Ecological Indicators 21 (2012) 134–144

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Original article

Delphi-based change assessment in ecosystem service values to support strategic spatial planning in Italian landscapes

Rocco Scolozzi[a,*], Elisa Morri[b], Riccardo Santolini[b]

[a] Sustainable Agro-ecosystems and Bioresources Department, IASMA Research and Innovation Centre, Fondazione Edmund Mach, Via E. Mach 1, 38010 San Michele all'Adige, (TN), Italy

[b] Department of Earth, Life and Environment Science, University of Urbino "Carlo Bo", Campus Scientifico "E. Mattei", 61029 Urbino, Italy

ARTICLE INFO

Article history:
Received 1 December 2010
Received in revised form 18 May 2011
Accepted 21 July 2011

Keywords:
Ecosystem services
Spatial planning
Environmental Impact Assessment
Strategic Environmental Assessment
Delphi survey

ABSTRACT

Most impacts on providing ecosystem services (ESs) are related to land use changes that may cause ecosystem fragmentation and loss of ecosystem functions. Spatial planning, focusing on sustainable landscape development, should consider the local potential for providing ESs as well possible impacts due to land use changes. The results of a large number of recent studies on ES assessment are difficult to replicate and integrate into landscape planning or definition of land use policies. The time and resource constraints, needed in data gathering and analysis, limit local administrations in the concern about ES.

Extending the approach of spatially explicit benefit transfer, we present an expert-based estimation of land use potential in providing ES. We enhanced ES estimations related to different land use classes (in terms of €/ha year) by considering "local conditions" such as elevation and distance from urban areas, assuming these spatial characteristics can affect ES provisioning. The adaptation for "local conditions" was performed through expert consultation, using Delphi survey and focus groups, involving 46 experts from 10 Italian research institutions. In detail, we used land cover maps for 1990 and 2000, and aggregated the results on the province scale. The results show spatial pattern of gains and losses in ES values for Italian provinces. The province level of aggregation seems to be effective for dealing with strategic phase of spatial planning. By eliciting possible consequences of land use changes, this knowledge framework may foster strategies for sustainable landscape planning and management.

1. Introduction

Over the past 50 years, humans have altered ecosystems more rapidly and extensively than in any comparable period in human history and wider impacts are expected (Daily, 1997; Millennium Ecosystem Assessment, 2003; Vihervaara et al., 2010). These changes are mainly related to land use changes and their impacts on adjacent lands. The CORINE land cover database shows significant changes in land use for Europe which have an impact on soil (EEA, 2006). The soil sealing is becoming a political issue (Blum et al., 2004) acknowledged a "serious problem" (EC, 2006). Between 1950 and 1990, in many regions of Italy, more than 20% of natural and agricultural land covers were sealed by urbanization and infrastructure development (EEA, 2002), and recently land consumption is occurring at an increasing rate (Barberis, 2006; Montanari et al., 2009). Little attention is paid to this theme, although it is so important for guiding spatial planning towards a more sustainable approach (Pileri and Maggi, 2010). Meanwhile, recent floods and landslides in many Italian regions call for better landscape management and protection (as claimed also by daily newspapers in 2010).

Considering the ecosystem services (ESs), potentially provided by the land uses/land covers (LC), is acknowledged as a necessary framework for linking human and natural systems in environmental management (Balvanera et al., 2001; Balmford et al., 2002; Howarth and Farber, 2002; Daily et al., 2009). The service approach to territorial management and planning entails a consideration of multi-functionality of LC, and of trade-offs between functions (Kienast et al., 2009; Nelson et al., 2009; Bolliger et al., 2010). The ES assessment may effectively support societal and political choices in the process of spatial planning (Farber et al., 2002), from data gathering to analysis, from participative assessment to ex-post appraisal, raising awareness of possible impacts on ecosystem goods and services due to land use changes (Kreuter et al., 2001; Wang et al., 2010; van Oudenhoven et al., 2012), helping to avoid or compensate these impacts (Deutsch et al., 2003). Although several studies are being internationally and nationally developed (an inventory is reported in www.natureavaluation.org), ES evaluations still seem to be lacking in impact assessments of land use changes (de Groot et al., 2010). We expect that ES assessment may play an important support to Strategic Environmental Assessment (SEA) as well to other environmental assessments.

* Corresponding author. Tel.: +39 3471 096921; fax: +39 0461 650956.
E-mail addresses: rocco.scolozzi@iasma.it, scolozzi.rocco@gmail.com (R. Scolozzi).

1470-160X/$ – see front matter © 2011 Elsevier Ltd. All rights reserved.
doi:10.1016/j.ecolind.2011.07.019

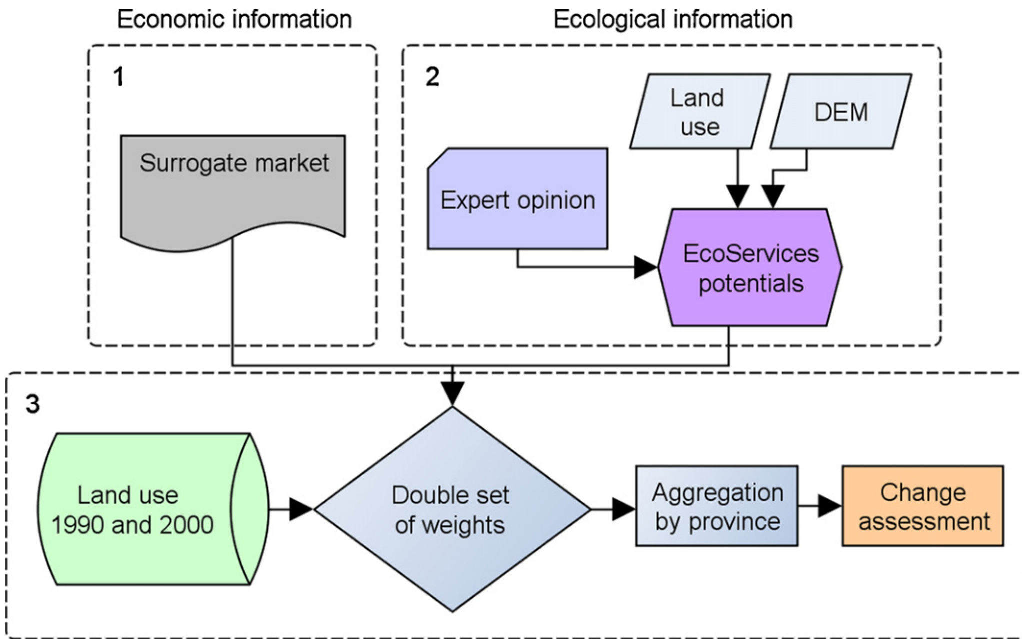

Fig. 1. Methodological steps.

The assessment and modelling of ESs is recently focused by an extensive research effort, after the seminal work by Costanza et al. (1997) the number of publications on ecosystem services has been increasing almost exponentially (Fisher et al., 2009), notwithstanding limits and criticisms (e.g. Norgaard and Bode, 1998; Toman, 1998; Gatto and De Leo, 2000). Several studies have been dedicated to ESs classification (e.g. de Groot et al., 2002; Millennium Ecosystem Assessment, 2005; Wallace, 2007; Costanza, 2008; Bastian et al., 2012), but the perspectives and concepts related to ESs are still debated (Boyd and Banzhaf, 2007; Fisher and Turner, 2008). The task is difficult due to the complexity of ecosystem connecting spatio-temporal dynamics (Kremen, 2005) on one hand, and on the other hand the socio-economic aspects related to the value definition and assessment (Howarth and Farber, 2002; Fisher et al., 2008; Busch et al., 2012). These may imply policy discourses, concerning multiple and rival goods and objectives, institutional issues, such as property regimes and user systems within social adaptive systems (Berkes et al., 2000), as well as cultural and psychological issues (Kumar and Kumar, 2008). Each ES assessment approach has a degree of uncertainty and generalization in terms of scale issues, specific method weakness (Plummer, 2009; Johnston and Rosenberger, 2010), drawbacks due to generalizations (Rosenberger and Stanley, 2006; Lindhjem and Navrud, 2008), imprecise classification or definition (Boyd and Banzhaf, 2007), economic inadequacy of results (Costanza et al., 1998).

The degree of precision and the amount of required data depend on the level and target of decision-making process we wish to support. Land cover, land use changes and ecosystem management decisions are all characterized by defined decision contexts, characterized by particular institutional setting, in terms of spatial level of governance, kind of institution (norms, rules, and laws), relevant interests, and involved stakeholders. For different decision contexts, different kinds of information with different degrees of precision may be beneficial; accordingly we can define a different degree of “acceptable” uncertainty in the results. For instance, strategic planning formulations do not require an exact budgeting of single ESs for the territory of concern, even if feasible. Assessment of ESs values for a region or province may constitute a reference in natural capital accounting and these may help in identifying the potential costs of losing present ESs. In this particular case, it is important to interpret potential losses or gains of ES, caused by land use changes, in monetary terms. This information may be really effective in cost–benefit valuation, enhancing discussion within planning and definition of priorities in environmental policies.

In this paper, we present an approach for an evaluation of ESs on the provincial level, aiming to put the issue on the decision table of spatial planning. The objective is to promote consideration of the ES provided by a territory and potential losses due to urbanization. We expect that this land use change impact assessment can effectively support environmental strategy definition, looking towards sustainable landscape management.

The assessment approach was developed using available land use data and economic valuations, involving experts from different backgrounds and research institutions. The assessment relies upon a modified version of benefit transfer method. The role of experts was to provide shared and reliable inferences about potential for ES provisioning by land covers. The results depict the potential of Italian provinces to provide a set of ten ecosystem services, framing an original assessment for Italy but applicable to other European countries. The methodology constitutes an ES assessment tool easily repeatable in scenario assessment, requiring relatively few data and easily updatable with new monetary data, potentially informative in SEA applications.

2. Methods

The methodology entailed three steps (Fig. 1):

1. Economic literature review to define a surrogate market, i.e. a collection (Table 1) of monetary values (€/ha) of ES provided by each land cover (LC).
2. Expert consultation, through focus group and Delphi survey, to estimate potentials of LCs to provide ES, considering local context, i.e. spatial characteristics such as elevation and distance to urban areas (see below).
3. Spatial-explicit estimation of the ESs values in the 1990 and 2000, aggregation of these values by provinces, calculation of changes in that period.

Table 1
The used definitions of ecosystem services.

Climate and atmospheric gas regulation	Role of biotic and abiotic processes to maintain chemical and climatic cycle in the atmosphere (e.g. CO_2/O_2 balance, ozone layer, SO_x regulation levels, etc.)
Disturbance prevention	Buffer function and protection from disturbances (e.g. storm protection, flood control, erosion, etc.)
Freshwater regulation and supply	Regulating runoff and river discharge, water recharge, avoided water purifying costs, provision of water for consumptive use (e.g. drinking, irrigation and industrial use)
Waste assimilation	Buffer function and human activities waste reduction (e.g. pathogens as *Escherichia coli*) and nutrients such as N, P, filtering of dust particles, toxic air compounds
Nutrient regulation	Recycling and spiralling of nutrients removed by plants (agriculture) and animals. Agriculture applies tons of fertilizers every year to maintain soil fertility
Habitat refugium	Continuum and functions of natural areas provided suitable living space (nursery, reproduction habitat, etc.) for wild plants and animals. Continuum of extended natural areas provided species survival useful (directly and indirectly) to human welfare
Recreation	Natural areas attract people for recreational activities (hunting, fishing, rowing, cycling, etc.) that provide direct benefits (tourist costs) but also indirect benefits (e.g. healthy population)
Aesthetic and amenity	Aesthetic and spiritual function, the value is referred to non-use, willingness to pay to maintain site quality and integrity
Soil retention and formation	Soil performs different functions, including filtering and storage of freshwater, waste absorption for plant growth. Natural systems create and enrich soil and prevent erosion during the rains
Pollination	The pollination by animal pollinators has great value for human activities, e.g. 80% of edible plants depend on pollinators. Natural pollination can only be replaced in some cases, with enormous costs

The assessment approach is a tailored version of the benefit transfer method (e.g. Costanza et al., 2006), where the value of ES_i for each land use polygon is calculated following Eq. (1):

$$VSE_{i,k} = A_k \cdot w_i(\text{LC, elevation, dist}) \cdot v_i(\text{LC}) \quad (1)$$

where $VSE_{i,k}$ is the value (€/year) of i-ES for a k-polygon of a defined land use, A_k is the k-polygon area (ha), w_i is a coefficient [0,1] for i-ES and specific LC considering context variables (elevation and distance to urban areas), v_i stands for monetary value of i-ES for defined LC (€/ha). Specifically, the term w_i means a sort of potential of a particular LC to offer a defined ES, from a negligible level (0) to a full capacity (1).

2.1. Spatial data

The basic spatial data was the CORINE land cover (CLC) vector map (1:100 000) for 1990 and 2000, and the digital elevation model (DEM, resolution 20 m) of Italy. CLC is a geographic land cover/land use database encompassing most of the countries of the European Community, it describes land cover (and partly land use) according to a nomenclature of 44 classes organized hierarchically in three levels (http://www.eea.europa.eu/publications/COR0-landcover).

Specifically, we utilized the third level classes of CLC classification (Bossard et al., 2000) related to inland zones (neglecting sea and coastal areas). In the ES potential estimation, these classes were associated to the more general land cover classes (7) used in surrogate market definition (Table 2).

The context of each CLC polygon was categorized in terms of variables (elevation and distance to urban areas, mentioned before and explained below), calculated from the same CLC dataset and DEM, and stored in fields of the tables associated to the vector maps.

2.2. Surrogate market

We used the term "surrogate market" to mean a hypothetical market for services and benefits for which valuations exist. The surrogate market consists of a collection of economic values derived by a literature review conducted for defined ecosystem services associated to definite LUs. In this review, we looked for monetary values per hectare for LC classes, beginning from a database updated to 2004 (Wilson et al., 2004). In this database, the ES monetary values were calculated by applying a variety of economic tools (e.g. willingness to pay, travel costs, avoided costs), and associated to seven land use types, namely: cropland, pasture, forest, urban green, fresh-water wetland, salt-water wetland, and fresh-water. The database referred to particular definitions of ES (Table 1). We used exactly these definitions in order to set and use a comparable database, even though other more precise and recent definitions are available (de Groot et al., 2002; Costanza, 2008; Fisher and Turner, 2008).

In our upgrading of the available database, 63 international studies were selected (out of about 900 reviewed), by criteria such as providing monetary valuations in terms of € (or other currency) per hectare of land use/cover, possibly for European territories. In effect, most of studies report specific and site-focused evaluations or too general estimations difficult to be converted in €/ha form. The monetary values were standardized to euro per hectares per year using 2007 as the base year. Table 2 shows the values for each land cover class concerning different ESs. The empty cells do not indicate no economic value for a specific ES provided by a particular LC class, rather they indicate that information is lacking (such as €/ha).

2.3. Expert-based assessment of ES potentials

The estimation of ES potential for the different LUs consisted of two phases, conducted over an 8-month period ending September 2010. In the first phase, in the course of a focus group (Carey, 1994), we identified meaningful local variables affecting the ES provisioning. These have to be spatial attributes characterizing the context of each LC-polygon, useful to qualitatively distinguish different ES potentials, and easily available or obtainable from diffused spatial data. The selected spatial variables were distance to urban areas and elevation (better described later in Section 3.2). The focus group involved (by web conference) 8 researchers with different backgrounds related to environmental monitoring or assessment, from four different Italian research institutions. These researchers met at an international conference on ES (Conference on Modelling Ecosystem Service, organized by ESP Ecosystem Services Partnership – May 27–30 2009, Lecce, Italy).

The second phase entailed a Delphi survey (Dalkey and Helmer, 1963) to assign appropriate weighs to LC (ES potentials) considering the selected variables. The Delphi method consists of an iterative process of individual expert consultation and knowledge accumulation repeated until a certain degree of judgement convergence is attained. This technique is applied in many cases of environmental assessment, in which relevant ecological knowledge is lacking, data are missing or are unsuited to empirical modelling (e.g. Hess and King, 2002; Curtis, 2004; Kangas and Leskinen, 2005; MacMillan and Marshall, 2006; Austen and Hanson, 2008; Geneletti, 2008). The participants were selected through a "snow ball" sampling method (Patton, 1990), by asking each experts to refer other

Table 2
Monetary values (€/ha per year) of ES for land-cover classes (range and literature references).

	Cropland	Pasture	Forest	Urban green	Fresh-water wetland	Salt-water Wetland	Fresh-water
Climate and atmospheric gas regulation	€23.00 (14–32) a	€7.50 (6–9) b,c	€124 (13–382) b,d,e,f	€623.3 (47–1519) g,h	€232 b	€117 b	
Disturbance prevention			€163 (97–228) i		€6346 b	€1 l	
Freshwater regulation and supply	€58.00 (39–77) a	€3 b	€3988 (8–20.550) d,i,m,n,o,p,q,r	€10 h	€4047.5 (1–9.740) l,s,t,u,v,z,br	€1672 (13–3.331) b	€640 (51–1.191) aa,ab,ac
Waste assimilation		€76 b	€76 b		€1454 b	€6779 (3–30 695) ad	€583 b
Nutrient regulation	€145.00 (102–187) a		€317 b		€212 ae		
Habitat refugium, and biodiversity	€1548 (781–2 315) af		€629.33 (3–1 720) ag,ah,ai		€80.37 (10–210) al,am	€288 (2–1 017) am,an,ao	
Recreation	€27.90 (15–40) a,bs	€2 a,b,bq	€107.46 (0.2–854) ap,aq,ar,as	€4609 bt	€1310 (0.7–2619) s,at,b,au	€30 (17–48) av,az,ba	€685 (1–2368) bb,bc,bd,be,ah,bg,bh,aa,bi
Aesthetic and amenity	€31.45 (15–48) az	€1 b	€1.51 bu		€3484 (54–7306) bl,b,u,bm	€219 (102–336) bn	€129 bo
Soil retention and formation	€3.97 (3.6–4.3) a	€5 b,bv	€9.05 b				€1 067 bp
Pollination	€30.62 (15–46) a,b	€24 b	€301.66 bz				

a: Shandu et al. (2008); b: Costanza et al. (1997); c: Sala and Paruelo (1997); d: Goio et al. (2005); e: Reyes and Mate (2004); f: AA.VV (2009); g: McPherson et al. (1998); h: McPherson (1992); i: Rein (1999); l: Farber (1987); m: Rich and Moffit (1992); n: Clinch (1999); o: Oster (1977); p: Berrens et al. (1196); q: Danielson (1995); r: Mathews et al. (2002); s: Lant and Tobin (1989); t: Creel and Loomis (1992); u: Hayes et al. (1992); v: Pate and Loomis (1997); z: Eftec (2007); aa: Piper (1997); ab: Henry et al. (1988); ac: Ribaudo and Epp (1984); ad: Breaux et al. (1995); ae: Gren et al. (1995); af: Christie et al. (2004); ag: Clayton and Mendelsohn (1993); ah: Shafer et al. (1993); ai: Kenyon and Nevin (2001); al: Vankooten and Schmitz (1992); am: Farber and Costanza (1987); an: Lynne et al. (1981); ao: Bell (1997); ap: Boxall et al. (1996); aq: Willis (1991); ar: Walsh et al. (1990); as: Willis and Garrod (1991); at: Thibodeau and Ostro (1981); au: Whitehead (1990); av: Farber (1988); az: Bergstrom et al. (1985); ba: Bergstrom et al. (1990); bb: Peters et al. (1995); bc: Rollins et al. (1997); bd: Kealy and Bishop (1986); be: Patrick et al. (1991); bf: Cordell and Bergstrom (1993); bg: Martin et al. (1982); bh: Kreutzwiser (1981); bi: Ward et al. (1996); bl: Mahan et al. (2000); bm: Doss and Taff (1997); bn: Anderson and Edwards (1986); bo: Young and Shortle (1989); bp: Dorfman et al. (1997); bq: Boxall (1995); br: Lant and Roberts (1990); bs: Feather et al. (1999); bt: Tyrvainen (2001); bu: Prince and Ahmed (1989); bv: Pimentel et al. (1997); bz: Hougner et al. (2006). The complete references are reported in Annex 2.

acknowledged experts to be included, starting from the ones involved in the focus group. Thus, 147 experts were contacted, covering different expertises and backgrounds (e.g. landscape ecology, forest ecology, landscape planning), including researchers as well professionals (such as agro-economists, architects and planners) from different Italian regions. Finally, 46 experts voluntarily participated, from 10 Italian research institutions from southern (e.g. Lecce, Napoli), middle (e.g. Rome, L'Aquila) and northern Italy (e.g. Milan, Bolzano). In order to appreciate the relevant differences in individual expertise, we classified each respondent according two categories: the "experts" as those acknowledged by other experts, and the "scientists" that have published scientific works about ES or related ecological processes, or economic assessments. In the elaboration, the answers from each "scientist", assumed to have more specific skills in ES assessment then in the definition of weights, were considered twice (i.e. as two questionnaires or interviews). In particular, the expert weighting was distinguished for each ES, thus, one respondent could be classified as "scientist" for a defined ES (e.g. habitat and biodiversity, being author of related scientific papers) but as "expert" in the estimation for other ESs (e.g. Recreation, being tourist in his/her free time).

Specifically, the Delphi survey implied two rounds of deliberation conducted in two different ways. The first round consisted in questionnaires sent by email. The second involved individual interviews applying a cumulative approach to the Delphi survey. The individual interviews allowed better interaction with the experts, making explicit their assumptions and collecting more detailed comments, which were useful in the following phase of data elaboration. The questionnaire provided by the experts included ES definitions, a table of monetary values (Table 2) and four questions, as follows:

1. Select one or more ES according to your expertise (recreational and aesthetic services are suggested to all experts, since they are citizens as well tourists).
2. Define the most relevant variable, among those proposed, to distinguish qualitatively different performances of a land use (LC) in selected ES.
3. For each land use class, define the minimum functioning area (if possible) referring to the different ESs, using the following average values: 0, 1, 5, 10, 25, or 50 ha. Where 0 (zero) means that some hundreds of square meters (i.e. less than 1 ha) are enough to provide some relevant benefits from defined ESs, and 50 ha means that there is need of at least a 50 ha area (of a defined land cover) to provide measurable benefits.
4. Define the potential of each LC for defined ES under different conditions, using qualitative categories, such as 'maximum', 'high', 'mediocre', 'low', 'very low' (numbered from 5 to 1). The context conditions are distinguished in terms of elevation (referred to the five intervals: 0–400 m, 400–800 m, 800–1200 m, 1200–1600 m, and >1600 m), and distance from urban areas larger than 100 ha (in five intervals: 0–1 km, 1–3 km, 3–5 km, 5–10 km, and >10 km).

The second question was omitted during the second round, presenting the defined variable for each ES (as shared information), and the knowledge accumulated up to that moment using in each interview. Specifically, in the second round, the definitions and

Table 3
Potentials for "habitat refugium, and biodiversity", considering distance from urban areas (with MAD values).

Land covers	CORINE land covers	Distance from urban areas (>100 ha)				
		0–1 km	1–3 km	3–5 km	5–10 km	>10 km
Urban green space	Green urban areas					
	Sport and leisure facilities					
Cropland	Non-irrigated arable land	0.4 ± 0.1	0.4 ± 0.2	0.4 ± 0.2	0.6 ± 0.2	0.6 ± 0.2
	Permanently irrigated land	0.4 ± 0	0.4 ± 0	0.4 ± 0.2	0.4 ± 0.2	0.4 ± 0.2
	Rice fields	0.5 ± 0.1	0.6 ± 0.1	0.75 ± 0.15	0.8 ± 0.2	0.8 ± 0.2
	Vineyards	0.4 ± 0.1	0.4 ± 0.2	0.6 ± 0.2	0.6 ± 0.2	0.6 ± 0.2
	Fruit trees and berry plantations	0.4 ± 0.15	0.6 ± 0.15	0.6 ± 0	0.6 ± 0	0.7 ± 0.1
	Olive groves	0.4 ± 0.2	0.6 ± 0.15	0.6 ± 0.2	0.8 ± 0	0.8 ± 0
	Annual crops associated with permanent crops	0.55 ± 0.15	0.6 ± 0.2	0.8 ± 0	0.8 ± 0.175	0.8 ± 0.2
	Complex cultivation patterns	0.6 ± 0.2	0.75 ± 0.15	0.8 ± 0.2	1 ± 0.2	1 ± 0.2
Pasture	Pastures					
	Land principally occupied by agriculture, with significant areas of natural vegetation					
	Natural grassland					
Forest	Moors and heath land	0.6 ± 0	0.8 ± 0	0.8 ± 0.2	1 ± 0	1 ± 0
	Sclerophyllous vegetation	0.6 ± 0	0.8 ± 0	0.8 ± 0.1	1 ± 0	1 ± 0
	Transitional woodland-shrub	0.6 ± 0.2	0.6 ± 0	0.8 ± 0	1 ± 0	1 ± 0
	Agro-forestry areas	0.6 ± 0.1	0.6 ± 0	0.7 ± 0.1	0.8 ± 0.2	0.8 ± 0.2
	Broad-leaved forest	0.7 ± 0.1	0.8 ± 0.2	1 ± 0	1 ± 0	1 ± 0
	Coniferous forest	0.6 ± 0.2	0.8 ± 0.2	0.8 ± 0.2	0.8 ± 0.2	0.8 ± 0.2
	Mixed forest	0.7 ± 0.1	0.8 ± 0.2	1 ± 0	1 ± 0	1 ± 0
Freshwater wetland	Inland marshes	0.8 ± 0	0.8 ± 0.2	1 ± 0	1 ± 0	1 ± 0
	Peatbogs	0.6 ± 0.2	0.8 ± 0.2	1 ± 0	1 ± 0	1 ± 0
Saltwater wetland	Salt marshes	0.8 ± 0.2	0.8 ± 0.2	1 ± 0	1 ± 0	1 ± 0
	Salines	0.6 ± 0.2	0.6 ± 0.2	0.7 ± 0.1	0.8 ± 0.2	0.8 ± 0.2
	Intertidal flats	0.6 ± 0	0.8 ± 0.2	0.8 ± 0	1 ± 0	1 ± 0
Freshwater	Water courses	0.6 ± 0.2	0.8 ± 0.2	1 ± 0.2	1 ± 0	1 ± 0
	Water bodies	0.6 ± 0.2	0.8 ± 0.2	1 ± 0.1	1 ± 0	1 ± 0

The shadows highlight classes of values.

Table 4
Potentials for "aesthetic and amenity" (with MAD values).

Land covers	CORINE land covers	Potentials
Urban green space	Green urban areas	
	Sport and leisure facilities	
Cropland	Non-irrigated arable land	0.4 ± 0.2
	Permanently irrigated land	0.4 ± 0.1
	Rice fields	0.6 ± 0.2
	Vineyards	0.8 ± 0.2
	Fruit trees and berry plantations	0.7 ± 0.1
	Olive groves	1 ± 0
	Annual crops associated with permanent crops	0.7 ± 0.1
	Complex cultivation patterns	1 ± 0
Pasture	Pastures	0.75 ± 0.15
	Land principally occupied by agriculture, with significant areas of natural vegetation	1 ± 0
	Natural grassland	1 ± 0
Forest	Moors and heath land	0.8 ± 0.2
	Sclerophyllous vegetation	0.8 ± 0.2
	Transitional woodland-shrub	0.6 ± 0.2
	Agro-forestry areas	0.6 ± 0.2
	Broad-leaved forest	1 ± 0
	Coniferous forest	1 ± 0
	Mixed forest	1 ± 0
Freshwater wetland	Inland marshes	1 ± 0
	Peatbogs	0.8 ± 0.2
Saltwater wetland	Salt marshes	0.8 ± 0.2
	Salines	0.8 ± 0.2
	Intertidal flats	1 ± 0
Freshwater	Water courses	1 ± 0
	Water bodies	1 ± 0

The shadows highlight classes of values.

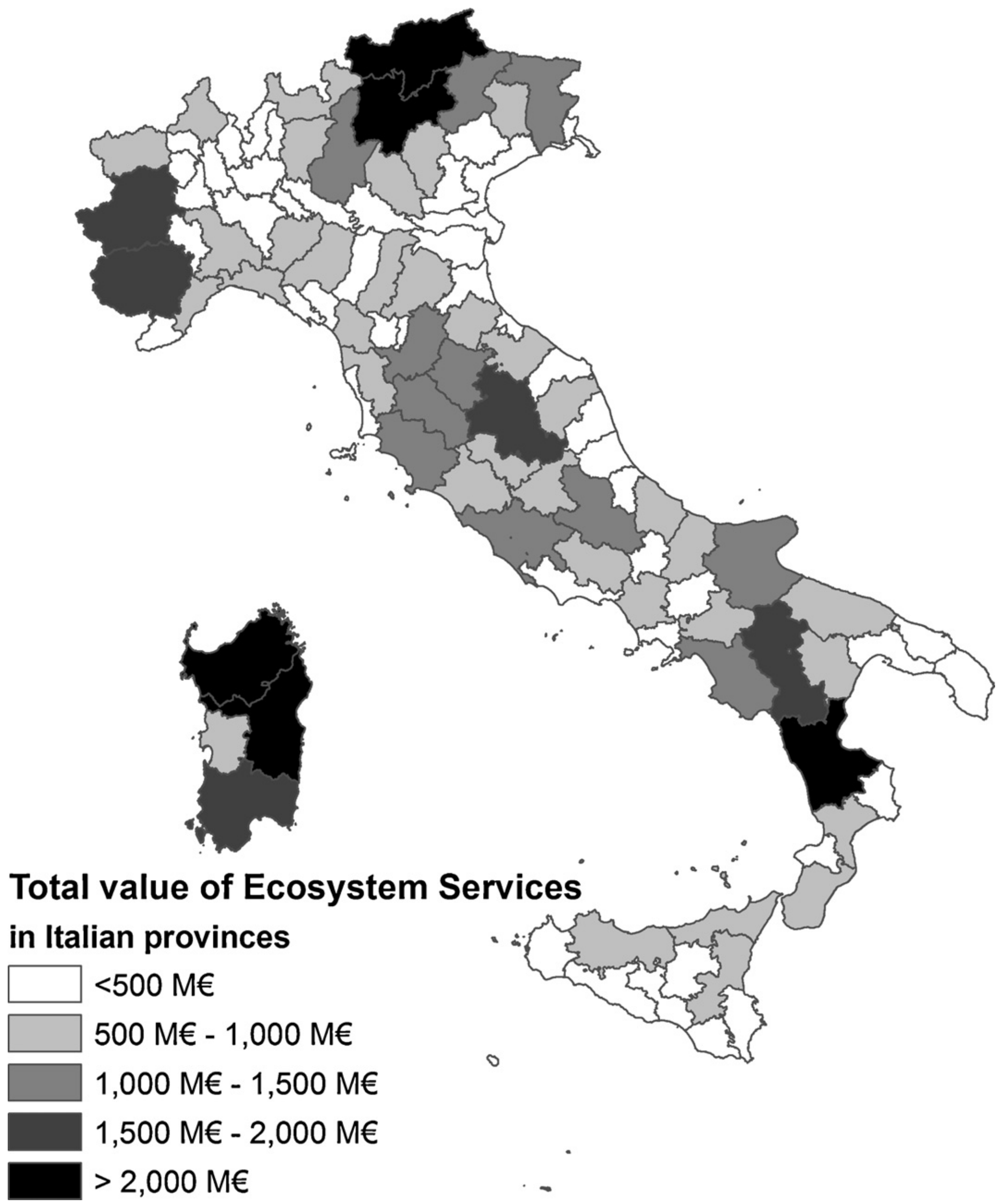

Fig. 2. Total value of ecosystem services, per province, in 2000.

the variable thresholds were modified according to experts' indications, improving the readability of the whole questionnaire and avoiding some ambiguities.

The minimum functioning area, in the third question, referred to a minimum area that may provide a valuable and measurable amount of ES. We tried the question being aware about the vagueness of the definition, in the attempt to disregard calculating values for too small polygons, assuming, for instance, that a very small patch (e.g. woodland) may not supply relevant values (e.g. for "waste assimilation"). In detail, the experts had to define a minimum area for each ES correspondingly to each land use (LC), bearing in mind that land cover may significantly differ in ES potentials per hectare (see Annex 1).

2.4. Spatial-explicit estimation of the ESs values and changes

All the experts' answers were collected and elaborated into matrices reporting the ES potentials, as values ranging from 0.2 to 1 (from a standardization of the qualitative scale), and the associated uncertainty, calculated as median of absolute deviation from medians (MAD). This uncertainty stands for the degree of convergence among the experts' judgements.

For each matrix cell an if-then rule was derived to be applied on spatial datasets. The (846) if-then rules were structured in a SQL-script and applied on CORINE land cover maps (1990 and 2000), using spatial database tools such as PostGIS and Postgresql (Refractions Research and Inc., 2010). The results of the SQL-script applied to the two periods were aggregated by provinces and compared to find the changes.

3. Results

3.1. The respondents

The respondents to Delphi questionnaires focused on few and different ESs according to their expertise. Therefore, the number

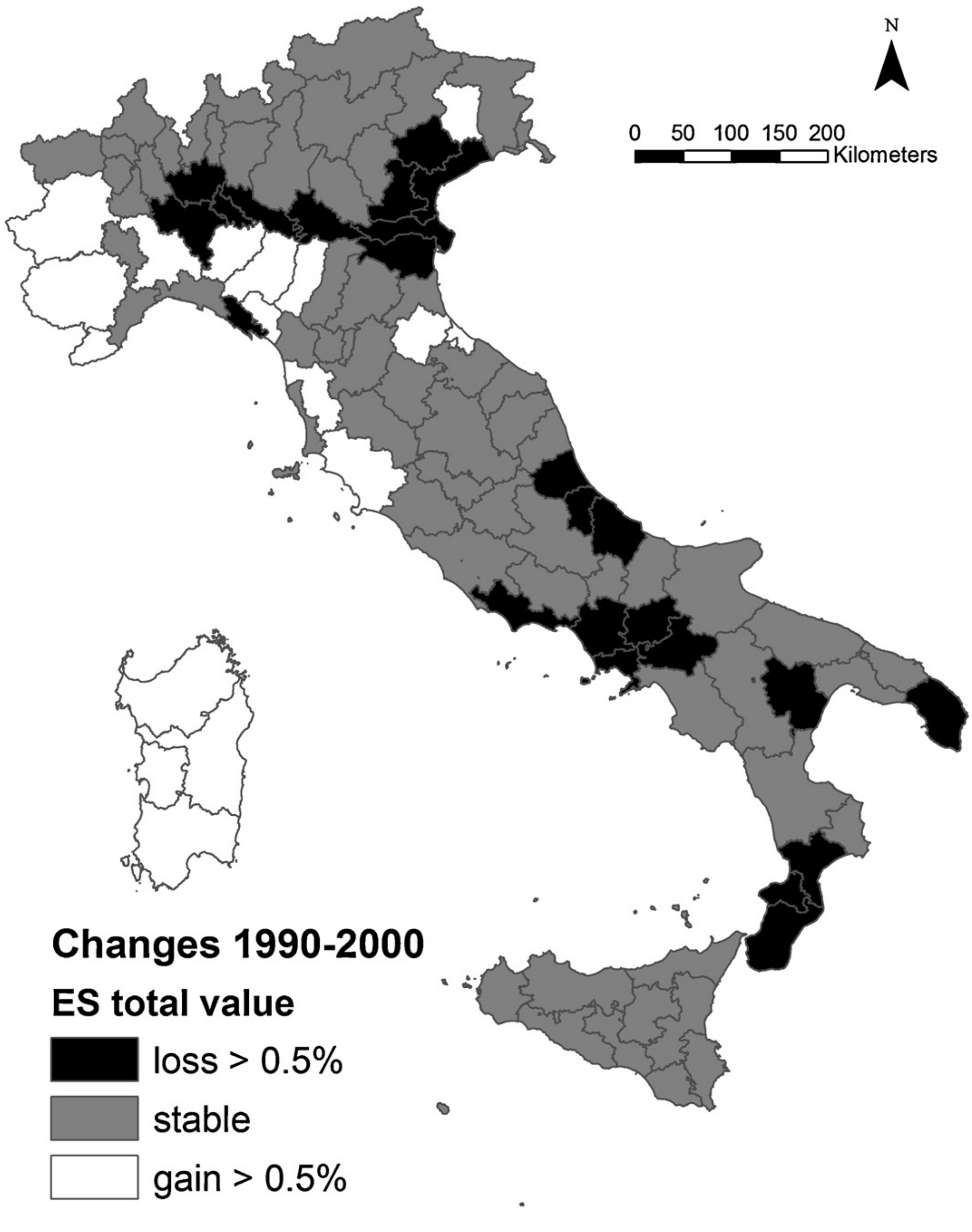

Fig. 3. Changes of ecosystem service values in the decade 1990–2000.

of estimations changed between ESs, ranging from 2 respondents about "pollination" to 33 about "aesthetic and amenity", with an average of about 16 respondents for each ES. Precisely, the number of estimations differed also among the LC classes, experts omitted evaluations for those LC classes about which they did not have enough knowledge (e.g. crops or wetlands).

3.2. *The relevant variables for ES provisioning by land uses*

The elevation and distance to urban areas (larger than 100 ha) were selected as relevant attributes of land use polygons, considering the available dataset for the whole of Italy. The elevation was meaningfully assumed to affect the potential for "climate and atmospheric gas regulation", according to the relations between ecological processes and elevation gradients (Becker et al., 2007). The distance from urban areas (considered by means of multiple-ring buffers), coherently with several studies (e.g. Andersson, 2006; Tratalos et al., 2007), was assumed to be relevant for "habitat refugium and biodiversity", in terms of disturbance (i.e. value increasing with distance from the urban area), and for "disturbance prevention", "waste assimilation", "recreation", considering value decreasing with the distance (i.e. ES supply sites far from demand sites are expected less valuable).

Concerning the other ESs, such as "fresh water regulation and supply", "nutrient regulation", "aesthetic and amenity", "soil retention" and "pollination", the experts did not agree about variables. Thus, the estimations for these ESs distinguished the provisioning only among the CORINE classes (see Annex 1). In these cases, the experts often stated that other variables should be considered, or

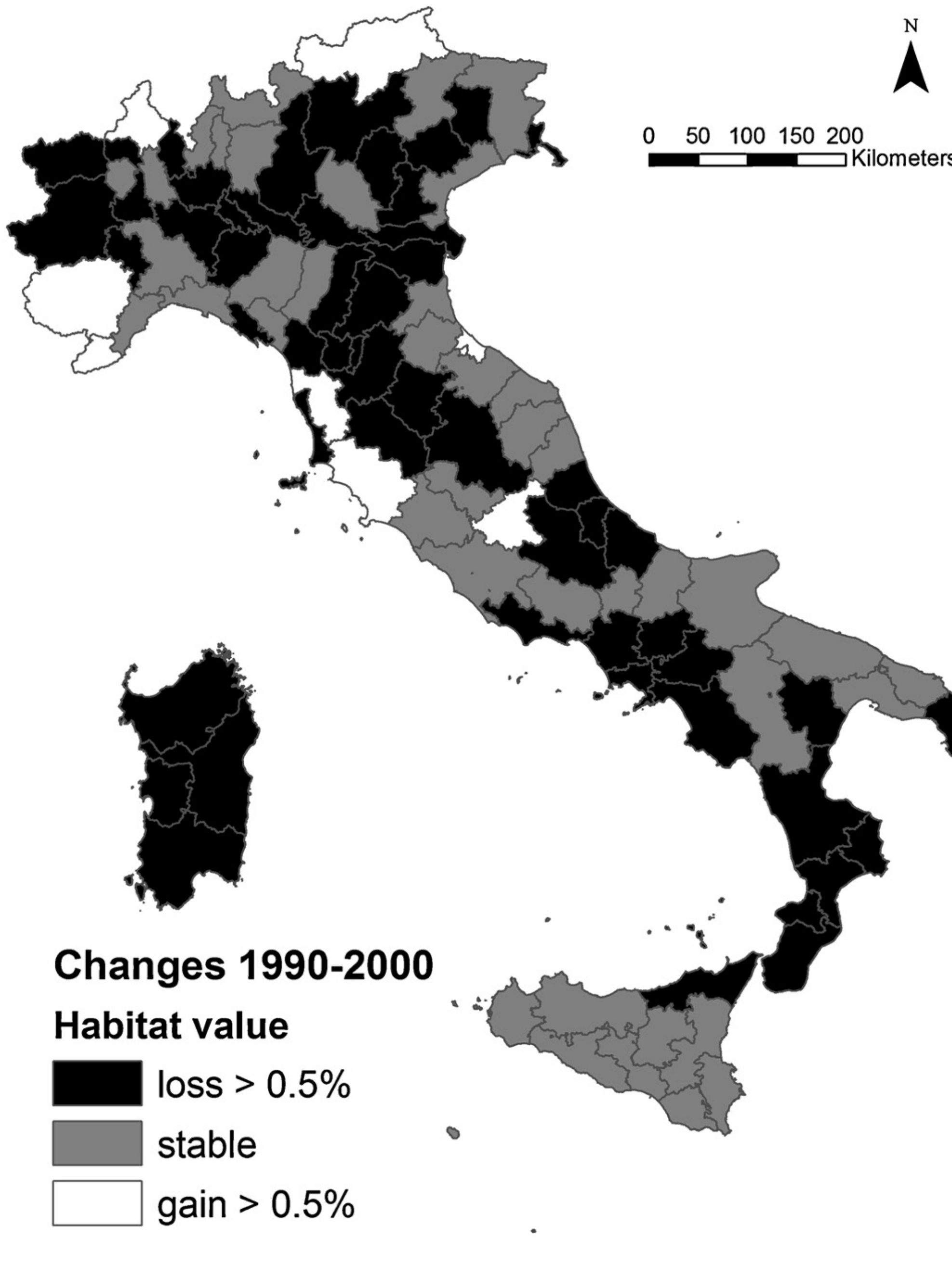

Fig. 4. Changes of "habitat refugium" value in the decade 1990–2000.

that the proposed variables may be interpreted in opposite manner considering different processes, thus, inadequate or even misleading. For instance, for "fresh water regulation and supply", the river catchment characteristics should be considered (such as morphology, geology, soil permeability, ratio impermeable/permeable surfaces).

3.3. Minimum functioning area

The estimations significantly varied among the experts and ES. "disturbance prevention", "climate and atmospheric gas regulation" and "aesthetic and amenity" seem to require quite large areas (more than 10 ha), in opposition to "pollination" for which less than 1 ha may be sufficient. The area thresholds significantly differ also within land cover classes, for example, at least 10 ha (better if more than 25 ha) of crops are required to have some "habitat refugium" values, while one hectare of forest or wetland could be satisfactory. In many cases the experts' opinions diverged, thus, this information was not used in the calculations.

3.4. Estimations of ES potential

The inferences on ES potential were expressed as weight values ranging from 0.2 to 1.0 (Tables 3 and 4). The lowest value is not zero because we considered at least a minimum value if recognized in the surrogate market. These weights were considered as nested in those "defined" by surrogate markets related to the general land cover patterns. This means that pondering an LC potential, the experts compared among CORINE classes (e.g. among broadleaves and evergreen woodland classes) included in the same

Table 5
Changes of ecosystem service values, in the period 1990-2000, for Italian provinces.

	Climate	Disturb.	Freshw.	Waste	Nutrient	Habitat	Recreat.	Aesth.	Soil	Pollin.
Max loss	−3.6%	−7.5%	−1.2%	−9.5%	−1.7%	−6.2%	−3.2%	−18.6%	−10.0%	−0.9%
Max gain	3.0%	10.1%	8.6%	20.7%	6.1%	1.1%	15.9%	15.5%	13.6%	4.8%
Mean	0.2%	1.6%	2.9%	4.7%	1.9%	−1.3%	5.1%	0.0%	2.3%	1.6%

general land cover (e.g. forest). Here only the results regarding the two ESs, "habitat refugium and biodiversity" and "aesthetic and amenity" are presented. For the other results see Annex 1.

3.4.1. Habitat refugium and biodiversity

A panel of 27 experts contributed to ponder the land use potentials for this ES. Table 3 shows evident differences among the CORINE classes, generally increasing with the distance from urban areas. Some LUs (forest types and water bodies) seem to be more sensitive to this distance, showing higher differences between near and far sites. Distance appears less meaningful for particular land uses with poor habitat potential (e.g. permanently irrigated land).

Several comments were used by experts to justify each estimation (not reported here). These enriched the ES estimation with a vast source of information. For instance, experts in their considerations included their knowledge about the use of agro-chemicals usually distinguishing different types of crops. Olive groves (in Italy often with very old trees), for example, likely have higher potential for a richer biodiversity than fruit plantations or vineyards. The empty cells mean that there are no usable monetary values for the specific ES and LU.

3.4.2. Aesthetic and amenity

This ES was pondered by 33 experts with very different backgrounds. Most of the simple "experts" recollected their perceptions in the assessment, like tourists remembering their holidays. The "scientists" tried to extrapolate, from their knowledge, what could be assumed as the most appreciated or preferred. High heterogeneity in valuations resulted for croplands and secondarily for forests and wetlands, meanwhile, both groups converged (see the lower median absolute deviation, MAD) for the cases with higher potential (Table 4).

3.5. Values of ecosystem services for Italian provinces and relative changes

The Italian ecosystems, we estimated, provide an amount of goods and benefits valued at €71,3 billion €/year. Some provinces appear richer in ES than others (Fig. 2). The distribution of lowest provincial values is mainly related to lowland and urban areas, as it is evident in the plan area along the Po river basin (the white polygons in the upper part of the map). The richest Italian provinces seem to be located in northern Italy (Trento and Bolzano, close to Austrian border) and southern Italy (Sardegna and Calabria regions), all dominated by mountain areas and with smaller urban areas. Virtually, citizens of these provinces may benefit from 6000 to 27,000 €each per year in terms of ecosystem services, in confront of about 128 €/year for the Milan province residents.

In the decade 1990–2000 the total monetary value of all considered ES increased by 2.0% (Fig. 3), likely due to a general increase of forests that in many cases is currently compensating for the impact of urbanizations. The main losses are measured for the province already scarce in ES, where the urban sprawl and urbanization rate has been relatively high in the last decades (EEA, 2002, 2010; Pileri and Maggi, 2010).

Broader changes can be recognized in single ESs (Table 5). Extreme changes are recorded for "waste assimilation" and "recreation" in terms of gain (+20.7% and +15.9%), and for "aesthetic and amenity" and "soil retention and formation" in terms of loss (−18.6% and −10%). On average, the primary ES loss concerns the "habitat refugium" (see Fig. 4 and Table 5).

4. Discussion and conclusion

This study is a first evaluation of the potential economic value of ecosystem services for Italian provinces. The approach presented is based on the common "benefit transfer" method (Bateman et al., 1999; Johnston and Rosenberger, 2010), already applied using geographical information systems (Costanza et al., 2006; Troy and Wilson, 2006). In particular, inspired by Troy and Wilson's work, we aimed to enhance that approach through a calibration, based on expert consultation, and considering local characteristics of any single land cover polygon. Although the presented calculations have clear limitations, the approach may form a valuable support to land use planning.

4.1. Limits

Most of the limits stem from the methodology applied; the benefit transfer implies generalization, which is a shortcut respect to data-driven ecological–economic modelling, susceptible to significant errors due to scarcity of empirical economic valuations. The available poor dataset challenges the methodological assumptions of comparability among studies and of contextual similarity for the resources being valued (Rosenberger and Stanley, 2006). Despite the sizable literature and the ubiquity of benefit transfer in policy analysis, the method remains subject to controversy (Plummer, 2009). The same economic valuation of ESs is constrained by the well-known limitations of non-market valuation methods, recognized since the first studies (Costanza et al., 1998) and still difficult to handle in operational decisions (e.g. Moilanen et al., 2009).

In our case, the lack of information (as €/ha) for some ecosystem services in some land cover types (e.g. beaches) makes our calculations a significant underestimation of the total ecosystem services value. In particular, the spatial data used (CORINE land cover) contributed to the underestimation being a low resolution land use map (1:100,000 scale). With a minimum map area of 25 ha we have neglected many of water bodies and most small wetlands that provide important and valuable ESs.

The results should be interpreted as a preliminary assessment. The several limits may have minor importance if we consider two factors. The changes in ES provisioning and their location may be more interesting for management and planning than absolute values. Secondly, diverse levels of decision making entail different degrees of information precision, our results are consistent with the resolution usually addressed by a strategic level of spatial planning.

4.2. Added value and potentials

An improvement of the presented approach, respect other similar studies (e.g. Kreuter et al., 2001; Liu et al., 2010; Tianhong et al., 2010) is the consideration of higher thematic resolution of land cover maps (third level of CORINE classes) and the expert-based spatial adjustment considering two variables related to ES provisioning. The expert consultation provided a significant information for a "local" adjustment that hardly may be replaced by data-based spatial modelling (e.g. Kienast et al., 2009), requiring a larger amount of data. Moreover, the assessment covers a comprehensive set of ecosystem services rather than focusing to one or few, as most studies usually do (e.g. Gren and Isacs, 2009; Vihervaara et al., 2010). Besides, the approach can be easily replicated, by applying the rule set (SQL-script) to other land use vector maps (e.g. scenarios), and updated, using new economic studies.

Even though underestimated and imprecise, this preliminary assessment may be effective in raising the problem to the level of environmental policy and planning debate. Often, ESs may not have a place in the mental models of decision makers simply because they lack general information on local flows of ES. In particular, the results support the identification of landscape functions and of impacts on ES sources. Hence, this preliminary assessment may support a screening analysis in order to define critical pressures or specific threatened ES on a definite region. Particularly, it may support screening and scoping phases within Strategic Environmental Assessment (Sadler and Verheem, 1996), helping a systematic consideration of environmental issues. Losses or gains in terms of €/year may ground criteria of impact assessment due to LC changes in landscape planning studies, fostering a wider consideration of the effects and the alternatives in strategic decision making (Fischer, 2003). As screening tool, this pilot assessment may also be useful to justify successively research investment and to guide more precise assessments, based on ecological and socio-economic data.

Acknowledgments

The authors are grateful to Maria Angela Cataldi and Nicola Zaccarelli (University of Salento) for their support in data analysis and valuable discussions, and to all experts contributed to the Delphi survey. The authors also acknowledge the Autonomous Province of Trento (PAT) for providing financial support for the research project "Public policies and local development: innovation policy and its effects on locally embedded global dynamics" (OPENLOC 2008–2011; www.openloc.eu).

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolind.2011.07.019.

References

Andersson, E., 2006. Urban landscapes and sustainable cities. Ecol. Soc. 11, 34.
Austen, E., Hanson, A., 2008. Identifying wetland compensation principles and mechanisms for Atlantic Canada using a Delphi approach. Wetlands 28, 640–655.
Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K., Turner, R.K., 2002. Economic reasons for conserving wild nature. Science 297, 950–953.
Balvanera, P., Daily, G.C., Ehrlich, P.R., Ricketts, T.H., Bailey, S.A., Kark, S., Kremen, C., Pereira, H., 2001. Conserving biodiversity and ecosystem services. Science 291, 2047.
Barberis, R., 2006. Consumo di suolo e qualità dei suoli urbani. APAT, Roma.
Bastian, O., Haase, D., Grunewald, K., 2012. Ecosystem properties, potentials and services – The EPPS conceptual framework and an urban application example. Ecological Indicators 21, 7–16.
Bateman, I.J., Lovett, A.A., Brainard, J.S., 1999. Developing a methodology for benefit transfers using geographical information systems: modelling demand for woodland recreation. Reg. Stud. 33, 191–205.
Becker, A., Körner, C., Brun, J.-J., Guisan, A., Tappeiner U, 2007. Ecological and land use studies along elevational gradients. Mt. Res. Dev. 27, 58–65.
Berkes, F., Folke, C., Colding, J., 2000. Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience. Cambridge Univ. Press, Cambridge, UK.
Blum, W.E.H., Büsing, J., Montanarella, L., 2004. Research needs in support of the European thematic strategy for soil protection. Trends Anal. Chem. 23, 680–685.
Bolliger, J., Bättig, M., Gallati, J., Kläy, A., Stauffacher, M., Kienast, F., 2010. Landscape multifunctionality: a powerful concept to identify effects of environmental change. Reg. Environ. Change 11, 203–206.
Bossard, M., Feranec, J., Otahel, J., 2000. CORINE Land Cover Technical Guide: Addendum 2000. European Environment Agency, Copenhagen.
Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecol. Econ. 63, 616–626.
Busch, M., La Notte, A., Laporte, V., Erhard, M., 2012. Potentials of quantitative and qualitative approaches to assessing ecosystem services. Ecological Indicators 21, 89–103.
Carey, M.A., 1994. The group effect in focus groups: planning, implementing and interpreting focus group research. In: Morse, J.M. (Ed.), Critical Issues in Qualitative Research Method. Sage, Thousand Oaks, pp. 225–241.
Costanza, R., 2008. Ecosystem services: multiple classification systems are needed. Biol. Conserv. 141, 350–352.
Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1998. The value of the world's ecosystem services and natural capital. Ecol. Econ. 25, 3–15.
Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the World's Ecosystem Services and Natural Capital. Nature 387, 253–260.
Costanza, R., Wilson, M., Troy, A., Voinov, A., Liu, S., D'Agostino, J., 2006. The Value of New Jersey's Ecosystem Services and Natural Capital. Gund Institute for Ecological Economics, University of Vermont, Burlington, VT.
Curtis, I.A., 2004. Valuing ecosystem goods and services: a new approach using a surrogate market and the combination of a multiple criteria analysis and a Delphi panel to assign weights to the attributes. Ecol. Econ. 50, 163–194.
Daily, G.C. (Ed.), 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, DC.
Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision making: time to deliver. Front. Ecol. Environ. 7, 21–28.
Dalkey, N., Helmer, O., 1963. An experimental application of the Delphi method to the use of experts. Manage. Sci. 9, 458–467.
de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Complex. 7, 260–272.
de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol. Econ. 41, 393–408.
Deutsch, L., Folke, C., Skånberg, K., 2003. The critical natural capital of ecosystem performance as insurance for human well-being. Ecol. Econ. 44, 205–217.
EC, 2006. Thematic Strategy for Soil Protection. Commission of the European Communities, Brussels.
EEA, 2002. Towards an Urban Atlas—Environmental Issue Report n. 30. European Environment Agency, Brussels.
EEA, 2006. Urban Sprawl in Europe. The Ignored Challenge. European Environmental Agency, Copenhagen.
EEA, 2010. The European Environment—State and Outlook 2010: Land Use. European Environment Agency, Copenhagen.
Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. Ecol. Econ. 41, 375–392.
Fischer, T.B., 2003. Strategic environmental assessment in post-modern times. Environ. Impact Assess. Rev. 23, 155–170.
Fisher, B., Turner, R.K., 2008. Ecosystem services: classification for valuation. Biol. Conserv. 141, 1167–1169.
Fisher, B., Turner, K., Zylstra, M., Brouwer, R., Groot, R., Farber, S., Ferraro, P., Green, R., Hadley, D., Harlow, J., 2008. Ecosystem services and economic theory: integration for policy-relevant research. Ecol. Appl. 18, 2050–2067.
Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68, 643–653.
Gatto, M., De Leo, G.A., 2000. Pricing biodiversity and ecosystem services: the never-ending story. BioScience 50, 347–355.
Geneletti, D., 2008. Incorporating biodiversity assets in spatial planning: methodological proposal and development of a planning support system. Landscape Urban Plan. 84, 252–265.
Gren, I.-M., Isacs, L., 2009. Ecosystem services and regional development: an application to Sweden. Ecol. Econ. 68, 2549–2559.
Hess, G.R., King, T.J., 2002. Planning open spaces for wildlife. I. Selecting focal species using a Delphi survey approach. Landscape Urban Plan. 58, 25–40.
Howarth, R.B., Farber, S., 2002. Accounting for the value of ecosystem services. Ecol. Econ. 41, 421–429.
Johnston, R.J., Rosenberger, R.S., 2010. Methods, trends and controversies in contemporary benefit transfer. J. Econ. Surv. 24, 479–510.
Kangas, J., Leskinen, P., 2005. Modelling ecological expertise for forest planning calculations-rationale, examples, and pitfalls. J. Environ. Manage. 76, 125–133.
Kienast, F., Bolliger, J., Potschin, M., de Groot, R., Verburg, P., Heller, I., Wascher, D., Haines-Young, R., 2009. Assessing landscape functions with broad-scale environmental data: insights gained from a prototype development for Europe. Environ. Manage. 44, 1099–1120.

Kremen, C., 2005. Managing ecosystem services: what do we need to know about their ecology? Ecol. Lett. 8, 468–479.
Kreuter, U.P., Harris, H.G., Matlock, M.D., Lacey, R.E., 2001. Change in ecosystem service values in the San Antonio area, Texas. Ecol. Econ. 39, 333–346.
Kumar, M., Kumar, P., 2008. Valuation of the ecosystem services: a psycho-cultural perspective. Ecol. Econ. 64, 808–819.
Lindhjem, H., Navrud, S., 2008. How reliable are meta-analyses for international benefit transfers? Ecol. Econ. 66, 425–435.
Liu, S., Costanza, R., Troy, A., D'Aagostino, J., Mates, W., 2010. Valuing New Jersey's ecosystem services and natural capital: a spatially explicit benefit transfer approach. Environ. Manage. 45, 1271–1285.
MacMillan, D.C., Marshall, K., 2006. The Delphi process—an expert-based approach to ecological modelling in data-poor environments. Anim. Conserv. 9, 11–19.
Millennium Ecosystem Assessment, 2003. Ecosystems and Human Well-being: A Framework for Assessment. Island Press, Washington, DC.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: A Framework for Assessment, Accessed on http://millenniumassessment.org/en/Framework.aspx.
Moilanen, A., van Teeffelen, A.J.A., Ferrier, Y., Simon, B.H., 2009. How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratios for impacted habitat. Restor. Ecol. 17, 470–478.
Montanari, I., Nucciotti, C., Cagnoli, P., Branchi, M., Carati, M., Santolini, R., Palazzini, M., Simonati, W., 2009. Natura e biodiversità. Annuario regionale dei dati ambientali 2009. ARPA Emilia-Romagna, Bologna.
Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front. Ecol. Environ. 7, 4–11.
Norgaard, R.B., Bode, C., 1998. Next, the value of God, and other reactions. Ecol. Econ. 25, 37–39.
Patton, M.Q., 1990. Qualitative Evaluation and Research Methods. SAGE Publications, Thousand Oaks, CA.
Pileri, P., Maggi, M., 2010. Sustainable planning? First results in land uptakes in rural, natural and protected areas: the Lombardia case study (Italy). J. Land use Sci. 5, 105–122.
Plummer, M.L., 2009. Assessing benefit transfer for the valuation of ecosystem services. Front. Ecol. Environ. 7, 38–45.
Refractions Research Inc., 2010. PostGIS. Refractions Research Inc., Victoria, British Columbia, Canada.
Rosenberger, R.S., Stanley, T.D., 2006. Measurement, generalization, and publication: sources of error in benefit transfers and their management. Ecol. Econ. 60, 372–378.
Sadler, B., Verheem, R., 1996. Strategic Environmental Assessment: Status, Challenge and Future Directions. Netherlands Ministry of Housing, Spatial Planning and the Environment, The Hague.
Tianhong, L., Wenkai, L., Zhenghan, Q., 2010. Variations in ecosystem service value in response to land use changes in Shenzhen. Ecol. Econ. 69, 1427–1435.
Toman, M., 1998. Special section: forum on valuation of ecosystem services—why not to calculate the value of the world's ecosystem services and natural capital. Ecol. Econ. 25, 57–60.
Tratalos, J., Fuller, R.A., Warren, P.H., Davies, R.G., Gaston, K.J, 2007. Urban form, biodiversity potential and ecosystem services. Landscape Urban Plan. 83, 308–317.
Troy, A., Wilson, M.A., 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. Ecol. Econ. 60, 435–449.
van Oudenhoven, A.P.E., Petz, K., Alkemade, R., Hein, L., de Groot, R.S., 2012. Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecological Indicators 21, 110–122.
Vihervaara, P., Rönkä, M., Walls, M., 2010. Trends in ecosystem service research: early steps and current drivers. AMBIO: J. Human Environ. 39, 314–324.
Wallace, K.J., 2007. Classification of ecosystem services: problems and solutions. Biol. Conserv. 139, 235–246.
Wang, X., Chen, W., Zhang, L., Jin, D., Lu, C., 2010. Estimating the ecosystem service losses from proposed land reclamation projects: a case study in Xiamen. Ecol. Econ. 69, 2549–2556.
Wilson, M.A., Costanza, R., Troy, A., 2004. The EcoValue Project, http://ecovalue.uvm.edu (accessed on line 20.05.10).