Abstract. This study aims to analyze how changes in land use influenced the delivery of ecosystem services in Cezava, a South Moravian agricultural region in the Czech Republic, in the period of 1845–2010. An observation of this period covering more than 160 years made it possible to reflect on social forces driving processes of transformation in the country. To capture the landscape multifunctionality and to indicate the environmental quality of the area under study, seven services provided in parallel by arable land, forests, and bodies of water were studied. The quantification of ecosystem services is based primarily on the transfer of values from the existing literature and on chronicle reviews and map analysis. Because looking back to the more distant past is a challenge and reliable information resources are lacking, a simple scoring method defining the functional features of the ecosystems was applied in order to evaluate the change of qualitative characteristics of the observed ecosystems. Besides that, the findings of these integrated assessments were supported by an analysis performed using landscape metrics. A comparison of service provision over the decades revealed that regulating and cultural services were significantly reduced, while provisioning services increased due to the proliferation of arable land, land consolidation, and agricultural intensification. However, a trend of improvement in the delivery of ecosystem services was introduced after 1990. Despite several uncertainties, this study demonstrates that it is possible to analyze long-term land-use trends to generate more meaningful, spatially explicit information, which can form the basis for landscape planning and ecosystem management.

Key words: Cezava, Czech Republic; ecosystems; ecosystem services; landscape; land use; long-term trends; transformation.

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Introduction

Agricultural management has already led to extensive transformation of about one-third of the global surface, and intensively utilized land is expected to expand in the future to meet human needs for living space, livelihoods, and food (Lambin and Geist 2006, Nelson et al. 2010). Concurrently, the conversion of natural vegetation cover to agricultural land (and urban areas) impacts water flows and the biogeochemical cycle and is closely linked to climate change (Schulp et al. 2008, Milad et al. 2011). The joint effects of land-use and climate change are perceived as the most important driver of biodiversity loss (Sala et al. 2000). Because biodiversity is known to represent a key prerequisite for the functioning of ecosystems and the delivery of ecosystem services (MA 2005, De Groot et al. 2010), a change of land use may undermine the regulatory capacities of the ecosystems, e.g., in terms of the ability to avoid and minimize hazards (Rockström et al. 2009, Preston et al. 2011). A number of risks initiated by land-use change or the consequences thereof originate in diminished land productivity, land degradation, disruption of the water regime, water contamination, or additional losses of biodiversity (Shao et al. 2005). As the provision of ecosystem services substantially depends on biophysical conditions and land-use changes over time and space, a more strategic allocation of intensively managed land-uses is necessary (Nelson et al. 2010, Burkhard et al. 2012). In addition to land-use
change, an important complementary factor influences the change in ecosystem service provision: shifts in the demand of consumers (Burkhard et al. 2012). Forces such as technical and scientific knowledge development, enhanced public awareness of the value of natural capital, or a threat of environmental changes and hazards (e.g., climate change) have considerable effect on the transformation of the demand for ecosystem services in time. For example, given the timescale of this case study, the demand for carbon sequestration barely existed back in 19th century because carbon dioxide emissions were much smaller and mostly came from renewable resources. Also, agricultural production was much more tightly linked with natural processes, relying on natural pest control, natural fertilizers, etc., whereas nowadays many of these services are substituted by agrochemicals and machinery. Even though we are fully aware of this factor, ongoing changes in land use and increasing concerns about the impact of such changes on the provision of ecosystem services remains to be the main motivation for this study. Therefore, the principal aim of the study is to analyze how changes in land use influenced the delivery of ecosystem services in Cezava, a South Moravian agricultural region in the Czech Republic, in the period 1845–2010.

Case study area

The Czech Republic is a Central European country where agricultural land is a dominant land-use category with coverage of more than 50% of the country’s territory (Food and Agriculture Organization of the United Nations Statistics Division; FAOSTAT; data available online).1 In relation to the examination of agriculturally utilized landscapes, Cezava was selected as an agricultural region with very favorable natural conditions (Pannonian mixed forests) and therefore intensively managed since historical times (Pokorny 2011). The region is situated ~15 km southeast of Brno and covers more than 15,000 ha (Fig. 1). Cezava is an alliance of 15 municipalities belonging to three districts (Brno-venkov, Hustopeče, and Vyškov), with a population density of about 129 inhabitants/m², which is comparable to the population density at the national level. Historically, the population decreased from the 1960s until the beginning of the 1990s, when it started to increase, especially after 2001. This trend is a result of suburbanization. In such agricultural regions, it is often the case that trade-offs take place and the environment is affected by intensive exploitation (Foley et al. 2005). Therefore, this issue provides a thought-provoking topic for research.

The next sections of the paper introduce the individual approaches that have been applied in the study. In particular, Methods introduces an analysis of long-term land-use changes, a landscape heterogeneity analysis and the environmental state of the case study area, followed by an analysis of ecosystem services with related changes in their availability over time. In this study, land-use change is regarded as a key driver that has had an impact on ecosystems and ecosystem service provision, which is analyzed from a historical perspective. Even though few regional studies of this type exist so far, they usually analyze a much shorter time period or report on either only a single service or a more specific driver of change (see, for example, Lautenbach et al. 2011, Li et al. 2007). Looking back to the more distant past is a challenge and reliable information resources are lacking; therefore, a simple scoring method defining the functional features of the ecosystems was applied in order to evaluate the change of the qualitative characteristics of the observed ecosystems. Finally, the findings of the study are discussed and summarized.

Methods

To understand how land-use change may alter the functioning of ecosystems and their capacity to provide ecosystem services, we first analyze long-term land-use changes in the case study area, applying data from the LUCC Czechia UK Prague database (available online).2 Land-use analysis is further developed with a determination of changes in landscape heterogeneity and an overview of the environmental state of Cezava. We then conduct an assessment of ecosystem services, including reflection on how provision changed during the observed period due to land-use change as the main driver under consideration.

Analysis of land-use change

Land-use changes reflect different phases of socio-economic development and political conditions, as well as environmental changes (Bičík et al. 2001). Current land-use in the Czech Republic has been greatly influenced by social forces and natural conditions. The evaluation of changes in the proportions of land-use categories was based on statistical data from the LUCC Czechia UK Prague database (Bičík et al. 2001, 2010). The land-use categories under observation were classified so as to be consistent with the categories in the database. Five land-use categories were analyzed: agricultural land, forests, water bodies, urbanized areas, and the remaining areas. The proportions (in ha) of particular categories were compared for five time periods: 1845, 1948, 1990, 2000, and 2010. These years, in fact, represent important milestones of modern Czech history (Bičík et al. 2001). The study compares land-use changes between 1845 and 1948 (under the conditions of a market economy), between 1948 and 1990 (under the conditions of a rigid centrally planned economy and a totalitarian regime),

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1 http://faostat3.fao.org/faostat-gateway/go/to/download/E/EL/E

2 http://web.natur.cuni.cz/ksgrsse/lucc/index.php?scn=2
and during the periods 1990–2000 (after the reintroduction of a market economy) and 2000–2010 (after accession to the European Union). To increase the accuracy of statistical data interpretation, additional resources concerning land-use change were consulted, such as historical maps (the first, second, and third Military Surveys dating from the 18th and 19th centuries), contemporary maps, aerial photos, and municipal chronicles.

Environmental state, landscape heterogeneity, and ecosystem functioning

An overview of the environmental state and problems is considered a straightforward way to indicate disturbed ecosystem functions, as limited ecosystem functioning undermines the provision of ecosystem services that support human wellbeing (Flynn et al. 2009). It has been recognized that intensive agricultural utilization modifies the quality of the environmental conditions and markedly influences the provision of bundles of ecosystem services (Metzger et al. 2008). Optimization of agricultural ecosystems for the provision of food, fiber, and fuel usually requires simplification of those ecosystems’ structure and intensification of their management. Consequently, landscape characteristics (e.g., heterogeneity) are changed. This triggers a chain of causal effects further influencing biological diversity, ecological functioning and, finally, the delivery of

Fig. 1. A map of Cezava, Czech Republic (source: Frelichova 2012), an alliance of 15 municipalities belonging to three districts (Brno-venkov, Hustopeč, and Vyškov).
regulation on which the harvest significantly depends (Zhang et al. 2007, Flynn et al. 2009, Frank et al. 2012). Due to this causality, attention is paid to changes in landscape heterogeneity as one of the symptoms of environmental change.

Changes in landscape heterogeneity can be described by Shannon’s diversity index (SHDI). The SHDI indicates the diversity of land based on information about landscape composition (the number of land-use categories present) and the relative abundances of different categories (Frank et al. 2012). The SHDI increases as the number of different patch types (i.e., patch richness) increases and/or the proportional distribution of the area among patch types becomes more equal (see the formula in Appendix A). Besides information about landscape diversity or ecological functionality, the index reflects the aesthetic value of the landscape, as the landscape’s naturalness, land-cover diversity, and heterogeneity contribute to people’s aesthetic perception of the landscape (Herbst et al. 2009). The SHDI was calculated for the case study area and for agricultural land subcategories in the period 1845–2000 (2010 was excluded because data on shares of meadows and pastures are not available separately). In addition to the SHDI, the sizes of fields during the observed time period are compared to indicate changes in the structure of arable land.

To further analyze environmental conditions and ecological functioning, the study focuses on soil erosion and water pollution, which point to negatively impacted energy flows within ecosystems. The level of soil erosion was adopted from a study by Havliček and Navrátilova (2005), in which erosion data were derived from the vulnerability of landscapes to water erosion according to Wischmeier and Smith’s Empirical Soil Loss Model as the mean potential annual soil loss in metric tons per hectare per year (t ha⁻¹ yr⁻¹).

To estimate water quality, basic chemical parameters (O₂, NH₄⁺, NO₃⁻, and overall P in mg/L) were checked in the seven main watercourses flowing through the studied area (Dunavka River, Hranečnický Brook, Moutnický Brook, Litava River, Otnický Brook, Říčka Brook, and Svrata River). These data are available from databases of the Agricultural Water Management Authority and the Czech Hydro-Meteorological Institute. Chemical parameters were referenced to the Czech norm of classification of surface water quality (ČSN 75 7221), and thus water quality was determined.

Analysis of ecosystem services

To capture the landscape multifunctionality and to indicate the environmental quality of the studied area, seven services in parallel by arable land, forests, and bodies of water were studied (Table 1). Although ecosystem services are also generated by ecosystems within the urban and remaining areas (see, for example, Bolund and Hunhammar 1999), this study is limited to only these three ecosystem types, which dominate the studied area, covering almost 90% of the territory. The main reason for the exclusion of the remaining areas, even though the category includes areas of significant ecological function, is the excessive heterogeneity of the category. Grasslands has also been left out of this type of the analysis due to data limitation.

The analysis follows MA (2005) classification of ecosystem services, particularly three categories: regulating services (air quality regulation, erosion regulation, disturbance regulation), provisioning services (food, fuel, medicine), and cultural services (recreation, cultural heritage, education). The quantification of ecosystem services is based primarily on value (benefit) transfer from the existing literature and on chronicle reviews and map analysis. Table 1 introduces indicators for service quantification in biophysical terms. The relevance of the selected services arises out of the agricultural character of the studied area, along with its land-use patterns and spatial context as introduced by Costanza (2008).

In addition to the biophysical valuation of ecosystem services, the total value of ecosystems and their services in economic terms is presented. The ecosystem values per hectare were adopted from Frelichova et al. (2014). Using basic value transfer and following the same methodology, the values of Cezava’s ecosystems were generated by an attribution of the total values of ecosystems to a land-use type (for details, see Appendix B). Monetary values are standardized to Euro currency, EUR ha⁻¹ yr⁻¹ using 2012 as the base year.

Changes in the delivery of ecosystem services

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

The three groups of ecosystem services (regulating, provisioning, and cultural) were attributed to three categories of landscape functions (ecological, production, and cultural). For each category of landscape function, four features were identified as indicators (these features are listed in worksheets for the ratings in Table 2). Based on the literature review (e.g., Zhang et al. 2007, Flynn et al. 2009, Meixler and Bain 2010, Wallenius et al. 2010, Frank et al. 2011) and the results from the case study, every feature was assigned a score ranging...
Table 1. Classification of ecosystem services and selected indicators.

| Ecosystem services                                      | Arable land | Forests | Water bodies |
|---------------------------------------------------------|-------------|---------|--------------|
| Regulating (regulation of biogeochemical cycles and biospheric processes) |             |         |              |
| Carbon sequestration (P)                                |             |         |              |
| - Annual amount of CO₂ stored in the biomass per 1 ha of ecosystem | x           | x       | ...          |
| - Annual amount of CO₂ stored in the soil per 1 ha of ecosystem  | x           | x       | ...          |
| Air quality regulation (P)                              |             |         |              |
| - Amount of pollutants removed (t·ha⁻¹·yr⁻¹)            | x           | x       | x            |
| Erosion control (P)                                     |             |         |              |
| - Amount of soil prevented from erosion (t·ha⁻¹·yr⁻¹)   | x           | x       | x            |
| Flood protection (P)                                    |             |         |              |
| - Amount of water stored (m³/yr)                        | x           | x       | x            |
| Provisioning (supporting humans with material benefits) |             |         |              |
| Food production (A)                                    |             |         |              |
| - Average crop yield (t·ha⁻¹·yr⁻¹)                      | x           | ...     | ...          |
| - Meat of hunted deer (t·ha⁻¹·yr⁻¹)                     | ...         | x       | ...          |
| - Fish production (t·ha⁻¹·yr⁻¹)                         | ...         | ...     | x            |
| - Mineral water collection (m³/yr)                       | ...         | ...     | x            |
| Material production                                    |             |         |              |
| - Average yield of lucerne and silage maize (A) (t/ha⁻¹·yr⁻¹) | x           | ...     | ...          |
| - Annual growing stock of timber (P) (m³/ha⁻¹·yr⁻¹)      | ...         | x       | ...          |
| Cultural (supporting humans with knowledge, information, cultural values, etc.) |             |         |              |
| Recreation and tourism (A)                              |             |         |              |
| - Cycling trails, length in km                          | x           | ...     | ...          |
| - Number of visitors per ha per year                    | ...         | x       | ...          |
| - Number of members of the fishing clubs/year           | ...         | ...     | x            |

Note: P is potential provision of ecosystem service, and A is actual provision of ecosystem service.

from −2 to 2, where 2 is strongly improves the functional feature, 1 is slightly improves the functional feature, 0 is neutral impact, −1 is slightly negative impact on the functional feature, and −2 is strong negative impact on the functional feature. Each category of landscape functions could receive a maximum of eight points (either positive or negative). In total, the maximum score for all functional categories was 24. To show the changes, the assessment was conducted for the appearance of Cezava’s ecosystems in 1845 and in 2010. Based on the scores, bar charts were produced to illustrate the relevant functional groups of features for each ecosystem type in the two time periods. Consequently, the categories of ecosystem services can be interpreted. Ecological features are presented as regulating services and, similarly, cultural features by cultural services. In the case of production features, two of them represent provisioning ecosystem services: food and material production. In addition to the illustration of the (multi)functionality of ecosystems, the bar charts (widths of the columns) reflect the shares of the ecosystem types in the studied region. The results of the assessment indicate the contributions of each feature to the overall multifunctionality of selected ecosystems.

**Results**

**Land-use changes**

Cezava covers about 0.2% of the total area of the Czech Republic and 0.4% of Czech agricultural land. Changes in the percentages of land-use categories in Cezava based on data from the LUCC Czechia UK Prague Database are shown in Table 3. The proportion of agricultural land and water areas dropped by 7% and 2% between 1845 and 2010, while the area of forests, built-up and remaining areas increased by 2%, 1%, and 7%, respectively. Among the five categories, the dominant change trends were conversion of agricultural land and water bodies to forests, built-up areas, or remaining areas.

Agricultural land shows only a slight downward trend in terms of total area during the years examined (Table 3). However, this category encompasses diverse types of land: arable land, permanent cultures, meadows, and pastures. When considering these subcategories, the heterogeneity of agricultural lands has decreased notably over time.

The area of forested land has been increasing since the 19th century. The most intensive afforestation period took place in the second half of the 20th century (Table 3). Until 1948, the forests exhibited a natural alluvial character with typical species composition (e.g., *Alnus* sp., *Salix* sp., *Populus* sp., *Quercus* sp.). Apart from the continuous forest cover, scattered islands of trees and a number of solitary trees were present in the landscape. According to the Military Survey maps, most roads were lined with alleys. The forests of today are managed at odds with the natural species composition, and the tree species planted (*Picea* sp.) mainly have a production function.
Table 2. The scoring worksheets (scored from $-2$ to $+2$) for functional attributes in 1845 and 2010 (compiled based on Lovell et al. 2010).

| Functional attributes | Arable land | Meadows | Pastures | Forest | Water | Total |
|-----------------------|------------|---------|----------|--------|-------|-------|
| **Year 1845**         |            |         |          |        |       |       |
| Ecological features   |            |         |          |        |       |       |
| Carbon sequestration  | $-1$       | $1$     | $1$      | $2$    | $0$   | $4$   |
| Wildlife habitat      | $-1$       | $1$     | $1$      | $2$    | $2$   | $4$   |
| Erosion control       | $-1$       | $2$     | $1$      | $2$    | $1$   | $4$   |
| Water quality         | $-1$       | $1$     | $1$      | $2$    | $2$   | $4$   |
| Total                 | $-4$       | $5$     | $4$      | $8$    | $3$   | $16$  |
| Production features   |            |         |          |        |       |       |
| Food production       | $2$        | $0$     | $0$      | $1$    | $1$   | $4$   |
| Material production   | $2$        | $1$     | $1$      | $2$    | $0$   | $4$   |
| Efficiency of input   | $1$        | $1$     | $1$      | $2$    | $1$   | $4$   |
| Economic value        | $1$        | $1$     | $1$      | $2$    | $1$   | $4$   |
| Total                 | $6$        | $3$     | $3$      | $6$    | $3$   | $21$  |
| Cultural features     |            |         |          |        |       |       |
| Recreation            | $0$        | $1$     | $1$      | $2$    | $2$   | $5$   |
| Visual quality/aesthetics | $1$    | $1$     | $1$      | $2$    | $1$   | $5$   |
| Education/research    | $1$        | $1$     | $1$      | $2$    | $1$   | $5$   |
| Living place          | $0$        | $0$     | $0$      | $0$    | $0$   | $0$   |
| Total                 | $2$        | $3$     | $3$      | $6$    | $5$   | $19$  |
| Performance sum       | $4$        | $11$    | $10$     | $20$   | $11$  | $56$  |
| **Year 2010**         |            |         |          |        |       |       |
| Ecological features   |            |         |          |        |       |       |
| Carbon sequestration  | $-2$       | $1$     | $1$      | $2$    | $0$   | $4$   |
| Wildlife habitat      | $-1$       | $1$     | $1$      | $1$    | $1$   | $3$   |
| Erosion control       | $-2$       | $2$     | $1$      | $2$    | $0$   | $4$   |
| Water quality         | $-2$       | $1$     | $1$      | $2$    | $1$   | $4$   |
| Total                 | $-7$       | $5$     | $4$      | $7$    | $2$   | $11$  |
| Production features   |            |         |          |        |       |       |
| Food production       | $2$        | $0$     | $0$      | $1$    | $1$   | $3$   |
| Material production   | $2$        | $1$     | $1$      | $2$    | $0$   | $4$   |
| Efficiency of input   | $1$        | $1$     | $1$      | $2$    | $1$   | $4$   |
| Economic value        | $1$        | $1$     | $1$      | $2$    | $1$   | $4$   |
| Total                 | $7$        | $3$     | $3$      | $6$    | $3$   | $17$  |
| Cultural features     |            |         |          |        |       |       |
| Recreation            | $0$        | $1$     | $1$      | $2$    | $2$   | $4$   |
| Visual quality/aesthetics | $-1$ | $1$     | $1$      | $2$    | $2$   | $4$   |
| Education/research    | $1$        | $1$     | $1$      | $2$    | $1$   | $4$   |
| Living place          | $0$        | $0$     | $0$      | $0$    | $0$   | $0$   |
| Total                 | $0$        | $3$     | $3$      | $6$    | $5$   | $17$  |
| Performance sum       | $0$        | $11$    | $10$     | $19$   | $10$  | $50$  |

Table 3. Changes in land use of the region Cezava from 1845 to 2010.

| Land use category         | 1845 Area (ha) | 1845 % | 1948 Area (ha) | 1948 % | 1990 Area (ha) | 1990 % | 2000 Area (ha) | 2000 % | 2010 Area (ha) | 2010 % |
|---------------------------|---------------|--------|----------------|--------|----------------|--------|----------------|--------|----------------|--------|
| Agricultural land         | 13 540        | 91     | 13 464         | 92     | 12 431         | 85     | 12 390         | 85     | 12 306         | 84     |
| Arable land               | 10 957        | 74     | 12 740         | 87     | 11 240         | 77     | 11 240         | 77     | 11 134         | 76     |
| Permanent cultures        | 363           | 2      | 374            | 2      | 1 012          | 7      | 1 010          | 7      | 962            | 7      |
| Meadows                   | 822           | 5      | 106            | 1      | 19             | 0      | 17             | 0      | 210†           | 1      |
| Pastures                  | 1 398         | 9      | 244            | 2      | 161            | 1      | 123            | 1      | ...            | ...    |
| Forests                   | 363           | 2      | 431            | 3      | 624            | 4      | 628            | 4      | 631            | 4      |
| Water areas               | 473           | 3      | 73             | 1      | 136            | 1      | 149            | 1      | 162            | 1      |
| Built-up area             | 167           | 1      | 190            | 1      | 298            | 2      | 309            | 2      | 313            | 2      |
| Remaining areas           | 285           | 2      | 412            | 3      | 1 180          | 8      | 1 180          | 8      | 1 245          | 9      |
| Total area‡               | 14 830        | 100    | 14 570         | 100    | 14 670         | 100    | 14 656         | 100    | 14 657         | 100    |

Note: Adapted from the LUCC Czechia UK Prague database (http://www.lucc.ic.cz).
† Shares of meadows and pastures individually are not available in 2010 due to aggregation of the categories into “grasslands.”
‡ Total area of Cezava slightly changes in time because of differenciations in borders of the cadastres.
In contrast to forest cover, the area covered by bodies of water and streams decreased significantly over the examined period. The first significant reduction in the number of bodies of water, particularly ponds, is evident in maps from the 18th and 19th centuries (maps from the first and second Military Surveys). The ponds were drained and replaced with fields. According to the chronicles, these fields were rich in nutrients and very fertile. For example, the chronicles mention that great sugar beet and lucerne yields were obtained for 20 years without any fertilization. Stream regulation and drainage, in response to regular spring floods and summer droughts, caused another reduction in the water area (e.g., in the municipality of Moutnice in the 1920s). The total area of bodies of water dropped to a minimum in the mid-20th century. Since then, the area covered by bodies of water has again increased, though many water courses were dredged or regulated. As Meixler and Bain (2010) mentioned, such management causes a decrease of the groundwater level and degradation of aquatic ecosystem functions.

An overview of land-use development in the studied area indicates that sociopolitical development has been the major influential factor acting as the driving force behind land-use changes over the past two centuries. The same factor is among the most important causes of land-use changes throughout the Czech Republic, typically resulting in a reduction of agricultural land and in the increase of forest and remaining areas (including built-up areas; Bičík et al. 2000). The only exception is the period from 1845 to 1948, when the area of agricultural land in the studied area increased. When considering changes within the agricultural land category, the studied area is again consistent with the primary types of change occurring at the national level. The most common type of change in the 1845–1948 period was the increase of arable land and permanent cultures, accompanied by the reduction of meadow and pasture lands. Later, from 1948 to 1990, the dominant type of change consisted in a significant decrease of arable land, meadows, and pastures. Permanent cultures, on the other hand, increased (Bičík et al. 2000). More recently, a remarkable sociopolitical development that significantly impacted land use in the Czech Republic was the Velvet Revolution (1989) and the subsequent transformation period (1990–2000). Since the 1990s, agricultural utilization of land has been concentrated more significantly in the lowlands, whereas extensive agricultural production in hilly and mountainous regions has been greatly reduced in order to increase the economic effectiveness of agricultural production (Bičík and Jančák 2005). Due to favorable natural conditions, the studied area continues to be utilized agriculturally, even though the area of arable land decreased by 1% in the decade after 2000. Permanent cultures also decreased slightly, though all other categories show an increase in area. Today, the landscape of Cezava, which had previously had a primarily agricultural function, has been transformed into a landscape with agricultural and residential functions.

Changes in landscape heterogeneity.—The results of the landscape heterogeneity assessment indicate that the studied area was the most uniform around 1948, when the value of the SHDI reached its lowest level (0.29). Since 1990, the SHDI has retained values around 0.5, which is even higher than in 1845 when the SHDI was 0.33. This is caused by a reduction of the agricultural land area and an increase of the shares of other land-use categories. However, another pattern is demonstrated by the agricultural land subcategory and related SHDI levels. The heterogeneity level was the highest (0.67) in 1845. The SHDI of agricultural land was 0.26 in 1948, 0.36 in 1990, and 0.35 in 2000. The 2010 time horizon was not analyzed because of changes in the classification of grasslands. Until 2000, all agricultural land-use subcategories represented at the beginning of the period remained present (also meadows in the value of 0.14% in 2000), and the results can be interpreted as a slightly increasing evenness of the structure of agricultural land despite the dominant share of arable land (roughly 76–77%). The proportion of arable land reached almost the same share as in 1845, but the average field size increased considerably, in line with the national trend. According to data from the Ministry of Agriculture (2009), the average size of fields reached approximately 0.2 ha in 1948 (before collectivization), as opposed to the 20-ha average size of fields today. High land-use intensity may also be demonstrated by the minimal size of the area of natural land cover types (Frank et al. 2012). In Cezava, protected landscape areas cover less than 1%.

Soil erosion and water pollution.—Most of the main water streams of the studied area show chemical concentrations corresponding to polluted, heavily polluted, or very heavily polluted water. The only exception with clean water is the Říčka brook. Regarding soil erosion, about 42% of the area is threatened by soil erosion (municipalities of Blučina, Nikolčice, Otnice, Těšany, and Židlochovice; Havliček and Navrátilova 2005). The arable land erosion rate is the average aggregate of erosion levels on arable land in individual municipalities (2.7 t·ha⁻¹·yr⁻¹), while for grasslands the national average rate calculated by Kráša (2010) was taken into account (0.18 t·ha⁻¹·yr⁻¹). Erosion on arable land and grasslands reached the lowest level in 1845 (29 800 tons). The period with the highest erosion rate was around 1948 (34 500 tons). After 1990, the erosion rate reached lower levels again, at around 30 400 tons. In 2010, annual soil loss was about 30 100 tons (1% more than in 1845). Besides that, soil quality has also changed. The dynamic properties of soil, soil structure, and soil biodiversity have been affected, resulting in soil degradation.
Ecosystem services provided in the present

Three dominant ecosystem types were analyzed in terms of provision of ecosystem services: arable land, forests, and bodies of water. These types cover about 80% of the studied area. Regulation, provisioning, and cultural services of the ecosystems were identified and, where possible, quantified in biophysical units (Table 4).

In addition to biophysical assessment of the ecosystem services, an economic valuation of the ecosystems was conducted. Based on Frelichova et al. (2014), it was possible to generate a map of the studied area, indicating the spatially explicit values of the ecosystems (Fig. 2). The explicit values for the key ecosystems, which were considered for the detailed analysis are shown in Table 5 (in EUR/ha).

Fig. 2 gives an illustration of the spatial distribution of the values of the ecosystems in Cezava. Similarly to the monetary valuation of ecosystems at the national level, the highest values are attributed to natural and close-to-natural ecosystems.

Changes in provision of ecosystem services

The changes in the provision of ecosystem services are described based on the two assessments. Firstly, alterations in the capacity to sequester carbon between 1845 and 2010 were quantitatively analyzed. Secondly, a qualitative assessment of landscape performance is introduced.

Changes in carbon sequestration capacity.—Net sequestration is the capacity of arable lands, grasslands and forests to sequester carbon. In the case of Cezava, the negative values of carbon sequestration of arable land push net sequestration into negative values (Fig. 3). Even though forests and grasslands capture carbon,
Fig. 2. A valuation map of ecosystems in Cezava, with currency in Euros. NA is not assessed.

Fig. 3. The change in net carbon sequestration rate by ecosystems in Cezava from 1845 to 2010.
highly representa- tion in the agricultural landscape does not outweigh the negative carbon values from the dominant croplands. From the long-term point of view, net sequestration has increased since 1948, though the studied area’s initial capacity to sequester carbon has not been reached again. The difference between initial carbon sequestration in 1845 and in 2010 is about 22% less carbon sequestered (whereas in 1845 it was 100%).

Changes in landscape performance and provision of ecosystem services.—The results of the landscape performance assessment are illustrated in Table 2, which introduces the worksheets for the ratings based on ecosystem features and land use in 1845 and 2010, respectively. Particularly, in 1845, arable land was managed less intensively than today and therefore a less negative impact on the erosion rate and water quality was indicated by the score. Also, the economic value was slightly lower (according to the results of the economic analysis). Another feature, visual quality, was importantly influenced by the transition from small-scale land-use to large-scale land-use.

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

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

The change of the composition of natural species was reflected in the lower score for wildlife habitat in 2010 compared with 1845.

Even though the character of grasslands (meadows and pastures) and their management has changed over the last 160 years, no specific information on this issue is available. Therefore, the same scores were attributed to meadows and pastures for both time periods. Based on the scores, bar charts have been produced (Fig. 4).

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

The findings also show that the level of landscape multifunctionality has declined since 1845. Meanwhile, all ecosystems possessed all three functional features in 1845; the cultural function of arable land declined considerably later in the observed period. Conversely, production on arable land is the only feature that increased with time. Other features either remained at the same level or declined. Besides that, the chart illustrates trade-offs (conflicts) that exist between particular groups of functions or services. The most notable trade-off occurs between the ecological and production functions (regulating and provisioning services). This also supports the findings from the biophysical assessment, particularly the identification of losses of benefits, such as air quality and soil retention capacity, or increased soil losses (Table 4). Another trade-off can be identified between provisioning and cultural services. While the production function remained the same over time, the cultural function of the ecosystems declined.

Discussion

Analysis of land-use change

The statistical data acquired from the LUCC Czechia database made it possible to analyze long-term land-use changes in the period from 1845 to 2010. The categorization of data on land-use changes into time intervals brings up several points for discussion, such as the unequally long intervals of observation and the hardly predictable intensity of a change within a given period, a change could have been quite sudden (e.g., within one year) or gradual (developing over several years or even decades). Another limitation lies in the fact that the statistical data are not spatially specific and do not provide qualitative information on ecosystems. This “black box” character of the gathered data limits their subsequent interpretation. For that reason, the statistical data were supplemented with historical maps from three Military Surveys and with aerial photos. Old maps and aerial photos provide relevant source data that enable assessments of landscape history (Engstová and Skaloš 2009), though they (particularly aerial photos) show only arbitrary snapshots in time (Hietel et al. 2004). The last type of specific historical data resource, the chronicles, partly aided in clarification of causes behind changes and helped shed light on ways that sociopolitical events have contributed to land-use development.
Drawing upon the available literature, major reasons for changes in land-use patterns are population growth, collectivization, and intensification of production followed by the introduction of diverse agro-environmental schemes after 1990 and accession to the EU. In the national context, Cezava is one of the regions where emphasis is placed on agricultural production. This prevailing function was determined based on the distribution of land-use categories in combination with an assessment of the ecosystems' functional features. Landscape elements such as meadows or pastures practically disappeared, even though they enhance the landscape's functioning (Fahrig et al. 2011). This phenomenon can be explained primarily as a result of ongoing agricultural intensification in the mid-20th century, strengthened by the introduction of new sociopolitical drivers that arose out of changes in the sociopolitical regime. The socialist government sup-

Fig. 4. The performance of landscape features in a) 1845 and b) 2010.
pressed private ownership and small-scale land-use (typically strip fields), providing for the livelihoods of individual farmers, rather transforming into large-scale land-use under the supervision of cooperative and state-owned farms (Sklenička 2005). The results reveal a distinct simplification of the agricultural land microstructure after the 1950s. Due to favorable natural conditions, the region has been attributed the agricultural function and remains agricultural to this day, as the efficiency of intensive management is higher than that of less favorable areas, e.g., those in higher altitudes and/or with less fertile soil.

The impact of environmental state on the provision of ecosystem services

To broaden the context of the findings, reflection on the environmental problems of Cezava is provided and the potential impact of these problems on the service availability is discussed. Shannon’s diversity index on agricultural land heterogeneity and the development of average field size indicates intensive agriculture in the studied area. The estimation of the SHDI represents a formalized assessment enabling rapid interpretation of particular land-use pattern development (Frank et al. 2012). In accordance with Frank et al. (2012), this exercise confirms the usefulness of the combination of landscape metrics and ecosystem services, even though landscape metrics are not able to evaluate all aspects of the service provision capacity of the landscape.

In agroecosystems, many of the natural ecosystem functions are replaced with human labor or fertilizers (Swift et al. 2004). Reduction of biodiversity in these systems results from the intensification of production with the specific purpose of maximizing production with minimal additional energy inputs. The conversion of land from complex natural systems to simplified agricultural ecosystems is a major cause of biodiversity loss (Flynn et al. 2009). Even though the effects of agricultural intensification on biodiversity are still poorly understood, it almost always results in fewer species with lower genetic variation and less functional groups (Swift et al. 2004, Sundseth 2009). Cezava is a region of major importance for birds. A decline in bird populations in the studied area has already been observed and described due to the disappearance of many of the semi-natural and natural habitats. As in other areas of the Czech Republic (and Europe), the number of farmland birds has decreased by approximately 50% since the 1980s (Vorišek et al. 2008, 2009). Examples of the bird species that are threatened today but were historically abundant in the studied area are the gray partridge (Perdix perdix) and the common quail (Coturnix coturnix). Another species of interest, the great bustard (Otis tarda), had appeared (though sporadically) in the studied area since the 1900s. This species indicates diversified, species-rich habitats and heterogeneous and sustainable landscapes (Škorpíková 2008). Its disappearance from the South Bohemia region is dated to the 1990s and is the consequence of further changes in crop selection and a substantial reduction of areas sown with alfalfa (Medicago sativa; Škorpíková 2008). The decline in bird species populations or their absence despite bird conservation efforts indicates inappropriate landscape management (Sundseth 2009). However, biodiversity is a condition for the delivery of ecosystem services (MA 2005) not only from agricultural ecosystems but also from all other ecosystem types which are not in isolation. Other than loss of biodiversity, reduced landscape heterogeneity means a monotonous landscape and limits the provision of cultural ecosystem services (Frank et al. 2012). Risks associated with reduced biodiversity may further lead to production losses through reduced diversity of pollinators or increased soil erosion. Increased levels of soil loss in turn threaten agricultural production and food safety, affect water quality (Otero et al. 2012), and contribute to soil degradation, which is considered to be one of the key environmental problems (Robinson et al. 2009, Adhikari and Nadella 2011, UK NEA 2011). Several areas in the Cezava region already lack a nutrient-rich layer in the soil horizon. The main reason for soil degradation is that a large part of land reclaimed for agriculture is not suitable for intensification practices and heavy machinery use (Elgersma et al. 2008).

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

Ecosystem service provision over time

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

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

The method provides for simple identification of dominant trade-offs among ecosystem services. However, data for the historical analysis were limited and one should therefore be careful with their interpretation. For example, details on the composition of forest species in the 19th century were not available despite a thorough search in several types of historical sources. The chronicles were expected to have the potential to contain the desired information, but they include primarily demographic or economic data, while ecological data are limited to extreme weather conditions. Therefore, the chronicles did not prove to be very relevant sources for this type of analysis, at least in this case study area. Because data restrictions are often a problem when analyzing past or current ecosystem services (see, for example, Costanza et al. 1997, Gaodi et al. 2006, Bíčik et al. 2010a), past ecosystem services were assessed based on land-use development and expert knowledge of political events and social life. Current ecosystem services were discussed in greater detail, qualitatively and, when possible, quantitatively, based on existing studies. Though transfer of data from the literature introduces some inaccuracies. For example, data on the actual rates of carbon storage and sequestration are based on measurements taken from only a few forests, from which the amounts of carbon for a given forest are then estimated. In reality, the actual amounts of carbon for a given forest type vary greatly depending on the plant community composition, age structure, soil fertility, and other local environmental conditions (Kremen et al. 2004). This study also assumes that the availability of ecosystem services is linear. In fact, the availability of services follows more sophisticated patterns than simple linear trends, as the ecosystems are complex and dynamic (De Fries et al. 2004, Koch et al. 2009).

Despite these limitations, we believe that the method presented by this study can contribute to the development of more comprehensive knowledge on long-term ecosystem service availability. A combination of multiple ecological and social characteristics and data sources enabled original interpretation of existing data and resulted in integrated landscape analysis. We were able to show how long term land-use trends affect ecosystems and ecosystem service availability. This can form the basis for landscape planning and ecosystem management either thanks to ecosystems value recognition or by revealed consequences of intensive management practices.

Besides, given the multifunctional aspect of the assessment, the method can be beneficial for an assessment of other types of multifunctional environment. Lovell et al. (2010), for example, suggest that urban and peri-urban farms offer unique potential for strong multifunctionality and ecosystem services provision. Therefore, urban gardens and farms represent suitable land-use categories for further research and the method application.

Even though we focused on availability of ecosystem services according to land-use changes specifically, we believe that the method offers the opportunity to be used also for a profound analysis of changes in the ecosystem service demand.

Conclusions

In this study, an assessment of landscape dynamic processes using long-term land-use data was presented. In combination with the concept of ecosystem services, this methodological approach helped to develop a framework reflecting changes in the landscape’s multifunctionality over a period of roughly 160 years. Being the subject of the research, the Cezava region is a case study area of great agricultural importance which shows symptoms of monofunctional over-use as demonstrated by the results of this study. From this perspective, its natural agricultural potential might be at risk in the future. Despite several uncertainties, the assessment clarifies the effects of land use on the environment, identifies the significance of particular services, and can potentially help in the assessment of the costs related to the loss of such services. This research also demonstrates that it is possible to analyze long term land-use trends to generate more meaningful, spatially explicit information, which can form the basis for landscape planning and ecosystem management. Though this case study presents a Czech example, the applied approach and the results will hopefully be supportive of other studies.
focused on the environmental impacts of long-term land-use change in different regions.

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Literature Cited
Adhikari B., and K. Nadella. 2011. Ecological economics of soil erosion: a review of the current state of knowledge. Annals of the New York Academy of Sciences 1219:134–152.
Bičík, I., and V. Jančák. 2005. Transformační procesy v českém zemědělství po roce 1990. UK, Prague, Czech Republic. [In Czech.]
Bičík, I., L. Jejček, J. Kabrda, L. Kupková, Z. Lipský, P. Mareš, L. Šefránek, P. Štych, and J. Winklerová. 2010. Vývoj využití ploch v Česku. Česká geografická společnost. Edice Geographica, Czech Republic. [In Czech.]
Bičík, I., L. Jejček, and V. Štěpánek. 2000. Major types of land use changes in the Czech Republic 1845–1948–1990. Pages 32–39 in M. Hwang, and Y. Himiyama, editors. Land use and land cover change: the contribution of geography. Journal of Geography Education, Special Issue for the IGU Study Group on Land Use and Land Cover Change. 29th International Geographical Congress, Seoul, Korea.
Bičík, I., L. Jejček, and V. Štěpánek. 2001. Land-use changes and their social driving forces in Czechia in the 19th and 20th centuries. Land Use Policy 18:65–73.
Boland, P., and S. Hunhammar. 1999. Ecosystem services in urban areas. Ecological Economics 29(6):293–301.
Burkhard, B., F. Kroll, S. Nedkov, and F. Müller. 2012. Mapping supply, demand and budgets of ecosystem services. Ecological Indicators 21:17–29.
Costanza, R., et al. 1997. The value of the world’s ecosystem services and natural capital. Nature 387:253–260.
Costanza, R. 2008. Ecosystem services: multiple classification systems are needed. Biological Conservation 141:350–352.
De Fries R.S., J. A. Foley, and G. P. Asner. 2004. Land-use choices: balancing human needs and ecosystem function. Frontiers in Ecology and the Environment 2(5):249–257.
De Groot, R. S., R. Alkemade, L. Braat, L. Hein, and L. Willemen. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity 7:260–272.
Elgersma, A. M., S.S. Dhillon, A. Arnoldussen, J. Fanta, and E. Boucniková. 2008. Understanding the quality of land in agricultural land use systems. Pages 217–236 in Brower, F., et al., eds. Sustainable land management. Edward Elgar, Cheltenham, UK.
Engstová, B., and J. Skalóš. 2009. Methodology for mapping non-forest wood elements using historic cadastral maps and aerial photographs as a basis for management. Journal of Environmental Management 91:831–843.
Fahrig, L., J. Baudry, L. Brotons, F. Burel, T.O. Crist, R.J. Fuller, C. Sirami, G.M. Siritiwardena, and J. Martin. 2011. Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. Ecology Letters 14:101–112.
Felton, A., M. Lindbladh, J. Brunet, and Fritz O. 2010. Replacing coniferous monocultures with mixed-species production stands: an assessment of the potential benefits for forest biodiversity in Northern Europe. Forest Ecology and Management 6:939–947.
Flynn, D. F. B., M. Gogol-Prokurator, T. Nogeire, N. Molinari, B. Trautman Richers B.B. Lin, S. Nicholas, M.M. Mayfield, and F. Declerc. 2009. Loss of functional diversity under land use intensification across multiple taxa. Ecology Letters 12:22–33.
Foley, J. A., et al. 2005. Global consequences of land use. Science 309:570–574.
Frank, S., C. Fürst, L. Koschke, and F. Makeschin. 2012. A contribution towards a transfer of the ecosystem service concept to landscape planning using landscape metrics. Ecological Indicators 21:30–38.
Frélichová, J. 2012. Integrated landscape assessment of Czevza region. Pages 59–66 in Y. Himiyama, I. Bičík, and J. Feranse, editors. Land use/cover changes in selected regions in the world, volume VII. Charles University in Prague. In Press.
Fryda, J., O. Švihla, and L. Chnuxia. 2006. Study on ecosystem services: progress, limitation and basic paradigm. Journal of Plant Ecology 30:191–199.
Havlíček, T., and M. Navrátilová. 2005. Nová krajina regionu Czevza: koncepce revitalizace krajiny na katastrech členských obcí. Atelier Fontes, Brno, Czech Republic. [In Czech.] Herbst, H., M. Förster, and B. Kleinschmit B. 2009. Contribution of landscape metrics to the assessment of scenic quality: the example of the landscape structure plan Havelland/Germany. Landscape Online 10:1–17.
Hietel, E., R. Waldhardt, and A. Otte. 2004. Analysing land-cover changes in relation to environmental variables in Hesse, Germany. Landscape Ecology 19:473–489.
Janssens, I. A., et al. 2005. The carbon budget of terrestrial ecosystems at country-scale: a European case study. Biogeosciences 2(1):15–26.
Koch, E. W., et al. 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. Frontiers in Ecology and the Environment 7(1):29–37.
Kremen, C. W., N. M. Williams, R. L. Bugg, J. P. Fay, and R. W. Thorp. 2004. Total area requirements of an ecosystem service: crop pollination by native bee communities in California. Ecology Letters 7:1109–1119.
Krása, J. 2010. Empirical models for water erosion in Czech Republic (In Czech: Empirické modely vodní eroze v ČR). Habilitation Thesis. Czech Technical University, Prague, Czech Republic.
Lambin, E. F., and H. J. Geist, editors. 2006. Land-use and land-cover change: local processes and global impacts. Springer-Verlag, Heidelberg, Germany.
Lautenbach, S., C. Kugel, A. Lausch, and R Seppelt. 2011. Analysis of historic changes in regional ecosystem services provisioning using land use data. Ecological Indicators 11:676–687.
Li, R. Q., M. Dong, J.Y. Cui, L.L. Zhang Q.G. Cui, and W.M. He.
2007. Quantification of the impact of land use changes on ecosystem services: a case study in Pingbian County, China. Environmental Monitoring and Assessment 128:503–510.
Lovell, S. T., S. DeSantis, C.A. Nathan, M.B. Olson, V.E. Méndez, H.C. Kominami, D.L. Erickson, K.S. Morris, and W.B. Morris. 2010. Integrating agroecology and landscape multifunctionality in Vermont: an evolving framework to evaluate the design of agroecosystems. Agricultural Systems 103:327–341.

MA [Millennium Ecosystem Assessment]. 2005. Ecosystems and human well-being: a framework for assessment. Report of the Conceptual Framework Working Group of the Millennium Ecosystem Assessment. Island Press, Washington, D.C., USA.
Main-Knorn, M., P. Hostert, J. Kozak, and T. Kuemmerle. 2009. How pollution legacies and land use histories shape post-Communist forest forest cover trends in the Western Carpathians. Forest Ecology and Management 260–70.
Meixler, M. S., and M.B. Bain. 2010. Landscape scale assessment of stream channel and riparian habitat restoration needs. Landscape and Ecological Engineering 6:235–245.

Métzger, M. J., D. Schröter, R. Leemans, and W. Cramer. 2008. A spatially explicit and quantitative vulnerability assessment of ecosystem service change in Europe. Regional Environmental Change 8:91–107.
Milad, M., H. Schaiacha, M. Bürgib, and W. Konolda. 2011. Climate change and nature conservation in Central European forests: a review of consequences, concepts and challenges. Forest Ecology and Management 261:829–843.

Nasi, R., S. Wunder, and A. J. J. Campos. 2002. Forest ecosystem services: Can they pay our way out of deforestation? Center for International Forestry Research for the Global Environmental Facility, Bogor, Indonesia.

Nelson, E., H. Sander, P. Hawthorne, M. Conte, D. Ennaanay, S. Wolny, S. Manson, and S. Polasky. 2010. Projecting global land-use change and its effects on ecosystem services provision and biodiversity with simple models. PLoS ONE 5(12):e14327.
Otero, J. D., A. Figueroa, F. A. Muñoz, and M.R. Peña. 2012. Loss of soil and nutrients by surface runoff in two agro-ecosystems within an Andean paramo area. Ecological Engineering 37(12):2035–2043.
Pokorny, P. 2011. Neklidné časy: Kapitoly ze společných dějin přírody a lidí. Dokořán, Prague, Czech Republic. [In Czech.]

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

Robinson, D. A., I. Lebron, and H. Vereecken. 2009. On the definition of the natural capital of soils: a framework for description, evaluation, and monitoring. Soil Science Issues 73:1904–1911.

Rockström, J., et al. 2009. Planetary Boundaries: exploring the safe operating space for humanity. Ecology and Society 14(2):32.
Sala, O. E., et al. 2000. Global biodiversity scenarios for the year 2100. Science 287(5459):1770.
Schulp, C. J. E., G.-J. Nabuurs, and P. H. Verburg. 2008. Future carbon sequestration in Europe: effects of land use change. Agriculture, Ecosystems and Environment 127:251–264.
Shao, J., J. Ni, Ch. Wei, and D. Xie. 2005. Land use change and its corresponding ecological responses: a review. Journal of Geographical Sciences 15(3):305–328.

Sklenička, P. 2005. Applying evaluation criteria for the land consolidation effect to three contrasting study areas in the Czech Republic. Land Use Policy 23:502–510.

Skorpiková, V. 2008. Drop velký: leštění problém české ochrany přírody. Ochrana přírody 5. http://www.casopis. ochranaprirody.cz/vyzkum-a-dokumentace/drop-velky/ [In Czech.]
Sundseth, K. 2009. Natura 2000 in the Pannonian region. European Commission, Environment Directorate General, Luxembourg.
Swift, M. J., A.-M. N. Izac, and M. Van Noordwijk. 2004. Biodiversity and ecosystem services in agricultural landscapes: Are we asking the right questions? Agriculture, Ecosystems and Environment 104:113–134.

UK NEA [UK National Ecosystem Assessment]. 2011. The UK National Ecosystem Assessment: synthesis of the key findings. United Nations Environment Program, World Conservation Monitoring Centre, Cambridge, UK.

Voříšek, P., et al. 2008. Agricultural intensification still threatens European field birds. http://www.cso.cz/index.php?ID=1756

Voříšek, P., A. Klvaňová, T. Brinke, J. Cepák, J. Flousek, J. Hora, J. Reif, K. Šťastný, and Z. Vermouzek. 2009. State of the birds of the Czech Republic. Sylvia 45:1–38.

Wallenius, T., L. Niskanen, T. Virtanen, J. Hottola, G. Brumelis, A. Angervuori, J. Julkunen, and M. Pihlström. 2010. Loss of habitats, naturalness and species diversity in Eurasian forest landscapes. Ecological Indicators 10(6):1093–1101.

Zhang, W., T. H. Ricketts, C. Kremen, K. Carney, and S. M. Swinton. 2007. Ecosystem services and dis-services to agriculture. Ecological Economics 64(2):253–260.