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Biodiversity and ecosystem services: Complementary approaches for ecosystem management?

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ABSTRACT

The concept of ecosystem services introduced a new view on the relationship between biodiversity and human well being. But are both concepts, biodiversity and ecosystem services, mutually beneficial? We will try to unravel part of this much debated question in three steps.

Firstly, a general hypothesis describing the present link between biodiversity, ecosystem services and land use intensity is tested for the region of Flanders (13.682 km^2). The spatial exercise illustrates that even in a densely populated area as Flanders (539 inhabitants/km^2), scores for biodiversity and ecosystem services show a clear decline with an increase in land use intensity. Despite the overall congruence of the rough indicators of biodiversity and ecosystem services used in this exercise, the need for new indicators integrating key structural and functional aspects of the ecosystem is highlighted.

In a second step a target scheme is proposed describing the relation between biodiversity and land use intensity, for the long-term future. In this conceptual framework the need for the conservation and restoration of biodiversity hot spots is emphasized. At the same time the importance of the search for an environmental 'bottom line' is stressed. What is the minimum amount of biodiversity needed to reach sustainable use? And what is the critical point for human impact beyond which society will largely depend on the import of ecosystem services to sustain or increase its well-being?

In a third step, ecosystem management is used as a concept to bridge the gap between the present and the target scheme. Ecosystem management is divided into three focal zones. (1) For the remaining zones with a high biodiversity and a low intensity of land use, a biodiversity conservation based approach is needed. (2) For most rural areas with multifunctional uses and a good state as a future perspective, a focus on ecosystem services is a good point of departure. (3) In the built-up areas and intensively used agricultural sites, beyond the minimum standards for land use intensity, a technological service based approach will be necessary.

A division of ecosystem management into these three zones confirms the complementarity of biodiversity and ecosystem services for policy and management strategies. The concept of this triple division can help to facilitate the discussion of a joint achievement of the goals for biodiversity and ecosystem services in the field.

1. Introduction

Over the last century humans have been changing ecosystems more rapidly than in any comparable period in history (MA, 2005; Vitousek et al., 1997). As a result biodiversity, or the variety of genes, species and ecosystems, has dropped rapidly (Balmford et al., 2003; Balmford and Bond, 2005). Since the Convention on Biological Diversity in 1992 in Rio de Janeiro, 193 nations subscribed the goal of achieving a significant reduction of the current rate of biodiversity loss at the global, national and regional level by 2010. This commitment should also contribute to a more equitable access to natural resources and to poverty reduction. However, a clear statement about the linkage between biodiversity loss and human well-being was made by the Millennium Ecosystem Assessment (MA) in 2005. The MA indicated habitat loss, pollution, overexploitation, climate change and invasive species as the main drivers of ecosystem changes. These drivers induce structural and functional changes within ecosystems, resulting in a loss of biodiversity and of ecosystem services (ES). The MA broadly defined ES as the benefits people derive from ecosystems (MA, 2005). The loss of biodiversity is at the same time an effect of these ecosystem changes as well as a cause of further changes and a decline in human benefits (Diaz et al., 2006).

Since the Millennium Ecosystem Assessment, ES has become a popular research theme and a conceptual framework for many science projects. Various classifications, mappings and valuations

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of ES on global, regional and local scales are proposed. The valuation of ES in economic terms became an increasingly popular approach not only to assess alternative land use strategies but also to demonstrate and justify the need for the conservation of biodiversity (Bayon and Jenkins, 2010; Chan et al., 2007; Costanza et al., 1997; de Groot et al., 2002; Fisher et al., 2009; Ghazoul, 2007a, 2007b; Ridder, 2008; Wallace, 2007). Ecological research confirms the positive correlation between biodiversity and ES. In a review, many researchers agreed about the positive effect of biodiversity on ecosystem functions (Hooper et al., 2005; Diaz et al., 2006). Scientific arguments underpin that ecosystem properties depend greatly on biodiversity in terms of the functional characteristics of the organisms present in the ecosystem. Many hypotheses, based on diversity in relation to functional compensation and resilience, are formulated (Bengtsson et al., 2003; Hector and Bagchi, 2007; Naeem, 1998; Tilman et al., 1998; Walker, 1992). Scientists conclude that, due to an increase in spatial and temporal variability, a large number of species with different functional characteristics may act as insurance, buffering ecosystem processes and their services and making the ecosystem more resilient.

The positive correlation between biodiversity and ES however is not uncontested in the literature (Chan et al., 2007; Naidoo et al., 2008; Ridder, 2008). Only limited field experimental proof is found on the importance of species richness for ES (Diaz et al., 2006; Hector and Bagchi, 2007; Hooper et al., 2002). Only for cultural services like ecotourism, resources for medicines and ethical or aesthetical reasons and for some regulating services executed by keystone species, a direct link can be described with species richness and the presence of endangered species (Begossi, 1996; Elmqvist et al., 2010; Hooper et al., 2005; Swift et al., 2004; Thomas et al., 2008). Scientists argue that many services depend only on a limited amount of species (Diaz et al., 2006; Ridder, 2008; Srivastava and Vellend, 2010). Ecosystem functions and services such as water purification, timber and food production, carbon sequestration, largely depend on a few dominant species, one functional group of species or some specific interactions between species (Hooper et al., 2002, 2005; Luck et al., 2009). Examples are reed beds for water purification, trees for wood production, bees for pollination and great tit populations for the control of caterpillar damage. Some of these services can also be delivered by alien invasive species such as Zebra mussels (*Dreissena polymorpha*) for filtering particulates from water or Asian lady beetle (*Harmonia axyridis*) to control the spread of aphids (Redford and Adams, 2009). Many studies revealed only weak associations (positive and negative) between priority areas for biodiversity conservation and different ES (Bennett et al., 2009; Chan et al., 2006; Naidoo et al., 2008). The assumed independence of species richness, means that optimizing ES can potentially lead to a decline of that part of biodiversity that is not required to perform these ES (Hooper et al., 2005; Redford and Adams, 2009; Ridder, 2008). But due to the complexity of ecosystem functioning, there is a great uncertainty about the role of many species. Therefore the precautionary principle is often used as an argument to protect all species and to avert catastrophes (Daily, 2000; Ridder, 2008).

Despite this limited concordance, win–win cases, local ecosystem management projects strengthening both biodiversity and ES optimization goals, can easily be found (TEEB, 2010; http://www.eea.europa.eu/atlas). Especially in densely populated areas where species and habitat protection tends to focus on managing semi-natural systems in a human-dominated landscape, the ES framework can be used to underpin or even advocate the importance of biodiversity conservation (Ghazoul, 2007b). The protection of a common 'open space' and 'environmental quality goal' can be seen as an important synergy between biodiversity and ES. However, does this mean that both concepts are simply interchangeable? Are biodiversity conservation with the aim of preserving biodiversity and ES with the goal of sustaining human well-being, mutually beneficial or are trade-offs always inevitable (Ghazoul, 2007b)? In this article we attempt to answer this much debated question.

Starting from the hypothesis of a gradual decline of biodiversity and ecosystem services in relation to land use intensification (Braat and ten Brink, 2008; Schneiders et al., 1993) we try to test these trends with some indicators for the Flemish region. This region of 13.682 km^2 in the north of Belgium is an interesting case because of its high land fragmentation. The small nature remnants, often with high cultural-historical values are strongly interwoven with intensively used human landscapes. A whole range from protected natural areas to industrialised and built-up areas is present. The population density is among the highest in Western-Europe (539 people/km^2). Due to the limited open space and multiple uses, there is a great chance that an optimization of ecosystem services will require trade-offs with biodiversity goals.

After confirming the hypothesis, an attempt is made to describe a desirable relationship between biodiversity and land use intensity on the long-term. Ecosystem management is used as a key to bridge the gap between the current and future scheme. Management options can be divided in three complementary zones: (1) biodiversity conservation, (2) multifunctional sustainable use and (3) technological engineering.

2. Materials and methods

The relation between biodiversity, ES and land use intensity is analysed by a comparison of grid maps with estimated values for each of the three variables. The common resolution for each of the maps is grid cells of 4 km × 4 km, corresponding with the resolution of the grid map of the flora in Flanders, which has among the variables in our analysis the lowest resolution. Grid cells at the border that cover less than 50% of the study area are not used for the analysis in order to exclude bias of local effects ($N = 857$).

2.1. Biodiversity

The attempts to quantify changes in biodiversity are manifold. Most of the measures focus on species abundance expected on different scales: an ecosystem, a region, a continent, the world (Balvanera et al., 2006; Feld et al., 2009; Gotelli and Colwell, 2001; Hector and Bagchi, 2007). To evaluate these measures different scoring systems are developed. On an ecosystem level scores often calculate the deviation from an expected species list in undisturbed or pristine systems (Alkemade et al., 2009; Maes and Van Dyck, 2005; Witte, 2002). On a regional or continental scale the focus often shifts towards scoring systems measuring the risk for local biodiversity loss using endangered species or habitat lists. This is often illustrated by species with international priority for conservation action, including the species of European interest, protected by the Birds and Habitats Directives[1] or conservation of Red List species (IUCN red list: http://www.iucnredlist.org/) and SEBI 2010 indicators (Streamlining European 2010 Biodiversity Indicators: http://biodiversity.europa.eu/topics/sebi-indicators) (Mace and Baillie, 2007).

Red Lists indicate the conservation status of species and show which species may become extinct in the near future (Brooks and Kennedy, 2004; Mace et al., 2008). The standard methodology to calculate the species conservation status is proposed by the IUCN and is based on rarity and rate of decline. Although initially

[1] Council Directive 79/409/EEC of April 1979 on the conservation of wild birds and Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

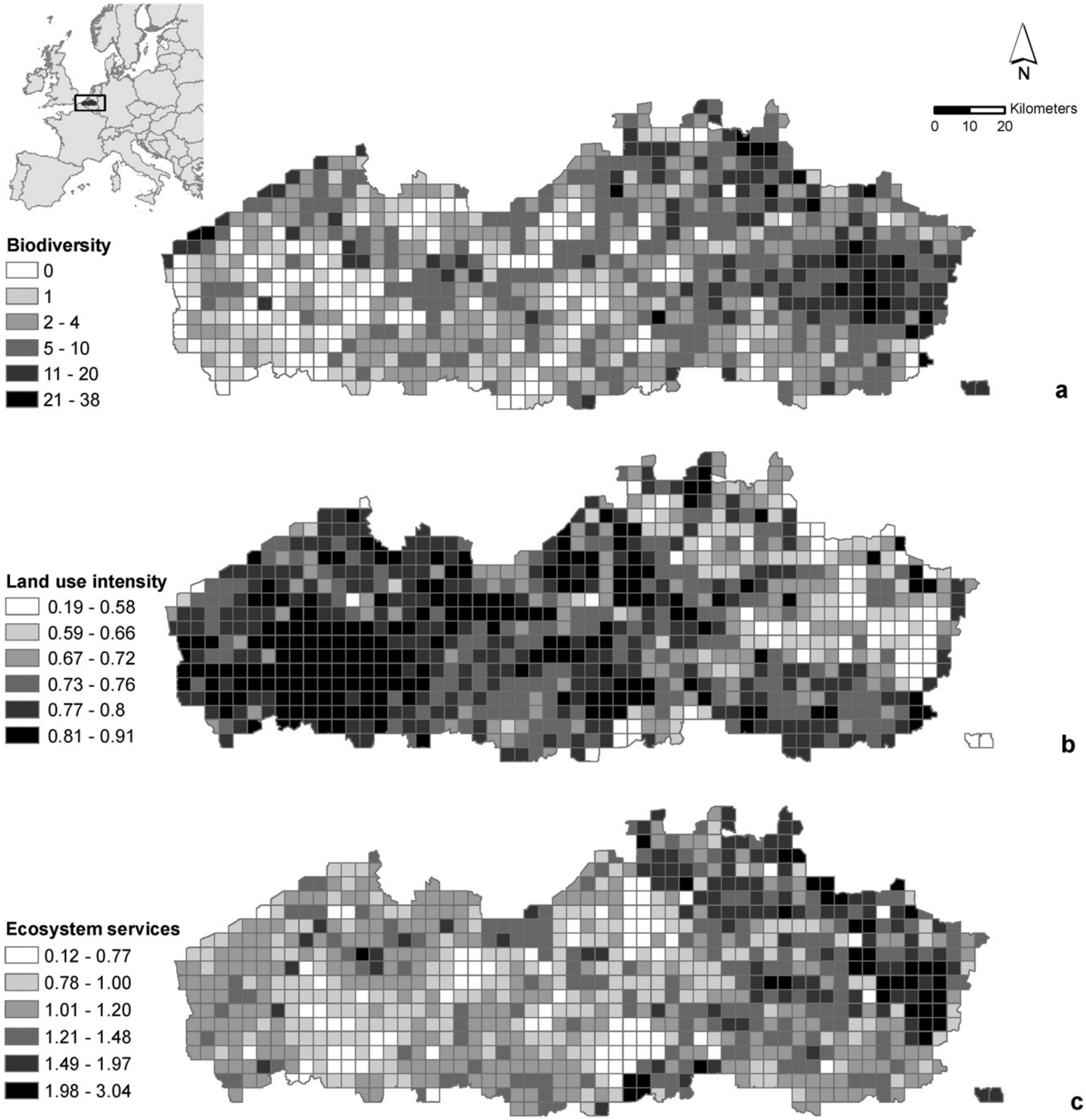

Fig. 1. Distribution of scores for biodiversity, land use intensity and potential ecosystem services in Flanders at a 4 km × 4 km resolution. (a) Biodiversity (number of Red List plant species per grid cell); (b) land use intensity score with a range from 0 (least intensive human use) to 1 (most intensive human use); (c) weighted mean ecosystem services score based on the ecosystem services ranking with a range from 0 (no landscape capacity to provide ES) to 5 (high landscape capacity to provide ES) (Burkhard et al., 2009).

the methodology has been developed for global purposes, the IUCN also provides guidelines for application of the methodology on a national or regional scale (IUCN Standards and Petitions Subcommittee, 2003). The IUCN regional Red Lists are an important tool for setting priorities for biodiversity conservation (Hoffmann et al., 2008; Rodrigues et al., 2006). The importance of Red List species in international and national policy context has induced an intensive monitoring system to classify Red List species and to look at their distribution.

In this case we used the accumulative number of vascular plant species of the regional Red List (Van Landuyt et al., 2006) recorded within each grid cell during the period 1972–2010 as a proxy for hot spots for biodiversity conservation (Fig. 1a). Within Flanders and the Brussels Capital Region, the geographical distribution of plant species is well known. Grid cells of 1 km^2 have been investigated over a long period of time, listing the presence of all wild vascular plant species observed. In each grid of 16 km^2 at least 4 smaller grids of 1 km × 1 km were prospected. The data were assembled into larger units of 4 km × 4 km to reduce the effect of spatial divergences in observation densities and recorder behaviour (Van Landuyt et al., 2008). Compared with other taxonomic groups of which the distribution in Flanders is well known such as amphibians and reptiles, butterflies, dragonflies, breeding birds and fish, vascular plant species are a rich taxonomic group. The number of species on the Red List of vascular plants outnumbers even the total number of species of each of these taxon groups. 108 species are listed as critically endangered, 69 as endangered and 40 species are vulnerable (Van Landuyt et al., 2006). This makes vascular plant

Table 1
Land use intensity scores (LUI) for different land use categories.

Land use type	Land use intensity
High nature value area with biodiversity management (including mud flats and salt marshes, coastal dunes, marshland, heathland, high nature value forest and grassland)	0.1
High nature value area without biodiversity management (including coastal dunes, marshland, heathland, high nature value grassland)	0.2
Multifunctional forest	0.3
Agricultural use with biodiversity target	0.4
Park, military facility	0.5
Agricultural use with environmental target	0.6
Recreation and sports terrain	0.7
Intensive agricultural use	0.8
Residential and commercial building, agrarian building, business and industrial site	0.9
Seaport, airport, infrastructure	1.0

species a good taxonomic group to use as a proxy for biodiversity and ensures that the overall result is not influenced by the ecological niche of a small number of species.

2.2. Land use intensity

Land use intensity scores are deduced from a land use map (150 m × 150 m resolution) with 27 categories (Gobin et al., 2009) (www.natureoutlook.be). The land use categories combine information on the main land cover type and the type of management (for example: grassland with nature management and grassland in agricultural use). This combined information makes it possible to group and rank the land use categories from 0 (low) to 1 (high), according to the intensity of human use and activities. The highest intensity score is attributed to large infrastructures such as airports and seaports followed by built-up areas with small green areas and intensively used agricultural sites. Intermediate scores are assigned to categories such as parks, agricultural sites with environmental measures, followed by agricultural sites with biodiversity measures (Aavik and Liira, 2009) and multifunctional forests. Low scores are assigned to sites with a high nature value like marshland and heathland. They are subdivided in protected and non-protected sites. The protected sites provided with a biodiversity management and restoration plan are considered to be the most natural sites with the lowest scores for land use intensity. An overview of the resulting land use intensity classes is given in Table 1. The range, based on expert judgement, is comparable with the intensity scale used in international exercises like GLOBIO 3 (Global Biodiversity Outlook 3) and COPI (Cost of Policy Inaction) (Braat and ten Brink, 2008; Alkemade et al., 2009).

The information of the land use map is upscaled to the coarse spatial resolution of the biodiversity map by averaging the land use intensity scores of all 150 m × 150 m cells within each 4 km × 4 km grid cell (Fig. 1b).

2.3. Ecosystem services

Many ES evaluation systems are being developed, for worldwide applications (Braat and ten Brink, 2008; Costanza et al., 1997) as well as for regional scales (Anderson et al., 2009; Burkhard et al., 2009). The spatial estimation of ES across all ecosystem types on a regional scale remains quite crude and difficult (Naidoo et al., 2008). It puts all services together with the same weight, while spatial relationships between service production and potential areas of benefits are often not taken into consideration (Turner et al., 2010).

For this exercise a general method for the assessment of multiple ES proposed by Burkhard et al. (2009) is used. The landscape capacity to provide ES was estimated by an ES ranking table. Based on expert judgement each land use category is ranked between 0 and 5 for each potential ES separately. The CORINE land cover classes used by Burkhard et al. (2009) could easily be linked to the categories of the Flemish land use map. Appendix A shows the rank scores for three ES groups: 'regulating ES', 'cultural ES' and 'provisioning ES'. The latter is subdivided in provisioning services linked with food (crops, livestock and fodder) and the remaining provisioning services (wild food, timber and wood fuel, biomass and energy, biochemicals and medicine, freshwater). An exhaustive list of the land cover types' capacities to provide the selected ES can be found online (http://www.landscapeonline.de/archive/2009/15/). The number of ES considered is very different between the ES groups. While regulating and provisioning services are based on 9 services each, the cultural ES group is only based on two services. In order to give equal importance to each of the three ES groups, the mean ES score is calculated within each of the ES groups. Based on the mean scores of the ES groups, a weighted mean ES score is calculated.

The information of the ES capacity map is upscaled to the coarse spatial resolution of the biodiversity map by averaging the ES scores of all 150 m × 150 m cells within each 4 km × 4 km grid cell (Fig. 1c).

2.4. Analysis

The average value of land use intensity and ES scores per grid cell of 4 km × 4 km was calculated with the zonal statistics tool of the spatial analyst extension in ArcMap 9.2 (ESRI, 2006). This tool calculates the average of all cells in the 150 m × 150 m grid that belong to the same 4 km × 4 km grid cell.

The statistical analysis was carried out with S-Plus 6.2 (Insightful Corp. 2003). Since not all the data are normally distributed, the non parametric Spearman rank correlation was used to calculate the correlations between the variables (Table 2). The software Mod_t_test (Legendre, 2000) was used to take into account the effect of spatial autocorrelation between the 4 km × 4 km grid cells (N = 857). This software employs the modified t-test method (Dutilleul, 1993) to correct the p-values by reducing the degrees of freedom in presence of spatial autocorrelation. The software can be used for both parametric Pearson correlation coefficients and non parametric Spearman rank correlations.

For presentation purposes the land use intensity, the mean ES score and number of Red List species in the legend of Fig. 1, and X-axes of Figs. 3 and 4 are binned into 6 classes. The classes were defined with a one-dimensional k-means clustering. The 6 classes for the Red List plant species in Figs. 1 and 4 were manually adapted in order to differentiate a separate class for 4 km × 4 km grids cells without any Red List species.

3. Results and discussion

Intensification of land use has led to a decline in biodiversity (Anonymous, 2010; MA, 2005). A schematic presentation of this gradual degradation is given in Fig. 2 (adapted after Schneiders et al., 1996). According to the parameters used for the calculation of the biological response to human impact, the relationship can be linear, exponential, threshold or sigmoid (Braat and ten Brink, 2008; Hooper et al., 2002, 2005; Karr and Chu, 1999; Norris and Thoms, 1999; Schneiders et al., 1996; Scholes and Biggs, 2005; Vandermeer et al., 2002). Many scientists from different disciplines have been trying to define the highest score for biodiversity corresponding with the pristine or natural state or state of reference (Karr, 1991; Karr and Chu, 1999; O'Keeffe, 1997; Schneiders et al., 1993, 1996; Vannote et al., 1980). Definitions for this natural state that have already entered the policy process are the indexes of integrity or 'high ecological status' developed for the European

Table 2
Pairwise correlation coefficients (Spearman rank correlation). All *p*-values are corrected for spatial autocorrelation using).

	Biodiversity: (number of Red List plant species)	Land use intensity	Ecosystem services					
			Cultural services	Provisioning services	Provisioning ES (food)	Provisioning ES (non-food)	Regulating services	Total mean ecosystem services
Biodiversity	1	**−0.58*****	**0.58*****	**−0.28***	**−0.47*****	**0.23***	**0.45****	**0.36****
Land use intensity		1	**−0.92*****	0.01	**0.36***	**−0.69*****	**−0.92*****	**−0.84*****
Cultural ES			1	**−0.20***	**−0.54****	**0.59*****	**0.88*****	**0.76*****
Provisioning				1	**0.87*****	**0.55*****	0.16	**0.40*****
Provisioning (food) ES					1	0.16	−0.22	0.01
Provisioning (non-food) ES						1	**0.84*****	**0.95*****
Regulating ES							1	**0.93*****

Significant values are in bold.
* $p < 0.05$.
** $p < 0.01$.
*** $p < 0.001$.

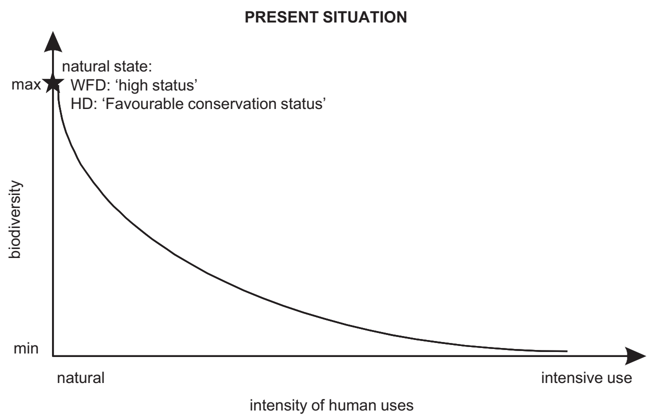

Fig. 2. Influence of human use on biodiversity: the present situation (adapted after Schneiders et al., 1996) WFD: European Water Framework Directive; HD: European Habitats Directive.

Water Framework Directive[2] or the definition of 'favourable conservation status' for the species and habitats of European interest (Habitats Directive, 92/43/EEC). At this time, most European member states are implementing and intercalibrating these new indicators (Breine et al., 2004).

The same generalised relationships are expected between most ES groups and the degree of land use intensity. Most ES are expected to show a gradual decline due to intensification of the land use. Only for food provisioning services another pattern (optimum curve) is postulated (Braat and ten Brink, 2008).

3.1. Case study Flanders

The hypothesis described in Fig. 2 is tested for the Flemish region, a very densely populated area. Sine the new set of integrity scores is not yet available on a regional scale for all ecosystem types, alternative scoring systems are proposed. Fig. 1 shows the three basic maps for (1) biodiversity, (2) ecosystem services and (3) land use intensity.

A first comparison of these maps suggests certain congruence between the Red List and ES indicator within the Flemish region. Land use intensity shows the opposite pattern. The contrasting zones within Flanders correspond well with the biogeographical classification (Van Landuyt et al., 2011). The northeastern part is characterised by a sandy region with a relatively low intensity of land use and with multifunctional forestry alternated with heathland and small rural centres. It is a zone with a high amount of Red List plant species and high ES scores. The south-west is a large sandy-loamy part with intensive agricultural sites. This part is characterised by a low amount of Red List species and low to moderate scores for ES, depending on the percentage of agricultural land. The central part of Flanders is characterised by large urban areas such as Brussels. Low ES are combined with low to moderate scores for Red List species. The number of the Red List species in these areas is

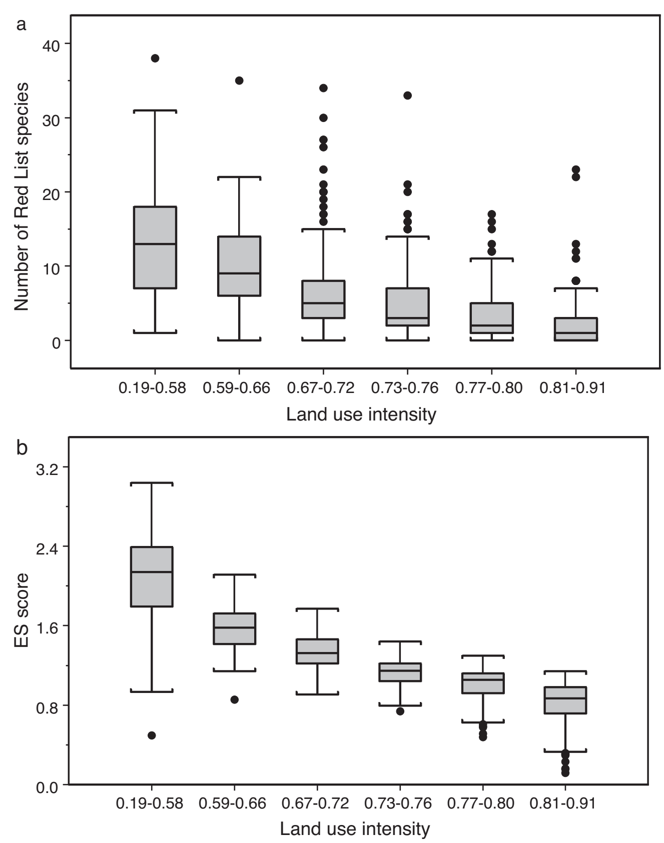

Fig. 3. Box-and-whisker diagram showing the relation between land use intensity (binned in 6 classes) and (a) biodiversity (number of Red List species) or (b) ecosystem services score (total weighted mean of provisioning, regulating and cultural services). The box indicates the interquartile range and the centre horizontal line the median value. The whisker caps indicate 1.5 times the interquartile range and the black dots are outliers.

[2] Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy.

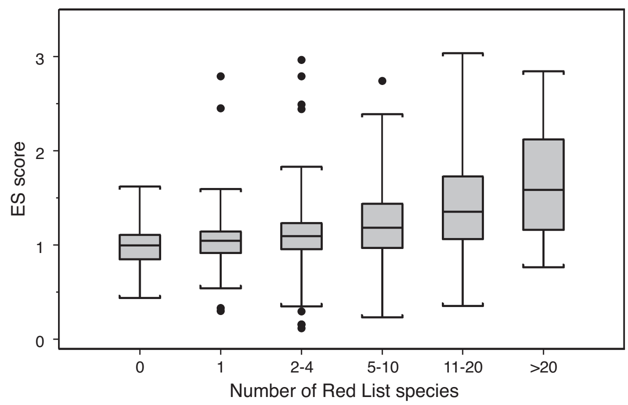

Fig. 4. Box-and-whisker diagram showing the relation between biodiversity (6 classes of Red List species) and total mean score for ES. The box indicates the interquartile range and the centre horizontal line the median value. The whisker caps indicate 1.5 times the interquartile range and the black dots are outliers.

affected by the large variety of microhabitats, soil types or specific urban dynamics (e.g. casual species arrived by transport).

The expected (dis)similarities of the three maps are confirmed by the graphs in Fig. 3 and by the correlation matrix (Table 2). Both the biodiversity and the ES scores show a clear gradual decrease with the intensification of land use, consistent with the hypothesis described in Fig. 2. Does this general pattern of decline implies that a decrease in biodiversity is automatically linked to a reduction in ES? Fig. 4 illustrates the link between the number of Red List plant species and the corresponding total mean ES score for Flanders. In general, areas with a high amount of Red List species indeed show a higher potential to generate ES. However, this pattern does not count for all ES groups. This is illustrated in Fig. 5. Regulating and cultural services have their optimum in (semi-)natural areas, with a gradual decline related to intensification of land use, parallel with the decline in biodiversity (Table 2, Fig. 5a and b) (Braat and ten Brink, 2008). Both ES groups are also correlated. The provisioning services score, according to Burkhard et al. (2009), is a combination of food (like livestock and crops) and non-food scores (wood production, biomass for energy production, freshwater production and genetic variation for medicines and agriculture). To show the difference in pattern, we split the provisioning group into a food and non-food part (Fig. 5c and d). The group of the food related provisioning scores shows a negative relationship with biodiversity and also with the other ES groups. This is also found in the literature (Hooper et al., 2005; Vandermeer et al., 2002). Intensification of agricultural land use will raise productivity to a certain optimum level, but always with a potential decline of other services (Braat and ten Brink, 2008).

These positive correlations between regulating ES, cultural ES and biodiversity are only based on the evaluation of the potential supply side for ES and not the demand side. The ES-score therefore does not present a direct measure of the actual contribution to human well-being, which would be affected by the spatial distribution of beneficiaries, patterns of population density and the availability of substitutes for the ecosystem services (Turner et al., 2010). On the other hand one can expect that in a generally densely populated area such as Flanders the distance between demand and supply will always be bridgeable.

Despite the general congruence between biodiversity and ES supply, both scoring systems are not interchangeable. Due to the rareness (and often the absence) of the group of Red List species

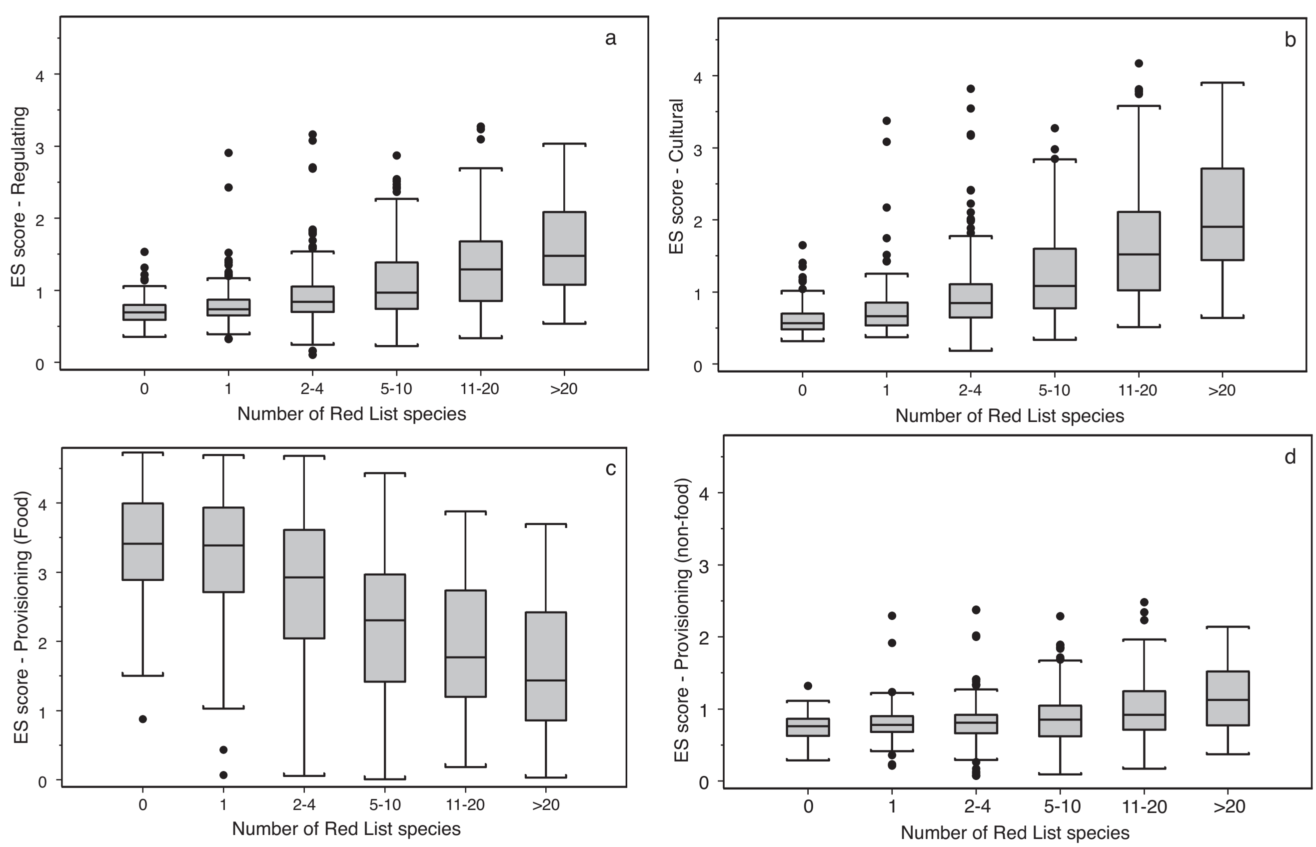

Fig. 5. Box-and-whisker diagrams showing the relation between biodiversity (number of Red List plant species) and (a) mean score for regulating services, (b) mean score for cultural services, (c) mean score for provisioning services (food) and mean score for provisioning (non-food). The box indicates the interquartile range and the centre horizontal line the median value. The whisker caps indicate 1.5 times the interquartile range and the black dots are outliers.

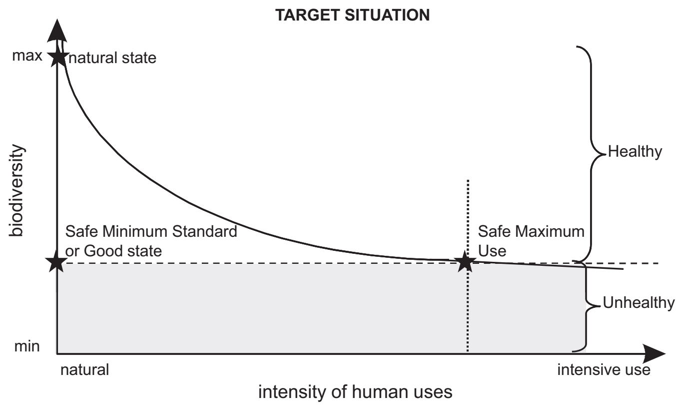

Fig. 6. Influence of human use on biodiversity: the target situation (adapted after Schneiders et al., 1996).

these scores cannot be used in all areas. In addition, ES are usually not provided by rare or endangered species. Often common and tolerant species, species that are resilient in the face of change or a specific group of species fulfilling certain functional criteria will fulfil most of the ES. This holds for ES such as water purification, timber and food production, carbon sequestration, etc. A number of examples illustrate that specific species extinctions have not resulted in an ecosystem collapse or in a fundamental decline of ES (Ridder, 2008). On the other hand, within a natural or a biodiversity hot spot a large supply for ES can often be expected.

Using new sets of biodiversity indicators such as the integrity scoring systems, and ES scores including the demand side, will help to better understand the synergies or trade-offs between the goals for biodiversity conservation and human benefits via ES.

3.2. Towards a conceptual framework for sustainable use

In Fig. 2 all attention is focused on the definition of the natural state or the 'highest' biodiversity level and on the distance to that target. In order to stop the loss of biodiversity this is an important point of discussion. For sustainable use, a consensus must be reached about the 'lowest' biodiversity score or the 'lowest' ES level accepted by society (Fig. 6). In the present situation (Fig. 2) the intensity of land use is often too high to allow sustainable use. Many sites are degraded or polluted. More and more the need for a new bottom line or 'safe minimum standard of conservation' (SMS) is recognised (Bishop, 1978; Ciriacy-Wantrup, 1968; Crowards, 1998; Fisher et al., 2008; Turner et al., 2010). SMS refers to an ecological threshold beyond which ecosystem changes may be irreversible, or may impose unacceptable social or economic costs. The SMS threshold increases with higher uncertainty regarding the underlying ecosystem dynamics (Fisher et al., 2008; Hein, 2010).

This bottom line or SMS is very difficult to define, and not only because of the complexity of the underlying ecosystem dynamics. It may also be located at different ecological levels, depending on the ecosystem service that is considered. In that sense it is also based on a value judgement and a political choice of which ecosystem services matter most to society. The decision must therefore be embedded in a broad understanding of the ecological, economic and social consequences of loss. The ecosystem exploitation or intensity of land use should not undermine the self-repairing capacity of the system. On the other hand, setting limits to ecosystem exploitation should also not impose 'unacceptable' social costs to specific stakeholders (Bishop, 1978). The valuation system should include ecosystem processes that are crucial for sustainability (Karr and Chu, 1999; Rapport et al., 1998).

All ecosystems above this SMS can be considered as healthy, below it as unhealthy. A healthy ecosystem should allow multifunctional use, providing ES and human well-being, not only for the current but also for future generations. It refers to a choice of alternatives within the envelop of desirable, sustainable states (Karr and Chu, 1999; Rapport et al., 1998). According to the European Water Framework Directive, the bottom line tolerated is called the 'good state' that should be reached in all aquatic ecosystems by 2015 (2000/60/EC). In a range of 0 (totally degraded system) to 1 (high or natural state), the good state is defined as a score of 0.6. European countries are now developing and intercalibrating biological indicators to define this bottom line. This 'good state' can be linked with the construct of a 'safe minimum standard' that is also used to define sustainable use of terrestrial ecosystems (Hein, 2010).

Together with the 'safe minimum standard' defined on the *Y*-axis, we should define a 'safe maximum use' for land use intensity on the *X*-axis. This point indicates the maximum tolerable human impact or the limit for sustainable ecosystem use. Beyond this point the biodiversity drops below the safe minimum standard and the stability and self-repairing capacity of the system are (or may be) undermined (Karr and Chu, 1999). Passing this critical point means entering a human dominated system. In such an environment you still can create a lot of ES, but these services will not determine the basic functioning of the human made system. These systems will always depend largely on ES import.

Many examples of these safe minimum standards can be defined. It can be the minimum area of wetlands needed to ensure flood control; or the maximum amount of fertilizer that can be used to avoid eutrophication and allow the good state in the river network within the watershed; or the marsh vegetation required for water purification; or the number of birds needed to reduce plagues of insects, etc. Mitigation measures can help to create a better functioning of ES in the man-made environment (such as building ecoducts, fish passages, green roofs, rainwater infiltration zones, using nest boxes for predatory birds) and they can shift the threshold value to the right.

Defining a safe minimum standard partly depends on what society tolerates. For example, it can be the point below which wild bee populations can no longer guarantee pollination services. Or it could be shifted (lower on the *Y*-axis and more on the right side of the *X*-axis) towards the point where farmers rely on managed honeybees. The critical point is then shifted towards the survival and reproduction capacity of the honeybees (Ghazoul, 2007b).

3.3. The role of ecosystem management

To bridge the gap between the present (Fig. 2) and the target situation (Fig. 6) ecosystem management is needed. According to Brussard et al. (1998) ecosystem management is managing areas at various scales in such a way that ecological services and biological resources are restored and conserved, while appropriate human uses are sustained. For a specific geographic area it is driven by explicit goals, based on the integration of ecological and social information, options and constraints (Christensen et al., 1996; Grumbine, 1994).

Starting from the three important points in Fig. 6 (high status, safe minimum standard, safe maximum use) we find arguments to divide ecosystem management into three focal zones (Fig. 7). In practice, they are likely to overlap to some extent. This threefold partitioning is not new; it was already recognised by Odum (1971). He provided the arguments to distinguish three major compartments: a protective environment, a productive and an urban-industrial environment. Among these compartments flows of energy and movement of organisms and materials take place.

In zone A of Fig. 7, ecosystem management focuses on the protection and restoration of areas with a high biodiversity value. This

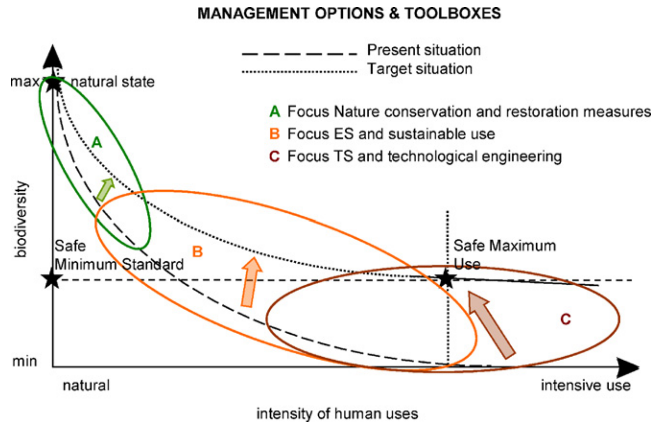

Fig. 7. Management tools for restoration and protection of biodiversity and ES (TS: technological services) case Flanders: zone A = 3%; zone B = 70%; zone C = 27% of the area.

is the core business of biodiversity conservation and the protection of intrinsic values. A whole package of policy instruments already exists. Natura 2000 is a European Network to protect and restore these 'islands of biodiversity' within an urbanizing and intensifying landscape in order to develop a network across Europe. On a lower scale (country or regional scale) high biodiversity value areas of national or regional importance are selected. In most countries a purchasing policy as well as a management and restoration policy for protected areas is developed. Within Flanders about 3% of the area has the status of nature reserve and about 6% is designated as Natura 2000 area.

Most natural sites have a high capacity to deliver ES, especially regulating and cultural services. They serve as genetic sources or suppliers of wild pollinators for the surrounding areas. Restoring and managing areas with a high biodiversity will support and enhance the natural ecosystem functioning, the environmental quality and amenity values. Restoration tools such as raising the ground water level in marshlands can increase the capacity of carbon storage or stimulate the self purification capacity of the area, etc. Valuing and visualising all these ES can help to increase the public support for the biodiversity restoration projects. But also the potential of disservices, and associated protests from stakeholders, should be mentioned. Certainly in densely populated areas it is possible that these natural sites deliver species causing economic losses for farmers (e.g. wild boar, starling, goose, etc.).

Zone B consists of multifunctional landscapes. In Flanders about 70% of the area belongs to this category. Here the main goal shifts from the restoration of the high state towards the restoration of the good state. This part of the curve is the playground of ES. The restoration programmes here are not limited to small islands with a high biodiversity value. Good state is a general goal for all zones in the open space. As mentioned above, this good state will always be influenced by local values for society and by political choices. The focal species in zone B are not necessarily endangered species, but species that play an important role in fulfilling the ES. Luck et al. (2009) defined these species as the 'Service Providing Units' or SPU. Examples are reed beds for water purification, trees for timber production, birds for pest control, etc. Most of the time these service providers are abundant or even dominant species. They determine the basic functioning of the ecosystem. But the more functions included, the higher the species richness needed to fulfil al these functions (Hector and Bagchi, 2007). Also parts of Natura 2000 areas, like the river Scheldt, are multifunctional landscapes belonging to zone B.

Carbon sequestration, food production, pollination, recreation, etc. all work on different spatial and temporal scales and the advantages and disadvantages of these services are not evenly spread across all stakeholders. The service producers might be separated in space and time from the beneficiaries (Chan et al., 2007). Local stakeholders do not necessarily refer to the same scale or values as the restoration managers or conservationists. Visualizing trade-offs among ES and the distribution of their social and economic impact can serve as a basis of negotiation and discussion among scientists, policy makers, community representatives and stakeholders (Corck and Proctor, 2005; Cowling et al., 2008; Ghazoul, 2007b).

Zones A and B are strongly interrelated. Nature management in zone A will enhance the natural functioning and the related ES in zone B. The improvement of the good status in zone B creates extra opportunities to reach the specific environmental goals within conservation areas of zone A needed for the protection of certain endangered species or habitats. Especially small conservation areas often suffer from environmental disturbance and degradation, making it impossible to restore part of the biodiversity goals. Investments in green infrastructure can not only generate an economic return within the rural area of zone B, increasing the amount of ES, but they will also create a migration network between the conservation sites of zone A and enhance the chances of creating more stable meta-populations (Aavik and Liira, 2009; Bobbink et al., 2010).

Zone C is situated in the most industrialised or built-up area or the most intensively used agricultural sites. In Flanders about 27% belongs to the first two categories. Depending on where to draw the critical point for agricultural use this part can increase up to 70% of the region. Within this zone the natural self repairing capacity of the ecosystems is exceeded and ES are low. Trees, parks, gardens, etc. increase the socio-economic viability of a city. They play a central role in many cultural services. And although the role of the garden network in the ecological functioning of urbanized regions may be underestimated, we expect built-up areas to depend largely on imported ES (Duvigneaud, 1974). In these areas the focus lies on services that make use of modern technology. They include the food chain, the delivery of energy, the distribution of potable water directly into homes, removing wastes such as sewage and trash from homes, etc. (Cairns, 1996). The focus in this part of the curve lies on technological innovation in order to lower the ecological footprint and to lift the present situation to a more sustainable level. Building passive houses, using solar or wind energy, changing lifestyles, all these measures lower the demand for the external import of ES. Certainly in densely populated regions these technological services penetrate far into the open space. Nest boxes help to reduce plagues in orchards as well as in cities. Cultivated beehives are used to pollinate many crops and fruit trees, cultivars are introduced to increase yields, etc. These kinds of technological interventions can even help to preserve certain ES in rural areas. Services such as biogas, hydro-electric power installations and windmills in rural areas deliver energy for the cities and they are more closely connected to the technological working space of a city or an industrialised area. The borderline between zones B and C is not always sharp and large overlaps are possible (and even advisable). A broader analysis of these technological services is not the scope of this article. We only included it as a necessary third pillar, in order to complete the story line.

Recognising these three focal zones does not mean that this concept of ecosystem management will solve all conflicts between the goals for biodiversity and ES. Certainly in a densely populated area like Flanders, the overlap is quite large so that conflicts and trade-offs remain. Examples are the pressure of recreation in nature protected areas, game damage in farmland, conflicts about the locations for windmills or hydro-electric power stations, etc. However, despite the potential conflicts the spatial arrangement can strengthen both strategies. An interesting research question is whether the nature network (with a mix of zones A and B) developed in many countries to halt biodiversity loss is the same as the resilient network needed to optimize a whole set of ES in order to reach a more sustainable land use. In addition they should both consider the opportunities as well as the potential conflicts with the technological services to reduce the impact of the built-up areas.

4. Conclusion

This paper started from the hypothesis outlined in Fig. 2 that biodiversity and ecosystem service supply both decline with land use intensification. We tested this hypothesis and based on the outcome we designed a new conceptual framework for a sustainable future.

The Flanders case confirmed that even in a very densely populated area, using rough indicators, an overall decline of biodiversity and ecosystem services in relation to land use intensity is expected. However, for the provisioning services, based on food production, trade-offs between biodiversity and ES are likely to happen. Despite the overall congruence with ES, the biodiversity indicator used has specific restrictions. Red List species are often linked with very specific and uncommon environments and are often absent. Together with this type of biodiversity indicators, other indicators highlighting the key structural and functional variables of the ecosystems should be developed. Such indicators are already partly worked out for the European Water Framework Directive. Using common metrics to measure and value biodiversity and ES will help to find a better alignment of both concepts.

Starting from the present framework a (desired) future framework is designed. Fig. 6 describes the relation between biodiversity and land use intensity on the long term. The curve shows three important points of reference: (1) the highest score for biodiversity, defined as the 'natural' state'; (2) the minimum score or 'safe minimum standard of conservation' that should be reached to ensure sustainable use, linked with (3) a 'safe maximum use', beyond which ecosystems and societies become largely dependent on import of ecosystem services. Developing the new set of biodiversity indicators it is important to define clearly this minimum score tolerated by society. Again the concept of 'good status' according to the European Water Framework Directive can serve as an example.

In the last part, we tried to bridge the gap between the present and the future state, using the concept of ecosystem management (Fig. 7). Looking at the different steps of ecosystem restoration, we find arguments to divide ecosystem management into three focal zones. Small patches focussing on restoration and management of the high ecological status are surrounded by large rural zones focussing on the concept of ES for the implementation of a sustainable use reaching the good state and by built up areas focusing on lowering the ecological footprint. These three zones do not only overlap, the management type proposed for each zone will contribute in reaching the goals in the other zones. A division of ecosystem management into these three zones can serve as a strategy to facilitate the joint achievement of the goals for biodiversity and ES in the field.

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Appendix A. ES values for each land use type. Adapted from (Burkhard et al., 2009). The number of services considered within each group is indicated between brackets

Land use map (Gobin et al., 2009)	Land use (CORINE)	Regulating (9)	Provisioning			Cultural (2)	Mean[a] (20)
			Provisioning all (9)	Provisioning food (3)	Provisioning non-food (6)		
Agricultural fabric and infrastructure	Discontinuous urban fabric	0.0	0.3	0.7	0.2	0.0	0.1
Agricultural grassland	Annual and permanent crops	0.8	2.3	5.0	1.0	0.5	1.2
Agricultural grassland (with an environmental or a biodiversity target)	Annual and permanent crops	0.8	2.3	5.0	1.0	0.5	1.2
Agricultural sites (not registered)	Pastures	0.8	2.3	5.0	1.0	0.5	1.2
Airport	Airports	0.0	0.1	0.3	0.0	0.0	0.1
Beaches and dunes (no biodiversity management)	Beaches, dunes and sand plains	0.7	0.2	0.0	0.3	3.5	1.5
Beaches and dunes (biodiversity management)	Beaches, dunes and sand plains	0.7	0.2	0.0	0.3	3.5	1.5
Business and industrial site	Discontinuous urban fabric	0.0	0.3	0.7	0.2	0.0	0.1
Cropland	Annual and permanent crops	0.8	2.3	5.0	1.0	0.5	1.2
Cropland with and environmental target	Annual and permanent crops	0.8	2.3	5.0	1.0	0.5	1.2
Cropland with biodiversity target	Annual and permanent crops	0.8	2.3	5.0	1.0	0.5	1.2
Forest (biodiversity management)	Mixed forest	4.3	1.7	0.3	2.3	5.0	3.7
Heathland (biodiversity management)	Moors & heathland	2.2	1.1	0.7	1.3	5.0	2.8
Heathland (no biodiversity management)	Moors & heathland	2.2	1.1	0.7	1.3	5.0	2.8
high nature value grassland (no biodiversity management)	Natural grassland	2.4	0.6	1.0	0.3	3.0	2.0
High nature value grassland (biodiversity management)	Natural grassland	2.4	0.6	1.0	0.3	3.0	2.0
Infrastructure	Road and rail networks	0.0	0.0	0.0	0.0	0.0	0.0
Marshland (biodiversity management)	Inland marshes	1.6	0.2	0.7	0.0	0.0	0.6
Marshland (no biodiversity management)	Inland marshes	1.6	0.8	2.3	0.0	0.0	0.8
Military facilities	Discontinuous urban fabric	0.0	0.3	0.7	0.2	0.0	0.1
Mud flat and salt marshes	Intertidal flats	0.8	0.0	0.0	0.0	2.0	0.9
Multifunctional forest	Mixed forest	4.3	2.3	0.3	3.3	5.0	3.9
Park	Green urban areas	1.2	0.2	0.0	0.3	1.5	1.0
Port area	Port areas	0.3	0.0	0.0	0.0	0.5	0.3
Recreation and sports terrain	Sport and leisure facilities	1.0	0.0	0.0	0.0	2.5	1.2
Residential and commercial fabric	Continuous urban fabric	0.0	0.0	0.0	0.0	0.0	0.0
Water	Water courses	1.1	1.0	0.0	1.5	5.0	2.4

[a] Mean = mean value of regulating, provisioning all and cultural.

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