of ecosystem services (ESS) is affected (MEA, 2005). Ecosystem services are defined as *‘the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life’* (Daily, 1997). They arise when a biophysical structure (e.g. vegetation cover) or function (e.g. slow passage of water) directly or indirectly contributes toward meeting a human need or demand. Such services (e.g. flood protection) generate benefits (e.g. serving health and safety) that contribute to overall well-being and can be valued by people (e.g. willingness to pay for protection) (MEA, 2005; Haines Young and Potschin, 2010; Potschin and Haines-Young 2011;) (Figure 1).

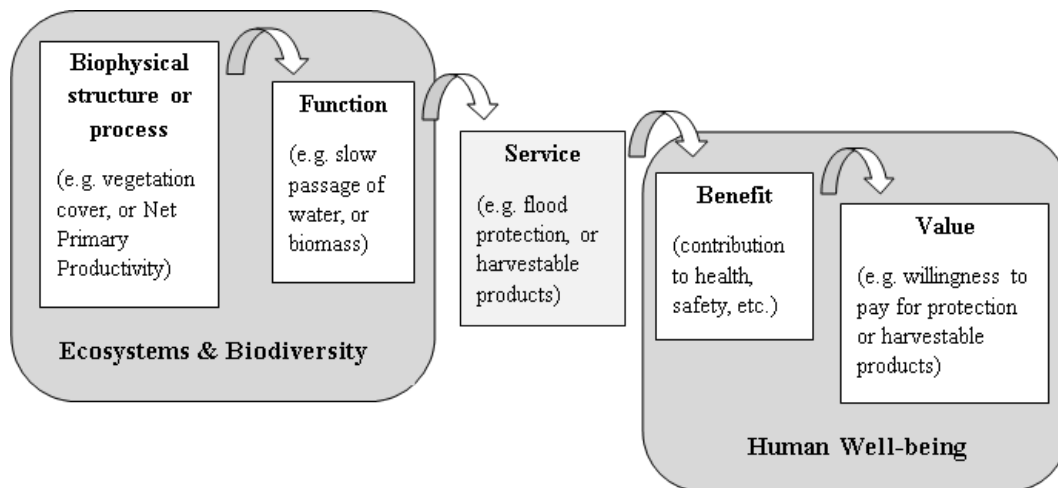


Figure 1: framework for linking Ecosystems & Biodiversity to Human Well-being (after Haines-Young and Potschin, 2010).

According to the Millennium Ecosystem Assessment (MEA) (2005) that advocated for a holistic approach to describe and categorize the relationship between ecosystems and society, ESS can be classified into four main categories: *provisioning* (e.g. food, fresh water), *supporting* (e.g. nutrient cycling, soil retention), *cultural* (e.g. aesthetic, spirituals experiences), and *regulating* (e.g. climate regulation, water purification) services. Cultural landscapes, for instance, are primarily managed to increase *provisioning* ESS of food, fibre, and fuel. At the same time they depend on a wide range of *supporting* and *regulating* services, such as climate, soil fertility and pollination (NCR, 2005) - factors that determine the underlying biophysical capacity of ecosystems (Wood et al., 2000). The production of over 75% of the world's most important crops that feed humanity and of 35% of the food is based on animal pollination (Klein et al., 2007). Recent shortages of bee colonies have led farmers in California to import honeybees all the way from Australia (Kremen and Ostfeld, 2005). Vegetation cover in upstream watersheds plays a major role concerning the amount, quality, and stability of the water supply to agriculture. Another important ESS is genetic diversity, which provides the basis for natural selection to make evolutionary adaptations. Many important crops could not be economical, competitive and value-creating without the regular genetic support of their wild relatives (de Groot et al., 2002). A suitable and stable climate, including temperature and precipitation regimes, relies on atmospheric regulation, which – like many other ESS – the functioning of multiple ecosystems is a prerequisite for. Thus, it is obvious that there is an urgent need to take actions, in order to preserve our life support system. Reducing the pressures that threaten natural systems requires a change in our approach. The awareness that natural resources are finite and cannot be exploited indefinitely must take place simultaneously at all social and political levels, since it depends on communities of all sizes (settlement, regional, national, EU, UN).

b) Ecosystem services – a promising approach

One possibility to provide the knowledge basis to meet these challenges can be the concept of ecosystem functions, goods and services (MEA, 2005). The ESS framework as proposed by the MEA extends prior models by encouraging people to re-examine the links between ecosystems and human well-being in a pragmatic way (Potschin and Haines-Young, 2011). The novel aspect of the idea is that by following an inter- and transdisciplinary approach, it is not only single resources but the full variety of services ecosystems provide for society that are taken into account. It is therefore easier to recognise the inter-linkages of ecosystems across the broad temporal and spatial scales over which they occur (Daniel et al., 2012). This approach is considered promising for heightening awareness of nature's contribution to human well-being. Emphasizing ecological, societal and economic dependence on natural assets may attract public interest on biodiversity conservation (see Westman, 1977; de Groot, 1987, Odling-Smee, 2005). To emphasise the political relevance of the ESS concept, it is important to point out the additional (marginal) value of ESS produced by conserving an area with diverging utilisation claims. The 'Economics of Ecosystems and Biodiversity' studies (TEEB, 2010) focus on the economic benefits, usually expressed in monetary units to communicate the marginal value. Monetizing ESS highlights the distribution of benefits, improves understanding of problems and trade-offs and can thus facilitate decision making (e.g. Aylward and Barbier, 1992; Salzmänn et al., 2001; de Groot, 2006). A case study in Florida, for instance, showed that the economical calculation of ESS (e.g. carbon offsets, stormwater management, and the maintenance of air and water quality) (Escobedo et al., 2010 and CITYgreen, 2010) led to a better understanding of the benefits ecosystems (green infrastructure is used as a synonym in that study) provide for urban areas. Based on the ESS analysis urban planners and policy makers became aware of the importance of green infrastructure for their city and included tree plantings in projects administered by the City of Miami (TEEBcase 2010). However, even if economic valuation encourages management options, decisions makers still have to take into account the overall objectives and limitations of economic valuation techniques (see Heal, 2000; Ludwig, 2000; Farber et al., 2002; Wilson and Howarth, 2002; Chee, 2004; Hein et al., 2006). Apart from the economic valuation model, other ways to evaluate the importance of ESS based on environmental and socio-cultural assessments are available (e.g. Paetzold et al., 2009; Willemsen et al., 2010, Burkhard et al., 2012b). Cultural services are regarded as a special case within the ESS evaluation process. Although the importance of cultural services has been mentioned consistently, they are characterized as 'intangible', 'subjective', and difficult to quantify in biophysical and monetary terms, which has led to their being ignored in many ESS assessments (MEA, 2005). Only few case studies address the assessment of cultural services and their socio-economic impact on well-being (e.g. Burkhard and Gee, 2012). In their review, Daniel et al. (2012) provide an overview and give examples how cultural services can be integrated more effectively into the broad ESS framework.

II. Landscape ecology and the human value

Although most progress in understanding the relationship between ecosystems and well-being is driven by research in discipline areas such as ecology, conservation and environmental economics (Egoh et al., 2007, Fisher et al., 2008, Seppelt et al., 2011), the concept of ESS can potentially also play an important part in landscape ecology. In decision making processes about land use change, services related to the ecological integrity of land (e.g. biodiversity, regulation and habitat services) are often regarded as non-compatible with economically driven services (e.g. production services). Although the principles of sustainable landscape development imply that in developing land, ecological, social and economic functions should be balanced in space and time to maintain their potential to provide goods and services to future generations (WCED, 1987, Linehan and Gross, 1998), short term goals related to economic values tend to dominate over public values related to landscape and ecosystem. To overcome this problem, ecological, socio-cultural and economic value domains have to be integrated in landscape planning processes by taking into account that each of them requires a different way of looking at the functions and pattern of landscapes (Opdam et al., 2006). Understanding these three domains regarding their interrelations, such as adverse effects as well as synergies and trade-offs, is seen as the fundamental basis for sustainable landscape development (Pope et al., 2004). Therefore, landscape scientific disciplines, especially landscape ecology are encouraged to extend their pattern-process approach (Forman and Godron, 1989; Turner et al., 2001) by including perceptions of value in their scientific scope (Figure 2). Consequently, the physical structure and functioning of landscapes should be connected with the values demanded by users (Termorshuizen and Opdam, 2009). To design and develop a landscape structure that functionally supports the provision of values demanded, inter- and transdisciplinary methods have to be applied. Valuation methods should be based on economics, psychology and sociology as well as natural science knowledge requiring a common communication concept (Nassauer and Opdam, 2008). In this context, the concept of ecosystem functions, goods and services may support this approach. As ESS are defined as benefits humans derive directly and indirectly from ecosystems (MEA, 2005), they can be used as a bridge between landscapes/ ecosystems and human values (Termorshuizen and Opdam, 2009) (Figure 1). Following the three dimensions of sustainability (ecological, social and economical) (IUCN, 1992), the concept of ESS may act as a conceptual framework connecting the central pattern-process paradigm of landscape ecology (Forman and Godron, 1989; Turner et al., 2001) to the social system to achieve sustainable land use. Employing the ESS concept therefore means supporting the development of policies and instruments that integrate ecological, socio-cultural and economical perspectives in order to provide insights into mid- to long-term human impact on ecosystems/ landscapes and the welfare effects of management policies (TEEB, 2010). Integrating ESS evaluations into spatial planning can help identifying where investments into the management and protection of ecosystems deliver the greatest benefits for landscape management and human-wellbeing.

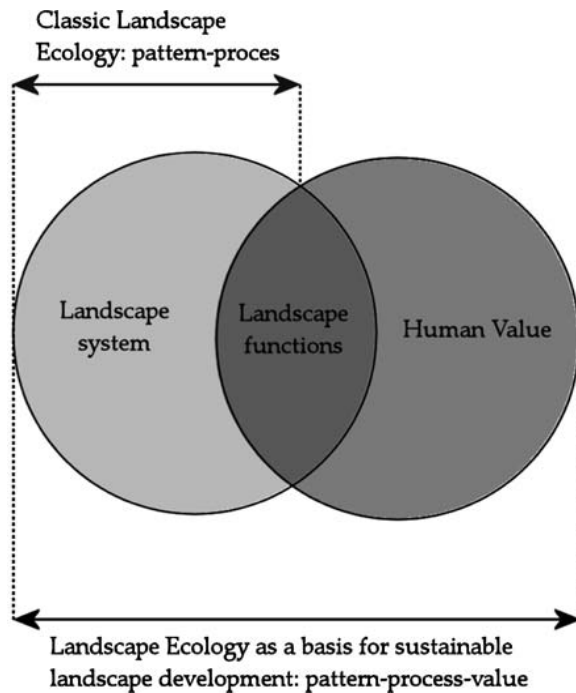


Figure 2: Landscape functions are part of the pattern–process paradigm in landscape ecology, but they can also be interpreted as values that humans attribute to landscapes. By including value into the pattern–process paradigm, landscape ecology can be seen as the basis for sustainable landscape development (Termorshuizen and Opdam, 2009).

III. Conceptual evolvement – ecosystem versus landscape services

Despite the fact that the realisation that ecosystems provide ecological, societal and economic values to humans has already taken place in the 1970s and 1980s (Westman, 1977; de Groot, 1987), the concept of ESS has attracted more attention in recent years (Fisher et al., 2009). Important milestones in the mainstreaming of ESS were Daily's book *Nature's Services: 'Societal Dependence on Natural Ecosystems'* (Daily, 1997) and the paper by Costanza et al. (1997) on the value of global natural capital. The monetary figures presented in the latter had a high impact on both science and policy making. Especially, after the release of the MEA (2005) several authors and projects have been dealing with analysing, assessing and evaluating ecosystem services (e.g. Turner et al., 2003, Nelson et al., 2009; de Groot et al., 2010).

In Europe there have been such approaches as well. In Central and Eastern Europe in particular, both the analysis of landscape patterns and processes and the evaluation of landscape functions essential for land use planning have a long historical tradition (Buchwald and Engelhardt, 1968). In contrast to landscape ecology, the central notion in landscape development has always been that people are part of the landscape and that landscapes are changed for their benefit (Linehan and Gross, 1998; Antrop, 2001; Gobster and Westphal, 2004; Nassauer et al., 2004). The concept of 'landscape functions', which are defined as *'the capacity of a landscape to sustainably fulfil basic, lasting and socially legitimised material or*

immaterial human demands' (Langer et al., 1985; Kiemstedt, 1992; Marks et al., 1992) originated from established concepts like 'the natural territorial potentials' developed by Neef (1966 – 'gebietswirtschaftliches Potential'), Langer (1970 – 'Naturpotential'), Haase (1978 – partielle Naturraumpotenziale), van der Maarel (1978 – landschaftliche Potenzen') as well as concepts based on Buchwald (1973), Bierhals (1988), Durwen (1995) and Grunewald and Bastian (2010).

The multifunctionality of forests and green spaces (Konkoly-Gyuró, 2011) is well established. The first papers on the environmental role and the beneficial functions of forests were composed as early as the 1960s (Speidel, 1966; Keresztesi, 1968; Heder and Meszoly, 1969; Thomasius, 1978; Mészöly, 1981). Hungarian foresters defined three basic functions (economic, environmental and recreational) that were accepted as a basic notion in forest management planning at the World Forestry Congress organized in Buenos Aires in 1972 (FAO, 1972; Lett, 2007). The multiple functions of green spaces were discussed in the book Environmental Planning (Konkoly-Gyuró, 2003) that documented the special ecological conditions of settlements and their various demands for the use of green spaces. The concept of agricultural multifunctionality originated from the first reform of the Common Agricultural Policy of the European Union in 1992 (Angyan et al., 1999) and aimed at implementing this concept into policy decisions. In this approach, the production, the social/cultural as well as landscape/nature-conservation role of agriculture was acknowledged (EEC, 1992). The European Charter of Rural Areas is another important document to emphasise the importance of multifunctionality in agriculture. Van Huylenbroeck et al. (2007) provide an overview of the great number of different publications regarding the topic of multifunctionality in agriculture.

Based on the concept of 'landscape functions' many studies dealing with different assessment methods, especially in the German speaking community, were carried out (Haber, 1979; Niemann, 1982; Bastian, 1996; Marks et al., 1992; Bastian and Schreiber, 1999; Leibowitz et al., 2000; Meyer and Grabaum, 2008). Bastian and Schreiber (1994), for instance, developed a framework for the assessment of landscape functions to support sustainable land use management.

It has to be stated that the definition of 'landscape function' as used in the European landscape planning discipline is similar to the understanding of the term 'ecosystem service' in the ESS concept; the former however, focuses more on the role of underlying values depending on specific human needs and societal norms (van Haaren and Albert, 2011). Landscape planning projects are therefore limited by valuation standards based on predefined values set by national governments. This normative understanding of 'function' differs from the term 'ecosystem function' in the ESS concept, which usually restricts 'functions' to natural capacities, properties and processes in ecosystems (Costanza et al., 1997, Daily et al., 2000) that are the basis for the provision of services (de Groot, 1992).

Due to the fact the international discussion about the concept of ESS applies similar terms with differing meanings; the terms have to be clearly defined in order to avoid confusion (Hermann et al., 2011). In this thesis the term 'function' follows the definition of Forman and Godron (1986): *'function is the interactions among the spatial elements, that is, the flows of*

energy, materials, and species among the component ecosystem' extended by the definition of de Groot (1992, 2006): *'functions are the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly'*. Based on a human ecological view (Linehan and Gross, 1998; Gobster and Westphal 2004; Nassauer et al., 2004) landscape, whether natural or transformed, provides a wide range of 'functions' for human society that can be translated into 'services' when they are actually used and valued by people (Termorshuizen and Opdam, 2009).

Among landscape planners and landscape ecologists as well as in the social sciences, the term 'landscape' is more popular than 'ecosystem'. 'Ecosystem' is usually not part of the language used in these scientific domains. The term 'ecosystem' was introduced by Tansley (1935). In the publication 'The use and abuse of vegetational terms and concepts' attention was drawn to the importance of the exchange of materials between organisms and their environment, regarding ecosystems as the basic units of nature (Tansley, 1947, Willis, 1997). Based on the definition of Tansley (1935) and subsequent developments, the Convention on Biological Diversity defined 'ecosystem' as follows: *'a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit'* (UN, 1992: Article 2).

More disciplines can identify with the term 'landscape' as it is seen as a broader concept than the term 'ecosystem'. However, the understanding of 'landscape' has been debated since the nineteenth century (cf. Riehl 1862; Seyfert 1903; Passagre 1920; Troll 1950; Schmithüsen 1976; Trepl 1995; Jessel 1998). In general, the scope of the definition is strongly influenced by the individual area of expertise. Turner et al. (2001) suggest a rather general description: *'a landscape is an area that is spatially heterogeneous in at least one factor of interest'*. This should ensure the independence of both scale (and size) and of human impact. The European Landscape Convention defines landscape as *'an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors'* (Council of Europe 2000: chapter I, article I). According to German landscape planning, 'landscape' is interpreted as *'a region of the Earth and aesthetic, cultural, ecological and economic system which reflects the human perception and realisation'* (von Haaren, 2001). As the term 'landscape' emphasises both the human perception and spatial aspects of the landscapes and conveys positive connotations, it seems more suitable for implementation into landscape planning projects.

Therefore, both terms 'landscape function' and 'landscape service' have recently become increasingly important in the ESS literature (Bastian and Schreiber, 1999; de Groot et al., 2010; Willemsen et al., 2010). In most cases the term 'landscape service' is just used as a specification of 'ecosystem' rather than an alternative. However, as the term 'ecosystem' is still more common in the ESS literature (Termorshuizen and Opdam, 2009), I decided to use this term in the remainder of the introduction chapter to avoid a permanent skipping between this two terms.

IV. Putting the concept into practice – limitations and challenges

Integrating the concept of ecosystem services in landscape planning decisions represents a great chance for a more balanced use of natural assets. However, the progress in this field, especially concerning the quantitative relations between services and the characteristics of landscape pattern, is not such that it can be fully incorporated into planning processes. Many limitations, obstacles and open questions regarding this process are documented and discussed in the relevant literature (de Groot et al., 2010; Hermann et al., 2011; Seppelt et al., 2011; Burkhard et al., 2012a). It has proven difficult to move from general announcements about the benefits people obtain from nature to credible, quantitative assessments of ESS. So far, there is no fully operational method of putting the concept into practice available. Due to the wide range of publications on the ESS concept, different approaches to and implementations of the concept occur (Hermann et al., 2011). This makes it difficult to assess the credibility of evaluation results and reduces the comparability of studies (Seppelt et al., 2011). One of the major shortcomings is the availability of useful data. Finding appropriate indicators related to the specific service and exploring how services are correlated with different management scenarios are questions yet to be resolved (Egoh et al., 2008; Willemen et al., 2008).

a) Landscape structure – service relations

Focusing on spatial pattern and scale, landscape ecology can provide useful input on how the spatial distribution of human activities influences important landscape processes and structures from which services are derived (Jones et al., 2008).

Landscape structure is dealing with the spatial relations between individual, distinct local ecosystems – the so-called landscape elements, as well as the distribution of energy, resources and species in relation to size, shape, number and type of these elements (Forman and Godron, 1986). **Landscape elements** are identified as the smallest units to be spatially mapped and can be delineated as ecologically homogeneous in regard to crucial land attributes like soil type, microclimatic and hydrological conditions, actual vegetation etc. Forman and Godron (1986) determined three basic types of landscape elements, according to spatial extent, shape and functionality: (1) the **matrix** consisting of the local ecosystems which cover a large part of the area and significantly influence the ecological functions of the landscape. This matrix holds the ‘background information’ which other important landscape features are interposed in; (2) **corridors** are characterized by their significant elongated shape as a line or a band depending on their width. They assume a crucial role in material and energy transport as well as in migration of animals and plants; (3) **patches** are incorporated in the matrix. These are small-scaled landscape elements that can be distinguished by their origin (regeneration, remnant, disturbance, resource dependant, introduced...), thus indicating significant differences in functionality and genesis of comparable types of mosaics.

The relationships between the different landscape elements might be most crucial in multifunctional, fragmented landscapes that have been heavily modified by human interventions and activities. In agricultural systems, which represent humankind's largest engineered ecosystem, spatial interactions between semi-natural habitats (e.g. field margins, fallows, and copses), intensively used ecosystems (e.g. fields, meadows), and elements of human origin such as footpaths and roads are causing synergies and trade-offs between different services (e.g. food production, local climate regulation, water supply, soil conservation). The flow of these services depends on how agricultural ecosystems are managed at the site scale as well as regarding the diversity, composition, and functioning of the surrounding landscape (Tilman, 1999). An example is the service '*pest control*' delivered by specific landscape elements (habitats) that either support parasitic and predatory insects by providing food and habitat or directly prevent the spread of the pest species (Hunter, 2002). Large-scale analyses reveal that the proportion of semi-natural habitats and patchiness of the landscape in agricultural systems may affect natural enemy populations and pest pressure in crops (Bianchi et al., 2006). Increased landscape complexity, which offers increased habitat and food sources for insects compared to mono-culture landscapes, is correlated with diversity of natural enemy populations (Thies and Tscharntke, 1999). Pest densities, however, may also respond positively to landscape complexity (Thies et al., 2005). As semi-natural habitats comprise many vegetation types, they have the potential to affect natural enemies and pests in a different way. Whereas the prime sources for most pest species are arable fields, semi-natural habitats may act as sources of pests for specific vegetation-pest species combinations. Thus, interaction between pest species and non-crop vegetation is seen as a key factor that can alter the outcome of landscape effects on natural pest control (Roschewitz et al., 2005; Thies et al., 2005). Although many investigations have been carried out to evaluate the effects of landscape structure on pests, there is a clear need for more studies to optimise management practices. So far, most studies have focused mainly on the stimulation of natural enemy populations as well as the colonization of arable fields by natural enemies. Only a limited number of studies have dealt with the relation between landscape structure and reducing pest densities and damage levels (see Thies & Tscharntke, 1999; Den Belder et al., 2002; Thies et al., 2003). Hence, there is a clear knowledge gap to be filled before the service '*pest control*' can be assessed in an adequate manner (Bianchi et al., 2006). In general, empirical evidence and well-developed models are in the early stages for most ESS, and more detailed case studies at the scale of typical land use decisions are needed (Kremen, 2005).

One of the most important research fields of landscape ecology, past and present, is in understanding the sustainable management of biodiversity (Poudevigne and Baudry, 2003). As biodiversity has multiple roles in relation to the delivery of ecosystem services its assessment represents a central component in the ESS research field (Kandziora et al., in press; Cardinale et al., 2012; Naeem et al., 2012; Mace et al., 2012). Biodiversity, for instance, enhances the efficiency of ecological processes such as primary production and decomposition. These processes are key determinants of ecosystem functions. Plant species

diversity in particular has an important role in structuring habitats, ecosystems and landscapes. Thus, sustainable conservation management in cultural landscapes needs indicators that provide quantitative links between landscape structure and biodiversity (Dauber et al., 2003). Empirical studies to date suggest that habitat loss, degradation and fragmentation have large, consistently negative effects on biodiversity (Fahring, 2003; Poudevigne and Baudry, 2003). Jeanneret et al. (2003) conclude that information on habitat mosaic provide more useful information on biodiversity than classical landscape metrics. Landscape ecologists have proposed many measures to assess the relationship between spatial heterogeneity and biodiversity in landscapes (Cullinan and Thomas, 1992; McGarigal and Marks, 1995). However, as data surveys are usually time-consuming and costly, nation-, or region wide biodiversity indicators are rare. Consequently, there is a need for new techniques to collect accurate data.

Remote sensing techniques, which can be partially supported by terrestrial field sampling, potentially provide an efficient means for regular indicator collection. According to SEBI 2010 (Streamlining European 2010 Biodiversity Indicators) (SEBI2010, 2004), the indicators among others could be 'ecosystems coverage', 'invasive alien species in Europe', 'fragmentation of natural and semi-natural areas', 'growing stock estimation' and 'dead wood'. Ongoing projects are investigating the feasibility and practical application of such methods to see whether they contain strong development potential in streamlining data acquisition over large areas.

b) Ecosystem service assessment framework

The data provided for decision making processes related to sustainable landscape development has to be relevant to the small scale at which local actors perceive their environment and make decisions on change. Therefore, information on service provision should be spatially explicit at the level of individual landscape elements, such as fields, margins and hedgerows. Depending on the composition of their elements different landscape mosaics provide different bundles of services to humans (Termorshuizen and Opdam, 2009). Different services are usually based on different service providing units at various scales (for instance, while provisioning services are often restricted to the local scale, cultural or regulating services are usually operating at a broader scale). It follows that a deeper understanding of the scales at which landscapes provide services to people is needed. The fact that ESS are generated at different scales (local, regional, global) has a big influence on the assessment by different stakeholder groups, on decision making and on the final management implementations. Hein et al. (2006) showed the differences between the spatial scale of ESS generation and the institutional scale of stakeholder benefits. To negotiate multifunctional goals, synergies and trade-offs between services have to be identified at different scales (Tippett et al., 2007; Opdam et al., 2008). As demonstrated by Zaccarelli et al. (2008), ecological cross-scale effects have to be included in ESS assessments. However, past studies tended to use just one single scale for the assessment of different

services with emphasis on regional or global scales, and/or simply provided aggregated values for large regions (Bastian et al., 2012). Although evaluation of trade-offs requires studies that include several services in one system, the majority of assessments are only related to a single service (Seppelt et al., 2011). The reason for this phenomenon is that detailed data and statistical relationships are often not available for a wide range of services. To overcome this problem, the explicit value transfer becomes a useful method (Bateman et al., 2002; Troy and Wilson, 2006; Brenner et al., 2010). Values and other data from the original study site are transferred to the designated policy site (Loomis, 1992). To give an example, in their paper Troy and Wilson (2006) presented a decision support system framework, which was built upon the value transfer methodology. Unfortunately, this approach also suffers from limitations, such as strength of the data and comparability between the source data and policy context (Troy and Wilson, 2006). Whereas some ESS provided at large scales (e.g. the avoided greenhouse gas costs of carbon sequestration) are easily transferable, other local scale services may have limited transferability (e.g. flood control values) (Farber et al., 2006).

Other studies made use of an assessment matrix, in which the capacities of land cover types (mostly CORINE data) to provide different services were assessed by expert knowledge or statistical data (e.g. Haines-Young and Potschin, 2008; Burkhard et al., 2009, 2012b). However, according to Eigenbrod et al. (2010) proxy data based on land use and land cover tend to be unreliable for deriving ecosystem service indicators compared to monitoring and field data. In their quantitative review of ESS studies Seppelt et al. (2011) stated that the methodological diversity employed illustrates that ESS research is currently a fragmented field. Setting up key research topics specific to ESS, they stressed the need for environmental data, model-based relations for ecosystem functions and feedback as well as the assessment of robustness and uncertainties. According to Barbier et al. (2008) a simplification of assessment methods is needed to effectively communicate with policy makers, but at the same time oversimplification is unsatisfactory for a comprehensive system description and can mislead decision-making. An intermediate approach seems to be the appropriate way to offer knowledge on ESS provision (see Nelson et al., 2009). Such an approach might be an innovative assessment framework that makes services spatially explicit by combining empirical data with spatial information (Willemsen et al., 2008; Kienast et al., 2009; Termorshuizen and Opdam, 2009). This framework should support higher relevance and comparability of ESS within sustainable landscape development and encourage interdisciplinarity. The resulting visualisation of ESS trade-offs at different scales might be used by decision makers, such as land managers and planners of ecologically sensitive regions, as a decision support tool for systematic evaluation of goal attainments and conflict detection.

OBJECTIVES AND OUTLINE OF THE THESIS

This thesis aims at contributing to the ongoing discussion on finding solutions for integrating the concept of ecosystem services into landscape ecology and decision making by

- (1) providing state-of-the-art knowledge on ecosystem service assessment in the field of landscape research as well as identifying specific knowledge gaps (Article 1)
- (2) improving the knowledge base on the relationships between landscape structure and ecosystem services with specific regard to the service '*pest control*' and introducing new ways of '*biodiversity*' assessment methods at landscape scale (Article 2, 3)
- (3) presenting and operationalising a spatially explicit assessment framework for various ecosystem services at different spatial scales applicable in transnational regions (Article 4, 5)

Chapter 1: The concept of ecosystem services regarding landscape research – a review

The first chapter presents a literature review about the assessment of ecosystem services in landscape research and provides a coherent knowledge base for further investigations. It includes both a description of the current state-of-the-art and an identification of knowledge gaps.

Article 1:

Hermann, A., Schleifer, S., Wrbka, T. (2011). The Concept of Ecosystem Services Regarding Landscape Research: A Review. *Living Rev. Landscape Res.*, 5, 1. Online version (accessed 10 October 2012): <http://www.livingreviews.org/lrlr-2011>

The paper initially provides the reader with an overview of the different ways of defining and classifying ecosystem services. Five selected classification systems that are commonly used in literature are discussed in detail. The second main part of this review focuses on up-to-date methods for quantifying and visualising ecosystem services and different valuation approaches. As there are still a lot of obstacles to be overcome regarding quantifying, mapping and putting a value on ecosystem services, the paper emphasises the importance of further research, initiatives and projects to improve the implementation of the ecosystem service concept in environmental planning and management at all levels of decision making.

Chapter 2: Landscape structure – service relations

This chapter focuses on the relationship between landscape pattern and process (landscape structure) and the provisioning of specific ecosystem services. Within an Austrian case study, correlations between landscape structure and the pest infestation level of wireworms, the larvae of click beetles, were explored, in order to gain more information for a successful wireworm risk assessment. In the second part of this section suitable instruments with special regard to remote sensing for surveying biodiversity indicators on the landscape scale are presented based on a literature review.

Article 2:

Hermann, A., Brunner, N., Hann, P., Wrbka, T., Kromp, B. (in press). Correlations between wireworm damages in potato fields and landscape structure at different scales. *Journal of Pest Science*

Wireworms, the larvae of click beetles, are world-wide pests of arable and vegetable crops. The aim of this study was to investigate whether different landscape variables, such as landscape composition, landscape complexity and ecological variables are related to wireworm infestation, represented by the proportion of wireworm damaged potato tubers. The study area was situated alongside the gradient from the lower warm-dry “Weinviertel” (region 1) to the cool-humid “Waldviertel” (region 2) in the North-East of Austria. Our results showed that wireworm damage was higher in region 2 than in region 1 and tended to be more damage in soils with higher sand content. Areas with a large proportion of grassy habitats, especially grassy field margins, were significantly positively correlated with wireworm damage. Consequently, farmers should pay more attention to field characteristics, such as sand content in soil and management like crop rotation.

Article 3:

Mücke, W. and **Hermann, A.** (2010). Estimation of biodiversity relevant quantities from airborne laser scanning data. *Österreichische Zeitschrift für Vermessung und Geoinformation*, Heft 4

Assessment of biodiversity is dictated by the European Union Fauna-Flora-Habitat directive in order to document the health of ecosystems. Remote sensing supported by terrestrial field sampling potentially provides an efficient means for a regular monitoring cycle. In this paper it is demonstrated how data collected with airborne laser scanning (ALS), especially with full-waveform (FWF)-ALS, can be used to derive quantities relevant for biodiversity assessment. These can be terrain surface features like roughness, as well as vegetation parameters like height, predominant tree type, location of fallen trees or forest layer structure.

Chapter 3: Providing a spatially explicit assessment framework

The third chapter presents a spatially explicit assessment methodology for a wide range of ESS based on expert knowledge and field surveys. Embedded in a spatial reference framework, the various services were evaluated with different approaches at different spatial scales dependant on their service providing units. However, the possibility exists to use only parts of the proposed methodology to generate information on selected services, if data availability is limited. Different approaches were tested in three case studies, two transnational Austria/Hungary and Germany/Czech Republic and one national study in the Czech Republic. The pros and cons of the different approaches were presented and discussed.

Article 4:

Hermann, A., Kuttner, M., Hainz-Renetzeder, C., Konkoly-Gyuró, E., Tirászi, A., Brandenburg, C., Allex, B., Ziener, K., Wrblka, T. (accepted). Assessment framework for landscape services in European cultural landscapes – an Austrian Hungarian case study. *Journal of Ecological Indicators*.

The aim of this study was to develop an integrative assessment framework for evaluating a wide range of landscape services at different spatial scales. The proposed methodology was applied within the cross-border region of Austria and Hungary. Embedded in a spatial reference framework, we assessed and mapped five main landscape services: *regulation, habitat, provision, information and carrier*. Considering location and spatial extent, three different approaches of service assessment were distinguished: (1) the Landform Approach was based on indicators that could be observed directly either by the use of Corine land cover or by clearly identifiable spatial indicators (2) The Broader Habitat Approach focused on assessment at the landscape element level. This was achieved by using of an expert driven capacity matrix as a basis, with values being revised by semi-quantitative data gained from field work. (3) The Socio-cultural Approach focused on the assessment of homogenous regions, representing a combination of relief, dominant land cover and land use intensity. For this approach, indicators were mainly defined based on geo-data. The results presented hot and cold spots of service provision at different spatial scales as well as the trade-offs between the different services.

Article 5:

Hermann, A., Tirászi, A., Bianchin, S., Skakanová, H. (in press). Services landscapes provide to society – regional case studies in the project regions North, Central North and Central South. In Csaplovics E., ed.: *Transnational Ecological Networks in Central Europe: Investigating natural and cultural values of landscapes without borders*. Rhombos-Verlag, Berlin.

Despite a multiplicity of conservation activities, valuable ecological networks are getting lost. Therefore, in order to enhance connectivity the status quo concerning ecological and socio-cultural values of these transnational networks has to be evaluated. To conduct comprehensive analyses, different levels of service assessment regarding spatial scale are distinguished. Whereas the studies in the project regions North and Central North derived their results only from connecting expert driven service values to spatial units, the study in Central South integrated habitat specific qualifiers into their assessments in addition to that, in order to be locally explicit. The final landscape maps may be used as a decision tool in landscape planning processes and might serve as a knowledge base for future conservation projects in order to enhance service delivery.

PROJECT FRAMES OF THE ARTICLES

BIOSERV

Articles 1 and 4 originated from the international project BIOSERV 'Biodiversity and Ecosystem Services as scientific foundation for the sustainable implementation of the Redesigned Biosphere Reserve Neusiedler See' that was funded by the Austrian Academy of Sciences. This 4 year project was implemented by the University of Vienna, Dep. Conservation Biology, Vegetation and Landscape Ecology (project leader), by the University of Natural Resources and Applied Life Sciences, Department of Landscape, Spatial and Infrastructure Sciences and by the University of Western Hungary, Institute of Forest resource Management and Rural Development.

Within this inter- and trans-disciplinary research project a scientifically sound baseline for a new delineation and zoning of the Biosphere Reserve (BR) 'Neusiedler See' according to the Seville criteria and for the description of management objectives with a special emphasis on stakeholder interests was built. Another aim of this study was to initiate a concept for the continuation of systematic ecological and socio-economic research. This has been achieved by using the scientific concept of "ecosystem goods and services" to identify, assess and communicate the ecological and socio-cultural values of the proposed enlarged Biosphere reserve.

Whereas article 1 *'The Concept of Ecosystem Services Regarding Landscape Research: A Review'* served as scientific basis for the development of the assessment framework used in this project, article 4 *'Assessment framework for landscape services in European cultural landscapes – an Austrian Hungarian case study'* presents the applied methodology as well as the final results in detail.

TransEcoNet

Article 1, 3, 4 and 5 aimed at contributing to the goals of the integrated project TransEcoNet ‘Transnational ecological networks – Landscapes without borders’ funded by Central Europe from 2009 until 2012. The overall objective of TransEcoNet is a comprehensive inventory and the protection of ecological networks along with their natural and cultural heritage. In total, 15 project partners from six different countries analysed and assessed parts of the Central European network within or between the wide-ranging ecological networks of the Alps, the Carpathians and the European Green Belt regarding (i) transboundary connectivity and existing gaps, (ii) land use changes since the end of the 18th century, (iii) ecological functionality of landscapes and services they provide to society and (iiii) national planning tools, regulations and transnational cooperation initiatives dealing with ecological networks. Consequently, the project results contribute to the pan-European interlinkage of valuable ecosystems and a sustainable territorial development.

The presented articles were produced in the framework of the workpackage 3 (WP leader: the University of Vienna), mainly targeting strategies for biodiversity conservation and a sustainable management of ecosystem services in transnational ecological networks.

Drahtwurm

Article 2 originated in the framework of the Austrian national project Drahtwurm ‘New approaches to regulate wireworms, with a special emphasis on organic farming’, led by the Austrian research institute Bio Forschung Austria. This four year project (2006 - 2010) was implemented by the project partners University of Natural Resources and Applied Life Sciences, University of Innsbruck and University of Vienna. It aimed at finding solutions for the increasing wireworm damages in organic maize production and other field crops. The main tasks were to explore the relationships between site-specific data, such as soil and crop rotation data, landscape parameters (see Article 2), climate data and wireworm infestation level. From 2007 to 2009, a total of 63 organic fields and 45 conventional fields situated in Lower Austria, Upper Austria, Burgenland and Styria were examined by means of wireworm-bait traps and damage ratings as well as. In addition to that, farmers were asked to provide information on crop rotation and cultivation methods for each field. Site-specific soil, climate and landscape parameters were collected from databases or field surveys. Due to uncertainty of morphological determination, the development of a molecular determination method for quick and confident identification of European *Agriotes*-species was a great challenge in this project. The project results were continuously disseminated to different target groups. Farmers and advisers were informed in single talks and seminars, the scientific community by presentations at congresses and workshops.

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CHAPTER 1

The concept of ecosystem services regarding landscape research – a review

Article 1

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Abstract

The awareness that natural and semi-natural ecosystems provide benefits to human society, which are of great economic, ecological and socio-cultural value, can be dated back to the mid-1960s and early 1970s. More recently, there has been an almost exponential growth in publications on the benefits of natural ecosystems to human society. However, despite the enhancing interest in ecosystem service research, still many open questions remain to fully integrate the ecosystem service concept in landscape research and decision making. The paper aims at providing the state-of-the-art of ecosystem service assessment regarding landscape research and to present a coherent knowledge base for further discussions. First the paper gives an overview of the different ways defining and classifying ecosystem services. Five selected typologies, very common in the literature are discussed in detail. The second main part of this review focuses on quantifying and mapping ecosystem services as well as on the different valuation approaches. As there are still a lot of challenges that have to be faced regarding quantifying, visualising as well as valuing ecosystem services the paper emphasizes the importance of further research, initiatives and projects to improve the implementation of the ecosystem service concept in environmental planning and management at all levels of decision making. To meet all these challenges research effort needs to be conducted side by side to understand underlying relationships and to improve ecological as well as socio-economic understanding.

Keywords: ecosystem services/functions, landscape services/functions, classifying, quantifying and mapping, valuation

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Contents

1	Introduction	5
2	Definitions of the different key terms	7
3	Classification systems and their different typologies	10
3.1	Presentation of five selected classification systems	10
3.2	Comparison of different typologies	13
3.3	The problem of double counting	14
3.4	Further developments of classification systems	14
4	Quantifying and mapping	15
5	Valuation	16
5.1	The ecological, economic and socio-cultural value of ecosystems	16
5.2	Different valuation methods	18
6	Discussion	22
6.1	Definitions and classifications – a challenge	22
6.2	Quantifying and mapping – their limitations	22
6.3	Multifunctionality	23
6.4	Different disciplines – advantages or hindrances?	23
6.5	Valuation and the future generation	24
7	Acknowledgements	25
	References	26

List of Tables

1	Comparison of five selected classification systems	11
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1 Introduction

The concept of ecosystem services is seen as a promising approach communicating the links between ecosystems and human well-being (MEA, 2005). Although the term “ecosystem services” was primary introduced by Ehrlich and Ehrlich (1981), the concept’s origin of the modern history dates back to the late 1960s and 1970s, highlighting the societal value on nature’s functions (King, 1966; Helliwell, 1969; Ehrlich and Ehrlich, 1970; Dee *et al.*, 1973; Ehrlich *et al.*, 1977; Bormann and Likens, 1979). In the 1970s and 1980s, it was already started to point out societal and economic dependence on natural assets in order to attract public interest on biodiversity conservation (e.g. Westman, 1977; de Groot, 1987). Important milestones in the mainstreaming of ecosystem services were on the one hand Daily’s book *Nature’s Services: Societal Dependence on Natural Ecosystems* (Daily, 1997) and on the other hand the paper by Costanza *et al.* (1997) on the value of global natural capital. The monetary figures presented in the last one resulted in a high impact on both science and policy making. Especially after the release of the Millenium Assessment (MEA, 2003), which focused on the benefits people derive directly and indirectly from ecosystems, the literature concerning ecosystem services has increased exponentially (Fisher *et al.*, 2009). Since then, several authors and projects have been dealing with classifying, quantifying, mapping and valuing of ecosystem services in order to integrate the concept into decision making processes (Costanza *et al.*, 1997; Wilson and Carpenter, 1999; Heal, 2000; de Groot *et al.*, 2002; MEA, 2003; Turner *et al.*, 2003; MEA, 2005; de Groot, 2006; Fisher *et al.*, 2009; de Groot *et al.*, 2010; Rounsevell *et al.*, 2010).

As landscapes are considered to be multifunctional and are subject to a wide range of land uses, the concept of landscape functions or services, used as synonym to ecosystem services, raised much attention in the field of landscape ecology and landscape planning. The central notion in landscape development has always been that people are part of the landscape and that landscapes are changed for their benefit (Linehan and Gross, 1998; Antrop, 2001). Because landscape sciences focus on spatial pattern and scale, they can provide useful insights into how the spatial distribution of human activities influences important landscape processes and structures from which services are derived (Jones *et al.*, 2008). Especially in Central and Eastern Europe both the analysis of landscape pattern and processes and the assessment of landscape functionality as a precondition for land use planning have a long tradition (Buchwald and Engelhardt, 1968). The idea of landscape function assessment (Bastian and Schreiber, 1994; Lee *et al.*, 1999) traces back to the multifunctionality concept of forests and green spaces (Konkoly-Gyuró, *in press*). Whereas the term “natural territorial potentials” (Troll, 1950; Neef, 1966) was too abstract for practical landscape planning, the concept of landscape functions arose. Bastian and Schreiber (1994) for instance, developed a framework for the assessment of landscape functions to support sustainable land use management. Based on this concept, many studies dealing with different assessment methods, especially in the German speaking community, were carried out (Haber, 1979; Niemann, 1982; Bastian, 1997; Bastian and Schreiber, 1999; Leibowitz *et al.*, 2000; Steinhardt and Volk, 2003; Palmer, 2004; Meyer and Grabaum, 2008).

However, despite the great interest in this research topic there are still remaining challenges which need to be addressed to fully integrate the concept of ecosystem services into landscape planning and decision making (de Groot *et al.*, 2010). The development of an integrative framework (Figure 1), which fully take the ecological, the economic as well as the socio-cultural values of landscapes into account, is still in process. Such a framework should be comprehensible, feasible and able to be applied at wide range of scales to different ecosystems or landscapes (Hein *et al.*, 2006). In the literature many limitations, obstacles and open questions regarding this process are documented and discussed (de Groot *et al.*, 2010). Because of the wide range of publications on the ecosystem service concept, different approaches to and implementations of the concept occur.

This paper aims at presenting the state-of-the-art of ecosystem service assessment regarding

landscape research. The target is to provide a coherent knowledge base contributing to the ongoing discussion process on finding solutions for integrating the concept of ecosystem services into landscape planning and decision making. The first sections of this paper address different key definitions used within the concept of ecosystem services and different classification systems of the services. The following sections illustrate various approaches and challenges of quantifying and mapping, different aspects of valuation methods and conclude with a discussion. Although the economic aspect within the concept of ecosystem services also remains as a main challenge, it is only marginally addressed in the present paper, because a review of economic valuation would go beyond the scope of this study. For more detail on this thematic, please refer to other reviews focusing on the economic approach (e.g. Peterson and Sorg, 1987; Pearce and Moran, 1994; Gómez-Baggethun *et al.*, 2010).

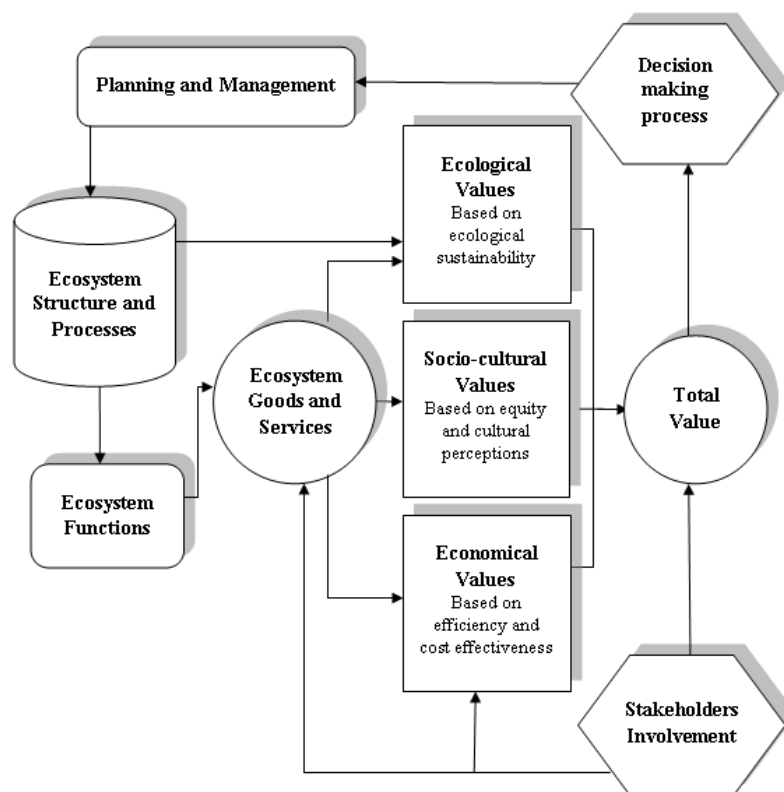


Figure 1: Valuation framework integrating the ecosystem service concept into sustainable landscape planning and management; Taking into account the total landscape value (including ecological, socio-cultural and economical values) in decision making processes effects indirectly the provision of services (adopted from de Groot, 2006).

2 Definitions of the different key terms

If the ecosystem service concept is designed to provide an effective framework for natural resource management decisions, ecosystem services have to be defined and classified in a way that allows comparisons and trade-offs amongst the relevant set of potential benefits. A number of scientists have attempted to construct typologies of ecosystem services (e.g. [Daily, 1999](#); [de Groot, 2006](#); [Boyd and Banzhaf, 2007](#)). However, ambiguity in the definitions of key terms – such as ecosystem processes, functions, services and benefits – makes it difficult to develop a coherent decision framework ([Wallace, 2007](#)). For meaningful comparisons across time and space, clear definitions of the key terms are required ([Boyd and Banzhaf, 2007](#); [Wallace, 2007](#)). However, according to [Boyd and Banzhaf \(2007\)](#) ecology and economics have failed to standardize the definition and measurement of ecosystem services. The following brief survey of definitions reveals multiple, competing meanings of the key terms used in the literature referring to the ecosystem service concept.

Ecosystem processes. According to the Elsevier's Dictionary of Biology ([Tirri *et al.*, 1998](#)) “process” is defined as “*a series of events, reactions or operations, achieving a certain definite result*”. Ecosystem processes are seen therefore as the complex interactions among biotic and abiotic elements of ecosystems, encompassing in broad terms material cycles and flow of energy ([Lyons *et al.*, 2005](#)). Although this definition is widely accepted, scientists interpret and classify processes in different ways. [Balmford *et al.* \(2008\)](#), for example, distinguish between “Core Ecosystem Process” (e.g. production, decomposition, nutrient and water cycling), “Beneficial Ecosystem Process” (e.g. biomass production, pollination, biological control, habitat and waste assimilation), and “Benefits” (e.g. food, fresh water).

Ecosystem functions. [De Groot \(1992\)](#) defines ecosystem functions as “*the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly*”. Functions therefore are the subset of biophysical structures and processes that provide services ([de Groot *et al.*, 2010](#)). They can refer variously to the habitat, biological or system properties or processes of ecosystems ([Costanza *et al.*, 1997](#)). Most authors agree that goods and services are generated by ecological functions (or processes) (e.g. [Costanza *et al.*, 1997](#); [Daily, 1997](#); [Farber *et al.*, 2006](#)). [Jax \(2005\)](#) notes that the term “ecosystem function” is considered as “capability”, but is often used more generally to refer to processes that operate within an ecosystem, like nutrient cycling or predation. Often the two terms ecosystem functions and ecosystem processes are commonly used as synonyms even within the same study (see [Costanza *et al.*, 1997](#)).

Ecosystem services. Ecosystem services can be simply defined as a set of ecosystem functions that are useful to humans ([Kremen, 2005](#)). They are consequences of supporting processes acting at various temporal and spatial scales ([Farber *et al.*, 2006](#)). These general definitions are widely accepted. However, when trying to classify services and applying this framework in decision making processes, several uncertainties are revealed. There exist various semantic classes of the term ecosystem services, depending on the specific goal or background ([Fisher *et al.*, 2009](#)). According to [Costanza and Folke \(1997\)](#) ecosystem services “*represent the benefits human populations derive, directly or indirectly, from ecosystem functions*”. In [Daily \(1997\)](#) ecosystem services (also referred to as nature's services) are the “*conditions and processes*”, as well as the “*actual life-support functions*”. Following [Eichner and Tschirhart \(2007\)](#), those biological resources are referred to as ecosystem services, which provide inputs into both production processes and consumers' well-being. The definition in the [MEA \(2003\)](#), which has been widely taken-up in the international research and policy literature, highlights the strong relation of ecosystem services to the benefits people derive directly or indirectly from ecological systems. Based on the MEA approach, the TEEB

(The Economics of Ecosystems and Biodiversity) project defines ecosystem services as direct and indirect contributions of ecosystems to human well-being (TEEB, 2010).

Boyd and Banzhaf (2007) provide an alternative approach. In their definition, ecosystem services are ecological components (including ecological structure) directly consumed or enjoyed to produce human well-being. Thus indirect processes and functions are not ecosystem services, but intermediate ecological components. For instance, recreational angling is seen as a benefit with multiple inputs. Whereas the water body and the target fish population are final services, the food web and water purification land uses on which the fish population depends, are intermediate components, because they are not directly related to the benefit (Figure 2).

Benefit	Final Services	Intermediate Components
Recreational angling	The water body	The water body's quality
	The bass population	
	The riparian forest	
Drinking water	The water body's quality	Wetlands, natural riparian land cover



Figure 2: Final Services vs. Intermediate Components regarding the benefits “recreational angling and drinking water”; Whereas Intermediate Components are indirect processes and functions, Final Services are directly consumed or enjoyed to produce human well-being (after Boyd and Banzhaf, 2007).

In contrast to the definition above, Fisher *et al.* (2009) suggest that ecosystem services are “the aspects of ecosystems utilized (actively or passively) to produce human well-being”. Therefore services encompass ecosystem organization and structure as well as process and/or functions if they are consumed by humanity either directly or indirectly (Figure 3).

Intermediate Services	Final Services	Benefits
pollination	clean water provision	drinking water; domestic use water
primary productivity	storm protection	property protection; decreased livelihood vulnerability
water regulation	constant stream flow	recreation; water for irrigation; water for hydroelectric power

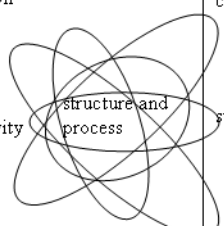


Figure 3: Conceptual relationship between Intermediate and Final Services; Structure and Processes become Intermediate Services if there are humans that benefit from them. Interactions among several Intermediate Services produce Final Services such as “clean water provision” and “storm protection” (after Fisher *et al.*, 2009).

Landscape services vs. ecosystem services. Another approach is to define functions and services at landscape scale to integrate the concept into land management decisions. The awareness that landscapes provide a multitude of functions and are subject to many possible land uses, gives rise to increasing research interest on the linkages between land use and land(sape) functions (see Bakker and Veldkamp, 2008; Pérez-Soba *et al.*, 2008; Verburg *et al.*, 2009). Therefore, recently the terms “landscape function as well as landscape service” have become more important in the literature (Bastian and Schreiber, 1999; de Groot *et al.*, 2010; Willemen *et al.*, 2010). As “landscapes” (contrary to “ecosystem”) may be more attractive to non-ecological scientific disciplines and may be associated with people’s local environment, the term “landscape services” is preferred as a specification (rather than an alternative) of ecosystem services. In addition, the terms “environmental” and “green” services are used in some articles (Termorshuizen and Opdam, 2009).

Within this present paper landscape functions and services are used as a synonym to ecosystem functions and services. As the debate on the definitions is still going on and several authors have different interpretations and preferences, we don’t follow specific definitions of the key terms in order to assure the provision of an overview.

Benefits. A benefit is something that directly impacts on the well-being of people (Fisher and Turner, 2008). Well-being is declared as the opposite end of a continuum from poverty, which has been defined as a “*pronounced deprivation in well-being*” (MEA, 2005). As well-being is dependent on one’s situation, cultural and ecological circumstances, benefits are spatially explicit (Boyd and Banzhaf, 2007). Resources of well-being encompasses factors like aesthetic, enjoyment, various forms of recreation, maintenance of human health, physical damage avoidance, and subsistence of food (Boyd and Banzhaf, 2007). Defined this way, benefits can be seen as the link between human welfare and ecosystems, on which theoretically an economic value can be put on. The benefits humans gain from ecosystems are derived from services (Fisher and Turner, 2008). As mentioned above, the MEA (2003) and also other scientists (e.g. Costanza *et al.*, 1997; Wallace, 2007) consider services and benefits to be the same.

Recently, another scientific discourse has suggested that human well-being is not only dependent on nature, but also on other landscape elements, which have therefore to be also taken into account (Carlisle *et al.*, 2009). Especially, in affluent societies well-being can be understood as socio-cultural constructions of modernity which comply with the demands of a capitalist economic system (Eckersley, 2005). A multidisciplinary and culturally informed focus on well-being is thus necessary to be able to realise that certain aspects of “modern life” affect the physical environment on which humanity depends.

Conceptual relationship among the “key terms”. Haines-Young and Potschin (2010) provide a valuation framework for linking ecosystems to human well-being, which has been used in several projects, for instance, the TEEB project (TEEB, 2010) (Figure 4). The proposed diagram makes a distinction between ecological processes and functions as well as the provided services and the outputs considered for humans as benefits. However, in the real world the relationship is not as simple and linear as illustrated in the diagram. Although the general structure of the suggested framework is widely agreed upon, the distinction between the terms “function”, “service” and “benefit” is still under discussion (de Groot *et al.*, 2010). Fisher *et al.* (2009), for example, propose a different conceptual relationship between the key terms (Figure 3). It shows how joint products (benefits) can stem from individual services. Intermediate services are based on complex interactions between ecosystem structure and processes and lead to final services, which provide human welfare benefits.

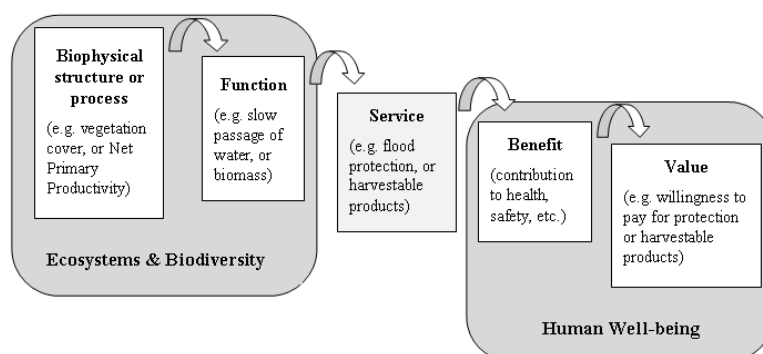


Figure 4: Conceptual relationship between Ecosystems & Biodiversity and Human Well-being (adopted from Haines-Young and Potschin, 2010).

3 Classification systems and their different typologies

3.1 Presentation of five selected classification systems

Although in the ecological literature, the key terms “ecosystem process”, “ecosystem function”, “ecosystem service” and “benefit” have been subject to various and sometimes contradictory interpretations, a wide range of authors have attempted to provide a systematic typology and comprehensive framework for integrated assessment and valuation of ecosystem goods and services (see Daily, 1997; de Groot *et al.*, 2002; MEA, 2005; de Groot, 2006; Boyd and Banzhaf, 2007; Fisher and Turner, 2008). However, because of the dynamic and complexity of ecosystems a single, consistent classification typology is difficult to develop (Costanza, 2008). There are many useful ways to classify ecosystem goods and services dependent on the different purposes of use.

Since a pluralism of typologies exists, we only illustrate some selected examples, which demonstrate different approaches and developments to classify ecosystem functions and services. Five different classification systems are presented, which are applied in many assessments and are used often as basis for further classification developments (Table 1). We have selected studies, which have shaped and differentiated the ecosystem service research community from the beginning (Costanza *et al.*, 1997; Daily, 1999; MEA, 2003) as well as typologies aiming at integrating the concept of ecosystem services into landscape planning and management within a European context (Bastian and Schreiber, 1999; de Groot *et al.*, 2010). In addition, two further classification approaches are presented, which show examples for further developments and adaption of the current typologies in the literature for regional as well as international integrated landscape planning projects.

Table 1: Comparison of five selected classification systems. Different typologies are presented, which are applied in many assessments and are often used as basis for further classification developments (Costanza *et al.*, 1997; Daily, 1999; MEA, 2003; de Groot *et al.*, 2010; Bastian and Schreiber, 1999).

Costanza et al. (1997)	Daily (1999)	MEA	de Groot et al. (2010)	Bastian and Schreiber (1999)
–	Production of goods	Provisioning services	Provisioning services	Production (economical function)
food production (e.g. fish, game, fruits)	food	food	food	renewable resources (herbal and animal biomass) / non-renewable resources (mineral raw materials and fossil fuel)
raw materials	durable materials (natural fiber, timber)	fibre	fiber, fuel, other raw materials	
	energy (biomass fuels)	biomass fuels		
	industrial products	–	biochemical products and medicinal resources	
	pharmaceuticals	bio-chemicals, natural medicines, etc.		
–	ornamental resources	ornamental species and/or resources		
genetic resources	genetic resources	genetic resources	genetic materials	
water supply	–	fresh water	water	available renewable resource: water
–	Regeneration processes	Regulation services	Regulation services	Regulation (ecological function)
gas regulation	cycling and filtration processes	air quality regulation	air quality regulation	regulation of material- and energy-cycles
water regulation		water regulation	water regulation	
waste treatment		water purification and waste treatment	waste treatment	
erosion control and sediment retention		erosion regulation	erosion protection	
pollination	translocation processes (dispersal of seeds, pollination)	pollination	pollination	
–	Stabilizing processes	–	–	
disturbance regulation	regulation of hydrological cycle	–	natural hazard mitigation	
	coastal and river channel stability	storm protection		
climate regulation	moderation of weather extremes	climate regulation	climate regulation	
	partial stabilization of climate			
biological control	control of pest species	human disease regulation	biological regulation	regulation and regeneration of population and biocoenose
	compensation of one species for another under varying conditions			

Table 1 – *Continued*

Costanza et al. (1997)	Daily (1999)	MEA	de Groot et al. (2010)	Bastian and Schreiber (1999)
–	–	Supporting services	–	–
nutrient cycling	–	nutrient cycling	–	–
soil formation	–	soil formation and retention	soil formation and regeneration	–
–	–	photosynthesis	–	–
–	–	primary production	–	–
–	–	water cycling	–	–
–	–	–	Habitat or supporting services	–
–	–	–	genepool protection	–
refugia	–	provision of habitat	nursery habitat	–
–	Life-fulfilling functions	Cultural services	Cultural & amenity	Habitat (social function)
recreation	–	recreation and eco-tourism	recreation and tourism	recreational function
–	–	–	–	human ecological function (e.g. filter- and buffer functions)
cultural	–	cultural heritage and diversity, sense of place	cultural heritage and identity	psychological function (aesthetic, ethic)
	aesthetic beauty	aesthetic values	aesthetic	
	cultural, intellectual and spiritual inspiration	inspiration	inspiration for culture art and design	
		spiritual and religious values	spiritual & religious inspiration	
	–	educational values	education & science	information function (science, education)
	scientific discovery	knowledge systems		
–	serenity	–	–	–
–	existence value	–	–	–
–	Preservation of options	–	–	–
–	maintenance of the ecological components and systems needed for future supply	–	–	–

Costanza et al. (1997) tried to estimate the current economic value of renewable ecosystem services for 16 biomes, based on published studies and a few original calculations. For the purposes of this analysis the selected ecosystem services were categorised into 17 major groups. According to **Costanza et al. (1997)** ecosystem services represent the benefits humans derive, directly or indirectly, from ecosystem functions. Some ecosystem services are the product of more than one function, and one single function can contribute to two or more services. The classified ecosystem services represent the basis for further studies (e.g. **de Groot et al., 2002**; **MEA, 2005**; **de Groot, 2006**).

According to **Daily (1999)** natural ecosystems and their related biodiversity are seen as capital assets that will yield a wide range of life-supporting goods and services over time. Benefits, which derive from ecosystems, will therefore enhance human welfare. In order to support sustain-

able ecosystem service management, [Daily \(1997\)](#) developed a conceptual framework for assessing ecosystem services and their trade-offs and revised it two years later ([Daily, 1999](#)). The “new” classification system encompasses the production of goods, regeneration processes, stabilizing processes, life-fulfilling functions, and conservation of options.

Using the definition of ([Costanza *et al.*, 1997](#)) [see Section 2 on definitions], the **Millennium Ecosystem Assessment** ([MEA, 2003](#)) provides a simple typology of services that has been widely taken-up in the international research and policy literature. Four broad types of service are suggested: “Provisioning services”, “Regulating services”, “Cultural services” and “Supporting services”. This classification is understandably not meant to fit all purposes, which has been pointed out for contexts regarding environmental accounting, landscape management and valuation, for which alternative classifications have been proposed (e.g. [Boyd and Banzhaf, 2007](#); [Wallace, 2007](#); [Fisher and Turner, 2008](#)).

Following [de Groot *et al.* \(2010\)](#) ecosystem functions are intermediate between processes and services and can therefore be defined as the “capacity of ecosystems to provide goods and services that satisfy human needs, directly and indirectly ([de Groot, 1992](#)). The provided typology is mainly based on the [MEA \(2003\)](#) and [de Groot \(2006\)](#). Four broad types of services are distinguished: “provisioning services”, “regulating services”, “habitat or supporting services” and “cultural and amenity services”. This classification concept was established aiming at integrating the concept of ecosystem services and values into landscape planning, management and decision making ([de Groot *et al.*, 2010](#)).

[Bastian and Schreiber \(1999\)](#), that are well known in the German speaking community base their classification approach on a long lasted research history in landscape functioning and management. The so-called landscape functions are divided into three groups: “production functions” (economic functions), “regulation functions” (ecological functions) and “habitat function” (social function). Each group is again classified into main-functions and sub-functions so that the cause and effect chains and interactions between land-use demand on the one hand and landscape structure on the other hand are observable ([Bastian, 1991, 1997](#); [Bastian and Schreiber, 1999](#)).

3.2 Comparison of different typologies

Whereas [Costanza *et al.* \(1997\)](#), the [MEA \(2005\)](#), and [de Groot *et al.* \(2010\)](#) focus on ecosystem services, [Bastian and Schreiber \(1999\)](#) refer to landscape functions (Table 1). [Daily \(1999\)](#), in comparison to them, includes in her classification both goods, processes and functions.

The typology of the ecosystem goods, services and functions is among these five broadly the same (except for the services of [Costanza *et al.* \(1997\)](#), which are often used as the basis for further developments). The groups “provisioning services”, “production of goods” as well as “production function” represent the presence of a large variety of living biomass, which provides many goods for human consumption e.g. food, raw materials and genetic material. “Regulation” or “regeneration processes” relate to the capacity of ecosystems to regulate essential ecological processes and life support systems. Whereas Daily separates the group “stabilizing processes” from “regeneration processes”, the MEA introduces the group “supporting services”. In contrast to the others, [de Groot *et al.* \(2010\)](#) include in their system the group “habitat or supporting” services, which are limited to two services (gene pool protection and nursery habitat). Thereby it is stressed that ecosystems provide refuge and reproduction-habitat that support ecological balance and evolutionary processes. Bastian and Schreiber also include “habitat function” but in the terms of social functions, that can be compared with the “cultural services” and “life-fulfilling functions” of the other authors. Although the typologies of these selected classification systems seem to be similar, the allocation of the services is varying, due to the different definitions of ecosystem goods, services, processes and functions and due to the different purposes of the assessments.

3.3 The problem of double counting

According to Wallace (2007), most of the proposed classification systems confuse *ends* with *means*. It should probably be distinguished between the benefit people enjoy and the mechanisms that give rise to that benefit. Assessed against these properties, any classification system containing both ecosystem processes and the outcomes of those processes within the same set will produce redundancy (Wallace, 2008). The fact that different ecosystem functions can deliver similar or equal services may lead to double counting in the assessment of the total value of ecosystems. Particularly, the regulation services are often still included in other services (Hein *et al.*, 2006). For instance, “pollination”, which is among others important for the maintenance of fruit production, is already included in the service “production of food”. Therefore, Hein *et al.* (2006) propose to include only regulation services if they provide a direct benefit to people living in the area or/and if they have an impact outside the ecosystem of consideration. Costanza *et al.* (1997) suggest establishing a general equilibrium framework that could directly incorporate the interdependence between ecosystem functions and services. Another approach to avoid double counting is distinguishing between final and intermediate goods, when valuating the total value (see Boyd and Banzhaf, 2007). Mäler *et al.* (2008) e.g. reorganized the MEA classification so that provisioning and cultural services are merged into a new category, final services, and the supporting and regulating services are merged into the category intermediary services. The reason for this is that both the cultural and provisioning services are affecting human well-being directly, whereas the two others are doing that only indirectly.

The TEEB project, which is mainly based on the MEA classification, shifted “supporting services” such as nutrient cycling and food-chain dynamics to ecological processes. The “habitat services”, instead, has been identified as a separate category to stress the importance of ecosystems to provide habitat for migratory species and gene-pool “protectors” (TEEB, 2010).

3.4 Further developments of classification systems

There exists a wide range of other useful ways to classify ecosystem functions, goods and services, like the suggestions from Costanza (2008) to classify by “spatial characteristics” or by the “excludability/rivalness” status of ecosystem services. The following presented classification systems demonstrate examples how the concept of ecosystem services can be applied to advanced international sustainability impact assessment projects as well as a comprehensive framework for analysing landscape functions in a coherent system.

The Integrated Project SENSOR (Helming *et al.*, 2008) aimed at developing ex ante Sustainability Impact Assessment Tools to support decision making on policies related to multifunctional land use in European regions and abroad. In the course of this project the concept of Land Use Functions (LUFs) (Pérez-Soba *et al.*, 2008), which are defined by the different land uses as the private and public goods and services, was developed. These functions include the most relevant economic, environmental and societal aspects of a region. Each LUF is characterised by a set of key indicators that assess the “impact issues” defined in the EU Impact Assessment Guidelines (European Commission, 2005). Nine LUFs were defined: The societal LUFs include “provision of work”, “human health” as well as “recreation and cultural functions”. Whereas the economic LUFs encompass “residential and land independent production”, “land-based production” and “transport functions”, the environmental LUFs cover “provision of abiotic resources”, “support and provision of biotic resources” and “maintenance of ecosystem processes”.

In comparison to other current classification systems a wide range of functions has been aggregated to three main function groups each again divided into three LUFs. On the one hand such a slim framework demonstrates a comprehensible communication tool to stakeholders, however on the other hand some loss of information has to be accepted. Great emphasis had put on reaching a balance between the main function groups within the assessments. However, this emerged very

difficult as the assessments of the functions groups societal, economic and environmental are based on different methods as well as within different spatial scales.

Recently a classification based on the Land Use Function concept has been provided including two main groups, namely the active and passive landscape functions (Konkoly-Gyuró, *in press*). Whereas the passive functions are divided into “regulating and life sustaining functions” of the natural systems (environmental regulation, habitat protection, biomass generation) and the “potentials” (biomass, raw material production and provision of territory for the different land uses and provision of information and aesthetics), the active functions are the services provided by human activities and artificial territories (settlements, infrastructure networks, recreation- and agricultural surfaces etc.). Considering the core idea of this concept, namely focusing on natural as well as human introduced landscape functions, it can be concluded that the benefits derived from non-natural landscapes transformed by human activities have also be taken into account into decision making. This coincides with the recently emerged approach that well-being can be understood as socio-cultural constructions of modernity, which often comply with the economic system (Eckersley, 2005). However, it is questionable if human transformed landscape functions are equally important as functions derived from natural ecosystems.

4 Quantifying and mapping

Dependent on data availability and spatial and temporal scales of assessments, different methods are available for quantifying and mapping landscape functions/ services. For assessments at global level as well as for rapid assessments, landscape functions and services can be determined directly by land cover or ecosystems using general assumptions from literature reviews. These methods are often applied when the economic value of the area is interesting (e.g. Naidoo and Ricketts, 2006; Troy and Wilson, 2006). However, a proper presentation of landscape functions/services would require also additional data beyond land cover observations. For example, the recreational function of a landscape is not only defined by the land cover of a specific location (e.g. natural area) but depends also on accessibility properties (e.g. distance to roads) and characteristics of the surrounding landscape (de Groot *et al.*, 2010). But in many cases this is only achievable at local or at least regional levels, because of data availability.

Kienast *et al.* (2009) present a framework for a spatially explicit landscape functions assessment at European scale, linking land characteristics with a high number of landscape functions. However, the assessments are often primarily based on area measurements and only marginally on measurements of quality (e.g. land use diversity, forest structure).

At regional or local scale a more data-driven method can be used. Function and service data are originated mainly from field observations, including census data, spatial policy documents and biophysical data. Willemen *et al.* (2008) present a methodological framework to quantify landscape functions and to make their spatial variability explicit. They distinguish three different methods depending on the measurable function: (1) linking landscape functions to land cover or spatial policy data, (2) empirical predictions using spatial indicators and (3) decision rules based on literature reviews (Willemen *et al.*, 2008). Whereas for some functions the exact location can be directly observed from the land-cover (e.g. wood for timber production), other functions such as recreation cannot be directly observed or only partially delineated and thus have to be empirically assessed based on landscape indicator analyses. If there does not exist any direct referenced information on the function's location (e.g. leisure cycling), we have to rely on landscape data based on expert knowledge, literature reviews or process models.

A lot of studies dealt with these challenges aiming at providing spatial datasets to map landscape functions (e.g. Chan *et al.*, 2006; Haines-Young *et al.*, 2006; Gimona and Van der Horst, 2007; Egoh *et al.*, 2008; Meyer and Grabaum, 2008). However, by doing the analysis major prob-

lems encountered. Finding appropriate indicators related to the specific service providing unit and exploring how functions and services are correlated with different landscape scenarios are still unresolved questions. To investigate the capacity of landscapes to provide services, landscape complexity and configuration analysis have to be addressed. Aspects such as size, form, and the border length between neighbouring land use types as well as the spatial connectivity of landscape units have to be taken into account. However, current landscape service indicators are still limited by insufficient data and an overall low ability to convey information (Layke, 2009).

Some indicators available are inadequate in characterizing the diversity and complexity of the services provided by landscape functions, especially concerning regulation as well as cultural services, which occur at various spatial scales. Ecosystems are complex, interrelated systems, in which processes take place over a range of spatial and temporal scales (Tansley, 1935) varying from competition between individual plants at plot level, via meso-scale processes such as fire and insect outbreaks, to climatic and geomorphologic processes at largest spatial and temporal scales (Clark *et al.*, 1979; Holling *et al.*, 2002). As service supply is dependent on ecosystem processes and functions, it may occur at different scales. Some services are even relevant at more than one scale. For instance regulation services can occur both at global scale (climate regulation) and plot-scale (biological nitrogen fixation) (de Groot, 1992). Also pressures on ecosystem services can have effects at different scales. In general physical processes on small scales are often driven by the impact on long period phenomena at large scales (climate patterns, hurricanes, fires) (Limburg *et al.*, 2002). However, large scale processes are also strongly influenced by smaller scale occurrences, for example, microbes respire enough CO₂ to keep many lakes and rivers supersaturated (Levin, 1992; del Giorgio *et al.*, 1997). Hence, for the analyses of the dynamics of ecosystem service supply it is very important to consider the drivers and processes at scales relevant for ecosystem service generation.

In addition, relevant to the time frame, ecosystems can act as service provider or suppressor (Martin and Blossey, 2009). For example, wetlands dominated by *Phragmites australis* can act as source and sink for greenhouse gases, depending on time scale (Brix *et al.*, 2001). The species assimilates atmospheric carbon dioxide through photosynthesis and through sequestration of organic matter produced in wetland soils. But it also emits methane into the atmosphere in a two stage process (Beckett *et al.*, 2001). Therefore, before an ecosystem can be seen as a service supplier, a time frame has to be defined for evaluation.

5 Valuation

5.1 The ecological, economic and socio-cultural value of ecosystems

Once the multifunctionality of landscapes and their services are identified, questions arise, like: How can we measure (value) the importance of these services, to get a basis for our decision making? How robust are the estimated values of ecosystem services? To answer these questions we have to address the terms “value” and “valuation”, which have different meanings in different disciplines:

Natural sciences. Most ecologists and other natural scientists would avoid to use the term “value”, except perhaps in its common usage as a reference to the magnitude of a number – e.g. “the value of a parameter” (Farber *et al.*, 2002), because ecosystems are seen to have an “intrinsic value”, which cannot be measured (Callicott, 1989). Nevertheless, some concepts of value are important in the natural sciences, and are commonly used to talk about causal relationships between different parts of a system: For example, referring to particular tree species and their value in controlling soil erosion in a high slope area, or to the value of fires in recycling nutrients in a forest (Farber *et al.*, 2002). Therefore, the ecological importance (value) of ecosystems is determined

by ecological criteria such as integrity, resilience and resistance (health). Ecological measures of value encompass parameters such as complexity, diversity and rarity (de Groot *et al.*, 2003). To integrate ecological values into landscape planning sustainable use-levels are often applied. Batabyal *et al.* (2003), for instance, propose to use a scarcity value which is described by ecological thresholds, as a measure for sustainable managing. Their study presents a formal model that explicitly analyses the connections between thresholds and ecosystem management (Batabyal *et al.*, 2003). The application of ecological modelling allows assessing the impact of environmental change and biodiversity loss on combined ecosystem services (Metzger *et al.*, 2006; Egoh *et al.*, 2008; Nelson *et al.*, 2009).

Another approach to value the impact of land use change on ecosystem services is the application of reference systems, e.g. the potential natural vegetation (PNV) (Tüxen, 1956). Tüxen emphasized the big value of PNV-maps for different purposes in landscape planning and nature conservation, particularly for forestry, agriculture and landscape management. However, maps of the potential natural vegetation are less useful for purposes of detailed planning on larger scales in cultural landscapes, where the reconstruction of the PNV has only hypothetical character (Zerbe, 1998).

Economy. In the economic context, the total value (TEV) of ecosystem services encompasses use values and non-use values. Use values include direct (consumptive and non-consumptive values) as well as indirect use values. Whereas direct consumptive values refer to ecosystem services like fish, fruits and some cultural services, direct non consumptive services refer for example to enjoyment of scenery or eco-tourism. Indirect use values relate to regulation services like pollination of crops, storm protection or flood prevention. The non-use values consider, for instance the importance people place on protecting nature for future use (option value) or because of ethical principles (bequest, existence and insurance value) (for more details see Pearce, 1991; Torras, 2000; TEEB, 2010).

To provide a common metric in which to express the benefits of diverse ecosystem services, the economic approach usually uses money as a general measurement unit. There exist many ways to translate the economic values into monetary terms. For details on valuation techniques see Dixon and Hufschmidt (1986); Peterson and Sorg (1987); Pearce and Turner (1990); Tietenberg (1992); Pearce and Moran (1994); Heal (2000); Turner *et al.* (2003) and the TEEB report (TEEB, 2010). Chee (2004), for instance, shows the principal methods for the monetary valuation and points out the pro and contra of these methods.

In general, there is a distinction between direct market valuation, indirect market valuation, contingent valuation and group valuation, each with its own associated measurement issues (de Groot *et al.*, 2002; MEA, 2003). Whereas services, which are directly linked to the market, can be easily valued according to their market price, non-market services are often valued using the “willingness to pay” or “willingness to accept” compensation methods encompassing “avoiding cost”, “replacement cost”, “factor income”, “travel cost” and “hedonic pricing” (de Groot *et al.*, 2002). In the last years “contingent valuation” and “group valuation”, which are based on an open public deliberation, have also become appreciate techniques for estimating values (Jacobs, 1997; Sagoff, 1998).

All these different methods have gained increasing attention concerning ecosystem service valuation and have become an applicable tool for estimating service values. Following proponents of monetary valuation techniques, these economic methods are able to illustrate the distribution of benefits, improve understanding of problems and trade-offs and can thus facilitate decision making (e.g. Aylward and Barbier, 1992; Salzmann *et al.*, 2001; de Groot, 2006). However, economic valuation of ecosystem services has reached its limits (e.g. Heal, 2000; Farber *et al.*, 2002; Wilson and Howarth, 2002; Chee, 2004; Hein *et al.*, 2006). Although it may encourage management options, decisions makers have to take into account the overall objectives and limitations of economic

valuation techniques (see [Ludwig, 2000](#)).

Socio-cultural sciences. Besides the ecological and the economical importance of ecosystems, natural and especially cultural landscapes offer a wide range of historical, national, ethical, religious and spiritual benefits, the so called socio-cultural values ([MEA, 2003](#)). However, although such cultural services play an essential part in the enhancement of human welfare, they are marginally present in the current research activities ([Benayas *et al.*, 2009](#)). This is considered as an increasing problem when the concept of ecosystem services is applied in cultural landscapes with typically long-lasting land use history, dynamic interactions of humans and nature, cultural patterns, and people's identities and values. Therefore, the ecosystem service approach should be expanded by the "cultural landscape paradigm", which includes humans as integral parts of landscapes, whereas other models in the present debate tend to see humans as impartial observers, as external drivers on ecosystems or as beneficiaries of environmental services ([Matthews and Selman, 2006](#)). Therefore, landscapes are seen as "social-ecological systems", in which social, economic and environmental components are closely interwoven ([Berkes *et al.*, 2003](#)).

While conceptual and methodological developments in monetary valuation have aimed at covering a wide range of values, including intangible ones, it can be stated that socio-cultural values cannot be fully evaluated by economic valuation techniques. A psycho-cultural perspective of valuation would strongly suggest a transdisciplinary dialogue ([Rist *et al.*, 2004](#)), aiming at cooperation between natural and social sciences research through debates on environmental ethics, tools and methods of social inquiry and socio-economic development as well as empowerment ([Kumar and Kumar, 2008](#)).

Since the last two decades many publications have dealt with different interpretations and implementations of the term "value" in the context of ecosystem services (e.g. [Costanza *et al.*, 1997](#); [Bishop, J.T., ed., 1999](#); [Odum and Odum, 2000](#); [Howarth and Farber, 2002](#); [Chee, 2004](#); [Farber *et al.*, 2006](#); [Kumar and Kumar, 2008](#)), which shows the big interest and importance of this topic. Following [Costanza \(2000\)](#) valuation is a basic need of human beings. Any choice and trade-offs between competing alternatives imply valuations, which are simply the relative weights given to the various aspects of decisions. Therefore, valuation ultimately depends on the specific goal or objective of an item ([Costanza, 2000](#)). For a long time the main focus has been on the utilitarian approach. However, individual utility maximization has become constrained when sustainability and social equity were also included as goals into the valuation concept ([Costanza and Folke, 1997](#)). According to the MEA and also to the TEEB approach the "total value" of an ecosystem and its services has to include three types of value domains, namely the ecological (environmental), economic and socio-cultural value ([Toman, 1998](#); [de Groot, 2006](#)). For example, hunting a game gives us food (health) and income but also cultural identity (as a hunter).

A special issue on valuation of ecosystem services, published in the journal *Ecological Economics*, discusses in detail the background, pro and contra of these three value approaches ([de Groot *et al.*, 2002](#); [Farber *et al.*, 2002](#); [Limburg *et al.*, 2002](#); [Wilson and Howarth, 2002](#)). One common problem in the valuation process is that information is often only available for some value domains and often in incompatible units.

5.2 Different valuation methods

Valuation can be conducted in many different ways ([Pagiola *et al.*, 2004](#)). The [MEA \(2005\)](#) and [TEEB \(2010\)](#) for instance focus on assessing the value of changes in ecosystem services resulting from management decisions or other human actions. This type of valuation is most likely to be directly policy relevant. The change in value can be assessed by either explicitly estimating the change value or by comparing the current value with the future value resulted by the alternative management regime. At landscape scale the (land use) change value approach proved also very

useful to present all the different stakeholder positions, and their linkages, in a rather objective and clear manner to support management discussions. Depending on the goal of the valuation and on data availability, monetary as well as non-monetary valuation approaches are applicable (Gómez-Baggethun *et al.*, 2010). In the further section we introduce some examples of valuation methods, which demonstrate important steps within the ecosystem service approach. As economic valuation has been implemented in many research studies and is also the main focus of the TEEB project, we provide also some important examples based on monetary valuation methods, although we do not place great emphasis on economic valuation within this review.

Economic valuation. Economic valuation has been often applied to assess the total value of services of a particular ecosystem or landscape at a given time (e.g. Adger *et al.*, 1995; Pimentel *et al.*, 1995; Costanza and Folke, 1997; Pimentel *et al.*, 1997; Hein *et al.*, 2006). This total economic value can be seen as an economic indicator, which provide as measure of gross national product or genuine savings policy-relevant information on the state of the economy (MEA, 2003). Costanza *et al.* (1997), for instance, whose publication presented an important milestone in the valuation process, attempted in their study to find the total economic value for a range of different ecosystem services at the global (biospheric) level. The current economic value of 17 ecosystem services for 16 biomes was estimated, based on published studies and a few original calculations. In general, they estimated unit area values for ecosystem services (in \$ ha⁻¹ yr⁻¹) and multiplied them by the total area of each biome. This approach has stimulated considerable debate and had not only to accept very sharp criticism from ecologists but also from economists (e.g. Opschoor, 1998; Turner *et al.*, 1998; Bockstael *et al.*, 2000; Xiaoli and Wie, 2009). Some of the core objections to Costanza's model can be summarized as follows (Xiaoli and Wie, 2009): the model did not adequately incorporate several factors which impact on ecosystem services, such as regional differences, spatial heterogeneity and social development. Neither can values estimated at one scale be expanded by a convenient physical index of area, such as hectares, to another scale, nor can two separate value estimates, derived under different contexts, simply be added together (Bockstael *et al.*, 2000). However it has to be stated, that the objective of this world wide study was not to present accurate values, but to show how valuable the natural world is (Pearce, 1998).

Since 1997, many studies were conducted to identify and quantify the value of ecosystem services. Whereas some of them based their results on Costanza *et al.* (1997) estimated values, others tried to modify Costanza's model by including new approaches (e.g. Sutton and Costanza, 2002; Williams *et al.*, 2003; Xiaoli and Wie, 2009).

To visualise that ecosystem services are spatially variable, and to identify key areas to be protected for the purpose of sustainable development, the "spatially explicit measure" represents a welcome method. It provides a mechanism for incorporating spatial context into ecosystem services evaluation (Chen *et al.*, 2009). Explicit value transfer becomes a useful method assessing ecosystems or landscapes, if valuation data is absent or limited (Bateman *et al.*, 2002; Troy and Wilson, 2006; Brenner *et al.*, 2010). Values and other data from the original study site are transferred to the designated policy site (Loomis, 1992). Troy and Wilson (2006), for example, presented in their paper a decision support system framework, which was built upon the value transfer methodology. In each case study a unique typology of land cover, to which ecosystem service estimates were available from the literature, was developed. Standardized ecosystem service value coefficients were broken down by land cover class and service type for each case study. Therefore, scenario and historic change analyses according to ecosystem services could have been conducted. However, this approach also suffers from limitations, such as availability of data, strength of the data and comparability between the source data and policy context (Troy and Wilson, 2006). Whereas some ecosystem services are easily transferable because they are provided at large scales (e.g. the avoided greenhouse gas costs of carbon sequestration), other local scale services may have limited transferability (e.g. flood control values) (Farber *et al.*, 2006).

Recognizing the limitations of value transfer, advanced research has focused more on spatially-explicit ecological and economic models, to explain the effect of human policies on ecosystem services and subsequently on human welfare (Naidoo and Ricketts, 2006; Barbier *et al.*, 2008; Polasky *et al.*, 2008; Nelson *et al.*, 2009). Such models show the spatial heterogeneity of service provision and supply a framework for regulatory analysis in the context of, for example risk assessment, non-point source pollution control, wetlands restoration and avalanche protection (Bockstael *et al.*, 1995). The application of integrated modelling supported by GIS to simulate environmental change scenarios, especially climate change, has become a useful tool to help decision-makers in selecting sustainable and economically feasible development strategies (see Bockstael *et al.*, 1995; Higgins *et al.*, 1997; Boumans *et al.*, 2002; Gret-Regamey *et al.*, 2008; Chen *et al.*, 2009). For example, in the Alpine region a study integrated into a single GIS platform several ecosystem process models simulating the provision of ecosystem services simultaneously with economic valuation procedures, in order to visualize climate change effects (Gret-Regamey *et al.*, 2008). However, modelling is costly of data and measurability requirements and therefore, studies often address relatively small spatial scales, at which it is achievable to develop ecological-economic models. In addition, most models usually focus only on a few ecosystem services and neglect the impact of biodiversity loss on combined ecosystem services. Only some authors tried to integrate the interactions between biodiversity and multiple ecosystem services in their studies (e.g. Metzger *et al.*, 2006; Egoh *et al.*, 2008; Nelson *et al.*, 2009).

The recent TEEB project, mainly based on economic valuation, concentrates on assessing the consequences of changes resulting from alternative management options, rather than for attempting to estimate the total value of ecosystems (TEEB, 2010). Within this project best practice examples from around the world are presented. However, the review of case studies undertaken by TEEB shows that, in many instances, more efficient but less precise methods have been used, hence the results must be interpreted with appropriate care. Especially, in more complex situations involving multiple ecosystems and services, and/or different ethical or cultural convictions, monetary valuations seems to be less reliable or unsuitable. Nevertheless, monetary assessments are important for internalizing so-called externalities in economic accounting procedures and in policies that affect ecosystems, especially where the alternative assumption is that nature has zero (or infinite) value (de Groot, 2006).

Non-economic valuation. Besides the economic valuation, other ways to analyse the importance of ecosystem services including environmental and socio cultural assessments are available. Assessing ecological quality the ecosystem service approach is seen as an applicable tool for supporting an environmental decision making process (Paetzold *et al.*, 2009). A specific Norwegian quality assessment, for example, evaluates current provision of services relative to their provision 100 years ago (Pereira *et al.*, 2005). Paetzold *et al.* (2009) propose to evaluate the status of an ecosystem in terms of its sustainable provision of ecosystem services in relation to the societal expectations. Thereby for each ecosystem service the quality is defined by the ratio of its sustainable provision to the expected level of service delivery. Thus, systems that provide services in a satisfactory and sustainable way can therefore be regarded as being of better quality than those that do not. One major challenge is to select, or develop appropriate indicators that, for example, assess the sustainability aspect of a service or societal expectations (McMichael *et al.*, 2005). In addition, it is difficult to obtain context-specific data on the provision and demand for many services (Chan *et al.*, 2006).

According to Martin and Blossey (2009) an ecosystem service cannot have a discrete value, because it depends on stakeholder preference and changes with quality and time frame. They suggest the following framework considering the quality of ecosystem services, the weighting, and the issue of time scale: $\mathbf{TV} = \int^t x_1 S_1 + y_2 S_2 \dots + z_n S_n$, where $\mathbf{TV}$ is the total value of a system; S_1 , S_2 , and S_n are service functions; 1, 2, and n include measures of quality; x , y , and z are

the respective weights of the service functions 1, 2, and n ; and t is the time frame considered. Habitat quality encompasses, for example, taxonomic diversity, suitability for rare species and historic composition of the site. The weighting of services depends mainly on the background and preferences of decision makers.

In the UK the merits of a “habitat, service and place based perspective” to the assessment of ecosystem services are emphasized (Haines-Young and Potschin, 2008). The habitat perspective is based on the use of a matrix of habitats and their related services. Pressures, respectively impacts on the services are additionally identified to assess state and trends of each service associated with England’s ecosystems. Since there is no commonly agreed terminology of pressures it is difficult to make such an assessment consistent. A clear advantage of using habitats as framework for representing the output of ecosystem services is that as distinct ecological units they could be seen in terms of “bundles” of services that they can deliver. It is generally known that most ecosystems are multifunctional, as structures and processes within them are capable of generating a wide range of different services (de Groot, 2006). The quality assessment of each habitat depends on the condition of their services and on the weighting of the service related indicators and their pressures. Although the habitat approach sounds very promising it also has its shortcomings, especially considering the multifunctionality of ecosystems. In most cases the links and interlinks between services might be overlooked. For policy relevance often costs-benefit analyses are conducted, because the exploitation of services usually has both costs and benefits for the society.

A wide range of studies illustrate that multifunctional landscapes are not only ecologically more sustainable and socio-culturally preferable but frequently also economically more beneficial than landscapes that only provide few ecosystem services (Balmford *et al.*, 2002; Turner *et al.*, 2003; Naidoo and Adamowicz, 2005). Therefore, Willemen *et al.* (2010) propose to assess landscape values by referring to the total potential provision of goods and services at multifunctional locations. For each landscape the capacities of all landscape functions are normalized and summed up (see Gómez-Sal *et al.*, 2003; Gimona and Van der Horst, 2007). Finally, a weighted value can be assigned to each landscape.

In the context of environmental assessment land use management decisions are often guided by some kind of transdisciplinary process, such as suggested by the concept ‘integrated planning assessment’ or more specifically the ‘quality of life capital’ approach (Potschin and Haines-Young, 2003; Haines-Young and Potschin, 2007). Thereby a “Leitbild” is used to describe what is viable in future, with regard to ecological sustainability and to the service preferences of society. Thus, the “Leitbild” concept can be applied as a reference system for service assessment in a given landscape.

To integrate in landscape planning not only environmental but also socio cultural values, great emphasis has to be placed on the expectations of inhabitants, tourists and the general public (Hunziker *et al.*, 2008). By integrating different social groups into the valuation process both conflicting and compatible views about landscape change may arise. However, these insights are important for steering landscape development in a stakeholder-related sense and for recognising and reducing conflicts of interest (see Backhaus *et al.*, 2007; Soliva *et al.*, 2008). The underlying idea is that an integrated and multi-dimensional approach will be more likely to capture the full range of values, including those which may be context specific (local, regional, national, and global). Schama (1995), for instance, show how landscape perception is over-formed by cultural and national identity.

In general, case studies of socio-cultural assessment methods are lacking (Benayas *et al.*, 2009). Christie *et al.* (2008) give an overview of non-economic techniques for assessing the importance of biodiversity to people in developing countries. Also, Pereira *et al.* (2005) provide some interesting non-monetary assessment methods.

6 Discussion

Although a lot of research effort regarding the investigation of ecosystem services has been done in the last years, it is still an innovative research field. Scientific models, frameworks and concepts for the evaluation of the benefits people derive from ecosystems have been provided. However, implementing the concept of ecosystem services into environmental planning and management at all levels of decision making still remains a big challenge and receives a lot of criticism.

6.1 Definitions and classifications – a challenge

In spite of the work done so far, there is still much discourse about definitions and classifications. According to Wallace (2008) a wide range of ways of evaluating trade-offs and synergies exist, but they need to be based on a coherent set of ecosystem services. However, maybe we should accept that no final classification can capture the myriad of ways in which ecosystems support human life and contribute to well-being. Since linked ecological-economic systems are complex and evolving, a ‘fit-for-purpose’ approach may be considered in creating clear classifications. Depending on the specific aim of applying a classification system the best suitable typology should be selected. Whereas some classification systems are more simple and thus well suited for educating a broad range of stakeholders (MEA, 2003), others are more complex focusing on the various spatial-temporal aspects of ecosystem services (Fisher *et al.*, 2009). While accepting that no fundamental categories or completely unambiguous definitions exist for such complex ecosystems, and any systematisation is open to debate, it is still important to follow some basic guidelines when developing a ‘fit-for-purpose’ approach: (1) defining the overall aim/purpose of the assessment as well as the area of interest (2) be aware of the target addresser (3) be clear about the meaning of the core terms used (4) think about which services and their related indicators are important for the final assessment (5) avoid double counting and (6) the final typology should be comprehensible and balanced between different function/service groups.

6.2 Quantifying and mapping – their limitations

Land management decisions usually relate to spatially oriented issues. To receive support for adequate choices, information on the spatial distributions of landscape functions and services is needed. A visualisation of landscape functions should also illustrate the spatial heterogeneity in quality and quantity of services provision, which is due to differences in biophysical and socio-economic conditions at different scale levels (Wiggering *et al.*, 2006; Meyer and Grabaum, 2008). However, although recently a large number of studies have been published dealing with various assessment methods of landscape functions and services (e.g. Kienast *et al.*, 2009; Brenner *et al.*, 2010; Haines-Young *et al.*, 2006; Willemen *et al.*, 2008) information on quantity and quality of spatially explicit services for policy relevant decisions is often lacking (Pinto-Correia *et al.*, 2006; Vejre *et al.*, 2007). The information that does exist remains fragmented, not comparable from one place to another, highly technical and unsuitable for policy makers, or simply unavailable (Schmeller, 2008; Scholes *et al.*, 2008).

Regarding the state-of-the-art, this paper shows, if the ecosystem service concept should be fully integrated into landscape planning issues, a better understanding of the interactions between land cover, use and function and methods to map and quantify land use and landscape function is needed (e.g. Verburg *et al.*, 2009). In some cases the state of ecological knowledge and the data availability allow using some direct measures of services, while in other cases it is necessary to make use of proxies. However, finding the appropriate proxy still remains a challenge (Egoh *et al.*, 2008; Willemen *et al.*, 2008). By searching for appropriate indicators and proxies several issues have to be faced, especially the relationship between services and scales. Synthesizing and

visualising different landscape function and services would require a spatial reference framework (Helming *et al.*, 2008), as different function groups usually occur at different scales (for instance, while provisioning functions are often restricted to a local scale, cultural or regulating functions usually operate at a broader scale). As the provision of landscape functions mainly depends both on the quantity and the spatial configuration of the landscape elements, special emphasis have to be put on defining thresholds indicating the change of function delivery when aggregating values within different spatial scales. For instance, a special function assessed on local level, cannot be assessed in the same way at landscape level. Therefore, to extrapolate function assessments to another level, special rules have to be defined. But, most of the existing assessments still tend to provide simply aggregated values for large regions, and thus data availability and disaggregation of spatial data are still one of the major limitations to the mapping of landscape functions and services.

6.3 Multifunctionality

As landscape functions do not equally interact with one another, multifunctional landscapes have different effects on service provision (Willemen *et al.*, 2010). Both negative and positive effects on ecosystem service provision can be observed. Whereas some functions seem to gain by the presence of other functions (e.g. plant habitat to tourism), others are affected negatively by multifunctionality (e.g. tourism to plant habitat). Although some research has already been done on the assessment of landscape function interactions, there are still remaining knowledge gaps (Willemen *et al.*, 2010). To analyse changes and trends in landscape function dynamics and to meet the challenges of trade-off analysis, more information on thresholds and optimum points is needed (Daugstad *et al.*, 2006; Groot *et al.*, 2007). In addition it would be very interesting to assess spatial and temporal scale effects on multifunctional areas (Hein *et al.*, 2006).

6.4 Different disciplines – advantages or hindrances?

Taking all these aspects into account the assessment of the full range of ecosystem values including the ecological, the economic and the socio-cultural, seems to be impossible. According to Norgaard (2010) scientists from different research fields are used to face complex issues within different models and most of which do not fit within a stock-flow meta-framework underlying the concept of ecosystem services. We should also be aware that different disciplines often try to attain different goals, including ecological sustainability, social fairness and the traditional economic goal of efficiency. Additionally, ecological and social phenomena happen on multiple scales and over different time periods that also match with the scalars of different social institutions (Wilson *et al.*, 1999; Folke *et al.*, 2005). However, that the three disciplines economy, ecology and applied social studies have to cooperate and produce new relations will also lead to positive effects. Economists are increasingly aware of the importance to integrate a social point of view into their ecosystem service valuation. Because the distribution of ecosystem services directly affects many people, carefully designed discursive methods, which involve small groups of citizens in the valuation process, will help ensuring the achievement of social fairness (see Farber *et al.*, 2002; Wilson and Howarth, 2002; Chee, 2004; Farber *et al.*, 2006). Whereas in former decades it has been focused on either ecological or economic modelling, in recent years an integrated approach is rising, which allows directly addressing the functional value of ecosystem services by observing long-term, spatial, and dynamic linkages between ecological and economic systems (Eichner and Tschirhart, 2007). By means of cooperation of these three different approaches every part can profit from the others, leading to an integrated ecosystem valuation concept. Economics, for example, could probably better understand the complexity of ecosystems and their effects on human well-being. While, on the other side the dynamics of markets and their information flows such as money and prices as well as the

trade-offs among ecosystem services are important to understand sustainable nature conservation. Thus, approaches to trade-off analysis can include multi-criteria (decision) analysis, cost-benefit analysis as well as cost effectiveness analysis. By including socio-cultural knowledge into the valuation concept, the analysis of human behaviour in its environment and human apperception of nature, will also help to enhance human welfare (Bockstael *et al.*, 1995).

Encompassing all three disciplines into the valuation process of ecosystem services could lead to the development of a well-balanced support tool for sustainable management decisions. However, the high number of unresolved issues in quantifying and mapping of ecosystem services as well as the valuation process itself remains still as major hindrances to the implementation of the ecosystem service concept in environmental policy and management.

6.5 Valuation and the future generation

Besides the scientific discourse about the different valuation methods, the question of valuation regarding to future generation will still remain as one of the main challenges. We have to be aware that evaluation of ecosystem services can only be made within the current specific political and economic context. As predictions about future developments are still limited, we just be able to suspect which resources are important for future generations (Norgaard, 2010). Current values of ecosystem services should thus be critically analysed in concordance with global, social and political change indices (Nowotny *et al.*, 2001). Whereas at the local and regional level the ecosystem service concept can act as a decision support tool for stakeholder to reach sustainable land use management, at global scale the valuation of ecosystem services can be seen as an alerting system and could encourage rethinking the global political systems to meet future challenges, like the climate and global change effects.

In conclusion, to meet all these challenges research effort needs to be conducted side by side to understand underlying relationships and to improve ecological as well as socio-economic understanding. Model-based research activities at local scale will take significant steps towards supplying policy makers with dependable and useable results.

As the ecosystem service research community is still very young compared to other research fields it could take some time to overcome the current barriers. However, on-going research studies, initiatives and projects, for instance the TEEB project, which are dealing with the ecosystem service approach raise hope, that the gaps get filled in the future and that the concept of ecosystem services can be integrated in environmental planning and management one time. The ecosystem service approach is an overarching concept that invites scientists from different disciplines to coordinate their research and guides their efforts towards outcomes more suitable for integration. To enhance the integration by coordinating collaborative efforts on ecosystem services at the global, national and local level a communication platform has been launched (<http://www.es-partnership.org>). Several international projects, for instance, the Global Earth Observation Biodiversity Observation Network GEOBON (Scholes *et al.*, 2008) or the World Resources Institute Mainstreaming Ecosystem Services Initiative (<http://www.wri.org/project/mainstreaming-ecosystem-services/tools>) are developing tools and approaches to model, map and value particular ecosystem services based on abiotic, biotic and anthropogenic indicators, as well as knowledge of relationships between these factors.

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CHAPTER 2

Landscape structure – service relations

Article 2

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Own contribution:

Study design 80%, implementation 60%, writing 80%

Correlations between wireworm damages in potato fields and landscape structure at different scales

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Abstract Wireworms, the larvae of click beetles are world-wide pests of arable and vegetable crops. Many studies have shown that arthropods are influenced by landscape context. Therefore, for successful wireworm risk assessment, it is important to gain more information on relationships between landscape structure and wireworm infestation level. The aim of this study was to investigate if different landscape variables are related to wireworm infestation, represented by the proportion of wireworm damaged potato tubers. Based on aerial photographs and field surveys landscape variables, such as landscape composition, landscape complexity and ecological variables were characterised around 14 randomly selected potato field plots alongside the gradient from the lower warm-dry “Weinviertel” (region 1) to the cool-humid “Waldviertel” (region 2) in the North-East of Austria at three scales (1, 9 and 25 ha). Relationships between landscape variables and wireworm damage ratings were described by Spearman correlations coefficients. Our results showed that wireworm damage was higher in region 2 than in region 1 and tended to be higher in soils with higher sand content.

Among the tested landscape variables only landscape composition, especially grassy field margins were significantly positively correlated with wireworm damage at the 25 ha scale. In areas with high proportion of such grassy habitats, farmers should pay more attention to field characteristics, such as sand content in soil, and management, like crop rotation. Further investigations on appropriate management regimes for grassy landscape elements suppressing wireworm populations but ensuring the maintenance of natural pest control should be conducted.

Keywords Wireworm damage · *Agriotes* sp. · Landscape variables · Landscape structure · Pest control · Spatial scales

Introduction

Wireworms, the soil living larvae of click beetle species (Coleoptera: Elateridae) are known as important pests of arable crops, especially potatoes and maize, in many parts of the world (Parker and Howard 2001). Although wireworm attack does not affect crop yield directly, it reduces crop quality considerably (Jansson and Seal 1994). As, in recent years, wireworm induced damage has become an increasing problem for farmers, the demand for a meaningful risk assessment and useful methods to restrict damage is rising. However, due to the cryptic habitat of the wireworms, pest control is very difficult and leads to unsatisfying results (Blackshaw and Vernon 2006). In Europe alone, up to 40 click beetle species occur, differing in their feeding behaviour, preferred environmental conditions and pest status (Cate 2007; Traugott et al. 2008). Generally, the duration of grassy vegetation at a site, soil type, texture, moisture and climate are known as important environmental factors

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influencing wireworm abundances in crop fields (Benefer et al. 2012; Gongalsky et al. 2009; Staley et al. 2007; Miles 1942). However, only few studies focussed on the species specific distribution in agricultural land (e.g. Blackshaw and Vernon 2006; Staudacher et al. 2011) which impedes the development of efficient pest control management methods (Parker and Howard 2001). Current strategies like using insecticides are restricted to conventional farming systems and often fail to achieve reliable control of wireworms in potato crop production (Parker and Howard 2001). Alternative approaches like trap cropping (Landl and Glauning 2011) or finding virulent strains of entomopathogenic nematods and fungi against wireworms (Ansari et al. 2009) are promising, but still in development.

In recent years, there is increasing recognition that based on their pattern and processes landscapes provide a wide range of beneficial functions (Hermann et al. 2011). Landscape structure (pattern and process) is dealing with the spatial relations between individual, distinct local ecosystems—the so-called landscape elements, as well as the distribution of energy, resources and species in relation to size, shape, number and type of such elements (Forman and Godron 1986; Turner et al. 2001). It is reflecting not only the natural settings of the landscape but also its history and the impact of mankind throughout the centuries (Antrop 2005; Ernoult et al. 2006). Accordingly, the present spatial patterns are unique results of former activities and processes in the landscape. An important economic landscape function is natural pest control (Bianchi et al. 2006; Wilby and Thomas 2002). Several studies showed that pest pressure was reduced within complex landscapes (Basedow 1990; Östman et al. 2001; Thies & Tschardtke 1999; Thies et al. 2003). Therefore, a diversified agricultural landscape mosaic may have enhanced environmental and economic benefits as it reduces yield loss without the negative consequences of pesticide use (Naylor and Ehrlich 1997; Östman et al. 2003). Landscapes composed of arable fields intermingled with non-crop habitats, like woodlots and field margins, support higher biodiversity than simplified landscapes mainly consisting of arable fields (Wrbka et al. 2008; Bianchi et al. 2006). Accordingly, such semi-natural landscapes are more likely to provide habitat for predatory birds, insects and nematodes than intensively used agricultural land (Fox and MacLellan 1956; Schalk et al. 1993; Gratwick 1989). However, benefits of natural pest control by complex landscapes are not self-evident, but depend on many factors and can easily be disrupted (Hunter 2002). Some predators, for instance are more susceptible to habitat fragmentation than are their prey, resulting in a higher density of pests (Abensperg-Traun and Smith 1999). The degree to which habitats function as source or sink for pests is determined by their quality and size in relation to the whole landscape context (Dunning et al. 1992). As a consequence, non-crop

habitats do not only enhance natural enemies, but also may act as reservoirs and sources for certain pest species (van Emden 1965), depending on their habitat requirements and migration abilities between crop fields and other landscape features. Regarding wireworm damage risk, complex landscapes might decrease wireworm pressure by enhancing predator density, while a high proportion of certain landscape elements might increase wireworm pressure, e.g. by providing adequate breeding habitats.

Many studies focus on the relationships between natural enemies of pests, for example carabid beetles and spiders, and environmental factors like land use intensity or local habitat properties (moisture or soil texture, e.g. Schweiger et al. 2005; Kromp 1999). However, investigations on the relation between landscape structure and the distribution of click beetle species are limited (see e.g. Benefer et al. 2010; Blackshaw and Vernon 2006). Therefore, the aim of our study was to gather more information on the relationship between landscape structure and wireworm pressure by correlating selected landscape variables with wireworm damage levels. The investigations were carried out within organically managed potato fields, because potatoes are especially susceptible to wireworm attack. We used a multiscale approach to determine the scales at which wireworm damage is associated with particular landscape variables. Our research questions were: (i) is the wireworm damage level related to the surrounding landscape structure taking into account climatic, soil and management parameters? and (ii) do the correlations differ between scales (1 ha/9 ha and 25 ha)?

Materials and methods

Study area and sample sites

The study area was situated in the North-East of Austria, Central Europe (Fig. 1). In order to investigate landscape-wireworm correlations across different environments, we determined two sampling regions with distinct environmental parameters (Table 1). Whereas, region 1 was situated in the lower warm-dry “Weinviertel”, region 2 was located in the higher, cool-humid “Waldviertel” in Lower Austria. The geomorphological features are characterised by lowland in region 1 (~150 m a.s.l.) and soft hills with an altitudinal range between 300 and 500 m a.s.l. in region 2. While the soils in region 1 were determined as limy fluvisols and tschernosems, region 2 was characterized by lime-free fluvisols, brown soils and pseudogleys (eBOD 2011). Within each region, sample sites were selected randomly out of a database on wireworm damage rates in potato fields which was compiled by Bio Forschung Austria (Kromp et al. 2009; six sites in region 1 and eight sites in region 2 = 14 sites total). The selected sample sites

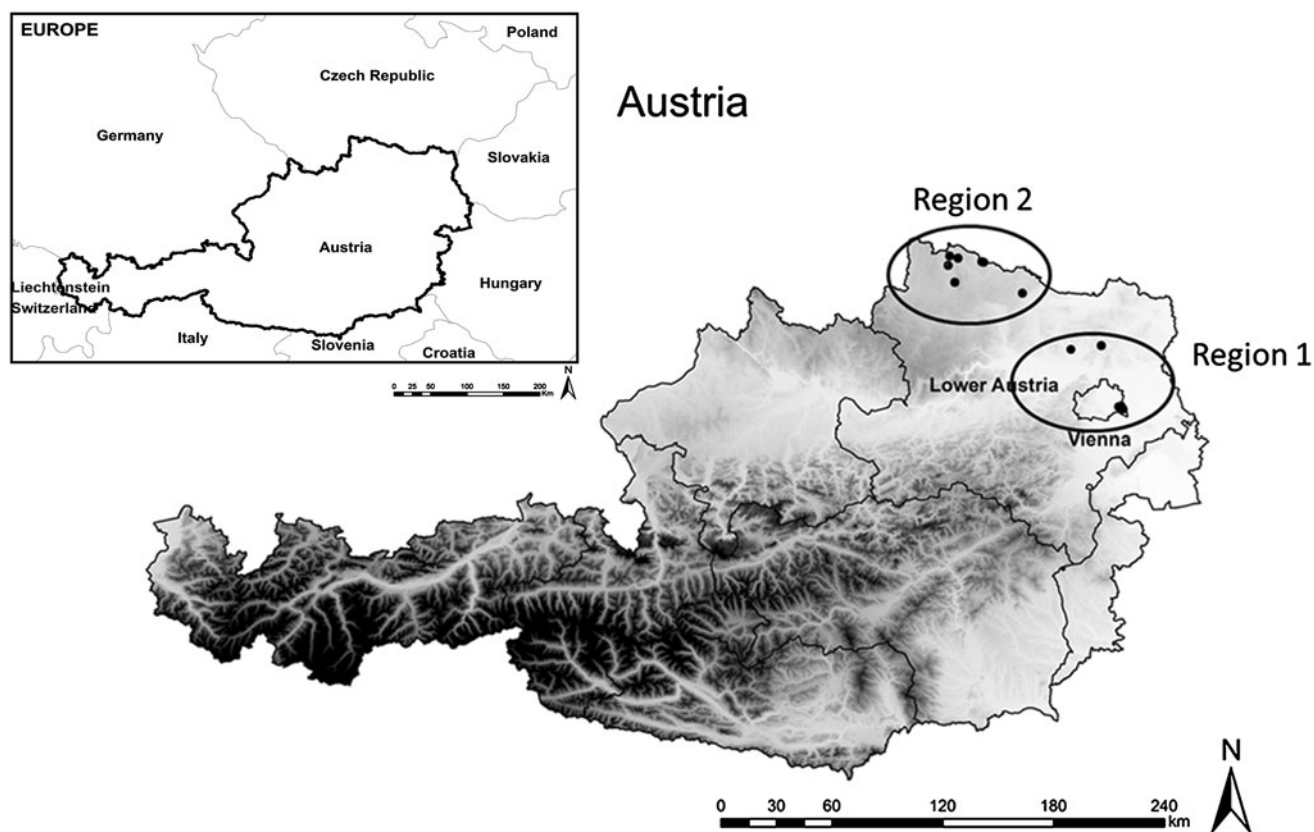


Fig. 1 Study area in the North-East of Austria; six sites in region 1, eight in region 2

comprised rather monotonous as well as more complex landscapes, but all being dominated by arable land.

Data on field sampling and wireworm damages

All sampled fields had reported histories of wireworm incidences, ranging from low to severe damages (Kromp et al. 2009). The sampling plots within the fields (a square of 450 m²) were located in areas, where the farmers did expect wireworm damage, according to observations in previous years. In each sampling plot, 20 potato plants were harvested randomly. The tubers were counted, washed and thoroughly inspected for wireworm feeding holes. Each potato tuber with one or more wireworm feeding holes was considered damaged. Based on these data, we calculated the proportion of damaged potatoes out of the 20 sampled potato plants. The damage samplings were conducted in 2007 and 2008 (Table 1). In region 1 (weather station coordinates: 48°11'58"N, 16°33'43"E) mean temperatures from April to September were 17.9 °C in 2007 and 17.1 °C in 2008; in region 2 (48°41'28"N, 15°21'58"E) mean temperatures were 4.8 °C in 2007 and 13.9 °C in 2008 (ZAMG 2011). Precipitation sum (April–September) in region 1 was 464 mm in 2007 and 447 mm in 2008; in region 2, the precipitation sum was 472 mm in 2007 and 502 mm in 2008. In both the

regions, we sampled several potato cultivars at differing harvesting dates within 2007 and 2008.

At 10 out of the 14 sampling sites, wireworm densities were measured by 6 baiting traps per sampling plot over a period of 2 weeks prior to the potato harvest (Kromp et al. 2009). The baiting traps were dug directly beneath the soil surface and attracted the wireworms through CO₂ emission from germinating wheat and maize grain. To investigate the relationship between wireworm density and damage, we calculated and tested the Spearman correlation coefficient between the number of baited wireworms and the percentage of wireworm damaged potatoes.

Site-specific parameters and landscape variables

The climatic characteristics of the different sampling sites (annual temperature sum, annual precipitation sum) were derived from the climate maps published by Harlfinger & Knees (1999). The maps are based on climatic data, measured from 1990 to 2000. The soil characteristics (soil type, estimated soil pH in A-horizon, sand = estimated percentage of sand content in A-horizon, estimated humus content level in A-horizon) were derived from the digital Austrian soil map (eBOD 2012). Management data (number of years with grassy crops in 5 years before potato

Table 1 Site characteristics and measured wireworm damages at the 14 sampling plots

Plot	Region 1					Region 2											Mean
	1	2	3	4	5	6	Mean	7	8	9	10	11	12	13	14		
Altitude (m) above sea level**	213	154	153	154	156	260	182	503	480	485	523	531	493	499	413	491	
Annual temperature sum (°C)**	3,499	3,643	3,655	3,635	3,641	3,409	3,580	2,597	2,587	2,549	2,566	2,511	2,695	2,698	3,137	2,668	
Annual precipitation sum (mm)**	529	532	534	533	533	533	532	593	599	604	583	615	582	584	494	582	
Years with grassy crops in 5 years crop rotation	2	0	0	0	0	0	0.3	2	2	1	2	2	0	0	1	1.3	
pH in A-horizon**	7.28	7.65	7.65	7.65	7.65	7.28	7.53	6.05	6.05	6.05	4.28	6.05	5.55	6.05	6.48	5.82	
Sand content in A-horizon (%) ^{ns}	15	41.25	41.25	41.25	41.25	30	35.0	55	55	55	15	55	15	15	35	37.50	
Humus content in A-horizon (%)	1.5–4.0	1.5–4.0	1.5–4.0	<1.5	1.5–4.0	1.5–4.0	1.5–4.0	1.5–4.0	1.5–4.0	1.5–4.0	>4.0	1.5–4.0	1.5–4.0	1.5–4.0	1.5–4.0	1.5–4.0	
Potato cultivar	Ditta	Agata	Agata	Agata	Agata	Ditta		Agria	Nicola	Ditta	Ditta	Ditta	Ditta	Ponto	Ditta		
Sampling dd/mm/year	04/09/07	15/07/08	31/07/08	15/07/07	31/07/07	18/09/07		27/08/08	27/08/08	27/08/08	16/08/08	27/08/07	28/08/08	27/08/08	26/08/08		
Proportion of wireworm damaged potatoes (%)**	17.40	1.70	8.10	15.70	2.90	14.30	10.02	79.90	96.60	55.60	25.50	87.10	34.50	13.00	7.00	49.90	

Region 1, “Weinviertel”; region 2, “Waldviertel”; grassy crops: clover-grass, clover, alfalfa, grassland; organic fertilisation: liquid and solid manure, compost, slurry; differences between the regions were tested by *t* test: ** $p \leq 0.01$, *ns* not significant

growing and potato cultivar) were inquired of the local farmers in course of the wireworm damage samplings. The potato cultivar was not included into further analyses, since three cultivars occurred only once.

For the acquisition of multiscale landscape data, the landscape sites were considered as squared windows of three different sizes centred on the wireworm sampling plots: 100 m × 100 m (1 ha, local scale), 300 m × 300 m (9 ha, medium landscape scale) and 500 m × 500 m (25 ha, large landscape scale). This method allowed investigating the scales at which wireworms are associated with particular landscape characteristics. When it comes to landscape ecology analyses different geometry of sample sites can be applied (Schindler 2010). However, to be able to cover the whole continuum of a landscape it has to be viewed as an aggregation of raster cells. Therefore, to be able to integrate the results into further landscape studies focusing on a broad landscape context, we needed to use square cells. The landscape analyses follow the “landscape pattern and process” paradigm (Turner et al. 2001). Thereby the basic unit of any landscape is the so-called landscape element. It is defined as being the smallest spatial unit, which can be identified and delineated as ecologically homogeneous in regard to crucial land attributes like soil type, microclimatic and hydrological conditions, actual vegetation, etc. (Forman and Godron 1986). Technically, a landscape element is the smallest mappable unit. In this study, it represents habitats with a specific land use and vegetation type.

At each scale, three main sets of landscape variables, namely landscape composition, landscape complexity and ecological variables of landscapes were determined, in order to measure the structure of the landscapes (Table 2). Landscape composition represents the number and areal proportion of the different land use types within a landscape. It provides a good overview of the features that characterise a landscape and influence its ecological functions. The more land use types are situated within a landscape, the more diversity it shows. In order to investigate the relevance of sustainable features for wireworm risk management, we focused on semi-natural elements. We distinguished between wood, total grassland, farmed grassland (meadows and animal pastures), unfarmed grassland (grassy field margins and fallow land) and grassy field margins alone within our analysis. The landscape complexity is described by the number of landscape elements indicating information on the landscape’s grain size (=mean size of all landscape elements) as well as by the landscape element edge density, which is an expression of the spatial heterogeneity of a landscape mosaic. High edge density indicates higher landscape pattern complexity (Schindler et al. 2008) and higher density on ecotones. The ecological variables are described by the concept of hemeroby (Naveh and Liebermann 1994) that is an indicator for ecological sustainability and by the

Table 2 Landscape structure: the three sets of landscape variables, estimated for the scales 1, 9 and 25 ha

Main set	Variables	Description
Landscape composition	proportion of the land use types per landscape site: wood, total grassland, farmed grassland (meadows and pastures), unfarmed grassland (grassy field margins and fallows), grassy field margins alone	Percentage of area occupied by certain land use type
	Number of land use types	Indicator for land use type diversity
Landscape complexity	Edge density	Total length of edge per unit area; high density = complex landscape, high number of ecotones
	Number of landscape elements	Indicator for landscape's grain size
Ecological variables	Hemeroby index per landscape site	Area weighted mean of hemeroby values of all landscape elements; index ranges between 1 (highest human impact) and 7 (no human impact)
	Number of vegetation types	Indicator for vegetation diversity

number of vegetation types. Hemeroby is defined as the magnitude of the deviation from the potential natural vegetation caused by human activities (Naveh and Liebermann 1994). An overview of the categories of the hemerobiotic state and their attributes is given in Naveh and Liebermann (1994) and Zechmeister and Moser (2001). The lower the hemeroby index, the higher the deviation (human impact). The number of vegetation types, including the crop types in the fields, is based on the dominating plant species composition (Mucina et al. 1993) and is used as indicator for vegetation heterogeneity.

For measuring the landscape variables, we used digitized landscape element maps (Forman and Godron 1986), derived from orthorectified aerial photographs with a spatial resolution of 0.25×0.25 m. The variables were computed by ArcGIS software (ESRI 2010) with V-Late 1.1, Vector-based Landscape Analysis Tools Extension (Lang and Tiede 2003). This provided a basis for the compilation of a geo-database which was updated and expanded by a recent field survey that was carried out between June and August 2009. During the field survey, to each landscape element a land use type was assigned (Peterseil et al. 2004) and the vegetation type was identified (Mucina et al. 1993). In addition, all landscape elements were assessed regarding the concept of hemeroby. The degree of deviation from the potential natural vegetation was determined by standardised criteria and expert knowledge (see Zechmeister et al. 2003). The final hemeroby index for one landscape site was calculated by the area weighted mean of hemeroby values of all landscape elements within a landscape site.

Statistics

The data were statistically analysed by SPSS 11.5.1 (SPSS Inc., Chicago, IL, USA). Differences between the regions

regarding climatic, soil and management characteristics of the sites were confirmed by *t* tests. Relationships between landscape variables and wireworm damage ratings were described by Spearman correlations coefficients. To avoid “statistical fishing” for significant correlations by multiple correlation tests, the original *p* values were multiplied with the number of landscape variables per scale (10), according to Bonferroni correction. Only correlation coefficients being significant on basis of the corrected *p* values were regarded as relevant (significant: $p \leq 0.05$). In addition to calculating correlation coefficients across both sampling regions, correlation coefficients were also calculated with the *z*-standardised region residues. Thus, the persistence of relevant wireworm-landscape relationships after eliminating general regional differences between the sites could be tested.

Interrelationships between landscape variables which were significantly correlated to wireworm damage, and the climatic, soil and management characteristics of the sampling sites were analysed by a principle component analysis (PCA). Wireworm damage was added as a passive variable to the biplot using its correlation coefficient to the first axis. Correlations between wireworm damage and environmental variables were confirmed by testing the Pearson correlation coefficients.

The explanatory power of the relevant landscape variables combined with field-related environmental variables (pH level, sand content in soil; occurrence of grassy crops in crop rotation), for the distribution of wireworm damages was tested by multivariate regression analyses. The results were shown as scatter plots.

Normal distribution for PCA and regression analysis was tested by the Kolmogorov–Smirnov test, homogeneity of variances by the Levene test. If necessary, data were transformed by natural logarithm or, in case of percentages, by arcsine transformation (Köhler et al. 2002).

Results

General

The number of baited wireworms was positively correlated with the proportion of wireworm damaged potatoes in the same field ($r_s = 0.750$, $p = 0.012$; $n = 10$). The wireworm damage rates were significantly different between region 1 and region 2 (Table 1). The plots with the lowest damage rates were located in the lower warm-dry “Weinviertel”, whereas the plots with the highest damages were located in the higher, cool-humid “Waldviertel” in Lower Austria. During the field survey we mapped 654 landscape elements in total. The far most frequent semi-natural land use type was field margin, followed by meadow (for all measured landscape variables per plot see Appendix A: Table A1–A3 in Electronic Supplementary Material). In total, we found 64 different vegetation types. The hemeroby values ranged between 2.05 (strong anthropogenic influence leading to a destruction of the biotope) and 4.58 (anthropogenic influence is lower, adaptations in part of the natural beings are possible) among the 14 landscape sites.

Correlations between landscape variables and wireworm damage

According to the Bonferroni-corrected p values of the correlation coefficients the landscape variables were significantly related with the proportion of wireworm damaged potatoes only at the 25-ha scale (Table 3). Only the presence of grassland (farmed grassland and grassy field margins) within 25 ha was positively correlated with the frequency of occurrence of wireworm damage. Contrary to the grassy margins alone, the unfarmed grassland (grassy field margins + grassy fallows) did not show any correlation with wireworm damage. These trends observed at the 25 ha were not significant at the 9 ha scale and no correlations between landscape variables and wireworm damage were present at the 1 ha scale.

Explaining wireworm damage distribution by landscape and field-related variables

The first two axes of the PCA accounted for 82, 84 % of total variance (Fig. 2). PC 1 was strongly correlated to percentages of the different grassland types within 25 ha, precipitation and—to a lesser extent—the frequency of grassy crops before potato growing. Mean annual temperature sum and pH value were also highly correlated to this axis, but in the opposite direction. The sampling sites were separated in two groups along this axis corresponding to the two sampling regions.

Table 3 Spearman correlation coefficients between the percentage of wireworm damaged potatoes and the measured landscape variables per scale (1, 9, 25 ha)

Wireworm damaged potatoes	Correlation coefficients (Spearman)		
	1 ha	9 ha	25 ha
grassland total	0.543	0.644	0.697
grassland farmed	0.347	0.664	0.721*
grassy field margins	0.486	0.688	0.767*
grassland unfarmed (margins + fallows)	0.495	0.345	0.459
wood	−0.632	−0.393	−0.284
number of landscape types	0.170	0.029	0.374
number of landscape elements	0.380	0.489	0.572
number of vegetation types	0.434	0.473	0.590
hemeroby	−0.172	−0.257	−0.130
edge density	0.178	0.600	0.521

The p values of the correlation coefficients were adjusted to the ten correlations per scale by Bonferroni correction; significant coefficients are boldly printed: $p * < 0.01$ $p \leq 0.05$

PC2 was strongly correlated with the sand content in the soil. This was the only parameter tested, that was not associated with differences between region 1 and 2 (PC1). Both axes (PC1 and 2) were correlated with the wireworm damage. Wireworm damage was higher in region 2 than in region 1, being associated with the general region differences, and tended to be higher in soils with higher than with lower sand content. Apart from the landscape variables, temperature sum, precipitation and frequency of grassy crops were significantly correlated with wireworm damage.

In Fig. 3a, b, the relationships between wireworm damage and the relevant landscape variables are shown as scatter plots. The percentage of grassy field margins in 25 ha showed a consistent positive relationship with wireworm damage across the two regions as well as within the two regions (Fig. 3a, black and white symbols). It was the only landscape variable that was significantly correlated to wireworm damage when calculated with region residues ($r = 0.666$, $p = 0.009$). A regression analysis showed that the percentage of grassy field margins alone could explain 50 % of the wireworm damage distribution (corrected $r^2 = 0.504$). A combination of grassy margins and the sand content in the soil increased the percentage of explained variance up to 72 % (p value for sand content when included into the regression model = 0.009; p value for the percentage of grassy field margins = 0.001). According to the regression model, fields that are surrounded by grassy field margins (>3 % coverage in 25 ha) and have a sand content in the soil of more than 35 % had the highest probability for serious wireworm damages in

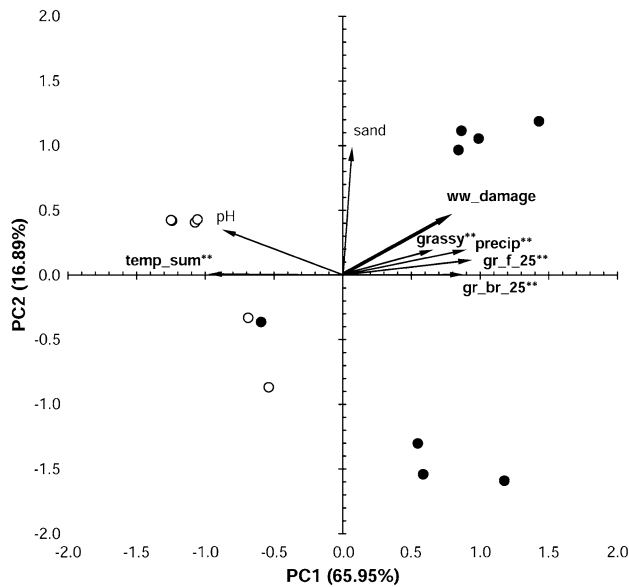


Fig. 2 Principal component analysis ordination of selected landscape, climatic, soil and management variables. PC = principal component + explained part of total variance in parentheses. The variables are shown as vectors, the sites are shown as circles (white sites in region 1, black region 2). Wireworm damage (ww_damage percentage of wireworm damaged potatoes) was added as passive variable in the plot by its correlation coefficients with the PCs (bold arrow). The landscape variables are gr_br_25 = percentage of grassy field margins in 25 ha surrounding the wireworm damage sampling plot, gr_f_25 percentage of farmed grassland; climatic variables: precip mean annual precipitation (mm), temp_sum mean annual temperature sum (°C); soil variables: pH estimated soil pH level, sand estimated sand content of the soil (%); management: grassy number of years with grassy crops (clover-grass, clover, alfalfa, grassland) in 5 years before potato growing. Environmental variables significantly correlated with wireworm damage are boldly printed and marked by asterisks: ** $p \leq 0.01$; in this figure the p values are not Bonferroni-corrected

potato crops. No other field-related variable (pH level, frequency of grassy crops) could significantly contribute explanatory power to the regression model.

The other relevant grassland percentage (farmed) showed less consistent relationships to wireworm damage than the grassy field margins (Fig. 3b). While in region 1 only very small percentages of farmed grassland could be measured, region 2 showed a remarkable higher level. Positive trends between wireworm damage and the respective landscape variable were evident only within region 2. Accordingly, the correlation was largely associated with general environmental differences between the two sampling regions and diminished when calculated with region residues. Since the relationships between this landscape variable and wireworm damage could not be confirmed within both sampling regions, multivariate regression analyses with field-related variables were discarded.

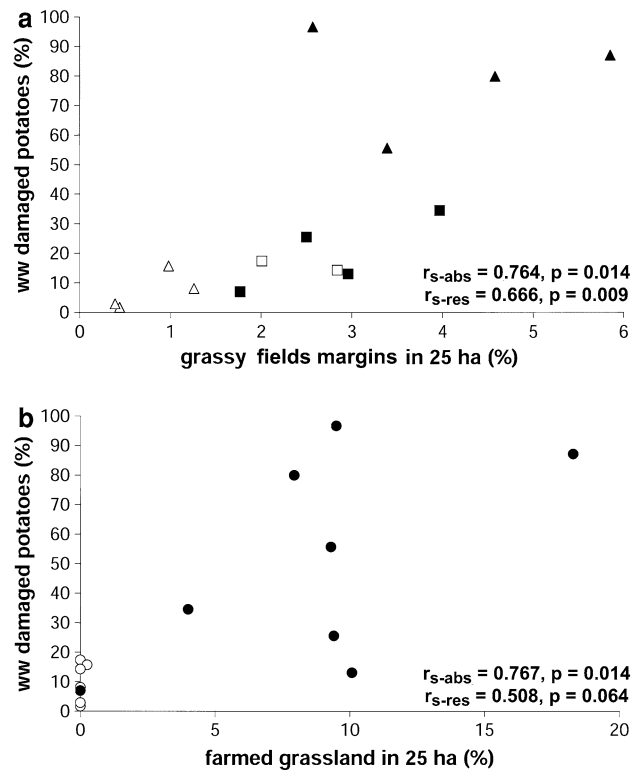


Fig. 3 Scatter plots of wireworm damage in potato crops versus significantly correlated landscape, soil and management variables. **a** Percentage of wireworm damaged potatoes (y axis) versus percentage of grassy field margins in 25 ha surrounding the damage sampling plot (x axis) and estimated percentage of sand in the soil (square: sand ≤ 35 %, triangle: sand > 35 %); **b** ww damage versus percentage of farmed grassland. The colour of the symbols indicates the region, which the sampling site was situated in: white region 1, black region 2; r_s Spearman correlation coefficients calculated across both regions with absolute values ($=r_{s-abs}$) and with region residues (r_{s-res})

Discussion

Our study demonstrated that the distribution of wireworm damages can be associated to landscape structure at a scale of 25 ha. Particularly landscape composition, namely the percentage of grassy habitats (farmed grassland and grassy field margins) in the vicinity of the sampling plots, was positively correlated with wireworm damage. The percentage of grassy field margins in 25 ha showed the most consistent relationship to wireworm damage distribution across the two sampling regions as well as within each region. They occur in most cropland regions in remarkable amounts but are also correlated to field size and structure, and hence to the amount of total grassland within a landscape. Consequently, the percentage of grassy field margins might be a key landscape variable, having high explanatory power for the distribution of wireworm damage risk.

Contrary to the grassy field margins alone, the combination of grassy field margins with fallows (=unfarmed

grassland) did not show any association with wireworm damage, which was incongruent to observations in other studies. A lot of investigations revealed that setting aside arable land increased occurrence of wireworms (Hancock et al. 1992; Lefko 1998; Parker and Howard 2000). However, according to a study in Czech Republic (Jedlicka and Frouz 2007) in older, well-established fallows, the abundance of wireworm decreased. On the one hand, the plant diversity in fallows might foster predators and on the other hand, elaterid egg deposition into the soil may be hampered by dense litter layers at the soil surface. In our study, young and old fallows were not differentiated which could be the reason why there were no significant relationship between unfarmed grassland (grassy field margins + fallows) and wireworm damage.

Together with the percentage of grassy habitats, wireworm damage was positively correlated to precipitation, and negatively to mean annual temperature sum and soil pH level alongside the gradient from the lower warm-dry “Weinviertel” (region 1) to the cool-humid “Waldviertel” (region 2). When combining the sand content in the soil, which is the only field-related variable being independent from regional differences, together with the percentage of grassy field margins in 25 ha, 72 % of the wireworm damage distribution could be explained (Fig 3a). This can be based on the fact, that soils with a higher sand content in the A-horizon are more prone to sudden drought. Hence, in sandy soils wireworms might more often seek to escape desiccation and burrow into the water-rich potato tubers. Generally, the movement of the larvae through the soil should be smoother in sandy soils. In a study of Parker and Seeney (1997), wireworm infested grassland sites were characterised by lower soil bulk densities, usually associated to higher sand contents in the soil. As stated above, landscapes with a high percentage of grassy habitats might support a high wireworm pressure on the fields. Therefore, if, in such regions, potato fields have sandy soils, the risk for serious wireworm damages might be even higher than for fields with lesser sand contents.

The close association of wireworm damages in potato fields to nearby grassy habitats corresponded well to observations in other studies, corroborating the hypothesis that grassland habitats are important to certain click beetle species (Parker and Seeney 1997; Blackshaw and Vernon 2006). Consequently, regions rich in grassland habitats might have higher wireworm potentials than monotonous regions or regions dominated by other features. They are areas of temporal stability in which soil-dwelling larvae can complete development without increased mortality caused by cultivation (Gough and Evans 1942; Blackshaw 1988; Parker and Seeney 1997; Blackshaw and Vernon 2006). Such spatial structures may also serve as corridors or stepping stones for adult species between the larger

farmed grassland areas and arable land (Blackshaw and Vernon 2006). Kabanov (1975) for instance, observed *A. lineatus* adults sitting on grasses and weeds along paths and riversides.

It is well documented in the literature that wireworms are influenced by soil type, temperature, texture and moisture in all developmental stages (Fox 1961; Cherry and Stansly 2008). At high temperature and low soil moisture, the larvae move downward into cooler and more humid soil regions (Jones and Shirck 1942; Falconer 1945; Lafrance 1968), where they cannot damage the potato tubers. Furthermore, Staudacher et al. (2011) showed differences between the wireworm species communities of the higher, cooler altitudes and the lower, warmer altitudes in Austria. The different species are considered to have different feeding preferences, which might also be a reason for the different damage levels in our two sampling regions. As shown in the PCA-ordination biplot (Fig. 2), there were various interrelations between landscape, climate, soil and management characteristics, corresponding to general differences between the sampling regions. Consequently, a distinct separation between the effects of landscape properties and the impact of other environmental factors remains impossible.

The ecological variables were not correlated to wireworm damage at any scale. Probably the method of hemeroby measurement was too rough to detect a relationship. In agricultural landscapes, the arable fields characterised by high human impact are very dominant and might have distorted the values. Our theory that higher phyto-diversity would favour wireworm predators could not be confirmed. Phyto-diversity was measured by counting the number of 280 vegetation types within an investigation area also including the crop types, which might have distorted the values.

Although it is well established that complex boundaries (edges) among habitat types can increase insect diversity (Hunter 2002), our results did not show any significant relationship between “edge density” and wireworm damage. The variables of landscape complexity include boundaries of various semi-natural features, such as grassland, hedgerows, old groves and small woodlots. Whereas grassy elements appear to be important for click beetle species (Parker and Howard 2001), hedges, for instance are about equally important for some predators (Kromp and Hartl 1991; Andersen 1997) and might decrease movement rates of beetles (Duelli et al. 1999). Opposing effects of both feature groups might have led to insufficient results.

Regarding scale size, significant correlations were evident not until the 25 ha scale. Generally, the 1 and 9 ha scale seemed to be too small to explain the wireworm potential of a landscape. One reason explaining why relevant landscape–damage correlations were restricted to the

largest scale might be a kind of measurement error. The time taken to complete larval development may vary considerably between the elaterid genera and depends on various environmental factors. But wireworms usually spend between 2 and 5 years in the soil before pupating (Miles 1942). Consequently, the female click beetles laid their eggs some years before the damage samplings in 2007 and 2008. In the period from oviposition to damage sampling, some landscape elements might have changed. At the small scale of 1 ha, the deviation might have been crucial, interfering significant correlations, whereas at the larger scales of 9 and 25 ha this error in measurement should have diminished continuously. When updating the aerial photographs, which were taken about 5–7 years before the damage samplings, in the field survey in 2009, no serious changes in landscape structure revealed in any scale.

Anyhow, the positive correlations between the percentages of grassy habitats in 25 ha (500 × 500 m) surrounding the sampling plots and the wireworm damage within the sampling plots indicated that colonisation by adult beetles from quite distant breeding and food source habitats is an important factor. Although click beetles are often considered to be lazy flyers and disperse over short distances only (e.g. Subklew 1935; Roebuck et al. 1947), Schallhart et al. (2009) showed that some species can migrate at least over a distance of 80 m. Crozier et al. (2003) explored *A. obscurus* and *A. lineatus* males and females flying on several occasions and both species seemed to be relatively strong flyers, travelling at a speed of ~5–10 km/h, while flying ranged from less than 1–100 m. According to the latter author, mass flights also occur. Boiteau et al. (2000) confirm the widespread use of flight by click beetles in arable land, with a peak of flight activity in the egg laying period. Elaterids were caught in heights of up to 14.3 m. Also Blackshaw and Hicks (2012) conclude that *Agriotes* spp. are able to disperse widely.

Potato variety is known to affect wireworm damage (see Johnson et al. 2008). As shown in Table 1, the sampling regions differ regarding the planted potato cultivars. But because of unbalanced sampling, the effects of region and potato cultivar cannot be separated. Accordingly, the prevalence of different potato cultivars may explain the regional differences in wireworm damage level. Nevertheless, the fact that the relationship between grassy field margins and wireworm damage was independent from region supports the importance of grassy habitats for wireworm damage distribution.

Another positive finding from our research was that the baited wireworms were positively related to wireworm damage rates. Since bait traps reflect activity/density rather than just density this is a potentially important contribution to developing monitoring programmes. Linking bait trap

counts directly to damage could bypass the need to estimate population size.

In general, the selection of sites favouring damage instils bias into the dataset. In the strict sense, the results of our study are only valid for sites that have a basic potential for wireworm damages. The influential variables can be interpreted as factors that enhance this basic potential. What cannot then emerge from the analysis is any evidence for landscape features that inhibit wireworm damage. It is also probable that inclusion of sites with zero damage would reduce the proportion of the variance explained in subsequent analyses. The absence of such sites does not invalidate the present results, but they should be recognised in continuative studies.

Our results support that potatoes grown in landscapes with high proportions of grassland might be subjected to higher wireworm pressures than potatoes in rather monotonous landscapes. In such high risk areas, farmers should exercise caution and pay more attention to field characteristics, such as sand content in soil, and management, such as the frequency of grassy crops in crop rotation. Further investigations on the specific habitat requirements of click beetles are strongly recommended. They should explore wireworm communities within semi-natural elements surrounding infested fields and the food preferences of the beetles and larvae of the different species. According to Traugott et al. (2008) wireworm feeding behaviour is very complex and needs further assessments. Also the knowledge on the dispersal behaviour of the female elaterid beetles and their preferred egg laying habitats is still fragmentary. In addition, spatial configuration (adjustments) and connectedness between the semi-natural elements should be investigated regarding their relation to wireworm abundances and migration.

Even though this study indicated that grassy landscape elements can increase wireworm pressure, the beneficial effects of green infrastructures in agro ecosystems through enhancement of pest enemies are well documented in the literature (e.g. Kromp 1999; Dennis and Fry 1992; Kromp and Steinberger 1992). Accordingly, cleaning the landscape from potential wireworm breeding habitats can lead to deficits in natural pest control with serious consequences for plant production. Grassy elements should rather be integrated into the landscape in a way that is spatially and temporally more favourable to natural enemies than to pests (Landis et al. 2000), e.g. by adequate management regimes.

Conclusion

This study supports the necessity of considering landscape variables for wireworm control. Landscapes rich in grassland habitats are indicated to have higher wireworm

potentials than monotonous regions or regions dominated by other features. Further research on the importance of grassy landscape elements for the wireworm pressure in crop fields is strongly recommended. As grassy landscape elements are well known as reservoirs for beneficial organisms, appropriate management regimes for grassy landscape elements suppressing wireworm populations but ensuring the maintenance of natural pest control should be investigated.

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Article 3

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Estimation of biodiversity relevant quantities from airborne laser scanning data

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Abstract

Assessment of biodiversity is prescribed by the European Union Fauna-Flora-Habitat directive in order to document the health of ecosystems. Remote sensing supported by terrestrial field sampling potentially provides efficient means for a regular monitoring cycle. Airborne laser scanning (ALS), also referred to as airborne LiDAR, is especially promising because of its ability to penetrate through gaps in the foliage and provide insight in the forest vegetation layer structure. Knowledge of the topography and the spatial distribution of the plant cover are considered an invaluable proxy for the estimation of biodiversity indicators. As ALS typically provides high point densities and good penetration rates, various biodiversity indicators can be estimated directly and reliably from these measurements.

In this paper it is demonstrated how data collected with ALS, especially with FWF-ALS, can be used to derive quantities relevant for biodiversity assessment. These can be terrain surface features like roughness, as well as vegetation parameters like height, predominant tree type, location of fallen trees or forest layer structure.

Keywords: full-waveform, laser scanning, biodiversity, digital terrain model, vegetation structure, classification

Kurzfassung

Die Bewertung von Biodiversität zur Dokumentation des Gesundheitszustandes von Ökosystemen wird von der Europäischen Union durch die Fauna-Flora-Habitat (FFH) Direktive vorgeschrieben. Moderne Fernerkundungsmethoden, welche von traditionellen terrestrischen Feldaufnahmen unterstützt werden, bieten Möglichkeiten für effiziente regelmäßige Datenerfassungszyklen. Das flugzeuggetragene Laserscanning (airborne laser scanning, ALS), welches oft auch als LiDAR (light detection and ranging) bezeichnet wird, bietet einen wesentlichen Vorteil gegenüber bildgebenden Systemen. Die Laserstrahlen können durch kleine Öffnungen in der Belaubung dringen und ermöglichen so eine Einsicht unter die Baumkronen und in die vertikale Struktur der Waldvegetation. Die Topographie im Wald sowie die räumliche Verteilung des Bewuchses werden von Biologen und Landschaftsökologen als wertvolle stellvertretende Kenngrößen für die Abschätzung der Biodiversität angesehen. Da ALS typischerweise hohe Punktdichten und gute Durchdringungsraten aufweist, können verschiedene Biodiversitätsindikatoren direkt und zuverlässig aus der Luft erfasst werden.

Im folgenden Text wird diskutiert, wie biodiversitäts-relevante Indikatoren aus den FWF-ALS Daten abgeleitet werden können. Solche Indikatoren können Eigenschaften des Geländes sein, wie zum Beispiel Krümmung oder Rauigkeit, oder aber auch Vegetationseigenschaften wie Höhe und Schichtigkeit des Waldes, vorherrschende Baumart oder die Position von umgefallenen Bäumen.

Schlüsselwörter: Full-waveform, Laserscanning, Biodiversität, digitales Geländemodell, Vegetationsstruktur, Klassifizierung

1. Introduction

The Habitat Directive, more formally known as "Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora" [1], was established by the European Union (EU) in 1992 for the conservation of wildlife and nature. The directive's main aim is the protection and conservation of habitats and species, both clearly described in the directive. One of the key terms in conservation is biological diversity, also referred to as biodiversity, which is defined as the variability among living organisms resident in all sorts of ecosystems (terrestrial and marine) [2]. This includes diversity within species, between species and of ecosystems. The Euro-

pean Commission initiated a global study named "The Economics of Ecosystems and Biodiversity (TEEB)" running from 2007 to 2010, which produced an interim report in may 2008 [3]. This report states that selected focal areas suffer from declining biodiversity as a result of human impacts like population growth, urbanisation and climate change. Subsequently, a loss of ecosystem services, such as water and air purification, climate regulation, flood and disease regulation, fisheries and timber production is imminent. It was found that some of these areas are being damaged beyond repair. As a consequence of this process, the existence of lots of species is endangered.

The awareness that natural and semi-natural ecosystems and landscapes provide benefits to human society can be dated back to the mid-1960's and early 1970's [4-6]. However, recently there has been an almost exponential growth in publications on the assessment and valuation of ecosystem services [7-13]. The SEBI 2010 (Streamlining European 2010 Biodiversity Indicators) [14] initiative launched in 2004 proposes a set of 26 appropriate biodiversity indicators, which can be used for the assessment and monitoring of ecosystem conditions. These indicators formulate the basis for the development of strategies for protection and recreation, which are paramount tasks for the conservation and management of a healthy natural environment.

Remote sensing techniques, which can be partially supported by terrestrial field sampling, potentially provide efficient means for regular monitoring cycles. Airborne laser scanning (ALS) is especially promising because a number of biodiversity indicators can directly be estimated from the measurements in an accurate and reliable way, especially if efficient data acquisition over large areas is needed. According to SEBI 2010, this among others could be the indicators ecosystems coverage (No.4), invasive alien species in Europe (No.10), fragmentation of natural and semi-natural areas (No.13), growing stock estimation (No.17) and dead wood (No.18).

The aim of this study is to provide an overview on how various existing methods can be used for the derivation of quantities relevant for biodiversity assessment. Chapter 2 gives an introduction on the measurement process of ALS with focus on the technology of full-waveform systems. Chapter 3 describes how full-waveform observables can be integrated in the generation of a digital terrain model (DTM), which serves as initial product for various further modelling approaches. In chapter 4 the derivation of biodiversity relevant quantities from ALS data sets

is addressed. A conclusion and an outlook are given in chapter 5.

2. Full-waveform airborne laser scanning

Modern ALS systems combine a position and orientation system (POS), a laser ranging module and a data-recording unit. The laser generator, as part of the laser ranging module, emits short pulses of infra-red light (typically 4 – 10 ns, wavelength 1064-1550 nm), which are deflected towards the earth surface, e.g. by a rotating, oscillating or nutating mirror [15]. The emitted pulse interacts with objects on the ground and a part of it is scattered back to the ranging module, where the reflection can be detected. The travelling time from the scanner to the ground and back is directly proportional to the distance covered and can therefore be computed [16]. Usually, ALS systems are carried by a fixed wing aircraft or a helicopter and most often mounted on the fuselage or below the wings. The forward movement of the airborne vehicle, which is called the flight path, is constantly tracked by the POS, which consists of a global navigation satellite system (GNSS) receiver and an inertial measurement unit (IMU). The IMU observes the movements of the aircraft along its axes with high frequency, e.g. 100 Hz, while the GNSS provides 3D positional information. The covered distance, which is also referred to as range, together with the deflection angles of the respective laser beam and the combined GNSS and IMU data are stored and a georeferenced point cloud, representing the scanned surface, can be produced in post-processing.

The latest development for commercially available ALS systems is the so-called full-waveform (FWF) processing. Conventional systems can record the measured range and backscattered energy of one or more consecutive discrete reflections, so-called echoes. In contrast, a FWF-system is able to detect and record the whole emitted and backscattered signal [17]. These

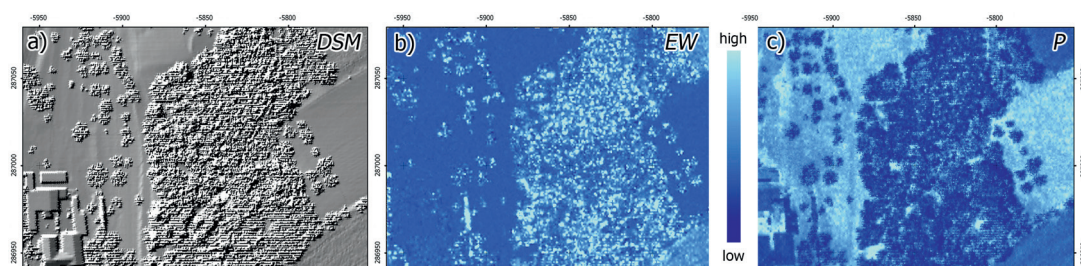


Fig. 1: (a) Digital surface model (DSM); (b) echo widths (EW), higher in vegetated (rough) areas; (c) amplitudes (P), lower in vegetated areas due to loss of backscattered energy because of multiple canopy reflections. [22]

stored signals, also referred to as waveforms, need to be analysed in post-processing. Recent papers propose different methods for ALS waveform analysis and echo extraction [18-20]. During the extraction process, the single echoes are detected and the range, as well as additional information can be derived. These additional observables are the amplitude, giving information about the reflectivity of the object hit by the laser beam, and the width of the signal, also referred to as echo width (EW), describing height distributions of small surface elements within the laser beam [16] (Fig. 1b,c). Moreover, the user-adaptable echo detection process in FWF-ALS allows for a larger number of echoes to be detected per laser shot [21] compared to discrete ALS systems, especially in vegetated areas.

3. Digital terrain model generation using full-waveform data

ALS has established itself as a very suitable measurement technique in environmental studies in the recent past. In forestry related applications, which feature strong thematic intersections with biodiversity relevant applications, ALS has proven very valuable, because most often efficient data acquisition over large areas is needed [23-25]. The biggest advantage of ALS over traditional photogrammetric measurements methods, including airborne and satellite based imagery in the visible and infra-red part of the spectrum, is its ability to "see" through small gaps in the forest canopy. Parts of the laser beam penetrate the foliage and are reflected by vegetation underneath, such as understory trees, bushes, herbaceous vegetation and also the forest floor. Hence, laser scanning over vegetation usually produces more than one echo per shot due to this penetration of the canopy. In the case of ALS, the single echoes in open and the last echoes from overgrown areas are the lowest and therefore considered best candidates to represent the terrain. Using these echoes and appropriate methods for classifying them into terrain and off-terrain points, a process that is also referred to as filtering, a digital terrain model (DTM) can be calculated even in overgrown areas [26-29]. However, there are certain situations when conventional filtering methods, which are solely based on geometric criteria and topological relations, fail to eliminate off-terrain reflections, e.g. from very dense near-ground vegetation [30-31]. Reflections from such objects, if not eliminated in the filtering process, can cause the DTM to run through the vegetation and decimeters above the actual ground surface. In such

cases, the usage of the additional FWF observables (e.g. amplitude and echo width) allows for the generation of more accurate DTMs. Wagner et al. [32] stated that the width of the backscattered echo is influenced by the vertical distribution of scatterers within the footprint area of the laser beam. Vegetation, due to the penetration of the foliage, usually features larger height distributions than flat terrain and consequently larger echo widths. This information can be used in the filtering process in order to increase the DTM quality. Lin and Mills [33] developed a point labeling process, determining terrain points using a threshold for the echo width, which is then applied to complement the single 3D points. This additional surface information is integrated in a DTM generation approach employing Axelsson's progressive densification method [27]. In Mandlbürger et al. [34], the echo width parameter is used to derive a-priori weights as input for the hierarchic robust filtering method [35]. Points featuring small echo widths are considered to stem from terrain and get high a-priori weights. On the contrary, the weight of points with larger echo width is decreased, as they are most likely to represent vegetation. Based on the approach of Mandlbürger et al. [34], Mücke [36] extended the method by assigning probabilities to the single 3D points based on the relation of their corresponding amplitude and echo width. These probabilities indicate whether an echo is likely to stem from terrain or not.

4. Retrieving biodiversity relevant quantities

4.1 Vertical structure

The quality of the derived DTM is essential for a number of secondary products relevant for the assessment of biodiversity, such as the normalized canopy model (nCM). Usually the first laser echoes per shot are used to calculate a digital canopy model (DCM), which represents the top most layer of the forest surface. The difference between the DTM and DCM (DCM minus DTM) is called nCM and it comprises the actual vegetation or tree heights. Due to the relatively small footprint size in ALS, usually about 10 – 50 cm depending on altitude, the laser beams frequently miss the tree tops. Consequently, the measured canopy heights are underestimated and the DCM is subjected to errors [37]. The influence of an incorrect DTM can therefore further decrease the quality of the nCM and products based on it. Hollaus [38] suggested that the stem volume can be estimated from the nCM. He stated that the canopy volume, which describes the volume

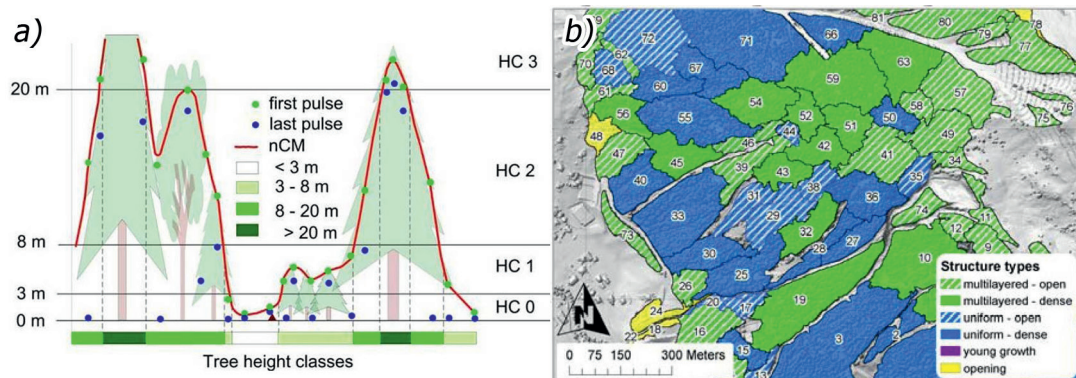


Fig. 2: (a) Classification of laser echoes into height classes according to nCM height; (b) derived structure type classification of forest sample area. [41]

between the terrain surface and the top most canopy layer, is related to the stem volume. It can therefore be defined by a linear function of the canopy volume. The nCM can also be employed to estimate the growth of the forest through comparison of ALS datasets from different acquisition times [39]. An increase or decrease in the amount of growing stock or stem volume can be a conclusive indicator for the effects of global changes, such as climate changes or natural disasters, on the condition of forested areas [40].

Alternatively, the nCM can be used for the definition of height levels above the forest floor for classifying all laser echoes according to their height above ground (Fig. 2a). In this way, information about the height distribution of scatterers and, subsequently, the forest or vegetation layer structure can be extracted (Fig. 2b) [41]. These vegetation layers are of major importance in terms of species diversity. According to Kati et al. [42], the vegetation height and structural complexity are the main environmental parameters determining species composition. For bird species diversity in forests, MacArthur and MacArthur [43] stated that the physical structure of a plant community, i.e. how the foliage is distributed vertically may be more important than the actual composition of plant species.

4.2 Horizontal structure

However, it is not solely the vertical, but also the horizontal complexity that is of importance. Robinson and Sutherland and Benton et al. [44-45] evidenced that changes at the landscape scale through the past decades have led to a decrease in spatial heterogeneity. According to Reichholf [46], especially consolidation farming has contributed to this fact by regulation of streams, straightening of country roads and re-

moving of so far unaffected corridors, seeking to improve agricultural efficiency. An enlargement of the intensive agricultural management units implies the damage of ecologically valuable elements in between, like boundary ridges, slopes and dense thicket [47]. As a result, former natural vegetation is cleared and the complexity of the land surface decreases. As a positive correlation exists between landscape complexity and biodiversity [48-49], the mapping of spatial heterogeneity by analysis of land cover with ALS seems a promising approach for evaluating biodiversity. Nevertheless, effects of spatial heterogeneity may vary considerably between species groups. Depending on their habitat requirements and mobility, heterogeneity can lead to positive or negative effects on species diversity [50]. For example, while forest gaps increase habitat heterogeneity for butterflies [51] and birds [52], they may fragment the habitats of ground beetles, causing disruption of key biological processes, such as dispersal and resource acquisition [53].

Fragmentations in natural areas can be caused by natural, geomorphologic processes, but, in a much shorter time scale, most likely by anthropogenic structures, like road networks. This is especially critical in forests, where roads together with clear fellingings are the main factors causing fragmentation [54]. Major ecological impacts of road networks are the disruption of landscape processes leading to loss of biodiversity [55]. Interrupting horizontal natural processes, e.g. groundwater flow, stream flow, fire spread, foraging and dispersal, fundamentally alters the way the entire ecosystem works [56]. The habitat loss by road construction, altered water routing and downstream peak flows, soil erosion and sedimentation impacts on streams, altered species patterns as well as human access in re-

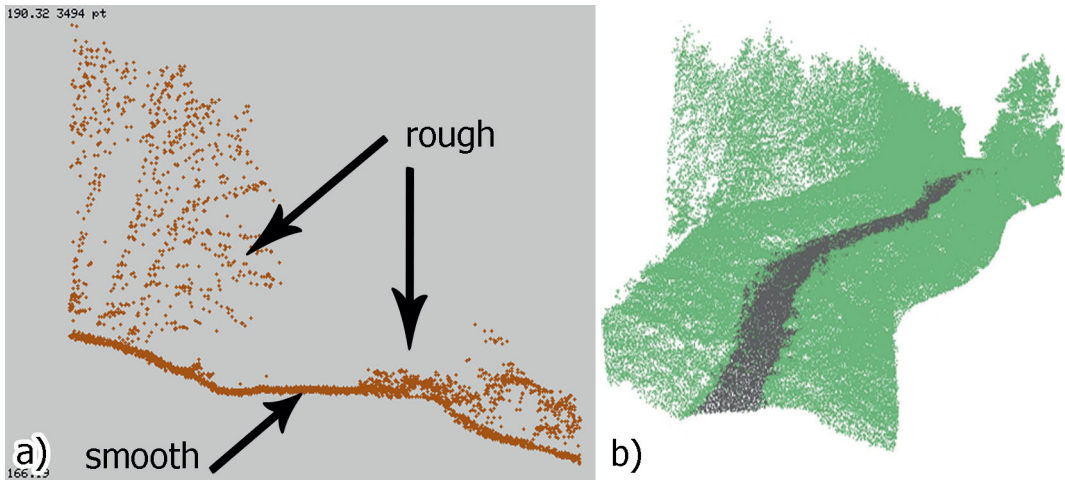


Fig. 3: (a) Profile of forest road (smooth surface) with roadside vegetation and overhanging trees (rough surface); (b) result of 3D segmentation of a forest road, black dots represent the road. [55]

mote areas is seen as a major ecological effect. Therefore, road density is often used as a proxy for forest intactness [57]. Forest road networks may create distinctive spatial patterns, such as converting convoluted to rectilinear shapes, decreasing core forest area, and creating more total edge habitats than logged areas [58-59]. Thus, interior species, species with large home ranges, rare native species and species dependent on disturbance and horizontal flows are affected by those structures. General spatial-process models illustrate that forest roads have the greatest ecological impact early in the process of land transformation, by dissecting the land, leading to habitat fragmentation, shrinkage, and attrition [60-61,57]. Locating and assessing these man-made barriers is of great interest for understanding the connectivity of habitats. In densely grown forests, the roads are often overgrown and occluded, therefore not easily visible for airborne measurement systems. To retrieve them from the 3D ALS point cloud, a measure of surface roughness in terms of height variation of scatterers can be used [62]. In this context, roads tend to be planar and smooth surfaces, compared to e.g. roadside vegetation, which usually features height variations and therefore appears coarser (Fig. 3a). Based on this assumption, roads can be detected directly and automatically in the 3D point cloud by using a 3D segmentation algorithm. Segmentation-based approaches are trying to produce homogenous groups of points. In this case, the homogeneity criteria is the planarity of the surface and the local roughness, both a-priori computed for each single point. A result

of the segmentation of a small scene can be seen in Fig. 3b, where the detected forest road is represented by the black points. The fragmentation of forests due to the road networks could subsequently be estimated by computation of road kilometres per hectare.

4.3 Corridors, patches and gaps

Apart from detecting the disturbances in the landscape connectivity, also the mapping of existing corridors is of concern in landscape ecology [63]. A corridor in terms of landscape ecology is a relatively narrow landscape element that differs from its adjacent areas on either side [64]. It usually connects habitat patches, which are homogeneous areas that differ from their surroundings, and provides routes for the movement of organisms between them [65]. As Vogt et al. [63] pointed out, the assessment of biodiversity indicators is a multiple-scale concept and assessment methods that allow multi-scale analysis should be preferred. They used morphological operations for the automated mapping of corridors, patches and gaps based on Corine Land Cover [66] data, stressing the fact that they were interested in regional to continental scale. Using a similar set of morphological image operations, these landscape elements could be extracted from ALS-based raster maps (e.g. vegetation maps), while extending to a finer, local scale level and exploiting the much higher resolution of the laser data. The penetration ability of ALS can further be used for a description of not only if two landscape patches are connected, but how this connection is composed in terms of vertical distribution of the foliage [67].

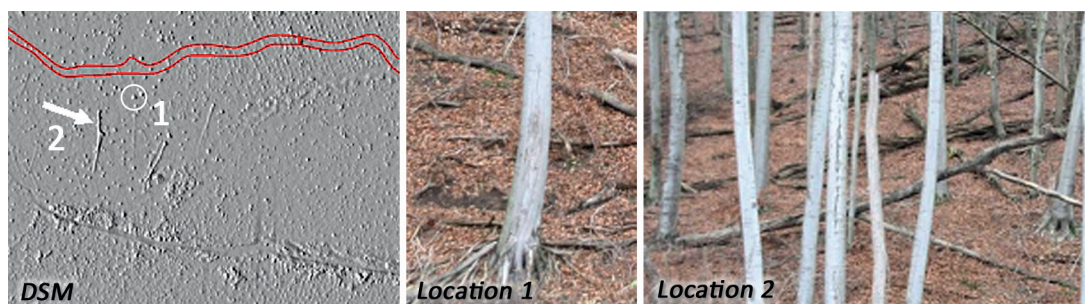


Fig. 4: DSM produced from full-waveform ALS point cloud cut of at 2 m height difference to DTM. Individual and fallen trees are clearly visible. The two images on the right were taken on-site and geo-referenced with GPS. [55]

4.4 Fallen trees and tree species

The high point densities, especially in FWF-ALS, allow for the detection of features that could not be accounted for in lower resolutions. A denser point cloud increases the probability of reflections from stems of single and fallen trees. By selecting points below a certain level of normalized height, e.g. 2 m, and computing a digital surface model (DSM) from the remaining point cloud, individual and fallen trees can clearly be seen in the model (Fig. 4). The presence of understorey or herbaceous vegetation would of course influence the result. Nevertheless, this could be controlled by adapting the height threshold with respect to the scrub layer.

Another way of locating fallen trees is by computing the relation of the number of points below 0.2 m and in the range of 0.2 to 2 m, which is a penetration rate for the near-ground zone (Fig. 5c). Hollaus [68] confirmed by visual examination that the linear structures in the resulting model correspond to broken-down trees. Dying and dead trees, either standing or fallen, provide habitats for a large number of rare and sometimes threatened species [47] and therefore have an important role in conserving forest biodi-

versity. They also influence the canopy, creating gaps when they fall. This allows for more sunlight to reach to forest ground, which can be used by different tree seedlings to grow faster [69]. Plant community composition depends upon both, the frequency of gap creation and the mode of gap-phase regeneration [70]. For example, gap-phase regeneration in tropical forests is dominated by lianas and stalled in a low-canopy state for many years, favouring the growth of a distinct suite of mature species and ultimately result in contrasting species composition [71].

The penetration rate can also be utilized for the recognition of different tree species. The method proposed by Hollaus et al. [72] is based on the assumption that the penetration differs between tree types and season. A deciduous tree loses its leaves during winter time, whereas coniferous trees usually keep their needles. Consequently, this is represented by the height distribution of the tree-wise laser echoes, which can be described by statistical measures and used for discrimination of tree types. For coniferous species like larch, which also drops its needles in winter, other discriminators provided by FWF-ALS can be used. Hollaus et al. [72] stated that the sig-

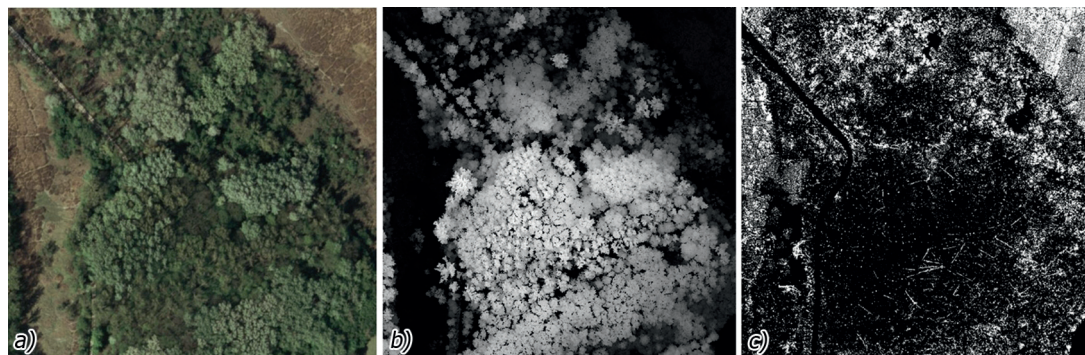


Fig. 5: (a) Orthophoto; (b) nCM (white = high); (c) ratio of points below 0.2 m and in between 0.2 and 2 m (black = only points below 0.2 m), linear structures correspond to fallen trees. [60]

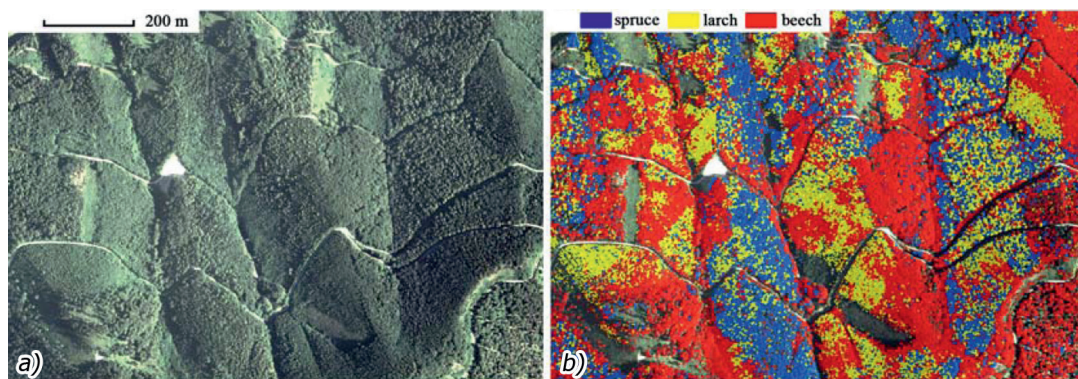


Fig. 6: (a) Orthophoto of forested area; (b) identified tree species spruce (blue), larch (yellow) and beech (red). [64]

natures of the FWF observables echo width and the derived backscatter cross section [18] show significant differences between species, which they used in a decision-tree-based classification method (Fig. 6). The naturalness of the tree species composition is a suitable indicator for the assessment of human impact on the forest by forest management practice [47].

4.5 Proglacial habitats

But it is not only ALS over forested or agricultural areas that can provide significant indicators for biodiversity assessment. In the past decade, ALS data were increasingly used for the mapping of glaciers [73-77]. Only few adapted species are able to live year-round and survive under the rather inhospitable conditions glaciers usually provide. However, within this living space they can exist and flourish because they are not endangered by competing species or predators. Glacier foreland, defined as the region between the current leading edge of the glacier and the moraines marking the latest maximum, are highly dynamic habitats, which are slowly populated by different plants and animals. Due to the melting of the glacier ice, these areas are extended and the few highly specialized species are displaced by others. Identifying areas where such processes take place is of importance for the estimation of the state of mountain ecosystems. Using the 3D point cloud and return intensities provided by ALS, maps of the spatial extent of glaciers including a classification into different surface types (e.g. crevasses, snow, firn, ice) can be created. Many glacial areas were covered by multiple scanning campaigns, so multitemporal analysis is possible. In this way, the data collected by ALS can be used for the monitoring of glaciers and provide efficient means for the

detection of natural processes and changes, like glacial retreat, melting of dead ice bodies (ice that is no longer connected to the active glacier) and development of proglacial habitats [78].

5. Summary and Outlook

This paper gives an introduction and overview on how ALS, and especially the additional observables from FWF-ALS, can be used in order to support biodiversity assessment. So far unequalled point densities offer unique possibilities for detection and modelling of features that could not be accounted for in conventional discrete ALS. As for the biodiversity indicators mentioned in the SEBI 2010 initiative, some can be directly derived with existing algorithms, like the forest stem volume or the forest layer structure (see section 4.1), fragmentation (see section 4.2 and 4.3) and dead wood (see section 4.4). While others can be estimated more indirectly, e.g. the occurrence of alien species, which could be found as the inverse of derived native species from tree species identification. For other indicators mentioned in the initiative, like the fragmentation of river systems, methods for derivation with the application of FWF-ALS are conceivable.

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CHAPTER 3

Providing a spatially explicit assessment framework

Article 4

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Study design 60%, implementation 50%, writing 70%

Assessment framework for landscape services in European cultural landscapes - an Austrian Hungarian case study

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Abstract

European cultural landscapes are characterised by a high level of anthropogenic fragmentation which is known as a major reason for the loss of biodiversity in industrialised countries. To receive support for adequate choices in sustainable landscape planning, information on the spatial distributions of landscape functions and services is needed. Therefore, the objective of this study was to develop an integrative assessment framework to evaluate a wide range of landscape services at different spatial scales. The proposed methodology was applied within the cross-border region of Austria and Hungary. Embedded in a spatial reference framework we assessed and visualized five main landscape services within the investigation area: *regulation, habitat, provision, information and carrier*. Considering location and spatial extent three different levels of service assessment were distinguished: (1) the Landform Approach was based on seven different Landform Types within the study area. All services were directly observable either by the use of Corine land cover or by clearly identifiable spatial indicators. (2) The Broader Habitat Approach focused on the assessment of services at the landscape element scale within randomly selected landscape sample sites. It was based on the use of an expert driven capacity matrix, which values were revised by semi-quantitative data gained from field work. (3) The *information* services occurring at a broader scale were assessed at the Landscape Character Type scale within the Socio-cultural Approach. Additional indicators mainly based on geo-data were defined. Finally, all services were extrapolated to the Landform Types revealing the actual landscape service provision within the study area. The results presented hot and cold spots of service provision at different spatial scales as well as the trade-offs between the different services. The landscape service maps might provide regional stakeholders with valuable information on service supply and can therefore be used as knowledge basis in cross-border landscape planning decision processes. Making landscape services spatially explicit and combining empirical data with spatial information presents an innovative approach to landscape research in the field of assessing and visualizing landscape services. This would enable the development of a decision support tool, which can be used for the systematic evaluation of goal attainments and conflict detection.

Keywords: landscape services, ecosystem services, assessment framework; mapping; capacity matrix, spatial reference framework

Abbreviations: LFT, Landform Type; LCT, Landscape Character Type; BHT, Broader Habitat Type; BHS, Broader Habitat(type) value; LESV, Landscape Element Service value

1 Introduction

European cultural landscapes are known to provide a wide range of functions and services that are useful for humans. However, the supply of ecosystem services and biodiversity is in danger, mainly caused by a high level of habitat loss and fragmentation (MEA 2003). One reason for the loss of ecosystems in cultural landscapes is the lack of integrating ecosystem service values in regional spatial planning projects. The ecosystem service concept is therefore aiming at supporting the development of policies and instruments by integrating ecological, socio-cultural and economical perspectives to provide insights into human impacts on ecosystems and the welfare effects of management policies (TEEB, 2010). This scientific concept has experienced increasing attention in the last decades as it provides the means of documenting the importance and benefits of ecosystems and landscape for human society. One of the most relevant publications is the Millennium Ecosystem Assessment (2003) which provides the basic framework for assessing the interactions between ecosystems and humans and how these can be measured, evaluated and strengthened for future human well-being. After the release of the Millennium Assessment (MEA, 2003), which focused on the benefits people derive directly and indirectly from ecosystems, the literature concerning ecosystem services has increased exponentially all over the world (Fisher et al., 2009). Several authors have been dealing with classifying, quantifying, mapping and valuing of ecosystem services in order to integrate the concept into decision making processes (e.g. Costanza et al., 1997; Daily, 1997; de Groot et al., 2010; Fisher et al., 2009; Hermann et al., 2011). However, despite the enhancing interest in ecosystem service research, still many open questions remain to fully integrate the ecosystem service concept in landscape research and decision making.

Because landscape sciences focus on spatial pattern and scale, they can provide useful insights into how the spatial distribution of human activities influences important landscape processes and structures from which services are derived (Jones et al., 2008). The central notion in landscape development has always been that people are part of the landscape and that landscapes are changed for their benefit (Antrop, 2001; Linehan and Gross, 1998). Especially, in Central and Eastern Europe both the analysis of landscape pattern and processes and the assessment of landscape functionality as a basis for land use planning have a long tradition (Bastian and Schreiber, 1994; Buchwald and Engelhardt, 1968; Lee et al., 1999). In recent years, the terms 'landscape function and service' have become more important in literature (Bastian and Schreiber, 1999; de Groot et al., 2010; Willemen et al., 2010). To receive support for adequate choices in landscape planning, information on the spatial distributions of landscape functions and services is needed. Although in the last years considerable progress has been made in assessing, quantifying and mapping a multitude of landscape services, implementing the concept into sustainable landscape planning and management still remains a big challenge (Hermann et al., 2011; Noorgard, 2010). Regarding the state-of-the-art, better insight into interactions between land cover, use and function and methods to assess and map land use and landscape function is still needed (e.g. Verburg et al., 2009). Visualisation should illustrate the spatial heterogeneity in quality and quantity of services provision, which is due to differences in biophysical and socioeconomic conditions at different scale levels (Meyer and Grabaum, 2008; Wiggering et al., 2006). Therefore, landscape services are to be addressed and assessed on various scales (Hein et al., 2006). Assessing and mapping the multitude of services provided by different landscapes at different scales is seen as prerequisite for sustainable landscape management (Verburg et al. 2009). This would enable the development of a decision support tool, which can be used for the systematic evaluation of goal attainments and conflict detection. As assessing and mapping of services is mainly dependent on data availability and finding the appropriate indicator, most publications focused either on selected landscape services and/or emphasised only on one assessment scale (e.g. Burkhard et al., 2009; Troy and Wilson, 2006; Willemen et al., 2008). An integrative framework that takes a wide range of

ecosystem/landscape services into account is still under development. Such a framework should be comprehensible, feasible and able to be applied at wide range of scales to different ecosystems or landscapes (Hein et al., 2006). We want to meet these challenges by the development of a framework which will link the processes in the landscapes with the services provided at different scales.

The aim of this paper is therefore to present a spatially explicit methodology evaluating a broad set of landscape services by meeting the following research objectives: (i) mapping the hot and cold spots of service provision within different landscape types (ii) visualizing the trade-offs between the services within the investigation area (iii) testing the concept of landscape services as an operational tool to evaluate sensitive regions.

2 Materials and methods

2.1 Study area

The project region covers the Austrian-Hungarian transboundary region of the Neusiedler See / Fertő (Figure 1) extending over 2000 km². It is part of the Small Hungarian Plain in Central Europe representing the westernmost extension of the Pannonian Basin. The shallow lake Neusiedler See which lies in a flat basin is dominating the landscape view. The southern Hungarian part (Fertő) is mainly lowland with gentle hills on the western side. The northern part belonging to Austria has contrasting western and eastern sides: the former is formed by the pronounced slope zone of a low mountain ridge, whereas the latter, the Seewinkel, represents the lowest land in Austria. The region is characterised by a hot, dry Pannonian climate with an annual precipitation of 700-800 mm and annual mean temperature of >9°C (ZAMG, 2002). In a relatively small area, plants and animals with Alpine, Asiatic and Mediterranean affinities, as well as northern species, are present, resulting in high species diversity. Thus, on both sides of the border, different protection categories were declared in the last decades: Nationalparks Neusiedler See-Seewinkel and Fertő-Hanság, Ramsar Sites, UNESCO World Heritage Site, Biosphere Reserves, various Natura2000-areas and different national conservation labels can be found throughout the region.

Today two main economy sectors dominate the area: intensive agriculture particularly crop-growing, wine growing and greenhouse-vegetable gardening and tourism, which is especially centered around the lake and focused on rather small areas. In the last decades the typical lake tourism changed to a more diversified tourism based on nature, the national park, cycling and other sports activities, cultural traditions and events. Nowadays severe problems arise from the growing conflict between these two demands caused by increasing requirement of land for their anthropogenic uses, additionally interfering with nature conservation issues. Because of the multifunctional landscape and the diverging claims on land utilisation we used that region to develop the proposed approach.

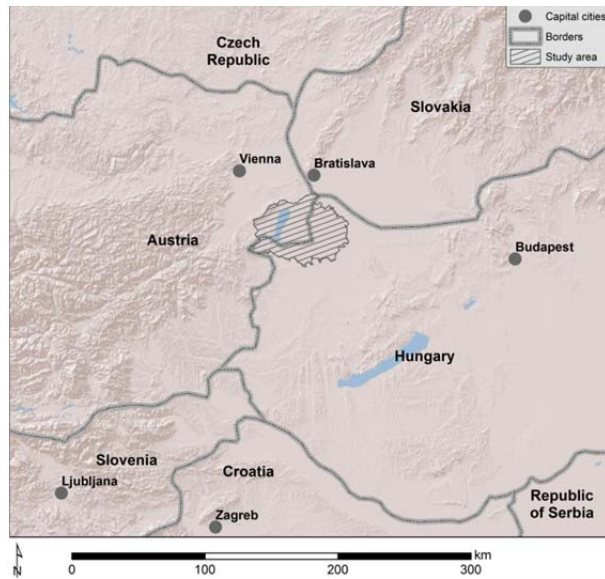


Figure 1: Location of the study area in the transboundary region of Austria and Hungary in Central Europe. Topographical map is made with Natural Earth and www.ArcGIS.com

2.2 Landscape services

In our study we used the concept of landscape services (Termorshuizen and Opdam, 2009). As ‘landscapes’ may be more attractive to non-ecological scientific disciplines in contrary to the term ‘ecosystem’ and may be associated with people’s local environment, we preferred the term ‘landscape services’ instead of ‘ecosystem services’. In contrast, ‘ecosystem’ may be related with natural processes and conservation instead of with human habitat and cultural patterns. In our definition landscape services are therefore all goods and services that landscapes provide for well-being. They include potentials, materials and processes of nature (e.g. biomass, raw materials, primary productivity, etc.) and services of cultural elements and constructions that come into being through human creation (e.g. buildings, settlements, infrastructure, etc.) (Konkoly-Gyuró, 2011b). Based on the list provided by de Groot (2006), a general set of landscape services was derived (Table 1). The individual sub-services are grouped into five main services (1) *regulation*, (2) *habitat*, (3) *provision*, (4) *information* and (5) *carrier*. Whereas the *regulation* services are related to the capacity of cultural landscapes to regulate essential ecological processes and life support systems through biogeochemical cycles, the *habitat* services describe the capacity to provide refuge and reproduction habitat to wild plants and animals. As the habitat requirements differ from species to species they are defined in terms of the carrying capacity and spatial needs (minimum critical biotope size) of the specific biotope type. The *provision* services are targeting the supply of natural resources concerning ‘edible wild plants and animals’, ‘raw materials’, ‘genetic’ and ‘medicinal resources’. Whereas the *information* services include all services referring to spiritual enrichment, cognitive development, recreation and aesthetic experiences, the *carrier* services describe the capacity of landscapes to provide suitable substrate (soil) for ‘cultivation’, ‘habitation’, ‘tourism facilities’, ‘energy conversion’ and ‘transportation’.

2.3 Overall methodology

The overall methodological framework (Figure 2) is based on data availability and spatial scales of assessments. Individual sub-services were assessed at the specific service providing unit and were finally aggregated to the main services (Table 1). Driven by the link between landscape service and service providing unit, three different scales for landscape service assessment were distinguished and

integrated into a spatial reference framework representing different homogenous assessment units (Renetzeder et al., 2008). Landform Types (LFT) by Konkoly-Gyuró et al. (2010) formed the common basis for our integrative assessment of Austrian and Hungarian landscapes. They represent the main geomorphological and hydrological features of the study area (Figure 3): ‘Lake Basin’ (LFT 1), ‘Marshlands’ (LFT 2), ‘River Floodplains’ (LFT 3), ‘Low lying terrace’ (LFT 4), ‘Elevated terrace’ (LFT 5), ‘Hilly area and hill range’ (LFT 6) and ‘Low and middle range mountains’ (LFT 7). Because spatially explicit data was not available for all sub-services, we selected landscape sample sites within the different Landform Types. This was done by a stratified random sampling method. In each Landform Type, six sample sites of 1 km x 1 km size according to the European Grid System (INSPIRE, 2009) were randomly selected after applying a list of sampling rules resulting in 41 landscape sample sites (Figure 3). These sampling rules included terms for the exclusion of grid cells with more than 1% not accessible areas, >30% water surfaces, or >10% artificial areas. The cells needed to be 99% within the same Landform Type and have at least 500 m distance to the border of the investigation area. The assessment of the *information* services that required another spatial scale was based on Landscape Character Types (LCT) representing a combination of relief, dominant land cover and land use intensity within the different Landform Types (Konkoly-Gyuró et al., 2010; after Swanwick, 2002).

This spatial reference framework served to link the individual landscape services and their assessment approaches on the different scales with each other in a hierarchical way. Dependent on the different service providing units at the different scales three assessment approaches, namely the (a) the Broader Habitat Approach based on the sample sites (b) Landform Approach based on the LFTs, and (c) the Socio-Cultural Approach based on the LCTs were distinguished (Figure 2).

- (a) Broader Habitat Approach: As the service providing units were located at the landscape element scale (Forman and Godron, 1986), this approach was carried out within the 41 landscape sample sites. The assessment was based on the use of a capacity matrix (Burkhard et al., 2009), which values were revised by qualifiers, that derived from field surveys.
- (b) Landform Approach: Assessment of landscape services that were directly observable for the entire Landform Types either by the use of Corine land cover CLC 2006 (EEA, 2006) , or by clearly identifiable spatial indicators.
- (c) Socio-cultural Approach: For the assessment of the *information* services we used the 14 LCTs as spatial units. Landscape metrics, biophysical and socio-economic landscape components were used to describe the location and the capacity of LCTs to provide services. These different landscape data were translated into spatial indicators and linked to the related services.

Finally, to compare the different landscape types regarding their provision of landscape services, all individual sub-services were extrapolated to the Landform Types and aggregated to the main services. The main services *regulation*, *habitat*, *provision*, *information* and *carrier* were plotted onto a 5 axes spider web diagram illustrating their trade-offs.

In the next sections the three approaches as well as the extrapolation and the aggregation of data are presented in detail.

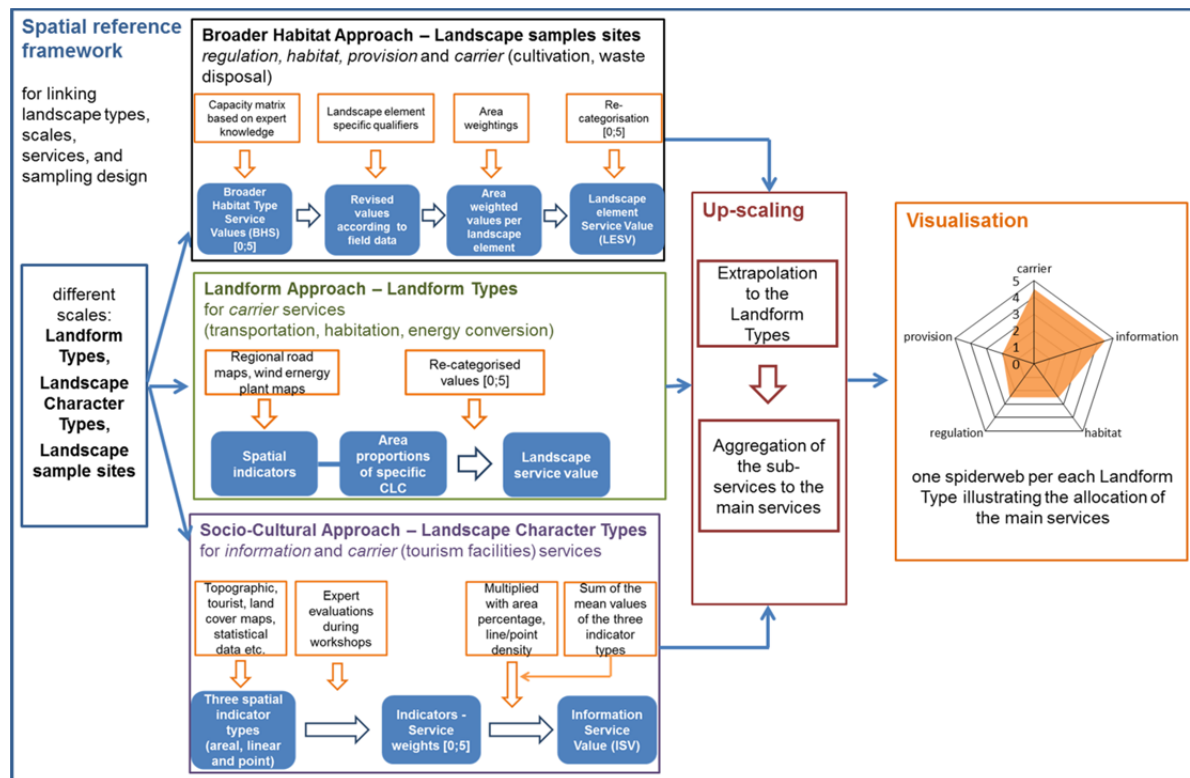


Figure 2: Methodological assessment framework based on three scales: Landform Types, Landscape sample sites and Landscape Character Types. Dependent on the different service providing units at the different scales the Landform Approach, the Broader Habitat Approach and the Socio-cultural Approach can be distinguished. Finally, the aggregated service groups were visualized at the Landform Type scale; CLC = Corine land cover CLC2006 (EEA, 2007)

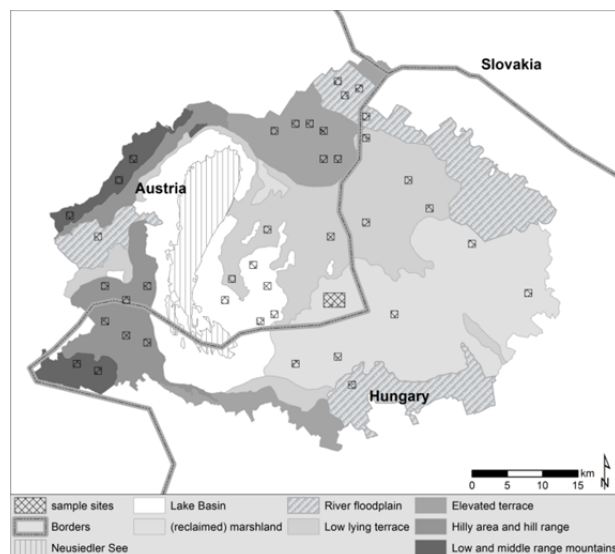


Figure 3: Location of 41 sample sites within the seven Landform Types as defined by Konkoly-Gyuró et al. (2010). Topographical map is made with Natural Earth and www.ArcGIS.com.

2.4 Assessing landscape services

2.4.1 Broader Habitat Approach

As data availability for specific service indicators (e.g. crops /ha; kJ/ha for cultivation service) was limited or often not comparable within our approach, *regulation*, *habitat*, *provision* and partly *carrier*

(only ‘cultivation’ and ‘waste disposal’) services (Table 1) were consequently related to a specific Broader Habitat by the use of a capacity matrix (after Burkhard et al., 2009; Haines-Young and Potschin, 2008). The list of about 181 Austrian Broader Habitat types (BHT) was based on the definition lists of Essl et al. (2002) and Pascher et al. (2010). During field work (May to September 2010) each landscape element in the 41 landscape site samples was assigned a BHT. All field data were stored in a personal geodatabase in ArcGIS 10 (ESRI, 2011). In principle, the subsequent work flow (Figure 2) consisted of four steps which were implemented within ArcGIS Modelbuilder in ArcGIS 10 (ESRI, 2011):

- (i) As a first step towards creating the matrix, the capacities of all 181 BHTs regarding their provision of individual landscape services were set up by expert evaluations during several workshops on different disciplines of ecology (Vegetation Science, Zoology, Pedology and Climate Science). The so-called Broader Habitat(type) Service (BHS) values ranged from categories ‘0’ (no relevant link) to ‘5’ (very high relevant link). The resulting capacity matrix (Table A 1) served as input table for subsequent analysis.
- (ii) To enhance the expert based approach, further attributes gathered by field mapping were used for a closer look on the actual service provision of each landscape element. During field survey, additional qualifiers on ‘broader habitat structure’, ‘management’, ‘pressure’ and ‘valuable attributes’ within the ‘landscape site samples’ were separately collected for each investigated landscape element. These qualifiers exerted either a positive (‘1’), negative (‘-1’) or no influence (‘0’) on the provision of a specific service (Table A 2) and served as a further input table within the modelbuilder environment by calculating a specific ‘landscape element service value’ (LESV). Thus, the former values from the capacity matrix were revised by increasing, decreasing or remaining constant for each landscape element depending on qualifiers found in the field.
- (iii) As the area of a landscape element has also an impact on the provision of a service (e.g. a large forested area has more impact on climate regulation than a small one), additional area-weighting was integrated into the assessment. Regarding waste disposal, area-weighting was not appropriate because direct relationships between area share and functional capacity could not be outlined.
- (iv) The resulting values for each service were re-categorised ranging from ‘0’ to ‘5’ based on 20%-percentiles and then each service-specific layer, including the final ‘landscape element service values’ (LESV) was stored in the geodatabase.

2.4.2 Landform Approach

Location and extent of the *carrier* services ‘transportation’, ‘habitation’ and ‘energy conversion’ were clearly distinguishable.

Transportation service: To measure the actual state of ‘transportation’ within the investigation area, absolute running meters of transportation networks were calculated for each Landform Type. For this reason, main and side roads as well as railroad tracks that derived from regional road maps were integrated into the assessment. Due to traffic densities, the lengths of the main roads were double-weighted. Resulting track lengths were divided by the total area shares of the single LFTs, resulting in area density values of the transportation network.

Habitation service: To comprehensively include *settlement areas* and *other man-made facilities* such as *industrial* and *commercial sites* or *sport* and *leisure facilities* into the assessment, Corine land cover CLC2006 (EEA, 2007) was taken as source layer. Area proportions of the aforementioned CLC

classes of interest were again separately calculated for all LFTs by multiplying class areas with class specific BHS values.

Energy conversion service: The ‘energy conversion’ service considers facilities for the conversion of wind energy into electricity (see Table 1). On the basis of a map sheet ‘Regionales Rahmenkonzept für Windenergieanlagen’ (‘regional framework for energy plants’), provided by the ‘GIS Koordinationsstelle, Raumordnung Burgenland’ all actual locations and suitable zones of wind power stations within the investigation area were detected and the area proportion was calculated and re-categorised. As this base layer was only available for Austria, wind power stations on the Hungarian side of the study region were mapped after visual interpretation of the latest aerial imagery available.

2.4.3 Socio-cultural Approach

Information services and the sub-service ‘tourism facilities’ of the *carrier services* were assessed on the level of the LCTs as defined by the spatial reference framework (see Figure 2). Indicators were developed for each sub-service and derived for the whole investigation area with the help of topographic-, tourist- and land cover maps, online Landmark Cadastral Register (Kollányi et al., 2012) database, as well as statistical data. The indicators for the ‘tourism facilities’ comprised *touristic nodes* (Ziener, 2003), *length of all touristic trails* and *water sport facilities* (the access of lake or pond and the existence of basic infrastructure like boat bridges). For the *information services* three main types of indicators have been developed: 1.) areal landscape elements (e.g. water bodies, wetlands, forests, etc.), 2.) linear landscape elements (e.g. forest edge, water edge, vineyard edge, etc.) and 3.) point landscape elements (e.g. churches, chapels, look-out towers, etc.). All data were stored and processed in a personal geodatabase in ArcGIS 10 (ESRI, 2011). First, based on expert evaluations each of the indicators was assigned a service weight between ‘0’ and ‘5’ reflecting the relevance of the indicator for the sub-services (Table A 3). In the next step, depending on the geometry of the data, the area percentage and the line/point density per LCT was multiplied with the service weight, then normalised between 0-5 and finally the mean of each indicator type was calculated. To reach one final *information service value* per LCT the sum of the mean values of the three indicator types were calculated. In the end the values were normalized again, in order to be comparable with the landscape service values assessed within the other approaches.

2.4.4 Extrapolation and aggregation of the different landscape services to the main services

The services assessed by the Broader Habitat Approach had to be aggregated onto the Landform Type scale in a stepwise approach, for further comparison with the services assessed within the Landform and Socio-Cultural Approach.

To receive one single value for each sub-service per sample site, area-weighted mean values of all LESVs were separately calculated for the different sub services. The outcomes can be interpreted as the actual state of each investigated sample site in the fulfilment of the individual services. These were extrapolated to LFT-level by calculating mean values for all six sample sites per LFT. Consequently, the main service values of *regulation*, *habitat* and *provision* were obtained by combining and calculating the mean values of the specific sub services on LFT level.

Hereby LFT 1, the ‘Lake Basin’, was regarded as special case, because representativeness of the sample sites for the up-scaling process was limited due to location of the ‘Neusiedler See’ itself plus its adjacent reed belt and satellite lakes within the LFT 1. However, these inaccessible areas make up

more than 60% of LFT 1 and therefore must be taken into account for landscape service provision. To overcome these difficulties, LFT 1 was split up into 4 subparts such as the terrestrial region, characterised by the Broader Habitat outcomes, the lake itself, the reed belt and the satellite lakes. For the latter three, provision of landscape services was derived by calculating area-weighted values from the BHS table and afterwards combining them with sample site based results for the terrestrial area according to their area shares.

The services evaluated by the Socio-Cultural Approach also had to be scaled-up on LFT level. As the different LCTs were embedded within the LFTs (refer to spatial reference framework), area-weighted values of the LCT-based main services were calculated for each LFT. Finally, the area-weighted main service values within each LFT were summed up to reach one final value for the *carrier* and the *information* main services.

3 Results

3.1 Individual landscape service maps – hot and cold spots

3.1.1 Broader Habitat Approach

Resulting from the GIS model the individual LESVs can be mapped for each landscape sample site within the different LFTs, in order to explicitly visualize local differences in landscape service delivery. As an example we present the ‘water regulation’ service within one sample site of the LFT ‘Lake basin’, ‘Low lying terrace’ and ‘Hilly area and hill range’ (Figure 4). Within the LFT ‘Lake basin’ the sample site shows grassland matrix interspersed by patches of arable land and vineyards providing medium to very high ‘water regulation’ values. Whereas the sample site of the ‘Low lying terrace’ represents an open and intensively used landscape supplying very low ‘water regulation’ service (cold spot), the sample site of the third LFT displays a heterogeneous land cover mosaic with very high landscape service values (hot spot) for the most part of the area.

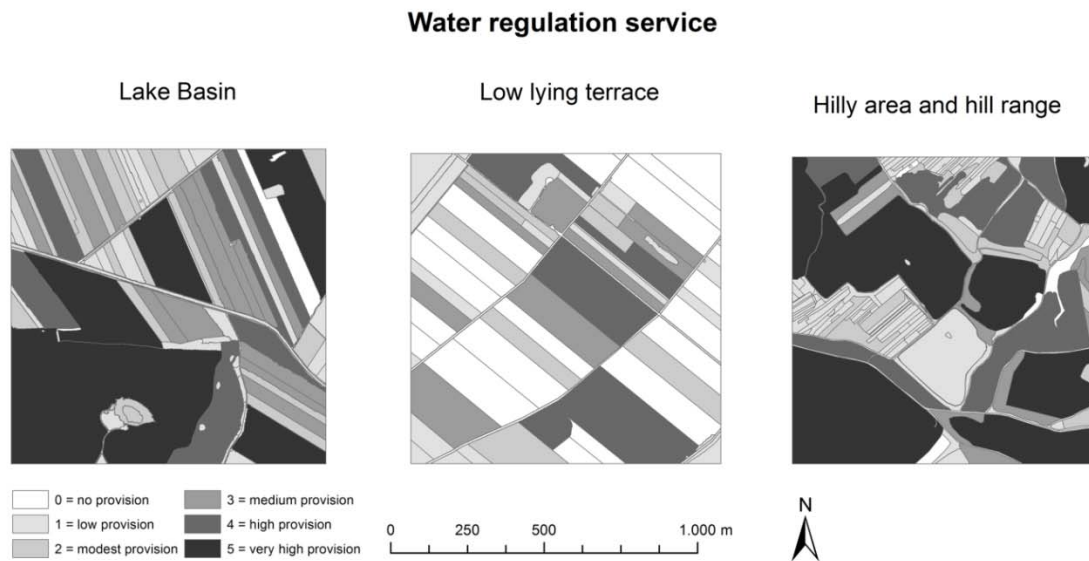


Figure 4: ‘Water regulation’ service maps of the LFTs Lake basin, Low lying terrace and Hilly area and hill range (one sample site per LFTs). 1=low service provision, 2 = modest service provision, 3 = medium service provision, 4 = high service provision, 5 = very high service provision

3.1.2 Socio-Cultural Approach

The values of the individual landscape service ‘tourism-facilities’ were presented at the LCT level (Figure 5). The very high value (hot spot) of the northern part of the lake basin which is dominated by open water (LCT 1b) resulted mainly from water sports. The big and middle divided touristic nodes (Ziener, 2003) on the lakeshore are only partly located in the lake basin (lake resorts), while the villages and towns are situated in the neighbouring LCTs. The hill areas on the western shore of the lake (LCT 3b) gained higher values than the flat areas on the eastern shore (LCT 1c and 2c), because of the diversified landscape pattern resulting in different structure of settlements and touristic nodes and a partly dense network of touristic trails. The marshland areas (LCT 1d and 2a) as well as the southern part of the lake basin which is dominated by the reed belt and wetlands (LCT 1a) showed the least ‘tourism-facilities’ values (cold spots). In former times the Hanság area was covered by extensive wetlands with the result that the settlements are very small. There, land use is dominated by agriculture and tourism is underdeveloped.

The distribution of the aggregated *information* services were visualised at the LCT level as well (Figure 6). In general, these values ranged between 1.4 and 3.3 (low to medium service provision). The hot spots of *information* services were found in the LCTs 3a, 3b, and 1c, where hill ranges and foothills of low mountains with medium or intensive human use are dominated by a mosaic of forest, grasslands and water surfaces. Relatively high values also occurred in low intensity human use areas, which are remnants of marshlands. The lowest values were provided by LCT 2b, where the similar visual appearance is due to the equally flat surface and to the overwhelming intensive arable land parcels.

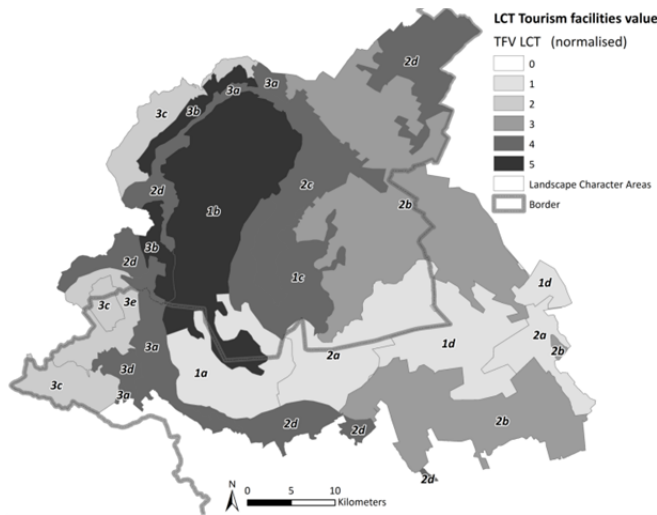


Figure 5: 'Tourism facilities' service values for each LCT; 0 = no service provision, 1=low service provision, 2 = modest service provision, 3 = medium service provision, 4 = high service provision, 5 = very high service provision

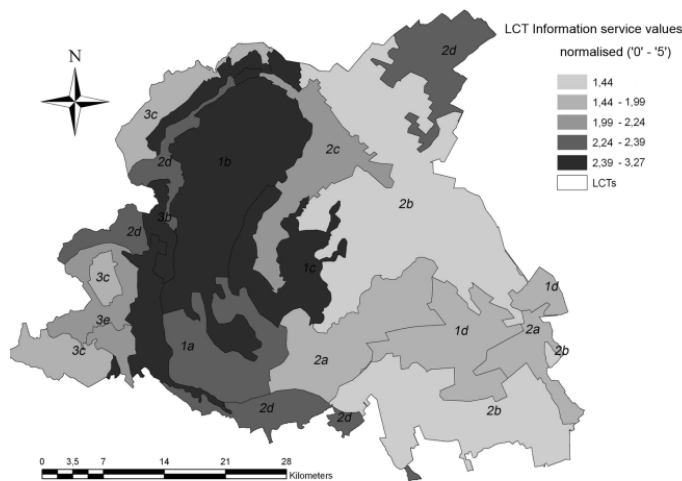


Figure 6: Information service values for each LCT. The single values were pooled into five groups displaying the range between the lowest value (cold spots) and highest value (hot spots) reached within the investigation area

3.2 Trade-offs between the main landscape services within the investigation area

The resulting values ranging [0;5] showed the actual provision of the main services *carrier*, *information*, *habitat*, *regulation* and *provision* within the seven Landform Types (Table 2). They represented the high diversity within the investigation area ranging from natural and semi natural areas such as the shallow lake and its immense reed belt, the remaining marshland and flood plains, the extensively used hilly area up to the intensive agricultural regions in the low lying terraces. Whereas most service values ranged between '0' and '3' the values calculated for the LFT 'Low and middle range mountains' mainly covered by forests resulted in very high *regulation*, *provision* and *habitat* service supply.

The diversifying LFTs 'Lake basin', 'Low lying terrace' and 'Hilly area and hill range' were selected to present the trade-offs between the main service categories (Figure 7). The *regulation* and *carrier* services comprised the most different landscape services based on very diversifying individual sub-service values within the three selected LFTs (see Figure 8, 9) The LFT 'Lake basin' presented a high provision of *habitat* and medium provision of *regulation* and *information* services, which was mainly

due to the dominating shallow lake surrounded by the reed belt as well as on the natural and semi-natural area in the southern and eastern part of the Landform Type. These areas are intrinsically providing conservation functions, as they serve as nesting and feeding habitats for colonies of reed-nesting birds (e.g. egrets, spoonbills). Within the *regulation* services the ‘water and nutrient regulation’ as well as the ‘disturbance prevention’ services were dominating (see Figure 8). Although ‘tourism facilities’ are very important in that area, the *carrier* services had only reached low values due to the low values of the other sub-services (‘energy conversion’ and ‘habitation’ services) in that category .

Compared to the other LFTs the ‘Low lying terrace’ at the same time presented the lowest values of the *provisioning* and *information* and high values of *carrier* services (Table 2). Predominant is the equally flat surface covered by intensive arable land parcels and peri-urban zones with growing horticultural establishments (see high values of ‘habitation’ and ‘cultivation services’ in Figure 9). These areas are less attractive, sharing only little recreational nature conservation potential.

The service values of the LFT ‘Hilly area and hill range’ were well balanced (Figure 7) reflecting a diversified landscape including both extensive and intensive rural areas accompanied by some semi-urban settlements. The higher values of *regulation* and *habitat* services were based on the semi open landscape on the western sandstone hill, mainly in Hungary. These landscapes are intensively used mainly covered by vineyards. Tourism is based on the wine culture and the dense cycling road network inserted into the landscape, which led to high *carrier* services.

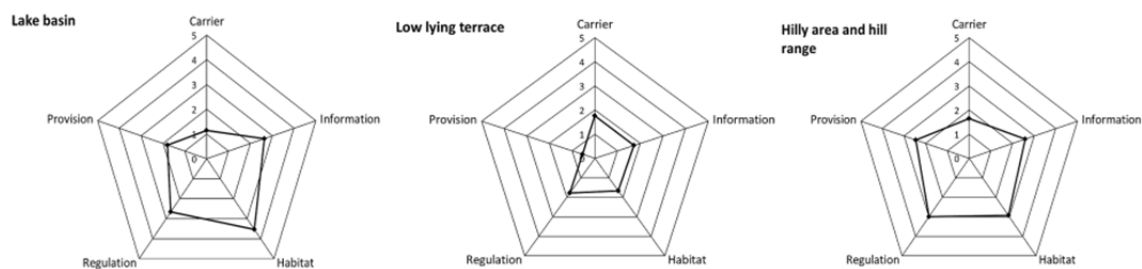


Figure 7: Trade-offs between the main landscape services: *carrier*, *information*, *habitat*, *regulation* and *provision* for the LFTs ‘Lake basin’, ‘Low lying terrace’ and ‘Hilly area and hill range’

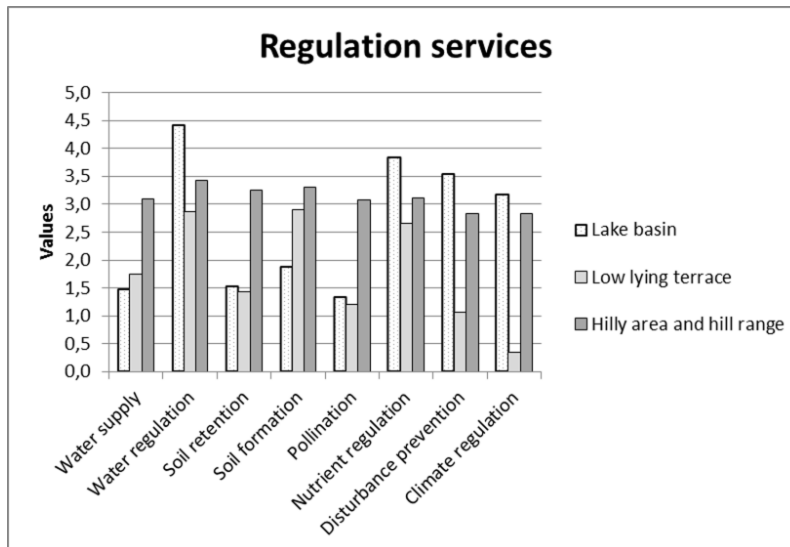


Figure 8: Distribution of the *regulation* sub-services of the LFTs 'Lake basin', 'Low lying terrace' and 'Hilly area and hill range'

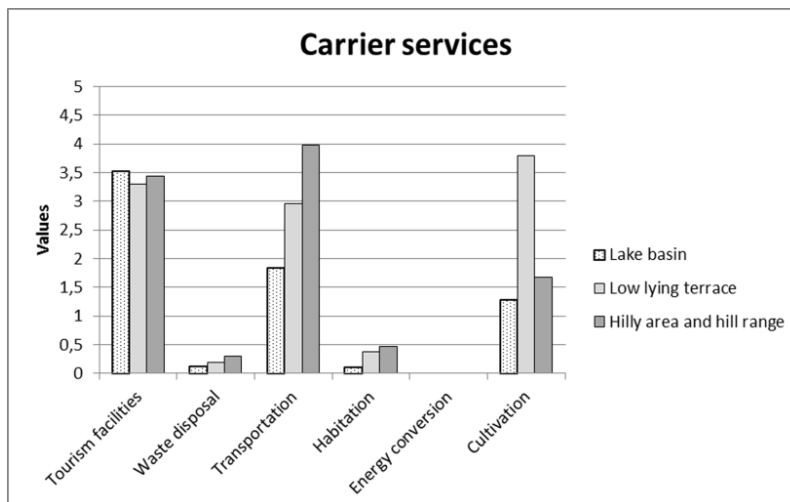


Figure 9: Distribution of the *carrier* sub-services of the LFTs 'Lake basin', 'Low lying terrace' and 'Hilly area and hill range'

4 Discussion

4.1 Evaluation of the methodology

Within our study we assessed a wide range of landscape services at different spatial scales to provide a good overview of the services people derive from the investigated landscapes. It has to be acknowledged that the proposed framework is a model of reality trying to reduce the complexity of human-environmental systems in an appropriate, logical and reproducible manner. Hence, generalizations and simplifications have to be tolerated in order to receive a holistic picture of complex systems. As different landscapes have different functions based on their structure and related processes the individual landscape service provision are strongly linked to natural conditions: e.g. water holding capacity, soil conditions, fauna and flora, elevation, slope and climate as well as anthropogenic influences, such as changing land use, greenhouse and aerosol effects. However, finding appropriate indicators related to the specific service providing unit and exploring how functions and services are correlated with different landscape scenarios are still unresolved questions

(Seppelt et al., 2011, Wallace, 2007). Current landscape service indicators are still limited by insufficient data and an overall low ability to convey information (Layke, 2009). Therefore, we used an expert driven evaluation in assessing most of the landscape services, extended by empirical data and spatial indicators. Earlier studies have already applied assessment matrices (e.g. Haines-Young and Potschin, 2008; Burkhard et al., 2009, 2011, Koschke et al. 2012), where land cover types were linked by hypothetical values to selected ecosystem service supply capacities. This method enables a rapid service assessment and supplies a good overview to see first trends for landscape service provision (Burkhard et al., 2009). Applying a relative five step scale enables a comparison between the different landscapes providing landscape service by harmonisation of the different indicators, and offers the opportunity to avoid value-laden units, such as monetary terms. However, we are aware, that these expert based values are dependent on the observer's ability to make appropriate evaluations and often lack objectivity. By including additional data from a field survey (for the Broader Habitat Approach) and spatial data (for the Socio-Cultural Approach) into our assessment, we tried to reduce this expert bias. The way forward to explicitly address each sub-service would be to develop indicator sets including data from monitoring, measurements, and computer based modelling targeted to each of these. But this would encompass the need of very detailed data and statistical relationships which are not available yet for the wide range of services.

Whereas most previous studies used CORINE data as service providing units (e.g.: Burkhard et al., 2009, 2011, Naidoo and Ricketts, 2006; Koschke et al., 2012), we applied land cover data of higher spatial resolution within our Broader Habitat Approach, in order to be locally explicit and further to be able to integrate our approach in regional landscape planning projects. Using habitat specific qualifiers, such as 'habitat structure', 'management', 'pressure' and 'valuable attributes' gives researchers the possibility to identify and quantify site-specific relations between landscape services and the environment. However, to fully assess services, both temporal and spatial scale effects additionally have to be integrated into the assessment framework (Bastian et al. 2012). For instance, *regulation* services can occur both at landscape scale (e.g. water regulation of the whole marshland area) and landscape element-scale (e.g. water holding capacity of a single mire) (de Groot, 1992). Also pressures on ecosystem services can have effects at different scales. In general, physical processes on small scales are often driven by the impact of long term phenomena at larger scales (climate patterns, hurricanes, fires) (Limburg et al., 2002). Hence, for the analyses of the dynamics of service supply it is very important to consider the drivers and processes at scales relevant for service generation. To integrate effects at broader scales e.g. the landscape scale, both spatial configuration of the landscape elements (see Frank et al. 2011), effects of neighbouring features (e.g. power plant) and temporal dynamics have to be considered in future projects.

Embedded within a spatial reference framework the individual landscape services of all three approaches were extrapolated to the Landform Type scale, in order to grasp the main differences between different landscapes and land cover types. Such a spatial framework enabled us to map locally explicit service provision as well as to visualise different service supply at the landscape scale. However, it has to be taken into account that the aggregation of the sub-services to the main 5 services as well as the extrapolation to the Landform Type scale blurred to some extent the picture of the services' provision due to the averaging. Also the effects on the Landform Type scale might be different from that at the Landscape Character Type or landscape sample site scale. Investigations on the extent of service delivery as well as on interactions among multiple landscape services across different scales (Willemsen et al., 2010) have to be undertaken by further investigations.

Our approach was based on the assessment of the single sub- services that were finally aggregated to the main service groups, which resulted in a loss of information. A sort of weighting of the single sub-services within one main service group could partly solve the problem. Such a weighting might be

carried out within a stakeholder workshop. In the present state our levelled [0-5] and aggregated service values only provided information on service provision with limited accuracy (Table 2). This may also be with the reason why the differences between the final main service values are not very big.

Another delicate issue is the proper assessment of socio-cultural (*information*) services. Although such cultural services play an essential part in the enhancement of human welfare, they are only marginally present in the current research activities due to yet unsolved assessment difficulties (Benayas et al., 2009; Vejre et al., 2010; Gee and Burkhard, 2010). Within our Socio-Cultural Approach, the indicator development for the *information* services was highly dependent on data availability. For linear elements, we currently used only the indicator of visually relevant edges in the landscapes. This set could be enlarged by integrating data on e.g. roads with panoramic view or tree rows. Areal indicators would profit from the inclusion of data on accessibility, visibility and diversity of land cover. Unfortunately the lack of data availability limited us to the actual set.

4.2 *The concept of landscape services as an operational tool to evaluate ecologically sensitive regions.*

By applying the proposed methodological framework, users will be provided with information on the landscape without intensive new data gathering. Although the methodology has been specifically targeted to our investigation area, the framework should be applicable in other regions as well. Especially in regions, where data availability on specific landscape services is limited or incomplete (e.g. transboundary areas are often lacking homogeneous data) it would deliver a good overview of the service provision of the different landscapes. Certainly, other areas will require an adaption of the spatial reference framework as well as of the service providing units and will be also dependent on a certain extent of available data (qualifiers for the Broader Habitat Approach and indicators for the Socio-Cultural and Landform Type Approach). However, there is the possibility to use only one of the proposed approaches to generate information of selected services, depending on the research question.

The service values resulting from the presented methodology may support spatial planners by providing information on service distribution within the target region. Whereas the aggregated results at the Landform Type supply an overview on total service provision, the distribution of the individual sub-services at larger scales (landscape sample sites and LCTs) can provide a detailed insight into the service provisions of landscapes. In this way, areas of high nature conservation value or potential for demographic development can be identified. Especially, for areas with high pressure on land resources, a visualisation of the actual service provision can help policy makers and spatial planners to make informed choices.

The implementation of our assessment framework into participating sustainable landscape planning has already started within our investigation area. We tried to integrate the landscape service concept into the establishment of the nature conservation and regional development project: ‘Biosphere Reserve Neusiedler See’. According to the Statutory Framework of the World Network of Biosphere Reserves, the recognition of new and already existing Biosphere Reserves requires, among other things, the involvement and participation of a suitable range of inter alia public authorities, local communities and private interests ... (UNESCO 1996: 17, Art.4, 6). Only the involvement and participation of the resident population and all stakeholders could increase the acceptance and traceability of decisions, strengthen the identification of citizens and stakeholders with decisions, as well as increase confidence in politics and public administration (Standards der Öffentlichkeitsbeteiligung, 2008). Therefore, to promote the expansion of the territory of the already

existing Biosphere Reserve ‘Neusiedler See’ and the implementation of the Seville Strategy, representatives of important interest groups, landowners and NGOs were involved in an opinion-forming process. The stakeholders were asked to discuss and evaluate some of the related services of different types of landscapes within the region ‘Neusiedler See’, in order to identify sensitive areas, where land use pressure is very high. Preliminary results of discussions showed that the landscape service concept has potential to be the basis of a participatory approach, but still needed to be further developed to be used as a successful communication tool. In following projects an advanced version of the proposed framework is planned to be developed, in order to be applied for finding an overall concept for the Biosphere Reserve ‘Neusiedler See’.

5 Conclusions

Embedded in a spatial reference framework a good overview of the distribution of the different landscape services within the transnational study area could be provided. Additionally Hot as well as Cold spots of the individual service provisions were identified. Thus, the concept of landscape services might be a useful tool to evaluate sensitive regions and provide a basis for cross border landscape planning decisions. Applying such a spatially explicit methodology provides useful information on how spatial patterns contribute to different service provision. Whereas in most cases homogenous landscapes only emphasise on one specific landscape service, complex landscapes reveal a more diversified picture. However, for more detailed analyses further investigations, especially on scaling effects and spatial characteristics as well as interactions between the different services have to be conducted. Making landscape services spatially explicit and combining empirical data with spatial information offers an innovative approach to landscape research in the field of visualising the provision of landscape services. While most other assessment methods only focus on selected services occurring at one scale, we present a first step in the methodological development to assess and map spatial variability of landscape services.

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Tables:

Table 1: List of main landscape services and their related sub-services with examples and information about the applied assessment approach (adapted from de Groot, 2006). The three approaches ‘Broader Habitat’, ‘Socio-cultural’ and ‘Landform Approach’ are dependent on the service providing unit and are explained in more detail within the section ‘overall methodology’.

Services	Definition	Examples	Applied approach
Regulation services			
Local climate regulation	Influence of broader habitat type on local climate	Maintenance of a favourable local climate (e.g. temp., moisture etc.) for e.g. human habitation, health, cultivation	Broader habitat Approach
Disturbance prevention	Influence of broader habitat on environmental disturbances	Storm protection and/or flood prevention (e.g. flood detention basin, shelter belt)	Broader habitat Approach
Water regulation	Role of broader habitat type in regulating runoff and river discharge	Drainage and natural irrigation	Broader habitat Approach
Water supply	Filtering, retention and storage of fresh water	Provision of water for consumptive use (e.g. drinking, irrigation and industrial use)	Broader habitat Approach
Soil retention	Role of vegetation root matrix and soil biota in soil retention	Maintenance of arable land; prevention of damage from erosion/siltation	Broader habitat Approach
Soil formation	Weathering of rock, accumulation of organic matter	Maintenance of natural productive soils	Broader habitat Approach
Nutrient regulation	Role of biota in storage (buffer) and recycling of nutrients (e.g. N, P and S)	Maintenance of healthy and productive ecosystems	Broader habitat Approach
Pollination	Role of biota in movement of floral gametes (is there any suitable habitat available for pollinators?)	Pollination of wild plant species and crops	Broader habitat Approach
Habitat services			
Refugium	Suitable living space for wild plants and animals	Maintenance of biodiversity, in particular	Broader habitat Approach
Nursery	Suitable reproduction habitat	Maintenance of commercially harvested species	Broader habitat Approach
Provision services			
Food	Conversion of solar energy into wild edible plants and animals	Maintenance of edible wild plants and fungi (not cultivated), game and fish	Broader habitat Approach
Raw materials	Conversion of solar energy into biomass	Material for human constructions (building and manufacturing), like lumber, fuel and energy wood	Broader habitat Approach
Genetic resources	Genetic material and evolution in wild plants and animals	Improve crop resistance to pathogens and pests and maintenance of old cultivated plants	Broader habitat Approach
Medicinal resources	Variety in chemical substances in natural biota	Drugs and pharmaceuticals	Broader habitat Approach
Information services			
Aesthetic information	Attractive landscape features and views	Enjoyment of scenery (scenic roads, housing etc.	Socio-cultural Approach
Recreation	Variety in landscapes with (potential) recreational uses	Travel to natural ecosystems for eco-tourism and (recreational) nature study	Socio-cultural Approach
Cultural and artistic	Variety in natural and cultural	Use of natural and cultural landscape	Socio-cultural

information	features with cultural and artistic value	elements as motive in books, film, painting, folklore, national symbols, architect, advertising, etc	Approach
Spiritual and historic information	Variety in natural and cultural features with spiritual and historic value	Use of natural and cultural landscape elements for religious or historic purposes (i.e. heritage value of natural ecosystems and features)	Socio-cultural Approach
Science and education	Variety in nature with scientific and educational value	Use of nature for scientific research	Socio-cultural Approach
Carrier services			
Habitation	Providing suitable space for human living	Living space (ranging from small settlements to urban areas)	Landform Approach
Cultivation	Providing suitable substrate for cultivation (actual available)	Cultivated food and fodder	Landform Approach
Energy conversion	Providing suitable substrate or medium for energy conversion	Energy facilities (only wind)	Landform Approach
Waste disposal	Providing suitable substrate for waste disposal	Space for solid waste disposal	Broader habitat Approach
Transportation	Providing suitable substrate or medium for transportation	Main and side roads as well as railroad tracks	Landform Approach
Tourism-facilities	Providing space and facilities for human activities related to tourism	Tourism and leisure activities (e.g. outdoor sports)	Socio-cultural Approach

Table 2: Aggregated service values of *carrier, information, habitat, regulation and provision* services for the seven Landform Types. The values can range between 0 (no service provision) to 5 (very high service provision).

LFT	Carrier	Information	Habitat	Regulation	Provision
Lake basin	1.15	2.64	3.53	2.66	1.80
Marshland	0.88	1.74	2.80	3.06	1.73
River floodplain	1.51	1.90	1.62	1.77	0.89
Low lying terrace	1.77	1.73	1.67	1.78	0.55
Elevated terrace	1.97	1.87	1.75	1.78	0.58
Hilly area and hill range	1.65	2.59	2.93	2.99	2.45
Low and middle range mountain	0.81	1.90	4.45	4.45	4.12

Appendices

Table Appendix 1: Matrix for the assessment of the 181 Broader Habitat types' capacities to provide different landscape services: Whereas on the x-axis the landscape services were displayed, the BHTs were placed on the y-axis. At the intersections, different BHTs' capacities to provide a certain landscape service were assigned. 0 = no service provision (orange colour), 1=low service provision (light green), 2 = modest service provision (grass-green), 3 = medium service provision (olive-green), 4 = high service provision (verdant-green), 5 = very high service provision (dark green). '*' indicates all BHTs that were used within the Austrian – Hungarian case study.

	Regulation services									Provision services			Carrier services			Habitat services					
	Local climate regulation	Disturbance prevention	Water regulation	Water supply	Soil retention	Soil formation	Nutrient regulation	Pollination		Refugium	Nursery	Food	Raw materials	Genetic resources	Medicinal resources	Carrier services	Habitation	Cultivation	Waste disposal	Transportation	
Broader Habitat Types (BHT)																					
Agriculture																					
Grain fields intensive *	0	0	2	0	0	1	1	0		1	0		0	0	0	0	0	5	0	0	
Grain fields medium intensive *	0	0	2	0		1	2	1	0	1	0		0	0	0	0	0	4	0	0	
Grain fields extensive	0	1	2	1		2	3	2	2	2	1		0	0	1	1	0	3	0	0	
Forage crops	0	2	3	1		2	3	1	3	1	0		0	0	0	0	0	5	0	0	
Root crop extensive	0	1	2	1		3	3	2	1	1	1		0	0	1	1	0	3	0	0	
Root crop intensive *	0	0	1	0	0	1	1	0		1	0		0	0	0	0	0	5	0	0	
Root crop medium intensive *	0	0	2	0	0	2	1	0		1	0		0	0	0	0	0	4	0	0	
Special types of fields *	0	1	2	1		2	2	1	2	1	0		0	0	0	0	0	5	0	0	
Intensive vineyard *	0	1	3	2		3	4	1	0	1	0		1	1	0	0	0	5	0	0	
Medium intensive vineyard *	0	1	3	2		3	4	1	1	2	0		1	1	0	0	0	4	0	0	
Extensively managed vineyards on limestone bedrock *	0	2	3	2		2	4	2	2	3	1		1	1	3	1	0	3	0	0	
Extensively managed vineyards on siliceous bedrock *	0	2	3	2		3	4	2	2	3	1		1	1	3	1	0	3	0	0	
Hops plantation	0	2	3	2		2	3	2	1	1	0		0	0	0	3	0	5	0	0	
Fruit plantation *	3	2	3	2		4	3	3	1	2	0		1	5	0	0	0	5	0	0	
Orchard, coppice culture *	0	2	2	2		3	3	2	3	1	0		1	1	0	0	0	5	0	0	
Vegetable garden	0	1	2	1		3	3	1	2	2	1		0	0	3	3	0	5	1	0	
Farmed grassland																					
Intensive meadow *	0	2	3	2		3	2	2	2	1	0		0	0	0	0	0	5	0	0	
Medium intensive meadow *	0	3	3	2		3	4	3	3	3	2		2	0	1	1	0	4	0	0	
Extensive meadow *	0	3	3	2		3	4	3	4	5	3		2	0	3	2	0	3	0	0	
Dry meadows on limestone bedrock *	0	2	2	1		2	1	3	3	5	5		1	0	1	1	0	0	0	0	
Dry meadows on siliceous bedrock	0	1	2	1		2	1	3	3	5	5		1	0	1	1	0	0	0	0	
Dwarf shrub heath	0	2	3	2		3	4	3	2	4	2		1	0	0	1	0	0	0	0	
Intensive Pasture *	0	2	3	2		2	3	2	2	2	0		0	0	0	0	0	5	0	0	
Medium intensive pasture *	0	3	3	2		3	4	3	3	3	2		1	0	1	0	0	4	0	0	
Extensive pasture *	0	3	3	2		3	4	3	4	5	3		1	0	3	1	0	3	0	0	
Old tree pastures with site specific composition of trees	1	3	4	3		4	4	3	3	4	3		1	1	1	1	0	2	0	0	
Young tree pastures with site specific composition of trees	0	2	3	2		4	4	2	2	2	1		1	0	0	1	0	2	0	0	

	Regulation services								Provision services		Carrier services				Habitat services			
	Local climate regulation	Disturbance prevention	Water regulation	Water supply	Soil retention	Soil formation	Nutrient regulation	Pollination	Refugium	Nursery	Food	Raw materials	Genetic resources	Medicinal resources	Habitation	Cultivation	Waste disposal	Transportation
Broader Habitat Types (BHT)																		
Farmed grassland																		
Old orchard meadows *	1	3	4	3	4	4	3	5	4	3	1	2	4	2	0	3	0	0
Young orchard meadows *	0	2	3	2	3	4	2	4	2	2	1	0	4	2	0	2	0	0
Intensively cultivated fruit tree plantation *	0	2	3	2	3	3	2	3	1	0	1	1	0	0	0	5	0	0
Other old grown tree pastures	1	3	4	3	4	4	3	3	4	3	1	1	1	1	0	2	0	0
Other young grown tree pastures	0	2	3	2	4	4	2	2	2	1	1	0	0	1	0	2	0	0
Other tree meadows, old *	1	3	4	3	4	4	3	4	4	3	1	1	2	2	0	2	0	0
Other tree meadows, young *	0	2	3	2	3	4	2	3	2	2	1	0	2	2	0	2	0	0
Gardens, parks *	2	4	3	3	4	4	2	4	3	2	1	1	0	2	0	0	0	0
Unfarmed grassland																		
Young fallow land *	0	2	3	2	3	3	1	2	2	0	1	0	0	2	0	0	0	0
Old fallow with tall herbs *	1	3	4	2	3	3	2	2	2	3	1	0	1	2	0	0	0	0
Old fallow land with shrubs *	2	4	4	2	4	4	3	3	3	3	3	2	1	2	0	0	0	0
Clearing, undifferentiated *	0	3	3	2	3	3	2	4	3	1	2	0	2	2	0	0	0	0
Large sedge stands *	2	4	4	4	3	5	4	1	4	2	1	1	1	0	0	0	0	0
Small sedge stands *	1	3	4	4	3	5	4	1	4	1	0	0	0	0	0	0	0	0
Salt marshes and salt dominated dry grasslands	0	2	3	0	3	3	3	2	5	2	1	0	0	0	0	0	0	0
Small semi-natural features																		
Ruderal areas covered with pioneer vegetation *	0	3	3	2	2	3	2	2	2	2	1	0	1	3	0	0	0	0
Ruderal areas sparsely covered with pioneer vegetation *	0	2	2	1	2	3	1	2	2	1	1	0	0	3	0	0	0	0
Ruderal field margin *	0	1	3	2	2	4	2	2	2	1	1	0	0	3	0	0	0	0
Field margins rich on perennial plants *	0	1	3	2	3	4	2	2	2	2	2	0	0	2	0	0	0	0
Tall herbs and Ruderalia *	0	2	3	2	4	4	3	2	2	2	1	0	0	1	0	0	0	0
Hedgerow of shrubs *	2	3	3	2	5	4	4	3	3	3	2	2	1	1	0	0	0	0
Hedgerow of trees *	3	4	4	3	5	4	4	3	3	3	2	2	1	1	0	0	0	0
Solitary shrub and coppice *	0	2	2	1	4	3	1	2	3	2	2	1	0	1	0	0	0	0
Old deciduous tree *	1	1	1	1	4	4	1	2	1	2	0	1	0	0	0	0	0	0
Young deciduous tree *	0	2	2	1	4	4	1	2	3	1	0	0	0	0	0	0	0	0
Old coniferous tree *	1	1	1	1	4	4	1	1	1	2	0	1	0	0	0	0	0	0
Young coniferous tree *	0	2	2	1	4	4	1	1	3	1	0	0	0	0	0	0	0	0
Old fruit tree *	1	2	2	1	4	4	1	4	3	2	1	1	3	0	0	0	0	0
Young fruit tree *	0	1	1	1	4	4	1	3	1	1	1	0	3	0	0	0	0	0
Copse composed of alien tree species *	4	3	4	3	5	4	4	3	2	2	2	4	0	1	0	0	0	0
Copse with site specific composition of deciduous trees *	4	3	4	3	5	5	4	3	4	3	2	4	1	1	0	0	0	0
Copse with site specific composition of coniferous trees	4	3	4	3	5	5	4	1	4	3	2	4	1	1	0	0	0	0

[illegible]

Broader Habitat Types (BHT)	Regulation services								Habitat services		Provision services		Carrier services				Cultivation				Transportation			
	Local climate regulation	Disturbance prevention	Water regulation	Water supply	Soil retention	Soil formation	Nutrient regulation	Pollination					Medicinal resources	Genetic resources	Raw materials	Food								
Forests																								
Seminatural riparian forest dominated by deciduous trees *	3	4	5	4	5	5	5	2			3	2		2	3	1	2			0	0	0	0	
Seminatural mixed oak and hornbeam-oak forests *	5	5	5	3	5	5	5	2			4	4		3	5	1	3			0	0	0	0	
Seminatural riparian forest dominated by alder trees *	3	4	4	4	5	5	5	2			3	2		2	3	1	2			0	0	0	0	
Seminatural deciduous forests dominated by ash – and maple trees *	5	5	5	3	5	5	5	3			4	4		2	3	1	3			0	0	0	0	
Seminatural spruce- and Spruce-fir forests	5	5	5	3	5	5	5	2			4	3		3	5	1	3			0	0	0	0	
Seminatural pine forests	5	5	4	3	5	5	5	2			4	3		2	5	1	2			0	0	0	0	
Semiantural speckled Alder forests	5	5	5	4	5	5	5	2			4	3		2	3	1	2			0	0	0	0	
Seminatural hop-beam forests	5	5	5	3	5	5	5	2			4	4		3	4	1	3			0	0	0	0	
Seminatural mixed forests, rich on chestnut-trees	5	5	5	3	5	5	5	3			4	4		4	3	1	3			0	0	0	0	
Seminatural larch- and Larch-Arolla Pine forests	5	5	5	3	5	5	5	2			4	3		3	5	1	3			0	0	0	0	
Seminatural riparian forest dominated by willows *	3	4	4	4	5	5	5	2			3	2		2	3	1	2			0	0	0	0	
Natural hardwood floodplain forests	5	5	5	3	5	5	5	3			5	5		3	4	2	3			0	0	0	0	
Natural floodplain forests, dominated by coniferous trees	5	5	5	3	5	4	5	2			5	4		2	4	2	2			0	0	0	0	
Natural softwood floodplain forests *	5	5	5	4	5	5	5	3			5	5		3	4	2	3			0	0	0	0	
Natural alluvial forest dominated by shrubby willows *	5	4	5	4	5	5	5	3			5	4		2	4	2	2			0	0	0	0	
Natural beech forests and spruce-fir-beech forests	5	5	5	3	5	5	5	2			5	5		3	5	2	3			0	0	0	0	
Natural riparian forest dominated by deciduous trees	3	4	4	4	5	5	5	2			4	3		2	3	2	2			0	0	0	0	
Natural mixed oak and hornbeam-oak forests	5	5	5	3	5	5	5	2			5	5		3	5	2	3			0	0	0	0	
Natural oak forests *	5	5	5	3	5	5	5	2			5	5		3	4	2	3			0	0	0	0	
Natural riparian forest dominated by alder trees *	3	4	4	4	5	5	5	2			4	3		2	3	2	2			0	0	0	0	
Natural deciduous forests dominated by ash – and maple trees	5	5	5	3	5	5	5	3			5	5		2	3	2	3			0	0	0	0	
Natural spruce- and Spruce-fir forests	5	5	5	3	5	5	5	2			5	4		3	5	2	3			0	0	0	0	
Natural pine forests	5	5	4	3	5	5	5	2			5	2		2	5	2	2			0	0	0	0	
Semiantural speckled Alder forests	5	5	5	4	5	5	5	2			5	5		2	3	2	2			0	0	0	0	

Broader Habitat Types (BHT)	Regulation services				Habitat services				Provision services		Carrier services				Transportation	Waste disposal	Cultivation	Habitation
	Local climate regulation	Disturbance prevention	Water regulation	Water supply	Soil retention	Soil formation	Nutrient regulation	Pollination	Refugium	Nursery	Food	Raw materials	Genetic resources	Medicinal resources				
Forests																		
Natural hop-beam forests	5	5	5	3	5	5	5	2	5	5	3	4	2	3		0	0	0
Natural mixed forests, rich on chestnut-trees	5	5	5	3	5	5	5	3	5	5	4	3	2	3		0	0	0
Natural larch- and Larch-Arolla Pine forests	5	5	5	3	5	5	5	2	5	4	3	5	2	3		0	0	0
Other natural riparian forests	3	4	4	4	5	5	5	2	4	3	2	4	2	2		0	0	0
Natural riparian forest dominated by willows *	3	4	4	4	5	5	5	3	4	3	2	4	2	2		0	0	0
Not natural riparian forests	3	4	4	4	5	5	5	1	2	1	2	2	0	2		0	0	0
Mire forests and forests on the edge of mires	5	5	5	4	5	5	5	2	5	0	2	3	0	2		0	0	0
Mixed scree forests and forests covering steep faces	5	5	5	3	5	5	5	2	5	2	2	2	0	3		0	0	0
Wet woodlands and peat swamps	5	5	5	4	4	5	5	2	5	5	2	2	0	2		0	0	0

Table Appendix 2: List of 131 qualifiers and their influence on service provision: positive ('1') (olive-green), negative ('-1') (red) or no influence ('0') (orange) on the provision of a specific service

[illegible]

[illegible]

	Regulation services								Habitat services		Provision services				Carrier services				
	Local climate regulation	Disturbance prevention	Water regulation	Water supply	Soil_retention	Soil_formation	Nutrient_regulation	Pollination	Refugium	Nursery	Food	Raw materials	Genetic resources	Medicinal resources	Habitation	Cultivation	Energy Conversion	Waste disposal	Transportation
QUALIFIER characterising the Broader Habitats in more detail																			
Threats																			
Overgrazing	0	0	0	0	-1	-1	-1	-1	0	-1	0	0	0	0	0	-1	0	0	0
Browsing by game	0	0	0	0	0	0	0	0	0	-1	-1	0	0	-1	0	-1	0	0	0
Bush encroachment	0	0	0	0	0	0	0	0	0	-1	1	0	0	1	0	-1	0	0	0
Large clear-cuttings	-1	-1	-1	-1	-1	0	-1	-1	-1	-1	-1	-1	-1	-1	0	0	0	0	0
Improperly forest thinning	-1	-1	0	0	0	0	0	0	-1	-1	0	0	0	0	0	0	0	0	0
Burning	-1	-1	0	-1	-1	0	-1	0	-1	-1	0	0	0	0	0	0	0	0	0
Drainage	-1	-1	-1	-1	0	-1	-1	0	-1	0	0	0	0	0	0	0	0	0	0
Clearing of Waterbodies	0	-1	0	0	0	0	0	0	-1	-1	-1	0	0	0	0	0	0	0	0
Desiccation	-1	0	0	0	0	0	0	0	0	-1	0	0	0	0	0	-1	0	0	0
Eutrophication	0	0	0	0	0	0	-1	0	-1	0	-1	0	0	0	0	1	0	0	0
Input of Biocides	0	0	0	0	0	-1	0	-1	-1	-1	-1	0	-1	-1	0	1	0	0	0
Sewages, Contamination of drinking water	0	0	0	-1	0	0	0	0	0	0	-1	0	0	0	0	0	0	0	0
Erosion	0	-1	-1	0	-1	-1	-1	0	0	0	0	0	0	0	0	-1	0	0	-1
Anthropogenic disturbance (e.g. tourism)	0	0	0	0	0	0	0	0	-1	-1	0	0	0	0	0	0	0	0	0
Water withdrawal	-1	0	-1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Intensification	0	0	0	0	0	0	0	0	-1	-1	0	0	-1	0	0	1	0	0	0
Invasion of alien species	0	0	0	0	0	0	0	0	-1	-1	0	0	-1	0	0	-1	0	0	0
Planting out alienus trees	0	0	0	0	0	0	0	0	-1	-1	0	0	-1	0	0	0	0	0	0
Underutilization	0	0	0	0	0	0	0	0	-1	0	0	0	0	0	0	-1	0	0	0
Area dominated by Neophytes (cover >60 %)	0	0	0	0	0	0	0	0	-1	-1	0	0	-1	0	0	0	0	0	0

Table Appendix 3: Assessment matrix for the *information* services illustrating the indicators' capacities to provide 'aesthetic information', 'cultural and artistic information', 'spiritual and historic information', 'science and education' and 'recreation'. The different indicators are grouped into point-, linear- and aerial elements.

Indicator groups		Indicators	S1	S2	S3	S4	S5	Data source
Point elements	Density of sacral buildings							
		churches	5	5	5	3	4	topographic (1) and tourist maps (2), landscape values data base (3)
		chapels	4	4	4	2	3	
		crucifixes	3	4	4	1	1	
		statues (sacral/non sacral)	3	4	4	2	1	topographic map
		cemeteries	4	4	5	1	1	
	Density of non-sacral buildings							
		castles	5	5	5	4	5	tourist map
		vinehouse, cave	4	4	3	3	5	tourist map, expert knowledge
		look out tower	5	2	2	4	5	tourist map
		archaeological sites	3	5	5	5	3	landscape values database
		museums	3	5	5	5	4	tourist map
Linear elements		study trails	4	2	2	5	4	tourist map
		research (visitor) centers	3	2	2	5	4	expert knowledge
	Visually relevant edges							
		any forest edge	5	4	1	2	5	satellite image
Areal elements		any water edge	5	4	1	2	5	
		any vineyard (permanent) crop egde	5	4	2	1	5	
	Land cover							
		water bodies	5	3	4	5	5	satellite image
		wetlands	4	3	5	5	2	
		forests	5	4	3	5	4	
		natural grassland	4	3	5	5	4	
		arable land	2	2	2	2	1	
		permanent crops	3	2	2	2	3	
		pastures	4	3	3	3	3	
		heterogenous agriculture	4	4	3	3	3	
		scrubs	2	2	2	2	1	

References:

- (1) Fertő, Lajta-hegység, Hanság. Turista-, kerékpáros és szabadidőtérkép. Wanderkarte mit Rad und Freizeitthematik. 1:80 000, Szarvas Kiadó
- (2) Collection of Landscape values TEKA <http://tajertektar.hu/hu/kereso>. Institute of Geodesy, Cartography and Remote Sensing
- (3) Topographic maps 1:100 000 EOVI (Uniform National Projection).

Legend

S1	Aesthetic information
S2	Cultural and artistic information
S3	Spiritual and historic information
S4	Science and education
S5	Recreation

Article 5

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Own contribution:

Study design 70%, implementation 35%, writing 35%

Services landscapes provide to society – regional case studies in the project regions North, Central North and Central South

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Keywords: landscape services, regional studies, capacity matrix, mapping

Abstract

European biodiversity and ecosystem service decline is still on-going, caused mainly by a high level of habitat loss and fragmentation. Despite a multiplicity of conservation activities, valuable ecological networks are getting lost. To enhance connectivity in order to preserve ecological as well as socio cultural values the status quo of transnational networks has to be evaluated. Therefore, the project TransEcoNet aimed at assessing the actual goods and services that landscapes provide for sustaining life. Within this project we developed an integrated approach for mapping and assessing landscape services based on spatial information as well as field data. In order to accomplish a comprehensive analysis, different levels of service assessment regarding investigation area and spatial scale are distinguished. Whereas the case studies in the project regions North and Central North based their assessment only on the use of a capacity matrix, those in the project region Central South extended their assessments by semi-quantitative data gained from field work. We successfully mapped and quantified five landscape service groups including a range of individual services: *regulation*, *habitat*, *provision*, *information* and *carrier* within the investigation areas. While the main service distributions were visualized as spider-web diagrams within each investigation area, the individual services delivery was displayed as maps. The final landscape service maps may provide regional stakeholders with valuable information on the service provision of transnational ecological networks and can therefore be used as a decision tool in landscape planning processes.

1 Introduction

Nowadays European cultural landscapes are characterised by a high level of anthropogenic fragmentation and habitat loss which are known as major reasons for the decline of biodiversity in industrialised countries. Countering this development by enhancing connectivity requires an evaluation of the status quo and trends of ecological valuable landscapes. One promising possibility to provide the knowledge basis to meet the needs of a sustainable development and management of transnational ecological networks can be the concept of ecosystem functions, goods and services (MEA, 2005). This scientific concept has experienced increasing attention in the last years as it provides the means of documenting the importance and benefits of ecosystems and landscape for human society. As local people define their environment more as a “landscape” than as an “ecosystem” we preferred the term “landscape services” as a specification of ecosystem services (Termorshuizen & Opdam, 2009). Within our assessment we include potentials, materials and processes of nature (e.g. biomass, raw materials, primary productivity, etc.) and services of cultural elements and constructions of human origin (e.g. buildings, settlements, infrastructure, etc.).

The main objective of this study was to implement the concept of landscape services on different spatial scales in selected transboundary areas along the ecological networks of the project regions North, Central North and Central South. We aimed at identifying the hot and cold spots of ecological as well as socio-cultural services.

2 Overall methodology

In our project the classification of landscape services is mainly adapted from de Groot (2006) and was grouped into five main categories including 25 sub-services: *regulation* (local climate regulation, disturbance prevention, water regulation, water supply, soil retention, soil formation, nutrient regulation, pollination), *habitat* (refugium, nursery), *provision* (food, raw materials, genetic resources, medicinal resources), *information* (aesthetic, recreation, cultural and artistic, spiritual and historic information, science and education) and *carrier* (habitation, cultivation, energy conversion, waste disposal, transportation, tourism facilities) services. Within this project we developed a methodological framework for the assessment and mapping of landscape services based on spatial information as well as field data. In order to accomplish a comprehensive analysis, different levels of service assessment considering location and spatial extent were distinguished within the different investigation areas. As service providing units we used the broader habitat types (BHT) (Bunce et al. 2009, 2011). BHTs were linked by expert knowledge about the different BHTs' capacities to provide various landscape services. The so called Broader Habitat Approach is based on a capacity matrix (Burkhard et al. 2009) and focused on the assessment of landscape services directly at the landscape element scale. Whereas on the x-axis selected landscape sub-services were placed, on the y-axis the BHTs were situated. At the intersections, different BHTs' capacities to provide landscape sub services were assigned. The so-called broader habitat (type) value (BHV) ranged from 0 to 5. The higher the value, the higher was the general relationship between BHT and service.

0 = no relevant link between BHT and specific service; 1 = low relevant link, 2 = relevant link, 3 = medium relevant link, 4 = high relevant link, 5 = very high relevant link

For the project region Central South these BHVs were additionally extended by habitat's specific attributes, so called qualifiers coming from a field survey. In further steps data were aggregated to the main service categories and extrapolated to different spatial levels within the investigation areas. Because of different data availability and specific study area characteristics within the three case studies (North, Central North and Central South) slightly different approaches were used to assess the broad set of landscape services. The individual methods and results are described in the following case study sections.

3 Project Region North

The investigation area Ore Mountains (Erzgebirge/ Krušné hory) and Elbe Sandstone Mountains border region (150 km²) was selected for the assessment of landscape services in the Project Region North. A more detailed description of the investigation area can be found in the first chapter of this section.

3.1 Methodology

The whole investigation area was overlaid by a regular raster dividing the surface into squares. We used the official European Grid system (INSPIRE, 2009). In this study, we used a basic grid size of 2x2 km (Figure 1). In total, 22 raster cells were analysed, including only raster cells which were situated completely within the investigation area. Altogether 7 raster cells are totally located in Germany, 6 are totally within the territory of the Czech Republic and 9 cells are transboundary with various percentages of area in Germany and the Czech Republic.

For the assessment of all of the 19 landscape services (because of lack of data among the main services *information* only "science and education" and among *carrier* only "habitation", "cultivation", "transportation" and "waste disposal") we applied the Broader Habitat Approach within the investigation area Erzgebirge and Sandstone mountains border region. Therefore a capacity matrix for the as-

assessment of the different links between the broader habitat types and the related services was developed. For the German part of the investigation area detailed land use and biotope type data from the free state of Saxony from 2005 (BTLNK 2005) and for the Czech part the ZABAGED 2007 data (Fundamental Base of Geographic Data/Základní báze geografických dat, scale 1:10 000) were used to identify the broader habitats.

To obtain a single value for a given sub-service in each raster cell area weighted mean of all BHVs ascribed to landscape elements was calculated. Then the mean of respective service values of all raster cells within a thematic raster group (German, Czech and Transboundary) was calculated to derive a single value of the specific service for a raster group. To get main values for the primary services' categories, a mean was calculated for related services for the whole investigation area separated for the German, Czech and Transboundary part.

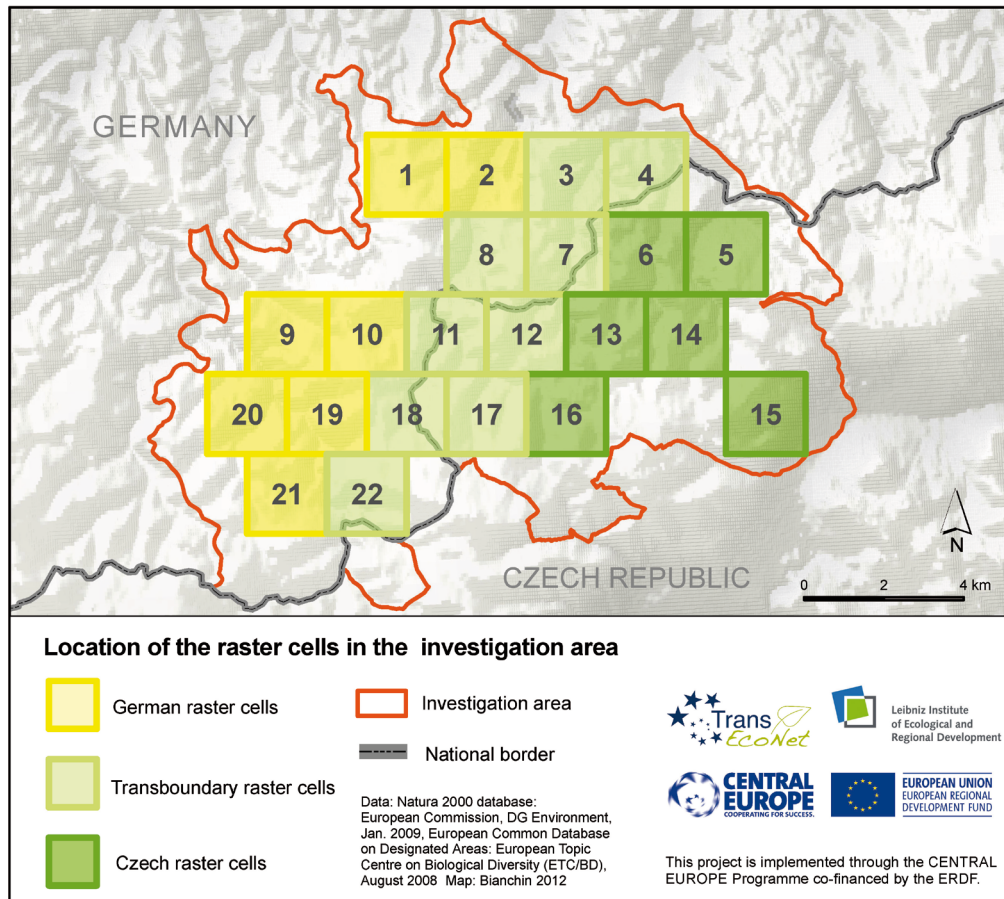


Figure 1: Location of the raster cells in the investigation area Ore Mountains (Erzgebirge/ Krušné hory) and Elbe Sandstone Mountains border region

3.2 Results

The values of the main service categories for the whole investigation area are visualised by a spider-web graph (Figure 2). The highest service provision occurred within the *regulation* service group mainly in the Czech part of the investigation area, in particular where the percentage of broad-leaved and mixed forest is large (Figure 2). The lowest value for the *regulation* service was found in the areas where the settlements and the agricultural fields are located, which is mostly in the north- and south-western area of the German part. In the transboundary raster cells the results of the service assessment were different from the results in the German and Czech part. Overall, the Czech part showed higher service provision in nearly all mean services except for the *carrier* service.

The land use of the German and Czech part differs substantially. The land use in the German part is a mixture of natural grassland, agricultural use and mixed forests. Artificial surfaces only have a very low

percentage. The Czech part is dominated by forest (broad leaved forest and mixed forest) as well as transitional woodland shrubs and natural grassland. The percentage of arable land is negligible as well as the percentage of artificial surfaces. The land use in the transboundary part is dominated by natural grassland, mixed forest and arable land. As the service analysis was mainly based on land use categories the service provision differed as well. An example of the spatial distribution of the single service soil formation (included in the main service *regulation*) is shown in Figure 3.

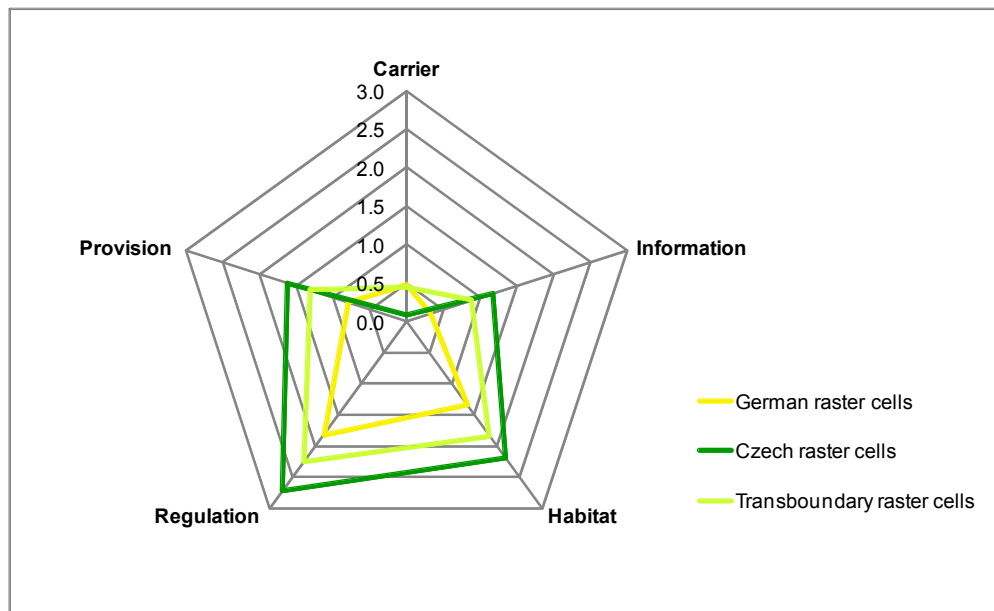


Figure 2: Allocation of landscape services in the investigation area Ore Mountains (Erzgebirge/ Krušné hory) and Elbe Sandstone Mountains border region

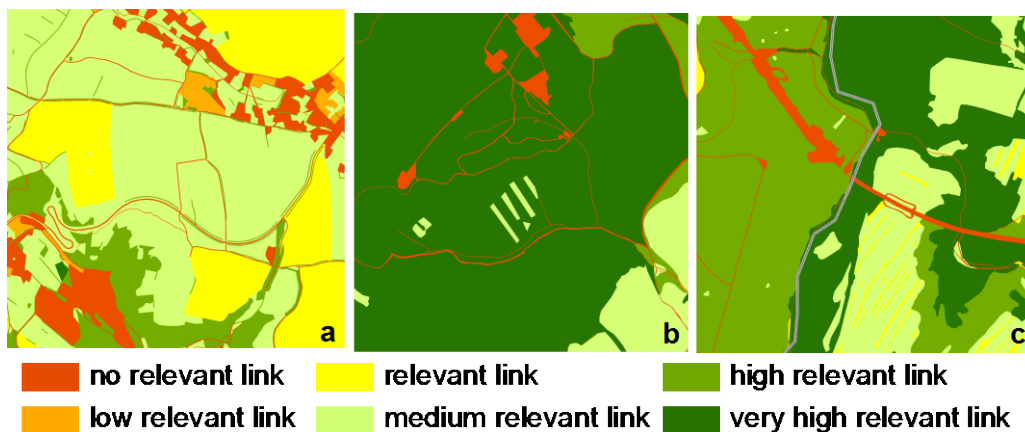


Figure 3: Spatial distribution of the service 'soil formation' within different raster cells, Germany (a), Czech (b) and transboundary (c). The different colours display the relevant links between the service 'soil formation' and the broader habitat types.

4 Project region Central North

The case studies of the project region Central North were grouped into three landscape types (LT) – alluvial forested LT, hilly agricultural LT and upland meadow forested LT. Their more detailed characteristics as well as spatial distribution is outlined in the first chapter of this section (*'Investigation areas*

used for the assessment of landscape functionality and landscape services in transboundary ecological networks').

4.1 Methodology

In total, 19 landscape services based on de Groot (2006) were assessed. Due to the lack of data among the *information* and the *carrier services* only the relevant sub-services "science and education" and "habitation", "cultivation", "transportation" and "waste disposal" could be analysed.

The assessment of landscape services was based on the Broader Habitat Approach, applying a matrix of broader habitats and their related services. The broader habitats were represented by 83 landscape elements distinguished on the basis of land cover classes and their ecological quality. The delimitation of the landscape elements is in more detail described in the chapter '*Ecological stability and landscape functionality assessment in Moravia*'.

The mean of all BHVs ascribed to landscape elements was calculated to obtain a single value for a given service in each case study. The mean of the respective service within these case studies was calculated to derive a value of the specific service for an individual landscape type. To get main values for the primary services' categories, a mean was calculated for related services on the LT level.

4.2 Results

The results of the landscape services assessment for all landscape types were visualised by a spider-web graph (Figure 4). The spatial distribution of the sub-service soil formation is shown as an example in Figure 5. Despite the different landscape characters of the selected landscape types, the spider-web values did not significantly differ from each other.

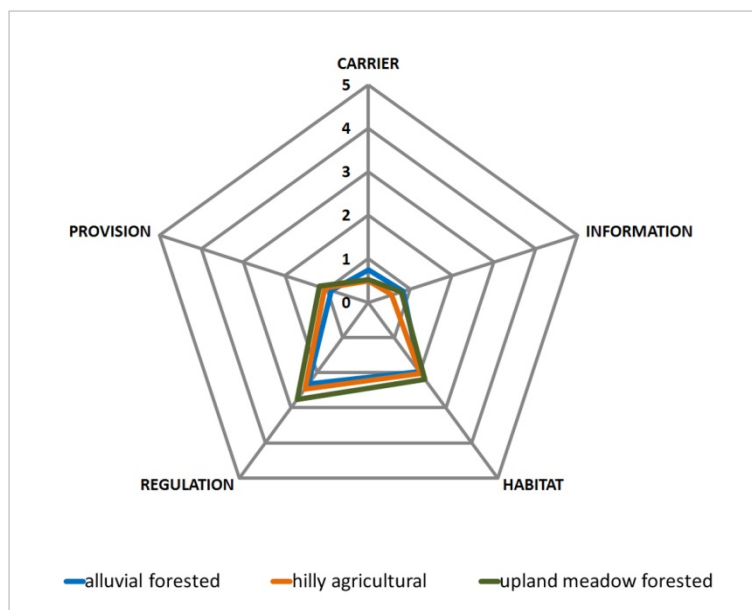


Figure 4: Allocation of landscape services in the landscape types: alluvial forested, hilly agricultural and upland meadow forested

Regulation and *habitat* services dominated in all landscape types. The highest values for these two services as well as for the *provision* service were found within the upland meadow forested LT. This is due to the vegetation types 'predominant oak-hornbeam', 'beech forests' and 'herb-rich meadows' which have higher capacities to deliver the sub-services: local climate regulation, disturbance prevention, soil formation, soil retention, nutrient regulation, raw materials, food and nursery.

The Hilly agricultural LT displayed higher values for soil formation, soil retention as well as water regulation, nutrient regulation (*regulation* services), refugium (*habitat* services) and cultivation (*carrier* ser-

vices. These results reflected both rather intensively used rural areas and more valuable landscape elements which were surrounded by these areas.

The highest values of *carrier* services calculated for the alluvial forested LT were caused by concentration of interconnected settlements in the river valleys. This was typical, in particular for the case studies Ostravice and Morávka. On the other hand, remaining floodplain forests together with wet grassland, especially in the Dyje case study, provided higher *regulation* as well as *habitat* services.

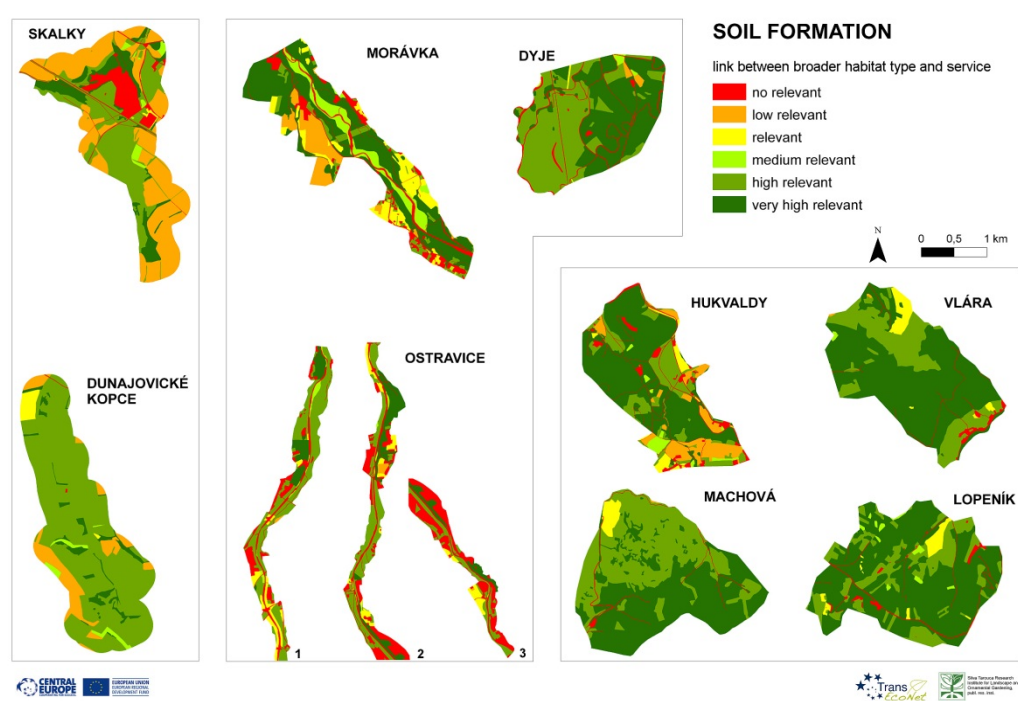


Figure 5: Spatial distribution of the service 'soil formation' in hilly agricultural LT (left), alluvial forested LT (middle) and upland meadow forested LT (right)

5 Project region Central South

The investigation area Central South covers more than 2000 km² in the cross-border region of Austria and Hungary. A more detailed description of that area can be found in the first chapter of this section.

5.1 Methodology

We assessed the five main services with all of the 25 sub-services embedded in a spatial reference framework. Therefore the investigation area was divided into seven Landform Types (LFT) representing the main geomorphological features of the study area: 'Lake Basin' (LFT 1), 'Marshlands' (LFT 2), 'River Floodplains' (LFT 3), 'Low lying terrace' (LFT 4), 'Elevated terrace' (LFT 5), 'Hilly area and hill range' (LFT 7) and 'Low and middle range mountains' (LFT 8) and 14 Landscape Character Types (LCT) illustrating a combination of relief, dominant land cover and land use intensity in the landscapes (Konkoly-Gyuró et al., 2010; after Swanwick, 2002). Driven by the link between landscape service and service providing unit, three different approaches of landscape service assessment, in terms of location and spatial scale, were distinguished:

- (i) Broader Habitat Approach, which is based on a capacity matrix linking 180 Austrian broader habitat types (Essl et al., 2002; Pascher et al., 2010) to the respective services. The capacity values of broader habitats to provide services derived from first expert evaluations that were extended by landscape element specific attributes, such as qualifiers on biotope structure, management, pressure and valuable attributes coming from a field survey. The resulting values were called "landscape element service value" (LESV). Thus, a locally

explicit landscape service evaluation was possible. For this approach, we needed to sample landscape sites. This was done by a stratified sampling method (see Kuttner et al. this section). In total, 41 landscape samples, each of 1x1 km size were selected.

- (ii) Socio-cultural Approach, which is based on different landscape data, such as landscape metrics, biophysical and socio-economic landscape components that were translated into spatial indicators and linked to the related services. As spatial units we used the 14 LCTs.
- (iii) Landform Approach: Landscape services were assessed at the Landform Type scale. They could be observed directly either by the use of Corine land cover or by clearly identifiable spatial indicators.

Based on the spatial reference framework the individual services assessed at the different scales could be aggregated to the main service categories at the Landform Type scale (for more detail regarding the aggregations within the different approaches see Hermann et al. submitted). To receive one single value for each service that was assessed within the Broader Habitat Approach, per sample site area-weighted mean values of all LESVs were separately calculated for the services. These values were extrapolated to LFT-level by calculating mean values for all sample sites per LFT. Consequently, the main service values were obtained by combining and calculating the mean values on LFT level.

5.2 Results

Three selected Landform Types were characterised concerning the allocation of the different main landscape services. The results were visualised by a spider web diagram. The figure represent the high diversity within the investigation area ranging from the natural and semi natural areas such as the remaining marshland and flood plains over the extensive used low and middle range mountains to the intensive agricultural regions in the elevated terraces (Figure 6).

The marshlands have a high capacity to provide *habitat* and *regulation* services (Figure 6). Lakes and some patches of wetland have remained in the deepest areas within the mosaic of forest and grassland supplying a range of *regulation* services. Forests are dominated by poplar plantation but also by some remains of the original *Alnus glutinosa* “marsh forest” with high nature value providing refugium and nursery habitat for wild animals. In comparison to the former services, the *information* and *provision* services reached lower values. This might be based on the fact that the former wetlands, covered by peaty soils, adjacent to the lake on the south-east are today dissected by artificial channel networks. The very low value of *carrier* services may be explained by the contradictory phenomenon, intensification versus extensification. The increasing biofuel crops and the expanding “plastic villages” of vegetable production on the one hand and the large set aside areas on the other hand led to different landscape characters.

The elevated terrace LFT is characterised by complete absence or insignificant presence of surface water due to the higher elevation of the terraces. Predominant is the equally flat surface covered by intensive arable land parcels and peri-urban zones as well as by growing horticultural establishments. Recently, energy production by means of wind turbines has increased in significance on the Plateau of Parndorf as well as in Hungary but to a lesser extent. These areas are less attractive and have neither recreational nor nature conservation potential. That is why this Landform Type provides the lowest values in *provision* (e.g. wild food, raw materials) and the highest values in *carrier* services at the same time. The relative high values in *regulation* and *habitat* services might be due to the well preserved nature conservation areas (wet grassland with high biodiversity) within this monotonous Landform Type.

The Landform Type “low and middle range mountain” is characterised by low mountains and foothills with low intensity human use, mainly covered by closed forests. The remarkable high values of *provision*, *regulation* and *habitat* services are based on the almost homogeneous oak-hornbeam forest with fringes of thermophilous downy oak associations with some infiltration of *Robinia pseudo-acacia* and small grassland patches on the hillsides of the deep valleys in the Leithagebirge. Although recreation and tourism play an important role, in particular in the vicinity of the towns, *carrier* (e.g. tourism facilities, transportation, habitation) and *information* (e.g. recreation, aesthetic, science and education) services only reached low values within this Landform Type.

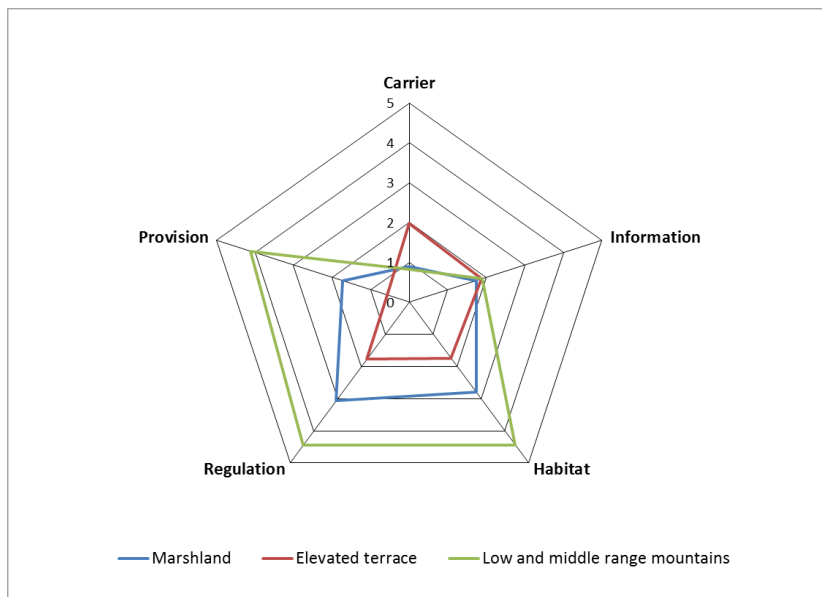


Figure 6: Distribution of the main landscape services within the LFTs marshland, elevated terrace and low and middle range mountain

Figure 7 shows the broader habitat types' capacity to provide the service soil formation within selected landscape sites in each of the three LFTs marshland, elevated terrace and low and middle range mountain. Whereas the sample site within the low and middle range mountain showed a very high capacity to provide soil formation, the elevated terrace sample site ranged between low and high service provision. The LFT marshland consisting of a more complex landscape structure showed between low and very high relevant links between broader habitat type and soil formation.

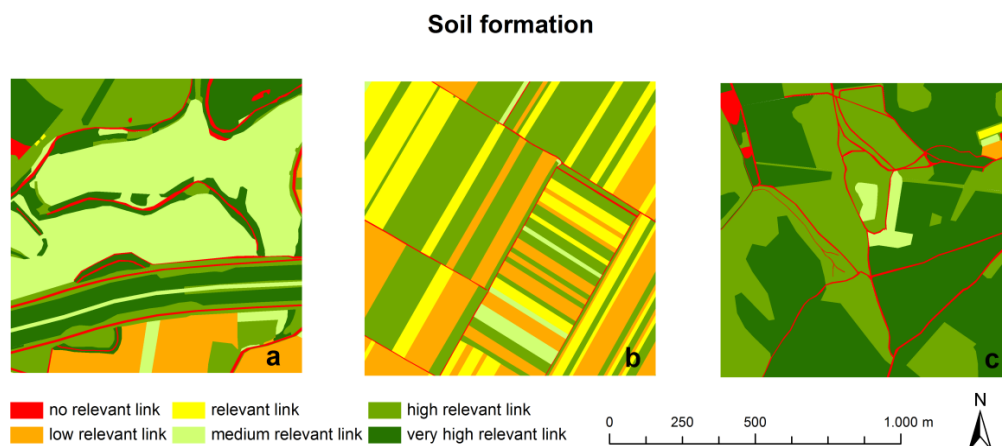


Figure 7: Spatial distribution of the service 'soil formation' within the LFTs marshland (a), elevated terrace (b) and low and middle range mountain (c). The different colours display the relevant links between the service 'soil formation' and the broader habitat types.

6 Discussion and conclusion

Within our project we assessed a wide range of landscape services at different spatial scales to provide a good overview of the services people derive from ecological networks in our investigation areas. As current landscape service indicators are still limited by insufficient data we used an expert driven assessment. Already previous studies have also used assessment matrices (e.g. Haines-Young and Potschin, 2008; Burkhard et al., 2009, 2012), where land cover types were linked by hypothetical values to selected ecosystem service supply capacities. This method enables a rapid service assessment

and shows a good overview to see first trends for landscape service provision (Burkhardt et al. 2009). Using a relative 0-5 scale provides a comparison of the different capacities of landscapes to deliver services by harmonisation of the different indicators. It also offers the opportunity to avoid value-laden units, such as monetary terms. Although economic valuation may encourage management options, decisions makers have to take into account the overall objectives and limitations of economic valuation (e.g. Heal, 2000; Farber et al., 2002; Hein et al., 2006). Whereas the studies in the project region North and Central North derived their results only from connecting the expert driven service values to the service providing units, the study in Central South integrated habitat specific qualifiers into their assessments in addition to that, in order to be locally explicit. Using habitat specific qualifiers, such as habitat structure, management, pressure and valuable attributes gives researchers the possibility to identify and quantify site-specific relations between landscape services and the environment. This method allows integrating the results into regional landscape planning projects.

Although connecting hypothetical values to service providing units has its drawbacks (Burkhardt et al. 2012, Hermann et al. submitted), this method provides a good opportunity to highlight the hot as well as cold spots of specific services within transnational ecological networks. Especially in transboundary areas, where data availability is very limited such a method enables to visualize and assess service delivery. The final landscape maps could provide stakeholders and planners with valuable information on the status quo of ecological and socio-cultural values. The service maps can therefore be used as a decision tool in landscape planning processes and might serve as a knowledge base for future conservation projects in order to enhance service delivery.

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