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CICLO XXX

"APPLICATIONS OF ECOSYSTEM SERVICES ASSESSMENT USING MODELLING, GIS AND REMOTE SENSING AT DIFFERENT SCALES"

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Abstract

Assessment and mapping of ecosystem services (ESs) is gaining greater attention among researchers and decision-makers because of their wide possibility of application.

Recently, different methods were developed to link land use and land cover to ecosystem service provision.

This work applied ES assessments to five case studies with the aim to answer to the following questions: i) are Protected Areas (PA) effective in maintaining the ES provision capacity? and ii) How ES assessments can be usefully implemented in environmental planning to support most efficient and sustainable solutions for human well-being?

The case studies applied the different ES mapping methods and detected (or dealt with) the different processes through which land use/land cover changes affect ESs.

The results showed that the considered PAs were not effective in maintaining ES supply because of their passive management. Even when they were effective to protect biodiversity, ES supply was not guaranteed. Moreover, the analysis highlighted some trade-offs among different ESs.

ES mapping can be transferred to theory to practice by the development of instruments, which can support decision-making process. ES mapping can be applied to different problems in different contexts, such as sustainability of renewable energies in agro-environment and cost-effective investments on urban green infrastructures.

Overall the finding of the study demonstrated that: i) Since the mere conservation of biodiversity does not guarantee ES supply, the governance of PAs need to switch from passive to active management, and ii) ES mapping can support decision-making process providing instruments for environmental planning.

Keywords: Ecosystem Services, Ecosystem Service Mapping, Land Use / Land Cover changes, Protected Areas, Environmental Conservation, Decision-making

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1. Introduction

1.1. Background

Despite the notion that human life depends on ecosystems was already understood during ancient times, the modern concept of Ecosystem Services (ES) gained popularity only during the last two decades, as suitable argument to link environmental conservation and human well-being.

The term "Ecosystem Services" was coined during the beginning of 1980s and was promoted on the background of sustainable development, before to gain popularity during the 1990s (de Groot et al. 2017).

Together with the growing interest of scientific community, different attempts provided definitions and classifications, with the aim to structure a scientific framework concerning ESs.

Daily et al. (1997) provided a first definition of ESs as "*the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life*".

In their pioneer work, Costanza et al. (1997) defined ESs as "*the benefits human populations derive, directly or indirectly, from ecosystem functions*", introducing the first global monetary evaluation attempt of ESs, under the concept of "Natural Capital" as the stock of natural assets generating flows of ESs.

The argument definitively reached its broader attention in 2005 with the publication of the Millennium Ecosystem Assessment (MEA), a four-year project launched by the United Nations that involved over 1300 scientists from 95 countries, which documented the dramatic magnitude of human impacts on ecosystems and definitively placed ES rationale in the policy agenda. The MEA report (2005) defined ESs as "*benefits people obtain from ecosystems*" and provided their classification in four classes: provisioning, regulating, supporting and cultural services.

Between the 2007 and the 2010, the UN Environment Programme launched a second international project called "The Economics of Ecosystems and Biodiversity" (TEEB). The TEEB adopted the ESs definition provided by de Groot et al. (2010): "*the direct and indirect contributions of ecosystems to human well-being*". The principal objective was to mainstream the values of biodiversity and ESs into decision-making at all levels, demonstrating their values in economic terms, and how to capture those values in decision-making.

The promotion of international communication and cooperation is currently claimed to meet the new biodiversity and ES targets. The need to put ES theory in practice resulted in the creation of several networks and independent bodies.

The European Union (EU) adopted an document on "Options for an EU vision and target for biodiversity beyond 2010", where there is an explicit reference to the mapping of ESs in high level policy document as a tool to define the scope of the maintenance and restoration efforts required to achieve biodiversity targets (Maes et al. 2017). On these bases, the European working group on Mapping and Assessment of

Ecosystem Services (MAES) (source: http://biodiversity.europa.eu/maes) provided a first standard set of indicators for mapping and assessment of ESs.

The evolution of the concept and its applications lead to ambiguity of definition and terms (La Notte et al. 2017) (see Tab.1.1.1 for glossary of terms adopted in this work). For instance, the use of the term "ecosystem function" in ES theory was often unclear. De Groot et al. (1992, 2002) identified ecosystem functions as "*the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly*". The ESs definitions provided by Kremen (2005): "*Ecosystem services are the set of ecosystem functions that are useful to humans*" and Hooper et al. (2005):"*Ecosystem goods and services are the subset of function of utilitarian value to human*" also involve ecosystem functions in ESs framework. However, the proper inclusion of ecological functions within the ES theory was affected by the uncertainties and ambiguity of the definition of ecological function and functioning itself (Jax 2005, 2016).

In order to model the paradigm of ES, Haines-Young and Potschin (2010) proposed the "cascade model" (Fig.1.1.1), where the word "function" is used "*to indicate some capacity or capability of the ecosystem to do something that is potentially useful to people*". An important assumption of the model is that an ES does not exist in isolation of people needs. The ecosystems are providers of ecological functions, which generate a flow of ESs only when there is a human demand. This also introduces the concept of beneficiaries (i.e. the end users of benefits).

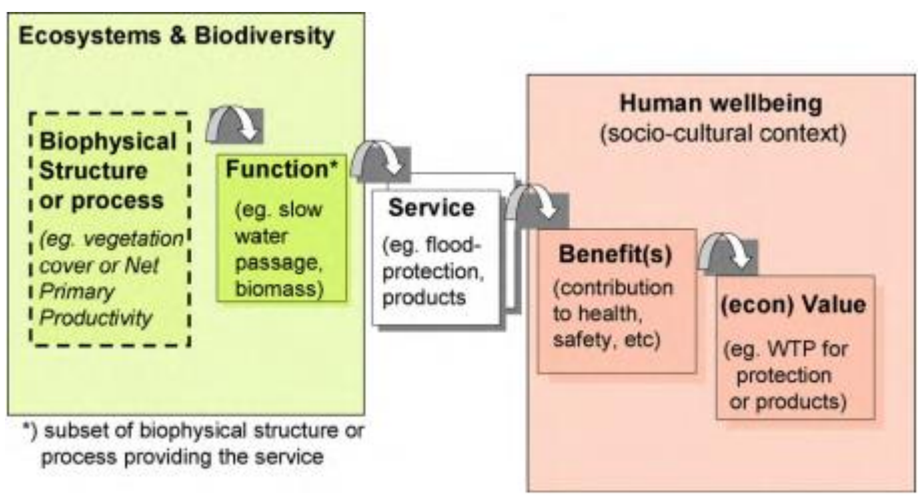

Fig.1.1.1: Cascade-model to link ecosystem properties to human wellbeing (De Groot et al. 2010)

The cascade model is sufficiently flexible to be developed and elaborated in ways that were meaningful for the different place-based studies (Potschin-Young et al 2017).

Thus, the conceptualization of the ES delivery can be modeled in: i) capacity to provide ESs, which attains to the biophysical attributes (i.e. ecological functions) of specific areas, in space and time, ii) demand, concerning socio-economic sphere (i.e. human needs) and iii) ES flows, representing the directly or indirectly use of the ecological functions by humans (i.e. the real use of ESs). This framework is fundamental to put ES mapping and/or assessment in practice (Palomo et al. 2013; Bagstad et al 2014; Felipe-Lucia et al 2015).

In general, such conceptual frameworks are suitable to clarify complex relationships and provide a template for empirical researches and operational strategies or applications (Potschin-Young et al 2017). As for the definition, many attempts of ESs classification were provided after the release of MEA. The different classification systems are usually structured according the final scope of ESs assessment. Some criticism were raised concerning the suitability to include supporting and cultural services within the classification (Fisher et al. 2009). The classification proposed by TEEB replaced supporting services with "habitat services" including "maintenance of life cycles" and "maintenance of genetic diversity". Fisher et al. (2009) argued that ESs "*are ecological in nature, in that aesthetic values, cultural contentment and recreation are not ecosystem services. Ecosystem services are ecological components, functions and/or processes, as long as there are human beneficiaries*", thus excluding cultural services from the ES framework.

In order to harmonize the ES classification for environmental accounting, the European Environmental Agency developed the Common International Classification of Ecosystem Services (CICES) (https://cices.eu/). This classification recognizes three ES categories: "provisioning", "regulating and maintenance" and "cultural" services. The "supporting" services are treated as a part of ecological functions that underpin structures and processes that ultimately provide final ecosystem services. In other words, supporting services are regarded as intermediate services, even though the relationship between supporting services (and particularly biodiversity) is not fully understood (see Section 1.5). The CICES is continuously under updating, as result of consultation with members of the different user communities. For example, some authors debate on the need to include "abiotic" services within the classification, as output provided by natural systems (van der Meulen et al 2016). However, it must be highlighted that the CICES does not aim to replace the other ES classifications, which are commonly used as well according to the final scope of the analysis. Overall, ES science suffers from several uncertainties concerning terminologies, definitions, classifications, ecosystem functioning and general conceptual models.

The role of the Ecology lies in the assessment of the ecological functions (i.e. ES capacity) at different scales, while a more integrated approach is required when the ES science has to be applied to governance and planning.

Tab.1.1.1: Glossary of terms used in this work.

Term	**Definition**
ES assessment	ES quantification using primary data or proxies
ES modelling	ES assessment using (a combination of) ecological production functions, which link available data with ES supply at different levels of complexity
ES mapping	Spatially-explicit assessment of ESs
ES delivery	Complete occurring of ES flows from ES capacity provision to satisfy the ES demand
ES capacity	Potential ES provision of a given area or ecosystem. This is not the real ES supply (flows), which instead is a function of ES capacity and ES demand.
ES demand	The need for specific ES by society, particular stakeholder groups or individuals
ES flow	Transfer of ES from ES capacity to ES demand. It is the real amount of ES that are actually used in a specific area and time

1.2. ES provision in space and time

The ES delivery is affected by spatial and temporal variation of ecological functions. Since human well-being depends on both their uneven spatial distribution and temporal changes, mapping such variations is fundamental to detect ES provision and to inform decision makers. On other hand, mapping efforts require the understanding of how ecological processes affect the ES provision capacity.

As different ecosystems provide different ecological functions, the spatial dimension of ESs is directly related to the spatial distribution of ecosystems. At landscape scale, the configuration of different units (catchments, administrative units, ecological zones delimitated by abiotic gradients, etc.) plays a fundamental role with respect to ES provision (Syrbe and Walz 2012; Mitchell et al 2015). In fact, the interaction among the different ecosystem patches distributed within the landscape results in complex outcomes that do not correspond with the mere sum of single units. For this reason, some authors proposed landscape metrics as indicators for ESs (Uueemaa et al 2013).

Moreover, ES capacity depends on local conditions involving biotic and abiotic properties, such as climate, soil, morphology and interaction among species. Climate conditions are particularly important for water-related services since they directly affect the local water budget (Lang et al 2017). Precipitation, temperature and evapotranspiration are the main climatic variables that determine a wide range of environmental factors, such as water availability (Milly et al 2005), mineralization rates (Rey et al 2005), abiotic stresses (Williams et al 2013) and species interactions (Tylianakis et al 2008).

Climate change has significant influences on ES delivery (capacity and demand), particularly for regulating services (Schröter et al 2005). Precipitation patterns (frequency and intensity of rain events) drive the demand for water storage and supply and/or flood prevention. Natural and semi-natural ecosystems (i.e. croplands and urban green) can satisfy such demand by regulating evapotranspiration and soil properties to increase or decrease water infiltration and absorption (Goulden and Bales 2014).

On the other hand, other climatic parameters (e.g. temperature) can affect ecosystem structures and functions, thus altering the capacity of ecosystem to provide such services (Lamarque et al. 2014). Rising temperatures increase the frequency of heat island phenomena in urban area (Patz et al. 2005). Urban trees can mitigate these events by shading impervious surfaces to decrease temperatures at local scale (Feyisa et al. 2014). Moreover, when urban settlements are located nearby wetlands, emergent aquatic vegetation can promote cooling effects and reflect solar energy by increasing evapotranspiration and albedo (Kiviat 2013).

Therefore, the projected climatic changes are expected to: i) affect the capacity of ecosystem to provide services through altering their resilience and ii) increase the human demand for ESs through increasing vulnerability of human settlements and health. The understanding of these trends is an important challenge in ES science.

Soil is an important regulator and provider of ecological functions. Soil characteristics regulate vegetation structure and consequently ecosystem functioning, for example through affecting nutrient and water availability (Bowles et al. 2014; Stefan et al. 2014) or supporting biodiversity (Orgiazzi et al 2016). Soil itself is also a provider of a large number of ESs. Due to the difficulty to clearly distinguish ESs mediated or directly provided by soil, the term "soil-related services" could be more appropriated to avoid ambiguity (as adopted in Chapter 3.5). The most valuable market-ES (i.e. direct monetary value) provided by soil is the food provision, which is the most relevant ES in agro-environments and strongly depends on soil fertility. Timber and biomass provision for raw material or energy purposes are other services related to soil properties and conditions. Soil is a fundamental natural buffer that protects ground and superficial water bodies from nutrient leaching and pollution (Castaldelli et al 2013), thus providing a valuable regulation service. This function depends on the retention of water-soluble pollutants, which, in turn, depends on soil characteristics, climatic conditions and land use practices (Jeppesen et al. 2011; Aschonitis et al. 2012). Soil cation-exchange capacity and pH are the most important descriptors for the evaluation of this service (Calzolari et al 2016), which are particularly affected by organic matter and clay content (Helling et al 1964). The pedosphere also provides habitat for soil organisms, which, in turn, participate to the provision of regulation services through their influence in organic matter dynamics and soil physical properties (Lavelle et al 2006). These properties can be considered a function (commonly named as habitat provision) and therefore a supporting ES per se (MEA 2005; Calzolari et al 2016) or processes participating to soil functioning (Dominati et al 2010), according the ES classification system adopted. Generally, regardless of the fact that some authors classify them as supporting functions or set of processes, the maintenance of soil biological activities and nutrient and water cycling are the core of soil formation (pedogenesis) (Dominati et al 2010), and is influenced by physical and chemical properties of soil, as well as by land use (Calzolari et al 2016). Since organic matter plays an important role in many functions and processes occurring in the pedosphere, soil organic carbon content can be used as a proxy for a wide set of soil ESs. The carbon amount sequestered and stored in soil significantly concurs to mitigate climate changes (i.e. climate regulation services), representing the most important terrestrial carbon pool (Lal 2004). The amount of soil organic matter depends on organic inputs and mineralization rates. The first mainly include in situ inputs derived by primary production (Bending et al. 2002), while the latter depends on temperature, nutrient availability, oxic/anoxic conditions, microbial and invertebrate communities (Cassman and Munns 1980; Frouz 2017). When considering agricultural land, management practices significantly affect soil organic dynamics (e.g. by tillage or removing crop residues) (Panettieri et al. 2014). Soil moisture content is another important parameter affecting the ES provision capacity both directly and indirectly. For instance, moisture content determines water availability for plants and consequently primary production (Chen et al. 2014). Soil moisture condition significantly affects the capacity of ecosystems to absorb water, which can result in different capacity to buffer flooding events and to regulate water fluxes (Massari et al. 2014; Yang et al. 2015).

Morphology adds additional spatial variation to ES provision. Ecosystems respond to altitudinal gradient by adapting their structure to the variation of climate conditions (temperature, solar radiation, atmospheric pressure, wind speed) and soil properties (Murphy et al. 1998; Becker et al. 2007). Slope is also an important factor affecting vegetation structure and soil; it is particularly relevant when assessing soil erosion prevention, since the erosion risk is higher where the slope is steeper and the vegetation coverage is less intense (Zaidat and Taimeh 2013). Morphology is also a leading variable to determine to distribution of human activities and related impacts affecting land use distribution (Becker et al. 2007).

Living organisms provide different ecological functions according to their abundance and their interactions. Generally, inter-specific interactions increase with the number of species that live in the ecosystem and are more sensitive to negative impacts than the latter (Valiente-Banuet et al 2015). Pollination, seed dispersal and biological control are the services that most directly depend on species interaction (Kremen et al 2007), supporting crop productivity and driving ecological successions (Verdu et al 2009).

Temporal changes influence positively or negatively ESs when involve phenomena that affect: i) climate conditions, ii) biological, chemical and physical properties of soil and iii) water bodies, iv) species and their interactions.

Temporal ES changes can be caused by different phenomena (both natural and human-induced), often occurring with synergic effects (Schröter et al. 2005). Among these, land use/land cover (LULC) change is probably the most complex and detrimental worldwide. LULC changes include transitions between land cover types, shift in management practices and loss of specific ecosystem attributes (see Section 1.3).

ESs provision is also affected by the intrinsic temporal variability of some ecosystems. For example, non-perennial water bodies (e.g. intermittent rivers, temporary ponds, etc) alternate the performance of different ecological functions. Organic matter transport, ecosystem metabolism, nutrient retention and species distribution in intermittent streams are strongly driven by drought and post-drought recovery cycles (Acuña et al 2005; Von Schiller et al 2008).

The invasion of non-native species in new ecosystems is also a type of temporal change, commonly occurring worldwide (McGeoch et al. 2010) and affecting the provision of ESs (Pejchar and Mooney 2009). ES losses can be due to alteration of community structure, which can lead to direct impacts on provision ESs with decline of economically valuable species or indirect impacts with the potential loss of resilience and resistance to perturbations (Charles and Dukes 2008). Impacts on regulation services, as nutrient cycling (Ehrenfeld 2003), disease regulation (Hogan et al. 2007), water quality regulation (Carlsson et al 2004) and cultural services, as recreation (Koel et al 2005), are also described in the literature.

Often, spatio-temporal changes determine the positive and/or negative variations of multiple ESs rather than a single service. This introduces to the relationships among ESs, which include three different possibilities: i) trade-offs, when the provision of one service results in the detriment of one or more services, ii) synergy, when the provision of one service results in the increase of one or more services and iii) neutrality, when the provision of one service has no effect on other services (Bennet et al. 2009). Generally, human activities boost the exploitation of few market-ESs with the complete or partial loss of other services (i.e. trade-offs). This is particularly evident in ecosystems which are more intensively manipulated by humans and whose natural components are intensively stressed, such as agro-ecosystems.

Depicting and managing trade-offs require analysis at various scales, since the ecological functions depend on local conditions but the interactions among ESs can occur at multiple scales. For this reason, ES analysis is often performed at the so called “landscape scale”. There is no an univocal definition of landscape scale, rather it is adopted to define a large spatial scale that address a range of ecosystem processes, conservation objectives and land uses (The Natural Choice 2011). Consequently, given the scale extension required, the spatio-temporal analysis of ES provision needs for the exploitation of LULC map sources and the use of geo-informatics tools as GIS (Geographic Information Systems) to process them.

1.3. Land Use/Land Cover changes and ESs

According to United Nations Food and Agriculture Organization (FAO), "*land cover is the observed (bio)physical cover on the earth's surface*", [while] *land use* [is defined] *by the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it*" (Di Gregorio et al 2000).

Despite the two terms have different meanings, they are commonly used as synonyms. This is often due to the difficulties to distinguish clearly human uses from the land coverage elements. This work will adopt the mixed term "Land Use/Land Cover" (LULC) when no clear discrimination between these two terms can be stated.

Among the causes for the variation of ES provision, the LULC change is probably the most relevant worldwide (Foley et al 2005; Metzger et al 2006; Costanza et al 2014). The consequences of LULC changes on ESs provision are widely described and studied in literature with growing interest during last years (Fig. 1.3.1).

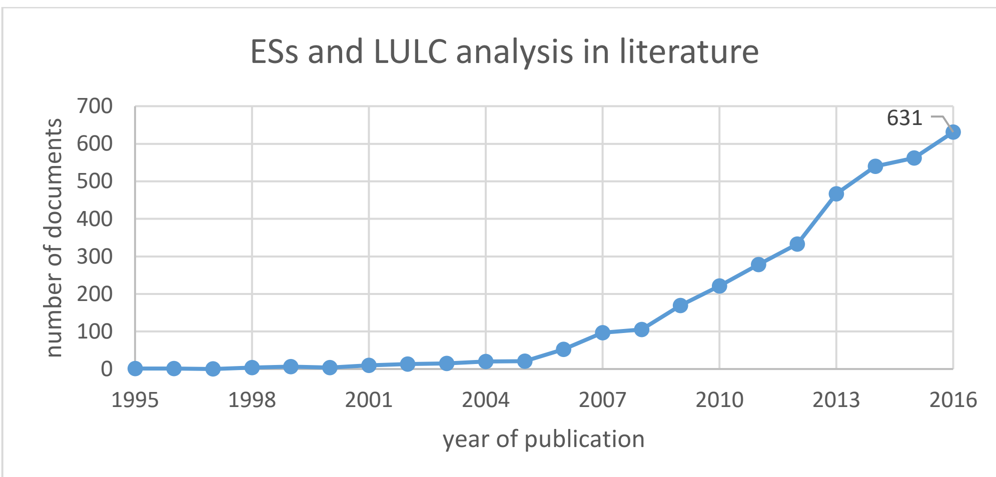

Fig. 1.3.1: Number of document published per year that include the terms "ecosystem services" and "land use" or "land cover" in title, abstract or keywords (source: SCOPUS, data of inquire 12/01/2018). The total number of indexed documents was 3550.

LULC changes can be either natural or human-induced, even though the latter are those largely more common.

Natural LULC changes are rare but can affect significantly ES provision capacity. For example, volcanic eruptions can influence ecological successions (Teramoto et al 2017) and primary productivity even after different years (Krakauer and Randerson 2003). Ecological successions led to changes in ecological structures and functions until the achievement of the climax condition (Stoy et al 2008; Whitfeld et al 2014), while natural eutrophication heavy alter water bodies affecting their ecological functions (Venugopalan et al 1998).

The human-induced alterations occur through changes in land use as a consequence of socio-economic drivers (Lambin et al 2001), often with controversial outcomes. The most common drivers for human-induced LULC changes are related to the increase of population (Meyer and Turner 1992), food (DeFries et al 2004), timber (Sierra 2001) and energy demands (Hoogwijk et al 2005), which lead to human appropriation of natural lands for the conversion to croplands or urban areas, and general detrimental impacts on natural ecosystem status and biodiversity (Dale et al 1994;Poschlog et al 2005; Guirado et al 2006).

The (mostly negative) impacts of LULC change on ESs can occur mainly through three different processes:

i) transition of one ecosystem type to another;
ii) intensification of land use;
iii) alteration of ecosystem attributes/functions.

The transition from one ecosystem type to another is the most drastic process of change. This results in the complete loss of the ecological functions of the previous ecosystem and the gain of new ones provided by the second. Human-induced transitions generally shape the landscape in order to obtain arable land and space for urban settlements, under the pressure of increasing socio-economic demands, particularly in developing countries. For example, the clearance of forested areas to obtain land for food, energy or cash crops were widely described in literature (e.g. López-Carr and Burgdorfer 2013). In this case, deforestation led to a complete loss of a wide set of ESs provided by forest in order to obtain a single provision service.

The intensification of land use involves the increase of impacts deriving from human activities on ecosystems functions. For instance, the production of wood for energy requires intense forest management practices, focused on the maximization of wood provision and with the removal of many natural attributes, such as understory vegetation and deadwood, and the detriment of soil properties (Routa et al. 2012). The negative effects of land use intensification are particularly evident in agro-ecosystems (Bommarco et al 2013). Wilby and Thomas (2002) defined "agricultural intensification" as the "*increased management intervention and increased external inputs with the intent of increasing agricultural yield*". The maximization of crop yields requires intense applications of chemicals that affect water quality and biodiversity (Johnston et al. 2011). The use of pesticides strongly alter pollinator population (e.g. bees) with important impacts on pollination service (Stanley et al 2015). Removal of vegetation in buffer zones, together with nutrient leaching due to fertilizer applications, causes the decrease of water quality regulation capacity (Sabater et al. 2000).In agricultural land the complete removal of crop residues and intense tillage induce a decrease soil quality and nutrient cycling by causing the loss of carbon storage. Loss of soil quality regulation can be the result of the detriment of soil organic

content and soil biota (Calzolari et al. 2016). Moreover, cultural services as aesthetic quality and sense of place can suffer losses because of simplification of agricultural landscapes (Dramstad et al. 2006).
Finally, humans can alter ecosystem attributes and functions, directly or indirectly, as consequence of their activities. This can be observed, for example, when a given ecosystem patch suffers impacts coming from the interaction with other patches throughout the landscape (i.e. effects due to changes in landscape composition). In case of water bodies, change in water parameters can affect ecological functions, even without LULC transitions or land use intensification (see Section 1.2).
This pathway of changes includes also natural processes as ecological successions. In this case, the ecological functions changes according the succeeding of different stages along time (Mitsch et al. 2005), without a clear complete transition to another ecosystem type (e.g. from grassland to forest).
LULC changes and relative responses in ES provision can be described and quantified using information technologies. Geographic Information System (GIS) softwares have proven to be useful for mapping and assessing the current, past and future distribution of ESs in different contexts and applications (Nemec and Raudsepp-Hearne 2013). GIS techniques can be used to create and analyze maps, perform spatial analysis and geoprocessing, run GIS-based models and support public participation (Troy and Wilson 2006; Grêt-Regamey et al 2008; Brown et al 2012). Moreover, advancements in GIS technology, together with the increasing availability and quality of spatial data, enhanced the potential and applicability of ES mapping (Bateman et al., 2002; Maes et al 2012). For these reasons, GIS applications provided a valuable support for advances in LULC and ES sciences during last years (Fig.1.3.2). It has to be mentioned that the number of documents considered in Fig.1.3.2 is likely to underestimate the real contribution of GIS applications, since many published researches implicitly apply GIS processing, without mention it in title, abstract or keywords.

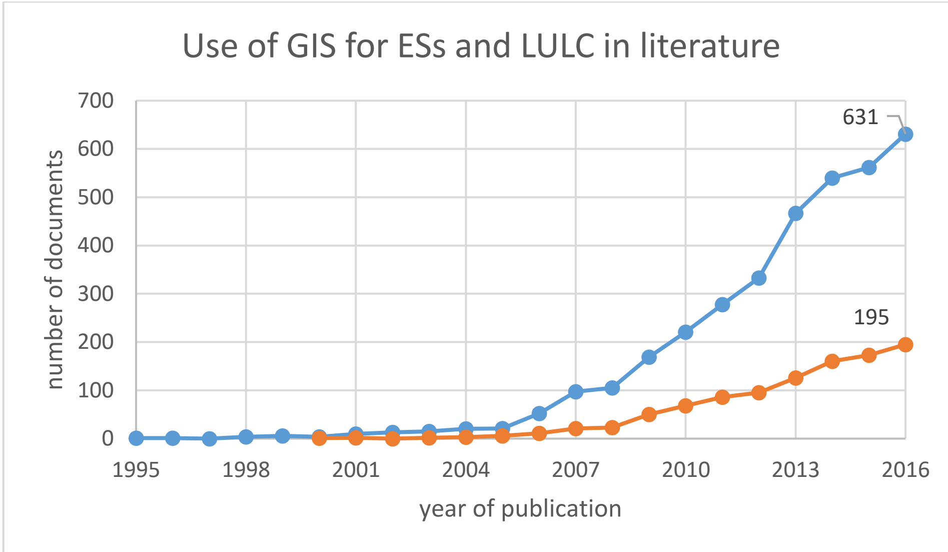

Fig. 1.3.2: Number of document published per year that include the terms “ecosystem services” and “land use” or “land cover” and “GIS” in title, abstract or keywords (source: SCOPUS, data of inquire 12/01/2018). The total number of indexed documents was 1022.

Application of remote sensing technologies lead to potential benefits in ES science (Fig.1.3.3), by improving the potential for spatially explicit assessments (de Araujo Barbosa et al 2015). Remotely sensed data provide information on vegetation and land coverage, which are often processed to obtain LULC maps.

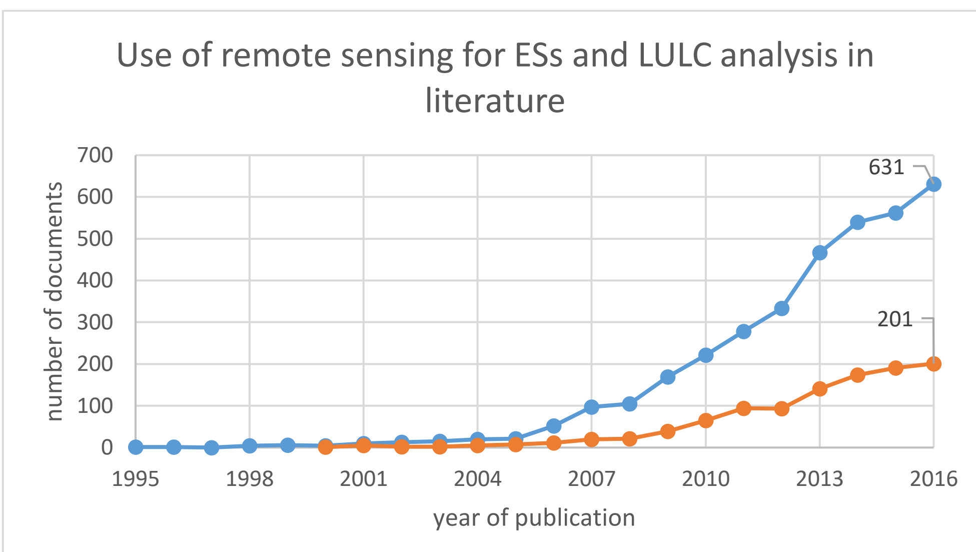

Fig. 1.3.3: Number of document published per year that include the terms “ecosystem services” and “land use” or “land cover” and “remote sensing” in title, abstract or keywords (source: SCOPUS, data of inquire 12/01/2018). The total number of indexed documents was 1074.

Temporal changes can be detected by processing remotely sensed data relative to different dates to obtain comparable products. Additional information on ecosystem attributes can be obtained by elaborating vegetation indexes (Krishnaswamy et al. 2009). Remote sensing technologies include the elaboration of images captured by passive sensors, as satellite scenes and aerial imagines, and active sensors. The latter can provide more accurate information, as three-dimensional images at finer resolution. For example, Lidar (Light Detection And Ranging) sensors directly measure the three-dimensional distribution of plant canopies, thus providing high-resolution topographic maps and highly accurate estimates of vegetation height, cover, and canopy structure (Lefsky et al 2002). Nonetheless, the application of active sensors is limited by the high cost of survey and the requirement of specific informatics competences.
Modelling ESs provision as a consequence of LULC changes is a fundamental step for ES mapping and assessment. The development and the application of method for modelling ESs gained great attention in literature (Fig. 1.3.4). The strength and limitations of the method directly affect the results of the analysis. Detailed analysis of the different methods are provided in Section 2.

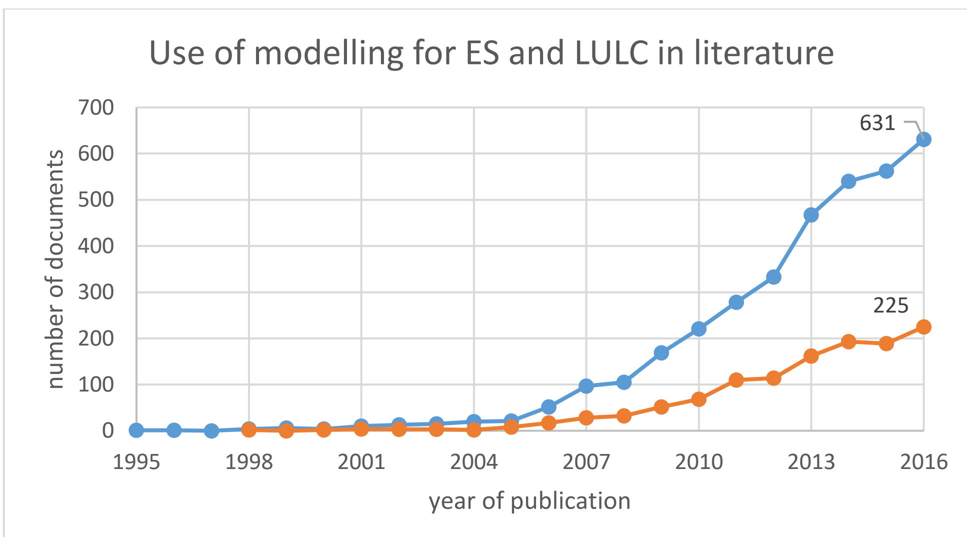

Fig. 1.3.4: Number of document published per year that include the terms “ecosystem services” and “land use” or “land cover” and “spatial model*” in title, abstract or keywords (source: SCOPUS, data of inquire 12/01/2018). The total number of indexed documents was 1214.

Overall, as can be observed in Fig.1.3.2, 1.3.3, 1.3.4, information technologies and spatial modelling supply a relevant contribution to LULC and ES assessment. Their applications can provide effective tools for monitoring ecological outcomes of environmental management and elaborate methodologies for include ESs in decision-making processes.

1.4. ES assessment applications: ES provision in Protected Areas and Tools for decision-making

The ES concept is based on an anthropocentric view of nature, since its main aim is to provide valuable arguments for linking nature and human well-being. For this reason, the practical application of ES assessment efforts is a key challenge in ES science (Burkhard et al 2013).

For example, ES assessments can be used to describe the effects of environmental conservation and address the future management of areas under protection. Protected Areas (PAs) are the most common conservation instrument adopted worldwide to arrest biodiversity loss and safeguard threatened species and ecosystems (Watson et al. 2014). PAs were originally established to safeguard iconic landscapes, biodiversity conservation objectives were considered only after the 70ies. During the last years, a new paradigm for PAs was proposed, adding the target to delivery ESs to humans (Watson et al. 2014). This can provide also valuable arguments for new financial support and for biodiversity conservation. Nonetheless, just a relatively limited number of studies concerning the ES provision in PAs is reported in scientific literature.

The understanding of the relationship between biodiversity and ESs is a fundamental issue in ES science, as well as to inform the management of PAs.

A general positive effect of biodiversity levels on provision and stability of ecosystem properties and services was observed in space and time (Tillman 1996, Naeem and Li 1997; Yachi and Loreau 1999, Hooper et al. 2005; Balvanera et al. 2006). However, biodiversity-ES relation is complex and generalizations among ecosystem types, properties and services or trophic levels manipulated or measured are difficult to sustain (Balvanera et al. 2006).

Several complexities emerge when investigating the relationships between biodiversity and ecosystem attributes. The effects of biodiversity alteration can be complex and differ at different ecological levels (Naeem & Wright 2003; Balvanera et al 2006). Naeem & Wright (2003) argued that determining a general relationship between taxonomic and functional diversity is neither necessary nor desirable. Balvanera et al. (2006) highlighted that biodiversity changes at community level have higher impacts on ecosystem processes when compared with those at ecosystem level, while changes at population levels seemed to exhibit negative effects.

The argument is even more complex when extending the relationship to ESs.

Harrison et al. (2014) found that biodiversity has positive correlations with water quality and flow regulations, mass regulation and landscape aesthetics (community and habitat diversity), pollination and recreation (species abundance), timber production and freshwater fishing (species richness). Conversely, water supply is negatively affected by biodiversity, while effects on pest regulation seem controversial.

In addition, data scarcity and mismatches between measured variables and ESs are additional sources of complexity and uncertainties (Balvanera et al. 2013).

Overall, even with the uncertainties due to the numerous interactions occurring in complex systems, biodiversity conservation (when effective) is expected to have general positive effects on ES provision capacity.

Despite the conservation efforts worldwide and the contribution of the scientific community to mainstreaming biodiversity-ES relationship, the achievement of the global 2010 targets for biodiversity conservation failed.

The tenth meeting of the Conference of Parties (COP 10) to the Convention on Biological Diversity (CBD), held in 2010, declared the adoption of a global Strategic Plan for biodiversity for the period 2011–2020. The ‘‘2020 Aichi targets’’ confirmed the previous biodiversity conservation targets with the addition of ES. In 2011, the EU adopted the Biodiversity Strategy specifically pointing out the mapping of ESs among its targets. Particularly, the target 2 states "*By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15 % of degraded ecosystems*"; while the target enounces that "*By 2020, the conservation of species and habitats depending on or affected by agriculture and forestry, and the provision of their ecosystem services show measurable improvements*".

Despite the awareness raising to the subject, the operationalization of ESs is yet poor (Gissi et al. 2015). Rall et al (2015) found that very few policy documents explicitly mention ESs, and that stakeholders show little exposure to ES modeling and major ESs initiatives. Cowling et al (2008) proposed an operational framework to include ES in decision-making, in three phases: assessment, planning, and management. The assessment phase provide information suitable for the other two phases. The ES assessment can provide suitable tools to solve complex conflict problems in spatial planning of environment and then contribute to implement ES theory into practice. Therefore, the implementation of ES assessment requires tools that are adapt to be transferred to the planning phase. Such tools need be to both scientifically correct and accepted by stakeholders. According to Wong et al (2015), decision-makers need credible and legitimate measures to evaluate decision for trade-offs.

In this respect, the development of tools for ES implementation needs to be case-specific in order to be responsive for the specific context (policy objectives, socio-economic conditions, scales).

1.5 Research questions

This study aims to provide applications of ES assessments to investigate the role of Protected Areas (PAs) in maintaining the provision of ES over time. Mapping ES provision capacity of PAs has been included among the requirements of EU Biodiversity Strategy to 2020, which aims to tackle the loss of biodiversity and ESs in the EU. Consequently, this investigation can provide suitable information to environmental managers on how to manage PAs to meet the targets fixed by the Biodiversity Strategy. The result can help non-EU decision-makers, according the paradigm shift concerning the role of PAs, from simple biodiversity hot spots and landscape amenities to important providers of ESs (see Chapter 1.5).

Moreover, ES assessment can support environmental planning, particularly in contexts with high levels of human impacts. Putting ES theory in practice requires the design of tools to support environmental planning decisions, in order to mitigate conflicts and harmonize the simultaneously achievements of different targets.

Overall, the research questions are:

- Are PAs effective in maintaining the ES provision capacity?
- How ES assessments can be usefully implemented in environmental planning to support most efficient and sustainable solutions for human well-being?

In order to provide evidence to answer to aforementioned questions, different case studies are presented in Section 3. Each case study is introduced by a brief presentation, which explain the relevance and the contribution to this investigation.

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2. ES modelling in changing landscapes

ES modelling (i.e. the way in which data sources are used to assess ES supply) links available information on landscape and ecosystems with ES assessment. This provides the use of ecological production functions with different levels of complexity.

The assessment of ES provision capacity includes: i) biophysical and/or monetary quantification and ii) spatial distribution mapping. ES modelling can detect variation of ES delivery when is applied as a function of LULC changes, providing important information for researchers and decision-makers.

However, all the different ES mapping methods have weaknesses and strengthens. The most suitable modelling method should be chosen according the final purpose of the analysis, the specific conditions and the scale of the study area and the data availability. To support the proper setting of ES assessment, the Mapping and Assessment of Ecosystems and their Services (MAES) project and Grey-Regameit et al (2015) suggested a tiered approach based on three levels. The higher is the tier level the higher are the complexity of indicators and the required expertizes for ES modelling, concerning GIS and GIS-based tools. The tier-1 is the simplest method and includes the modelling using available data. The most common indicators are derived from LULC maps (e.g. Corine land cover). The tier-2 level includes the link of different indicators with the LULC data according to known relationships between land use and ES provision, to obtain indicators that are more complex. The tier-3 level requires the modelling of biophysical processes in a GIS or in other software, instead of linking indicator data through simple relationships.

Different authors reviewed and classified the different methodologies adopted to model ESs provision. Martínez-Harms and Balvanera (2012) classified mapping researches in primary and secondary data sourced. The latter are further divided in three methods: “look-up tables”, “expert knowledge” and “causal relationship”.

Egoh et al (2012) classify three groups: “primary data”, “proxies” and “process models”.

Schägner et al (2013) proposed a classification in five methods: “proxies”, “non-validated models”, “validated models”, “representative data” and “implicit modelling”. Englund et al (2017) categorized mapping methods in: “direct mapping”, “empirical models”, “simulation and process models”, “logical models”, “extrapolation” and “data integration”.

The proposed classifications of different methods are forcedly ambiguous, since the assessment methods are often not clearly distinguishable and study cases not attributable to a specific class.

This work classifies ES modelling in three categories, listed according increasing complexity levels as following:

i) Benefits transfer
ii) Indicators

iii) Model tools

These methodologies roughly correspond to the three tier levels proposed by Gret-Regameit et al. (2015) are described in detail in the following sections.

2.1. Benefits transfer

The benefit transfer method is defined as the practice of extrapolating levels of ES provisioning (per unit area) measured for a mapped class in one location to all occurrences of that class in a study extent, which may differ from that of the original measurement (Andrew et al 2015). The accuracy of the results largely depends on the representativeness of initial study (Plummer 2009). The estimates, derived either within the study area or, more typically, elsewhere, are spatialized by linking attributes to mapped landscape units (typically LULC) (Andrews et al 2015).

The benefit transfer method roughly corresponds to the tier-1 level of MAES framework and to the "look-up tables" of Martínez-Harms and Balvanera (2012) classification.

This method is the most adopted in literature because of its simplicity and adaptability. In fact, the benefit transfer requires only two inputs (i.e. area extensions for each LULC types and ES value per unit area), and, therefore, enables ES mapping in regions where primary data are lacking (Eigenbroad et al 2010).

The benefit transfer can be applied to monetary (Costanza et al 1997; Sutton and Costanza 2002; Troy and Wilson 2006) or biophysical values (Balthazar et al 2015).

On the other hand, several limitations have to be considered. The relationship between areas and a given ES (or a set of ESs) is assumed linear, despite the fact that many factors related to landscape patterns and local conditions of different patches (see Section 1.3) affect the ES provision. Moreover, even when accurate, initial values can be quickly dated because of the temporal variations of bio-physical properties (i.e. ES provision capacity) and socio-economic dynamics (i.e. ES demand).

The relevance of these values on the total ES values can be assessed with sensitivity analysis in order to test the weight of possible bias. Nonetheless, sensitivity analysis in benefit transfer studies are often based on erroneous assumptions derived from economic theories (See Section 1.5.1.1).

However, the benefit transfer may be suitable for identifying general trends of variation in ESs at different scales to inform decision-makers and raise public awareness (Costanza et al 1997; 2014). It can be suitable when higher levels of modelling (i.e. higher tier-level in MAES) are not applicable for lack of data or for intrinsic ecological complexity of study area. The method can also be applied as a screening technique to determine if a more detailed study should be conducted.

2.1.1. "Criticism on elasticity-sensitivity coefficient for assessing the robustness and sensitivity of ecosystem services values"

When benefit transfer analysis involves monetary values (i.e. values transfer), the analysis goes beyond the only assessment of ES provision capacity and involves socio-economic dynamics, which add the dimension of human demand. In these cases, sensitivity analysis are often performed to test the "robustness" of the analysis, by the use of Coefficient of Sensitivity (CS). The latter is based on the concept of economic elasticity, to describe the sensitivity in demand of a specific good or service in response to changes in its price. The CS is widely applied in hundreds of scientific ES studies.
This Section provides criticism to the use of such practice, providing evidence for uncorrected interpretation of concepts imported from economic sciences with the aim to incorporate human dynamics into ES science.

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Abstract:

The Coefficient of Sensitivity CS (or coefficient of elasticity) is used to determine the sensitivity and robustness of prices (coefficients) in the analysis of Ecosystem Services (ESs). The common CS approach is applied based on a specific % change of an ES coefficient keeping constant the coefficients of the remaining ESs. This approach assumes that when CS value is >1 then the estimated ES value is non-robust because it is elastic. The aim of this study is to show that the common approach of CS used in ESs studies is erroneously applied and interpreted. A simplistic calculus is provided which shows that the CS values of ESs a) are always in the range between 0 and 1 leading always to the conclusion that the applied coefficients by the users are robust, and b) are always independent by the % change of an ES coefficient defined by the user. Other reasons which question the validity of the common approach are that the CS values a) are always positive which is unrealistic in real market since it always violates the "law of demand" and b) can be manipulated by the user by changing the boundaries of the study area.

Introduction

The inclusion of Ecosystem Services (ESs) approach (Costanza et al., 1997) to assess the direct and indirect economic contribution of ecosystems to human welfare has given significant merit in decision making related to environmental management (Kareiva et al., 2007; Fisher et al., 2009). The robustness of the approaches used for ESs assessment is strongly based on the use of realistic ESs prices (coefficients) provided by the researchers. Due to the large uncertainty of these coefficients (Schmidt et al., 2016), many studies use the simplistic approach of sensitivity coefficient CS of Mansfield (1985) as proposed by Kreuter et al. (2001) in order to determine their sensitivity and robustness.

The CS coefficient is based on the concept of elasticity, which is used in economics to describe the sensitivity in demand of a specific good or service in response to changes in its price (Gwartney et al., 2006). The analysis of elasticity is based on the ratio between the percentage change in quantity demanded and the percentage change in price of a good/service (this ratio is equivalent to CS). When the absolute value of the ratio is <1, the demand is considered inelastic, which indicates that changes in price have a relatively small effect on the quantity of the good/service demanded. When the absolute value of the ratio is >1, the demand is considered elastic, which indicates that changes in price have a relatively large effect on the quantity of a good/service demanded.

The aforementioned approach is applied in the case of ESs where the threshold of unity is also considered as a measure of robustness for the ESs values (CS<1 defines robust and inelastic coefficients) (Kreuter et al., 2001). Taking into account the international literature, the use of the aforementioned approach has been applied in many studies where the CS values are always in the range 0<CS<1 (Li et al., 2007; Hu et al., 2008; Tianhong et al., 2010; Yoshida et al., 2010; Hao et al. 2012; Wang et al., 2014a, b; Zhang et al., 2015a, b; Fu et al., 2016; Crespin & Simonetti, 2016; Fei et al., 2016; Kindu et al., 2016).

The aim of this short communication is to present a simplistic calculus and other justifications, which show that the common CS approach used in ESs studies is erroneously applied and interpreted.

The common CS method for ESs sensitivity analysis

The CS is usually applied based on a specific percentage change of an ES coefficient keeping constant the coefficients of the remaining ESs. In the context of ESs framework, the CS is calculated by the formula (Kreuter et al., 2001; Mansfield, 1985):

$$CS = \frac{\left(ESV_j - ESV_i\right) / ESV_i}{\left(VC_{j,k} - VC_{i,k}\right) / VC_{i,k}} \quad (1)$$

where ESV is the total estimated value of all ESs (in monetary units per year), VC is the value coefficient (monetary units per year per unit area), i and j represent the initial and adjusted values, respectively, and k represents the land use category. For the calculation of Eq.(1) a predefined % change is usually used for all coefficients (e.g. ±50%). In this study, we don’t use a fixed value of change but a general value equal to x. The value of x is used here as coefficient and not as percentage (e.g. -30% change of VC corresponds to x=0.7 while for +30% of change x=1.3).

Results

If we assume that the initial VCi,k of a land use category k is changing based on the x coefficient and the VC values of the remaining land uses are constant, then the adjusted values of VCj,k and ESVj of Eq.1 are equal to:

$$VC_{j,k} = x \cdot VC_{i,k} \quad \text{and} \quad ESV_j = ESV_i - (1-x) \cdot VC_{i,k} \cdot A_k \qquad (2a,b)$$

where Ak is the area of land use k (in area units).

Taking into account Eq.(2a,b), then Eq.1 is readjusted according to the following:

$$CS = \frac{(ESV_j - ESV_i)/ESV_i}{(VC_{j,k} - VC_{i,k})/VC_{i,k}} = \frac{\left[(ESV_i - (1-x) \cdot VC_{i,k} \cdot A_k) - ESV_i\right]/ESV_i}{(x \cdot VC_{i,k} - VC_{i,k})/VC_{i,k}} = \frac{-(1-x) \cdot VC_{i,k} \cdot A_k}{(x-1) ESV_i} = \frac{VC_{i,k} \cdot A_k}{ESV_i} \qquad (3)$$

The final result of Eq.3 has the following attributes: a) is independent by x and consequently by the % change of the VC value selected by the user and b) is always in the range between 0 and 1.

Discussion

The use of Eq.1 in the ESs framework either as elasticity or simply as sensitivity index should no longer be used following the approach of section 3 for the following reasons:

- when Eq.1 is used to examine the elasticity of the VC coefficients, the CS values range always between 0-1 leading to the conclusion that the used VCs are inelastic and consequently robust. This finding questions by itself the validity of the formula for this purpose.
- According to economic theory, CS application to real market conditions usually yields a negative value, due to the inverse nature of the relationship between price and quantity demanded, as described by the "law of demand" (Gwartney et al., 2006). Despite the fact that the "law of demand" can be violated

in many cases (e.g. Veblen and Giffen goods) in real market (Choo et al., 2007), in the ESs framework the law is always violated since CS is always positive. This suggests that the results of the common CS approach in the ESs framework are unrealistic especially for those services directly related to the market (e.g. food production).

• when Eq.1 is only used to examine the sensitivity of the VC coefficients, the CS values are always independent by the % change of the VC coefficient selected by the user as indicated by the final form of Eq.3. Eq.1 can not be considered as sensitivity formula in ESs framework but only as a ranking index that defines which land use is more important in the total ESV.

• Either as elasticity or sensitivity or ranking index, Eq.1 can be manipulated by the user because its results are related to the geographic extent of the land uses. When one land use has an extremely large % coverage in a study area or a large VC coefficient, its CS value is expected to be proportionally high. The user can reduce the extent of this land use by changing the boundaries of the study area in order to manipulate the CS values. Again, this suggests that the CS approach can not be used for assessing the robustness of ESs values. The arbitrary delineation of the boundaries of the study areas used in ESs services affects not only the results of the CS but also the results of all the other compartments related to ESs analysis. For this reason, rules for the delineation of the study areas should be adopted in the ES framework. Some suggestions to avoid such criticism could be the use of boundaries related to administrative units (e.g. provinces, prefectures, cantons etc) (Gaglio et al., 2016; Gissi et al., 2016), because they constitute economic entities of the states, or physical boundaries such as natural hydrologic basins (Tian et al., 2016) because they constitute the most common base for development of environmental management strategies.

Conclusion

The study provided proofs and justifications, which show that the common approach of elasticity-sensitivity coefficient used in many ESs studies is erroneously applied and interpreted. Our observations suggest that this approach should be abandoned for assessing the robustness and sensitivity the ESs coefficients.

2.2. Indicators

Indicator method represents the second level of ES modelling and can be considered within the category "causal relationship" of Martínez-Harms and Balvanera (2012) and "proxies" of Egoh et al (2012).

An ecosystem service indicator is information which communicates the characteristics and trends of ecosystem services, making it possible for policy-makers to understand the condition, trends and rate of change in ecosystem services (Layke et al 2012). In this case, indicators include a no simple linear relation with LULC type.

These indicators can be obtained by the combination of causal variables. Unlike the benefit transfer method, they take into account local conditions that concur to ES provision. Indicators incorporate existing knowledge about how different layers of information related to ecosystem processes and services to create a new proxy layer of the ESs (Martínez-Harms and Balvanera 2012).

For instance, Calzolari et al (2016) modelled the provision capacity for habitat for soil organisms by combining maps of bulk density and soil organic matter content. Schulp et al (2014) modelled the pollination service capacity by combining bee potential habitat and the distance to pollinator-dependent crops.

ES modelling using indicators requires GIS skills for geo-processing and spatial analysis. They provide more accurate and finer information to researchers and decision-makers than benefit transfer approach.

2.3. Model tools

The use of model tools represents the finest method for ES assessment. ES models (*sensu stricto*) are representations/simulations of complex ecological processes occurring at different scales.

They differ from logical indicators by increasing complexity of ecological production functions. Often, spatially explicit models are based on more ecological production functions, which are combined in order to obtain simulations and predictions of given ecological phenomena.

Model tools can be directly related to ESs, or specifically built to assist analysis in other disciplines.

Models directly related to ESs are applied for the assessment of multiple ESs, with the aim to describe ES variation in space and time as a consequence of human intervention and/or climate changes (Nelson et al 2009), as well as to identify priority areas for the environmental management.

ES-related models usually work at landscape scale and can be applied at different landscape types. For example, the InVEST model (Sharp et al 2016) is widely applied for these purposes. It provides a set of tools specific for each ES (e.g. climate mitigation, water regulation, erosion control, etc) designed with aim to inform decisions about natural resource management. Other tools belonging to the same category are ARIES, MIMES, LUCI and many others (Bagstad et al 2013). Additionally, some tools were specifically developed to specific type of ecosystem, e.g. i-Tree can be applied for urban landscapes.

ES analysis can be carried out also by using non-directly related ESs tools. Such models were not originally developed for ES assessment purposes but can support such studies providing analysis on ecosystem functionality supporting single services. For instance, hydrological models as SWAT were used to model soil and water-related ESs (Francesconi et al 2016). SWAT studies include the analyses on water yield, nutrient and sediment transport related to different applications and drivers of changes (Krysanova and White 2015).

Specific hydrological models can provide more accurate outputs but require a huge amount of data inputs and specific expertise on hydrology. When more ESs have to be assessed simultaneously (and generally this is the case of ES assessment), including water- and non-water related services, ES-tools can be more appropriated in capture information needed to decision-makers (Vigerstol and Aukema 2011).

Applications of model tools are limited by the facts that they are time-consuming and generally require high competences in GIS analysis and geoprocessing. Posner et al (2016) demonstrated that the use of InVEST model tools among countries is related to the capacity to use the tools. Therefore, formal training efforts are fundamental to build the capacity to use decision tools.

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3. Case studies

In order to investigate the two research questions stated in Section 1.5, different case studies are presented in this section (Tab.2). The analysis described in Sections 3.1, 3.2 and 3.3 concern case study on LULC changes in PAs, with aim to evaluate the effects of environmental conservation on ESs. Sections 3.4 and 3.5 provide possible applications for ES mapping in spatial planning.

The five case studies provide evidence for the research questions using the different mapping methods described in Section 2 at different scales. All the pathways of changes for ESs listed in Section 1.3 were observed. A brief presentation of each case study is provided at the beginning of each section, to describe its contribution to the analysis.

Section	**Mapping method**	**Scale**	**Environment**	**Administrative unit**	**LULC change pathways (directly detected or the process dealing with)**		
					transition from one ecosystem type to another	intensification of land use	alteration of ecosystem attributes/ functions
Section 3.1	Benefit transfer (Monetary)	Landscape (13 000 ha)	Transitional	PA	X	X	X
Section 3.2	Benefit transfer (Biophysical)	Landscape (40 000 ha)	Tropical Mountains	PA	X		
Section 3.3	Model tools	Landscape (800 ha)	Floodplain	PA	X		X
Section 3.4	Model tools	Municipality	Urban	Municipality			X
Section 3.5	Indicators	Regional	Agricultural land	Region		X	

Tab.2: The five different case studies and relative details presented in Section 3.

3.1. “Land use change effects on ecosystem services of river deltas and coastal wetlands: case study in Volano-Mesola-Goro in Po river delta (Italy)”

This section presents a case study in a protected area of Po river delta (Northern Italy), which was subjected to extensive land reclamations in the past. The effects of land use changes on ESs during two different periods are discussed to assess the effect of environmental conservation on ESs. The benefit transfer method, based on monetary values, was applied for ES mapping at landscape scale.
Wetland and aquatic vegetation losses were found to be the main causes of ES decrease, as examples of two ES change pathways: transition from one ecosystem type to another and alteration of ecological attributes/functions, respectively.

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Abstract

The landscape of river deltas and coastal wetlands is under a continuous alteration due the combined effects of human and natural factors. The aim of the study is to analyze the Land Use/Land Cover (LULC) changes and associated Ecosystem Services (ESs) of a protected wetland area in the Po river delta (Northern Italy). A combination of methodologies which take into account both the assessment of socio-economic benefits (approach of ESs) and the monitoring of ecosystems attributes (LULC change analysis using transition matrices TMs) were used by comparing the changes observed during two periods (1954-1976 and 1976-2008) described by different environmental protection governance. The period 1954-1976 is described by extensive land reclamations while the period 1976-2008 by significant efforts for applying environmental protection measures. The results highlighted an extensive loss of vegetated wetlands due to direct human interventions (croplands and urban areas expansion) during the first period. The direct human intervention was significantly reduced during the second period. However, vegetated wetlands losses did not follow an analogous reduction probably due to indirect human interventions and natural factors. TMs identified the exact LULC conversions while the ESs approach highlighted the significant economic impact of vegetated wetlands’ losses. Waste treatment was the most important ES of the specific system providing approximately 70% of the estimated natural capital value. The proposed combination of the selected methods (TMs and ESs) provides a detailed description of landscape changes and their economic impact, which can be used as decision support tool for landscape conservation policies.

Introduction

The economic evaluation of the environment has found a scientific dimension through the concept of ecosystem services (ESs) (Gissi et al. 2015). ESs have become a central issue for researchers and decision makers since the late '90s, and particularly after the publication of Millennium Ecosystem Assessment (MEA 2005), filling the gap between natural protection and human welfare (Fisher et al. 2009).

Despite the increasing awareness, during the last decades, that human well-being strongly depends on natural ecosystems (Egoh et al. 2007), human populations continue to alter the landscape and natural lands, as a consequence of socio-economic and socio-ecological phenomena (Lambin and Meyfroidt 2010; Lambin et al. 2001), at extraordinarily high rates (Lambin et al. 2003). Therefore, Land Use/Land Cover (LULC) changes are considered the major form of anthropogenic pressure on the environment (Ellis et al. 2010), causing changes in ESs patterns, and a loss of biodiversity affecting ecological functions (Daily 1997; MEA 2005; Haines-Young and Potschin 2010).

River deltas are very sensitive environments since they are subjected to intense natural pressures associated with interactions between terrestrial, coastal and marine factors, which significantly increase their spatial and temporal heterogeneity (Wang and Yu 2012; Yang et al. 2013; Liu et al. 2014; Lane et al. 2015; Shen et al. 2015). The inclusion of human activities as an additional factor of pressure makes these systems even more vulnerable, leading to a high alteration rate of their landscape in time (Mikhailov and Mikhailova 2003; Litskas et al. 2010; 2014; Wang et al. 2012; Glenn et al. 2013; Aschonitis et al. 2012; 2013a; 2014). River deltas are usually formed downstream in large rivers where strong flooding phenomena, in combination with the interaction with the sea, lead to the formation of various aquatic ecosystem types such as coastal wetlands, coastal lagoons, salt marshes etc (Stanley and Warne 1993; Bhattacharya and Giosan 2003; Xie et al. 2013). These natural ecosystem types can co-exist with non-natural LULC types (e.g. agricultural lands and urban areas) providing a significant amount and variety of services to support human welfare (Rashleigh et al. 2012; Sawut et al. 2013; Aschonitis et al. 2013b; Li et al. 2015). The aforementioned natural aquatic ecosystems, and especially wetlands, provide a wide range of valuable services such as nutrient filtration (Knox et al. 2008), fisheries (Engle 2011), storm and flood protection (Costanza et al. 2008), recreation (Shrestha et al. 2002), highlighting their important ecological and economic value (Woodward and Wui 2001; Camacho-Valdez et al. 2014).

Despite their ecological and economic value, wetlands and river deltas suffered significant losses during the last decades (Coleman et al. 2008) particularly from the expansion of croplands and intensification of agricultural activities (Huang et al. 2012; Myers et al. 2013). The Ramsar Convention of 1971 was the first global nature conservation tool, which was adopted to protect wetlands and to tackle human impacts. The Ramsar Convention underwent a significant evolution in the following decades. The initial conservation approach was superseded in the early 1980s by the principles of 'wise use', according to which ecosystem properties should be maintained and restored (Hettiarachchi et al. 2015). Moreover, since the Ramsar Convention, several environmental policies and tools have been applied with the scope to mitigate the negative effect of anthropogenic pressures. Among these, the establishment of Protected Areas (PAs) was one of the most important protection strategies adopted. PAs are mainly focused on habitat and biodiversity conservation at any cost to the well-being of local and nearby populations (Petrosillo et al. 2009). For this reason, a combination of methodologies, which take into account the assessment of socio-economic benefits and the monitoring of environmental-ecological attributes and functions, is needed in order to evaluate the outcome of environmental policies and strategies for wetlands protection (Clare and Creed 2014; Marino et al. 2015). Apart from biodiversity, which is one of the most important indicators of ecosystems' health, the monitoring of natural capital and its sustainable use can be a suitable and strategic indicator of PAs effectiveness (Petrosillo et al. 2009), both in terms of environmental protection and governance (Marino et al. 2015). In this case, economic evaluation must not be considered as a pure monetary issue, rather as a proxy in order to describe and quantify the spatial-temporal trend of ESs pattern (Petrosillo et al. 2010).

The aim of the study is to analyze the LULC changes and associated ESs in the case of a wetland PA in the Po river delta (Northern Italy). A combination of methodologies which take into account both the assessment of socio-economic benefits (approach of ESs) and the monitoring of ecosystems attributes (LULC change analysis using transition matrices TMs) were used by comparing the changes observed during two periods (1954-1976 and 1976-2008) described by different environmental protection governance.

Study area and data

The study area is the administrative unit of Volano-Mesola-Goro (VMG) with coverage of 13,730 ha and altitude between -4 to 10 m above sea level (Fig.1). The VMG is part of the Po river delta (Italy). Po river is one of the larger rivers in Europe and its delta constitutes one of the most important UNESCO "World Heritage Sites". The VMG study area constitutes the transitional zone between the ancient and current location of the delta as it was formed after the excavation of the channel near Porto Viro performed by Venetians in 1604. The ancient delta is at the south of the current location (Bondesan et al. 1995; Castiglioni et al. 1999). During the last two centuries, the Po river delta has been strongly altered by both natural (e.g. coastal sedimentation) and anthropogenic activities (e.g. land conversion, water resources regulation) (Bondesan 1990; Cencini 1998; Simeoni and Corbau 2009). Land reclamation has also heavily altered the landscape during 1870-1960. The first reclamation actions were performed in the southern part of delta in 1872, where wide wetlands were present in the areas of the ancient Po riverbanks. Reclamation works were mainly financed by the Italian Kingdom with the aim to support economic development and to confront malaria. Reclamation work continued during the 1930's with the Fascist Government and was completed by the Republican Government after World War II in the context of agrarian reform. In the ancient delta, where reclamations were more intensive, 98% of freshwater marshes and more than 70% of salt marshes have been converted mainly to croplands since the beginning of the century (Cencini 1998).

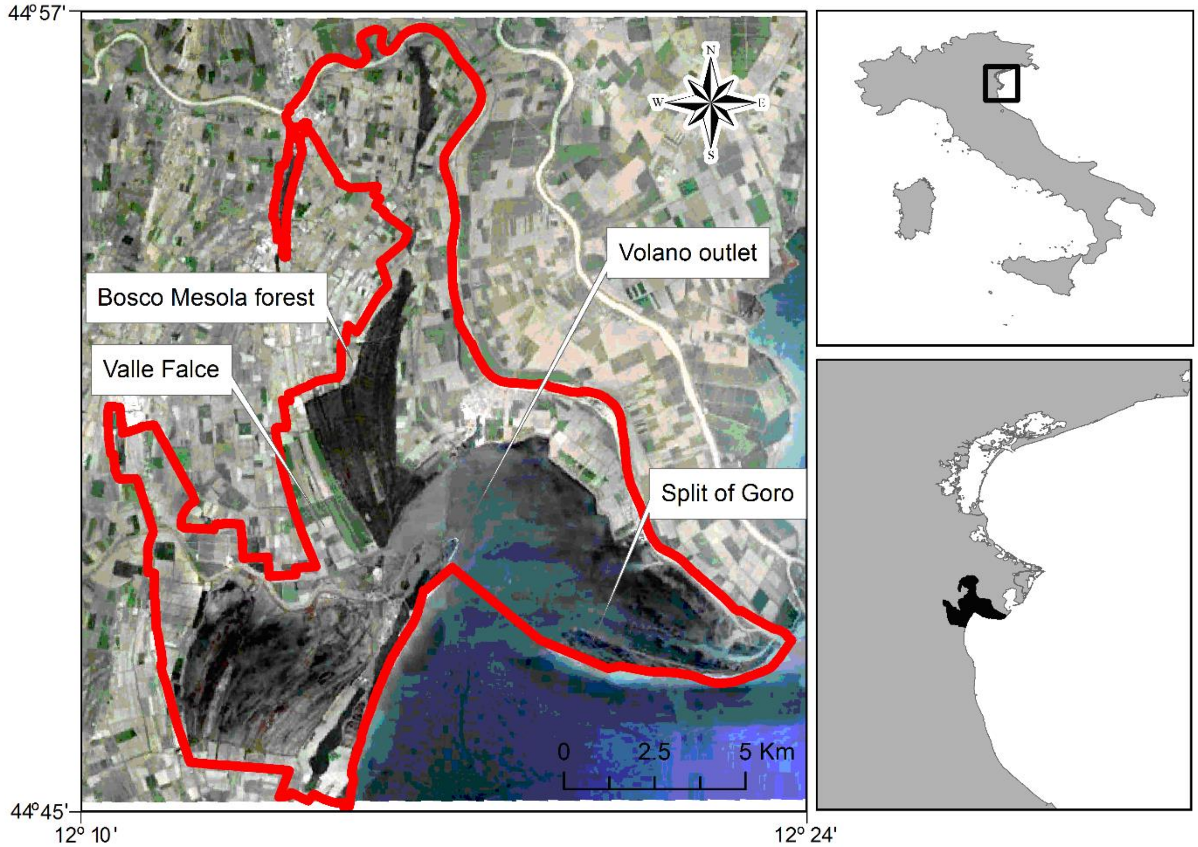

Fig. 1: The study area of the Volano-Mesola-Goro (VMG).

Three land cover maps from 1954, 1976 and 2008 were used, in order to describe the spatio-temporal changes in LULC and ESs in the VMG area. The maps include the VMG administritative boundaries (13,730 ha) plus additional areas (438 ha) of Sacca di Goro Lagoon bench which existed only in 1954 map. The maps were produced by Cartographic Archive of Emilia-Romagna Region using ortho-rectified aerial photos at the geographic scale of 1:25,000. The land use maps were further corrected in order to provide a more detailed description of the wetlands types and their associated aquatic vegetation using more detailed maps and aereal photos provided by IGM (Istituto Geografico Militare) (for the year 1954 and 1976), satellite images from Google Earth for the year 2008, habitat map surveys of aquatic vegetation from Noferini et al. (2005) (the spatial resolution of datasets ranged between 5-30 m). The final maps include 12 different LULC types (Table 1). The last LULC type (No.12 in Table 1) merges two types (the unvegetated coastal wetlands and the transitional marine lagoon environment) which have similar characteristics and functions. The analysis was performed by comparing changes between the two periods 1954-1976 and 1976-2008 associated to two different landscape management conditions: a) the period of 1956-1976 of extreme land use conversion due to land reclamations and b) the period 1976-2008 which corresponds to the application of environmental protection measures.

Table 1: Description of the 12 LULC types in VMG region, their typical vegetation characteristics, main ecological functions and total ESs value.

No.	LULC type	Description	Typical vegetation species	Main ecological functions	Total ESs value US$2007 ha^{-1} $year^{-1}$ (*)
1	Closed valley	Closed valley with variable salinity. water recharge both from sea water and rivers, water level is regulated artificially	*Lamprotanium papulosum, Ruppia cyrrhosa*	Water regulation, fish farming, recreation (hunting and birdwatching)	140,174
2	Coastal wetlands with emergent vegetation	Salt marshes along or nearby the coastline in intertidal flats. Vegetation is composed by species which emerge from the water, while roots are permanently or temporarily submerged	*Mainly Phragmites australis, rarely Bolboschoenus maritimus*	Nutrient regulation, shoreline protection, habitat for aquatic species and birds	193,843
3	Coastal wetlands with submerged vegetation	Salt marshes along or nearby the coastline. The vegetation is constantly submerged by water	*Ruppia cyrrhosa,* sometimes associated with algae *(Ulvales, Chaetomorpha, Cladophora, Ceramium).*	Habitat for aquatic species, sediment stabilizing	28,916
4	Croplands	Arable land and permanent crops, including woody plantations	Mainly cereals (e.g. maize, wheat), orchards, vineyards. poplar plants	Food production	5,568
5	Dunes, beaches and sand	Beaches and dunes constantly above the sea level	*Cakile maritima, Salsola kali, Puccinella palustris, Ammophila arenaria, Agropyron pungens, Spartinea juncea, Salicornia veneta*	Shoreline erosion prevention, recreation	8,944
6	Forest	Woodlands (tree coverage >30%)	*Quercus ilex, Q.robur, Carpinus betulus, Fraxinus oxycarpa* in Mesola forest. *Pinus spp.* in coastal woodlands	Habitat for species, carbon storage and sequestration, recreation	3,137
7	Grassland/Rangeland	Shrubs (tree coverage < 30%)	*Prunus spinosa, Crataegus monogyna, Ligustrum vulgare*	Habitat for species, climate regulation, food provision	4,166
8	Inland wetlands	Freshwater swamps and plains flooded by rivers, or river meanders where the water flow is slow (regulated water flow)	*Mainly Phragmites australis. Typha angustifolia, Potamogeon pectinatus, Ceratophyllum demersum, Trapa natans*	Water and disturbance regulation, nutrient regulation, habitat for species	25,681
9	Rivers/Lakes	Rivers and lakes both natural or artificial, including banks	*Populus sp., Salix sp. and P.australis* along the banks.	Water regulation, fishing	12,512
10	Urban	Residential areas, infrastructures, industrial sites and ports	-	No functions are considered in this study	0

11	Urban green	Parks, villas and non-covered urban areas	Native and non-native ornamental species and grass	Climate mitigation, recreation	6,661
12	Non vegetated coastal wetlands and marine	Marine habitats and coastal wetlands with bare sediment (very scarse pleustophitic vegetation)	*Mainly Ulva rigida. Chaetomorpha*, Cladophora, Ceramium	Climate regulation Habitat for species	1,368

*The total values are estimated using the Table S.1 in the supplementary material.

Land reclamation period 1956-1976

The period 1956-1976 covers the more recent land reclamations (Valle Giralda in 1958 and Valle Falce in 1969) performed after WWII by Ente Delta Padano (a public body created *ad hoc* within the framework of the agrarian reform). During the following decades, reclamation of the Valle Falce was blamed for lowering the groundwater table and the associated die-off of the most ancient trees of the Bosco della Mesola region (Munda et al. 1995). This period corresponds to changes in the Italian demographic and economic conditions, which led to an increasing demand for arable land and food production. This demand also forced agriculture activities towards more intensive and industrial practices, with massive use of fertilizers and pesticides. During this period, the landscape was strongly reshaped and fragmented by the construction of roads and dense canals networks.

As a consequence of the improvement of social and economic conditions, coastal areas were also subjected to increased tourist activities. Beaches and related natural habitats suffered coastal urban growth with a severe loss of natural ecosystems such as coastal meadows and sandbanks (Bondesan et al. 1995). Additional human-induced factors such as groundwater withdrawal together with salt water intrusion (Teatini et al. 2006) and gas extraction (Bau et al. 1999) intensified from 1938 to 1961 (Antonellini et al. 2008; Bondesan et al. 1995; Cencini 1998) reducing water provisioning and soil productivity for crops, in contrast with the aim of agrarian reform.

In the first period, the only local tools for urban planning were municipal building codes under National Law 1150/1942 without considering rural areas and the Law n.1497 of 29th June 1939 which had the aim to safeguard aesthetic and cultural values derived from landscape and nature. This law was the first attempt to deal with environmental conservation from an aesthetic perspective (Settis 2010).

Environmental protection period 1976-2008

After the previous period of intense land conversions, environmental protection actions were designed and implemented in the EU by its member states including Italy (Ramsar 1971; UN 1972; Birds Directive 79/409/EEC). Many areas in the wider Po river delta were included in the lists of protected areas conventions (Habitat Directive 92/43/ECC; Natura2000 network) such as the Natural State Reserve of Bosco della Mesola in 1977, a forested area of 1058 ha inside the study area of VMG and the Regional Park of the Po Delta in 1988 (Fig.1). During the second period (1976-2008), various laws and masterplans were developed for natural landscape protection such as the Provincial Territorial Coordination Plan of Province of Ferrara (PTCP) (R.L. 47/1978) and the Regional Landscape Plan of Emilia Romagna (RLP) (L. 431/1985, 20/2000, 42/2004).

Methods

Analysis of LULC changes using transition matrices (TMs)

The analysis of LULC changes was performed using LULC transition matrices (TMs). Probability-based transition matrices, such as Markovian models or cross-tabulation matrices, are often obtained from area-based transitions which are used as tools in landscape ecology studies (Mas et al. 2004; Takada et al. 2010). In our study, TMs were developed directly by the LULC changes between 1954, 1976 and 2008 without using probabilistic approaches in order to show the exact change from one LULC type to another (Wang et al. 2014). TMs compare the extent of LULC types between two time intervals (e.g. t_1 and t_2) providing the area of each LULC type which remained intact and the specific changes to other LULC types during t_1-t_2. More details about TMs attributes are given directly as footnotes on TMs tables. The LULC maps of 1954, 1976 and 2008 correspond to three time intervals and for this reason, three TMs were built that correspond to the periods 1954-1976, 1976-2008 and 1954-2008. TMs were developed after overlapping the three shapefiles of LULC maps of 1954, 1976 and 2008. The overlapping was performed in ArcGIS (ESRI) environment using the Union tool (in Analysis Tools → Overlay) that creates a final shapefile with an attribute table, which contains the LULC information of the three previous layers. The final shapefile was used to calculate the area coverage of each polygon. The attribute table of the final shapefile was exported in Excel in order to create a pivot table (in Insert tools→ Pivot table) which provides the information for TMs development.

Analysis of Ecosystem Services (ESs) changes

The relationship between LULC and ESs changes was analyzed based on ESs values obtained from Costanza et al. (2014) which describe 17 ESs for 16 ecosystem types (referred as biomes). These values were used as a proxy to describe ESs changes in VMG. Although this method has several limitations (Rosenberger and Loomis 2000), it represents the most comprehensive and suitable method to estimate both the magnitude and the flow of ESs in space and time (Kreuter et al. 2001). The economic values (US\$2007 ha^{-1} $year^{-1}$) that were attributed to each of the 17 ESs for each ecosystem type in VMG are given in Table S.1 of the supplementary material. The spatio-temporal change of natural capital provided by ESs is estimated by the following formula:

$$NCV = \sum_{i=1}^{M} VC_i \quad \text{where} \quad VC_i = \sum_{j=1}^{N} \left(V_{ij} \times A_i \right) \tag{1a,b}$$

where NCV is the total natural capital value (US\$ year^{-1}), Vij is ES value (US\$2007 ha^{-1} year^{-1}) for the j ecosystem service in the i LULC class, A_i is the area coverage (ha) of i LULC class. M and N are the maximum number of observed LULC classes and ESs, respectively. The ESs values in literature for urban areas were accounted for green urban areas, while no ESs values were accounted for urban zones (Scolozzi et al. 2012).

Annual rates of change for each land use type and ES are calculated with the following function (Puyravaud 2003):

$$r = 100\left[1/\left(t_2 - t_1 \right) \right] \times 1n\left(A_2 / A_1 \right) \tag{2}$$

where r is the annual rate of change (expressed as %) of a given LULC type or ES, A_2 and A_1 are LULC type coverage or ES value at time t_2 and t_1 respectively.

A sensitivity analysis was also performed in order to assess the change in NCV when the VC value (Eq.1b) of a specific LULC class is adjusted by ±50% keeping constant the VC values of the rest LULC classes (Li et al. 2007; Hu et al. 2008; Tianhong et al. 2010; Kindu et al. 2016). The coefficient of sensitivity (CS) was also calculated according to the standard economic method (Mansfield 1985; Kreuter et al. 2001) after its simplification by Aschonitis et al. (2016):

$$CS_{t,i} = \frac{VC_{t,i} \cdot A_{t,i}}{NCV_t} \tag{3}$$

where NCV is the total natural capital value (US\$ year^{-1}) of all ESs from all LULC classes at t year (US\$ year^{-1}), $VC_{t,i}$ is the total value of ESs provided by the i LULC class at t year (US\$ year^{-1}) and $A_{t,i}$ is the area coverage (ha) of the i LULC class at t year. It has to be mentioned that Eq.3 gives the same results with the function proposed by Kreuter et al. (2001) when there are no changes in the prices of ESs (Aschonitis et al. 2016). Kreuter et al. (2001) also suggested to use the threshold of unity for comparing the obtained CS values in order to extract conclusions about the robustness of ESs values/prices. Aschonitis et al. (2016) found that the use of this threshold for assessing the robustness of ESs values based on CS is wrong since CS is always <1 when the prices don't change. For this reason discussion about the robustness of ESs values is not performed and the CS values are used to rank the importance of each LULC class.

Results

LULC change and TMs

The LULC maps of the VMG region for 1954, 1976 and 2008 are given in Fig.2. Table 2 provides the % coverage, the total % change for each period and the annual rate of change *r* (Eq.2) for each LULC type. Results were also derived merging the vegetated wetlands (LULC No.1,2,3 and 8 in Table 1,2), which are of special interest (last row of Table 2). Vegetated wetlands were the most abundant LULC types in 1954 covering the 31.4% of the total area while in the next years their coverage was reduced reaching 19.41% in 1976 and 14.95% in 2008 (Table 2). After 1976, croplands became the most dominant LULC type in the study area. The coastal wetlands with emergent and submerged vegetation were the most impacted loosing more than 80% of their coverage, the Closed valley was slightly impacted (-13.6%) while the inland wetlands showed an increase of their coverage in both periods (40.9% increase during the total period 1954-2008). The direct anthropogenic impact associated to the changes in croplands, urban and urban green (LULC types No.4, 10, 11) showed higher rates of increase during the period 1954-1976 in comparison to 1976-2008. Especially for croplands, the annual rate of change *r* was reduced from +1.33% (period 1954-1976) to +0.04% (period 1976-2008).

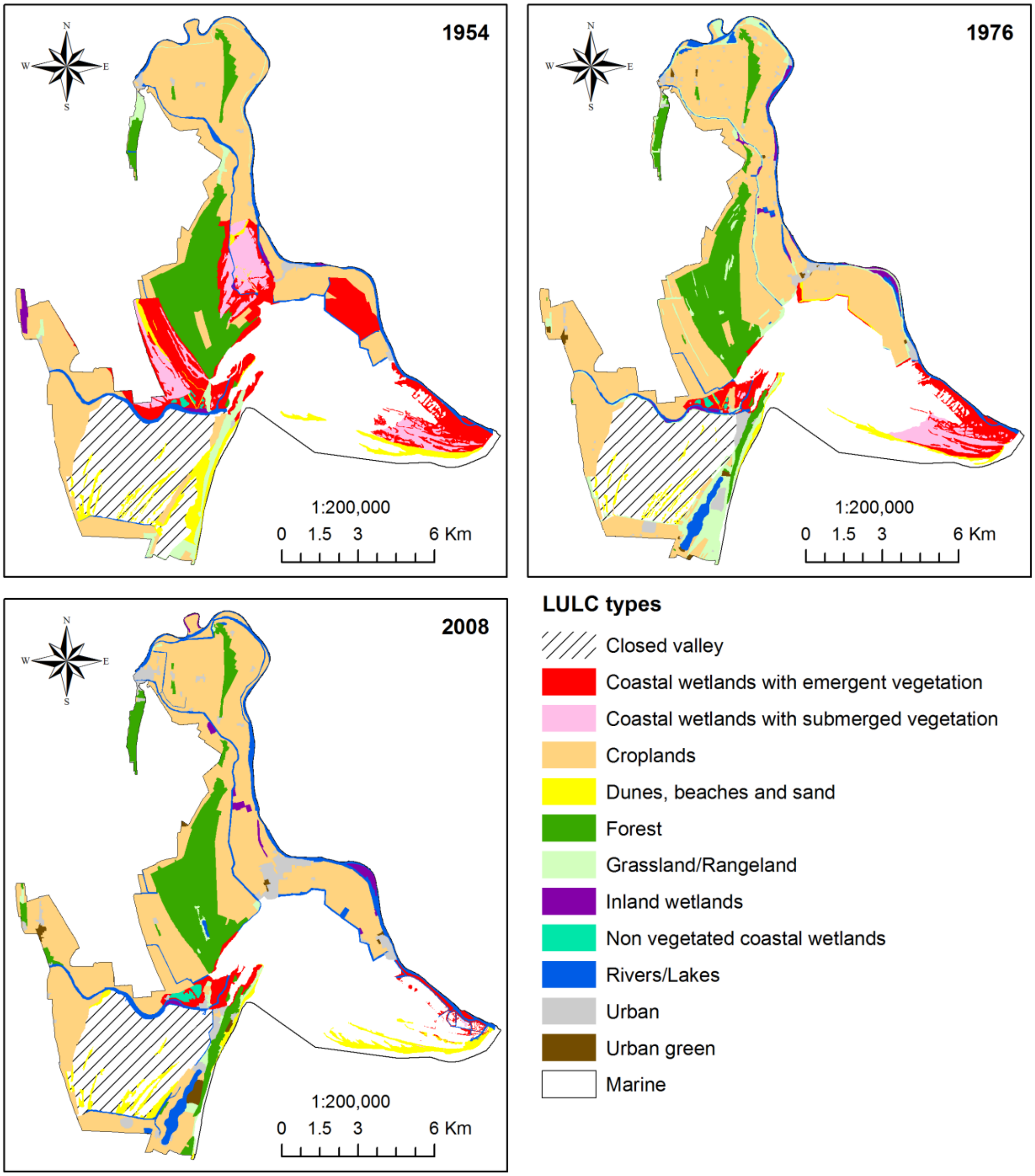

Fig. 2: LULC maps of the VMG for the years 1954, 1976 and 2008.

Table 2: % coverage, total % change and annual rate of change *r* (Eq.2) for each LULC type for the periods 1954-1976, 1976-2008 and 1954-2008.

LULC type	% Coverage			Total % change			Annual change *r* (%)		
	1954	1976	2008	1954-1976	1976-2008	1954-2008	1954-1976	1976-2008	1954-2008
1. Closed valley	14.27	12.89	12.33	-9.66	-4.33	-13.57	-0.46	-0.14	-0.27
2. Coastal wetlands with emergent vegetation	11.31	4.04	1.90	-64.24	-53.01	-83.20	-4.67	-2.36	-3.30
3. Coastal wetlands with submerged vegetation	5.32	1.85	0.03	-65.33	-98.49	-99.47	-4.81	-13.09	-9.72
4. Croplands	26.77	35.86	36.27	33.95	1.16	35.51	1.33	0.04	0.56
5. Dunes, beaches and sand	4.09	1.78	2.85	-56.44	59.87	-30.36	-3.78	1.47	-0.67
6. Forest	9.20	9.23	10.60	0.38	14.82	15.25	0.02	0.43	0.26
7. Grassland/Rangeland	2.27	5.25	0.79	131.30	-84.94	-65.17	3.81	-5.92	-1.95
8. Inland wetlands	0.49	0.63	0.69	29.40	8.90	40.92	1.17	0.27	0.64
9. Rivers/Lakes	3.30	3.17	5.52	-4.00	74.02	67.07	-0.19	1.73	0.95
10. Urban	0.61	2.00	2.77	229.15	39.04	357.67	5.42	1.03	2.82
11. Urban green	0.00	0.28	0.51	-	82.66	-	-	1.88	-
12. Unvegetated coastal wetlands and marine habitats	22.38	23.02	25.73	2.88	11.80	15.02	0.13	0.35	0.26
Total vegetated wetlands (Sum of no. 1,2,3 and 8)	31.39	19.41	14.95	-38.16	-22.99	-52.38	-2.18	-0.82	-1.37

Table 2 is useful to detect the general changes in land uses coverage but it cannot provide information about the exact LULC conversions. For this reason, the TMs for the periods 1954-1978, 1978-2008, and 1954-2008 were developed and they are provided in Table 3a, b and c, respectively.

Regarding the TM of the first period (1956-1976) (Table 3a), vegetated wetlands lost 1,697.2 ha (sum of totals of 1954 for No.1,2,3,8 LULC types minus the respective sum of totals of 1976) from which 87.2% (1,480.6 ha) was associated to anthropogenic interventions and more specifically due to conversions principally to croplands (1,430.2 ha)[1] but also to urban (41.9 ha) and urban green areas (8.5 ha). The rest 12.8% (216.7 ha) of vegetated wetlands loss was due to natural factors and indirect human interventions associated to a) the expansion of grasslands/rangelands (160.1 ha), unvegetated coastal wetlands and marine habitats (78.8 ha), rivers' surface (61.4 ha) and forests (4.4 ha) and b) the net positive contribution (88.1 ha) of dunes, beaches and sand areas to vegetated wetlands. The exact positions of vegetated wetlands losses for the period 1954-1976 are given in Fig.3a.

Taking into account the TM of the second period (1976-2008) (Table 3b), vegetated wetlands lost 632.4 ha (sum of totals of 1976 for No.1,2,3,8 LULC types minus the respective sum of totals of 2008) from which only 4.2% (26.4 ha) was associated to direct anthropogenic interventions and more specifically due to conversions to urban areas (19.8 ha) and croplands (6.5 ha). The rest 95.6% (606 ha) of vegetated wetlands loss was due to natural factors and indirect human interventions associated to a) the expansion

[1]These values are calculated taking into account both the area which was lost but also the area which was gained. For example, 1,490.2 ha of vegetated wetlands were converted to croplands, but also 61.0 ha of croplands were converted to vegetated wetlands. This leads to a net loss of vegetated wetlands equal to 1,430.2 ha due to croplands for the period 1954-1976 (Table 3a). As vegetated wetlands are count the LULC codes 1, 2, 3 and 8.

of unvegetated coastal wetlands and marine habitats (448.3 ha), dunes, beaches and sand areas (125.1 ha), rivers' surface (75.1 ha) and b) the net positive contribution (42.6 ha) of grasslands/rangelands to vegetated wetlands. The exact positions of vegetated wetlands losses for the period 1976-2008 are given in Fig.3b.

Finally, using the TM of the total period (1954-2008) (Table 3c), a ranking scheme of the LULC types which were expanded versus the vegetated wetlands was developed as follows: croplands (1,474.6 ha) > unvegetated coastal wetlands and marine habitats (482.2 ha) > rivers (160.8 ha) > urban areas + urban green (105.5 ha) > dunes, beaches and sand areas (57.6 ha) > forest (27.8 ha) > grasslands/rangelands (21.0 ha).

Table 3: LULC transition matrices for a) 1954-1976 (land reclamation period), b) 1976-2008 (environmental protection period) and c) 1954-2008 (total study period) (values expressed in ha).

a)	Period 1954-1976													
LULC type	Closed valley	Coastal wetlands with emergent vegetation	Coastal wetlands with submerged vegetation	Croplands	Dunes, beaches and sand	Forest	Grassland/ Rangeland	Inland wetlands	Rivers/ Lakes	Urban	Urban green	Unvegetated coastal wetlands and marine habitats	LOSS	Total of 1954
1. Closed valley	**1,732.7**[a]	0.5[b]	0.0[b]	23.7[b]	52.5[b]	0.3[b]	84.5[b]	0.9[b]	96.6[b]	25.4[b]	4.4[b]	0.0[b]	288.8[c]	2,021.5[d]
2. Coastal wetlands with emergent vegetation	0.0[e]	**398.6**[a]	94.0	887.1	13.0	18.1	53.6	0.0	11.1	11.2	0.0	115.9	1,203.9	1,602.5
3. Coastal wetlands with submerged vegetation	0.0[e]	24.9	**147.8**	548.8	0.2	0.0	16.5	0.0	12.6	2.1	0.0	1.2	606.3	754.1
4. Croplands	6.9[e]	9.1	0.0	**3,251.5**	7.8	39.5	190.4	45.0	81.9	145.3	15.2	0.0	541.2	3,792.7
5. Dunes, beaches and sand	84.6[e]	53.8	15.5	66.4	**89.5**	11.0	149.5	0.0	7.3	19.4	5.4	77.0	489.8	579.3
6. Forest	0.0[e]	14.4	0.0	100.4	0.0	**1,130.2**	56.1	0.0	0.5	1.0	0.0	0.3	172.8	1,302.9
7. Grassland/ Rangeland	0.0[e]	4.4	0.0	85.4	1.7	106.4	**87.2**	1.9	2.9	13.3	5.8	12.7	234.5	321.8
8. Inland wetlands	0.0[e]	4.6	0.0	31.7	0.0	0.4	11.8	**7.2**	5.8	3.3	4.1	0.2	61.8	69.0
9. Rivers/Lakes	2.0[e]	28.6	0.0	65.2	14.8		85.3	34.1	**222.3**	7.9	0.0	8.1	245.9	468.2
10. Urban	0.0[e]	0.0	0.0	19.3	0.7	1.2	4.7		1.5	**53.7**	4.8	0.0	32.3	85.9
11. Urban green	0.0[e]	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**	0.0	0.0	0.0
12. Unvegetated coastal wetlands and marine habitats	0.0[e]	34.1	4.2	1.0	72.2	0.7	4.6	0.1	6.9	0.2		**3,046.1**	124.0	3,170.1
GAIN	93.5[f]	174.4	167.5	1,828.9	162.9	177.7	657.0	82.1	227.2	229.1	39.6	215.3		
Total of 1976	1,826.2[g]	573.0	261.5	5,080.3	252.4	1,307.9	744.2	89.3	449.5	282.8	39.6	3,261.4		14,168.1[h]

[a]The diagonal bold values show the area coverage of a LULC type, which remained intact during the period 1954-1976.

[b]The values of each row, except the bold ones, show how many hectares of a specific LULC type were converted to another LULC type (for example: 23.7 ha of Closed valley were converted to croplands during 1954-1976).

[c]LOSS: The total sum of the values of each row, except the bold ones, which provides the total area of a specific LULC which was converted to another LULC types (for example: 288.8 ha of Closed valley were converted to other LULC types during 1954-1976).
[d]Total area of a LULC type at the beginning of the study period (for example: the total coverage of Closed valley was 2,021.5 ha in 1954).
[e]The values of each column, except the bold ones, show how many hectares of a specific LULC type were gained (for example: Closed valley gained 6.9 ha after conversion from croplands to the specific LULC during 1954-1976).
[f]GAIN: The total sum of the values of each column, except the bold values, which provides the total area which was gained for a specific LULC (for example: Closed valley gained a total area of 93.5 ha during 1954-1976).
[g]Total area of a LULC type at the end of the study period (for example: the total coverage of Closed valley was 1,826.2 ha in 1976).
[h]Total coverage of the study area.

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b)	Period 1976-2008													
LULC type	Closed valley	Coastal wetlands with emergent vegetation	Coastal wetlands with submerged vegetation	Croplands	Dunes, beaches and sand	Forest	Grassland/ Rangeland	Inland wetlands	Rivers/ Lakes	Urban	Urban green	Unvegetated coastal wetlands and marine habitats	LOSS	Total of 1976
1. Closed valley	**1,691.4**	0.2	0.0	7.6	100.0	0.0	0.2	0.0	26.2	0.6	0.0	0.0	134.8	1,826.2
2. Coastal wetlands with emergent vegetation	0.0	**191.2**	4.0	1.0	62.4	6.3	0.9	0.0	52.1	22.2	0.0	232.9	381.8	573.0
3. Coastal wetlands with submerged vegetation	0.0	8.7	**0.0**	0.0	14.4	0.0	0.0	0.0	3.2	0.0	0.0	235.1	261.5	261.5
4. Croplands	1.2	2.8	0.0	**4,648.7**	1.4	86.2	37.8	14.5	146.7	125.4	15.7	0.0	431.6	5,080.3
5. Dunes, beaches and sand	44.4	7.4	0.0	3.7	**101.6**	2.9	5.8	0.0	18.9	4.8	1.7	61.0	150.7	252.4
6. Forest	0.0	6.1	0.0	30.7		**1,259.7**	4.8	0.0	0.0	4.9	0.0	1.8	48.2	1,307.9
7. Grassland/ Rangeland	9.2	15.6	0.0	281.2	15.0	136.2	**40.2**	18.8	168.8	31.8	12.6	14.8	704.0	744.2
8. Inland wetlands	0.0	3.9	0.0	16.3	0.0	0.0	0.0	**36.5**	32.5	0.0	0.0	0.0	52.8	89.3
9. Rivers/Lakes	0.0	11.5	0.0	79.3	0.1	0.3	2.4	27.5	**321.2**	6.3	0.0	0.8	128.3	449.5
10. Urban	0.9	2.1	0.0	61.7	0.0	5.4	5.1	0.0	9.4	**164.1**	34.0	0.0	118.7	282.8
11. Urban green	0.0	0.0	0.0	9.0	0.0	3.2	0.0	0.0	0.6	18.5	**8.3**	0.0	31.3	39.6
12. Unvegetated coastal wetlands and marine habitats	0.0	19.7	0.0	0.0	108.4	1.4	14.9	0.0	2.6	14.6	0.0	**3,099.8**	161.6	3,261.4
GAIN	55.8	78.0	0.0	490.6	301.8	242.0	71.9	60.7	461.0	229.1	64.1	546.4		
Total of 2008	1,747.1	269.2	4.0	5,139.3	403.5	1,501.7	112.1	97.2	782.2	393.2	72.4	3,646.1		14,168.1

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c) LULC type	Closed valley	Coastal wetlands with emergent vegetation	Coastal wetlands with submerged vegetation	Croplands	Dunes, beaches and sand	Forest	Grassland/ Rangeland	Inland wetlands	Rivers/ Lakes	Urban	Urban green	Unvegetated coastal wetlands and marine habitats	LOSS	Total of 1954
	Period 1954-2008													
1. Closed valley	**1,695.0**	1.1	0.0	96.5	74.5	0.4	1.5	0.3	125.3	9.4	17.6	0.0	326.6	2,021.5
2. Coastal wetlands with emergent vegetation	0.0	**191.5**	2.8	865.1	28.7	19.5	19.0	4.0	70.6	49.8	3.4	348.0	1,411.0	1,602.5
3. Coastal wetlands with submerged vegetation	0.0	5.7	**0.0**	547.1	10.5	0.0	0.0	0.0	19.9	12.8	2.1	155.9	754.1	754.1
4. Croplands	1.0	0.0	0.0	**3,277.8**	5.6	73.0	24.9	52.5	158.9	188.2	10.8	0.0	514.9	3,792.7
5. Dunes, beaches and sand	50.4	4.6	1.1	121.4	**182.8**	32.5	33.3	0.0	17.4	15.6	26.9	93.2	396.5	579.3
6. Forest	0.0	12.4	0.0	84.8		**1,199.9**	1.9	0.0	2.6	0.1	0.0	1.2	103.0	1,302.9
7. Grassland/ Rangeland	0.0	5.5	0.0	78.4	1.4	151.0	**18.6**	0.0	26.1	15.2	7.4	18.1	303.1	321.8
8. Inland wetlands	0.0	7.1	0.0	19.4		20.3	6.0	**2.8**	2.9	10.5	0.0	0.2	66.2	69.0
9. Rivers/Lakes	0.8	19.6	0.0	42.9	0.3	2.5	0.0	37.5	**347.1**	10.8	0.0	6.7	121.1	468.2
10. Urban	0.0	0.0	0.0	5.4	0.5	2.1	0.0	0.0	2.1	**71.6**	4.3	0.0	14.3	85.9
11. Urban green	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**	0.0	0.0	0.0
12. Unvegetated coastal wetlands and marine habitats	0.0	21.8	0.0	0.5	99.0	0.6	6.8	0.2	9.4	9.2	0.0	**3,022.6**	147.5	3,170.1
GAIN	52.2	77.7	1.1	1,861.6	220.6	301.8	93.4	94.5	435.1	321.5	72.4	623.5		
Total of 2008	1,747.1	269.2	4.0	5,139.3	403.5	1,501.7	112.1	97.2	782.2	393.2	72.4	3,646.1		14,168.1

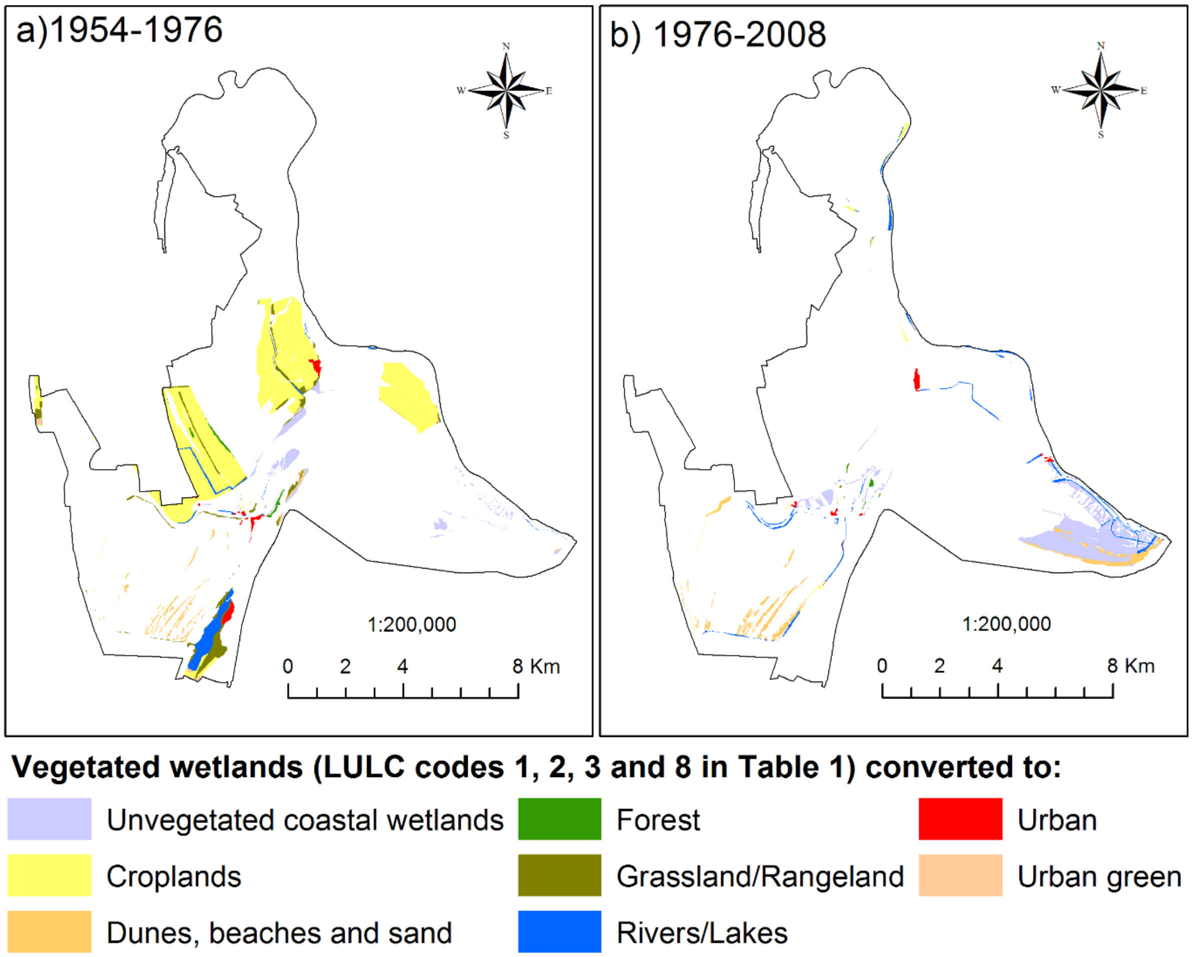

Fig. 3: Areas of vegetated wetlands which were converted to other LULC types in the VMG region during a) the reclamation period 1954-1976 and b) environmental protection period 1976-2008.

Ecosystem Services (ESs) change

Table 4 presents the values (US$2007×$10^3$), the total % changes and the annual rates of change for each of the 17 ESs in VMG area for the periods 1954-1976, 1976-2008 and 1954-2008.

With respect to the first period 1954-1976, the LULC changes reduced the total *NCV* by -35.56%, with an annual change rate *r* of -2.00%. The value of the total provisioning services was increased by 3.41% with an annual rate *r* equal to 0.15% mainly due to croplands expansion. On the other hand, the total values of regulation, supporting and cultural services were reduced by -38.34%, -33.36% and -19.72%, respectively, with respective annual rates *r* equal to -2.20%, -1.85% and -1.00%, mainly due to the losses of vegetated wetlands. The most impacted ES types during this period, in terms of monetary units and % change, were the waste treatment, erosion control and habitat refugia. Significant positive % change was observed for genetic resources, gas regulation, soil formation and pollination but with small contribution to the total *NCV*. It is indicative that the great expansion of croplands during this period was not followed by an analogous increase of food production service since significant amount of this ES was lost by the respective loss of vegetated wetlands.

Regarding the second period 1976-2008, the LULC changes reduced the total *NCV* by -17.09%, with an annual change rate *r* of -0.59%. The total values of provisioning, regulation and supporting services were reduced by -6.73%, -18.64% and -17.92%, respectively, with respective annual rates *r* equal to -0.22%, -0.64% and -0.62%, mainly due to conversion of vegetated wetlands to unvegetated coastal wetlands and marine habitats. The total value of cultural services was slightly increased by 3.18% (*r*=0.10%) with insignificant contribution to the total *NCV*. The most impacted ES types during this period, in terms of monetary units and % change, were again the waste treatment, erosion control and habitat refugia due to vegetated wetlands losses. Positive % change with significant contribution to *NCV* was observed only in the water regulation mainly due to the expansion of rivers coverage. Despite the fact that direct human intervention in the VMG was not significant during the period 1976-2008, the *NCV* was reduced due to the loss of vegetated wetlands due to the expansion of unvegetated coastal wetlands and marine habitats

Taking into account the overall period 1954-2008 and the ESs values (US$2007×$10^3$) of Table 4, it was observed that the waste treatment is the most important service providing the 73.88%, 70.39% and 68.52% of the total *NCV* for 1954, 1976 and 2008, respectively. The waste treatment together with erosion control and habitat refugia services provided 80-88% of the *NCV* in the VMG during all periods. These three ES types were mostly associated to the vegetated wetlands highlighting the significant economic contribution of their ecological functions.

The sensitivity analysis and the *CS* calculation for each LULC type, which was performed separately for each year (1954, 1976 and 2008) (Table 5), gave more detailed information about the contribution of LULC types in ESs assessment. Table 5 showed the importance of specific vegetated wetlands such as the closed valley and the coastal wetlands with emergent vegetation in the final *NCV*. Due to the significant progressive loss of coastal wetlands with emergent and submerged vegetation, their effect is also reduced while the closed valley seems to be the most important LULC type with a progressive increase of its effect on the final *NCV*. A significant increase in the effects of croplands, rivers and forests is also observed but of lower magnitude. The rest LULC types present small effects on the final *NCV* values.

Table 4: ESs values in 1954, 1976 and 2008, total % change and annual rate of change *r* (Eq.2) for the periods 1954-1976, 1976-2008, 1954-2008.

ES type	US$2007×$10^3$			Total % change			Annual change *r* (%)		
	1954	1976	2008	1954-1976	1976-2008	1954-2008	1954-1976	1976-2008	1954-2008
Food production	15,793.13	16,674.32	15,236.64	5.58	-8.62	-3.52	0.25	-0.28	-0.07
Raw materials	2,741.56	2,538.35	2,452.07	-7.41	-3.40	-10.56	-0.35	-0.11	-0.21
Genetic resources	5,762.74	7,060.71	6,256.45	22.52	-11.39	8.57	0.92	-0.38	0.15
Water supply	6,548.84	5,624.34	5,806.40	-14.12	3.24	-11.34	-0.69	0.10	-0.22
Total provisioning services	30,846.26	31,897.72	29,751.56	3.41	-6.73	-3.55	0.15	-0.22	-0.07
Gas regulation	2.90	6.70	1.01	131.30	-84.94	-65.17	3.81	-5.92	-1.95
Climate regulation	2,925.72	3,157.94	3,097.72	7.94	-1.91	5.88	0.35	-0.06	0.11
Disturbance regulation	18,201.93	11,806.73	9,870.00	-35.13	-16.40	-45.78	-1.97	-0.56	-1.13
Erosion control	37,168.06	18,628.61	11,501.81	-49.88	-38.26	-69.05	-3.14	-1.51	-2.17
Waste treatment	487,221.81	299,142.81	241,426.20	-38.60	-19.29	-50.45	-2.22	-0.67	-1.30
Biological control	1,177.60	1,180.26	1,202.55	0.23	1.89	2.12	0.01	0.06	0.04
Water regulation	7,522.39	7,147.87	9,549.90	-4.98	33.60	26.95	-0.23	0.91	0.44
Nutrient cycling	3,390.72	2,740.14	3,081.41	-19.19	12.45	-9.12	-0.97	0.37	-0.18
Total regulation services	557,611.12	343,811.05	279,730.60	-38.34	-18.64	-49.83	-2.20	-0.64	-1.28
Soil formation	2,036.58	2,722.54	2,755.37	33.68	1.21	35.29	1.32	0.04	0.56
Pollination	94.70	137.82	116.99	45.53	-15.11	23.54	1.71	-0.51	0.39
Habitat refugia	54,612.85	34,952.54	28,163.68	-36.00	-19.42	-48.43	-2.03	-0.67	-1.23
Total supporting services	56,744.13	37,812.89	31,036.03	-33.36	-17.92	-45.31	-1.85	-0.62	-1.12
Recreation	12,620.31	9,898.62	10,377.65	-21.57	4.84	-17.77	-1.10	0.15	-0.36
Other cultural	1,681.50	1,582.53	1,468.73	-5.89	-7.19	-12.65	-0.28	-0.23	-0.25
Total cultural services	14,301.81	11,481.15	11,846.37	-19.72	3.18	-17.17	-1.00	0.10	-0.35
TOTAL (NCV) (Eq.1a)	659,503.32	425,002.82	352,364.57	-35.56	-17.09	-46.57	-2.00	-0.59	-1.16

Table 5: Change in total *NCV* (%) and sensitivity coefficient (*CS*) after adjusting ESs values by ±50%. in the VMG for the years 1954, 1976 and 2008.

LULC type	1954		1976		2008	
	±%	CS	±%	CS	±%	CS
Closed valley	21.48	0.430	30.12	0.602	34.75	0.695
Coastal wetlands with emergent vegetation	23.55	0.471	13.07	0.261	7.41	0.148
Coastal wetlands with submerged vegetation	1.65	0.033	0.89	0.018	0.02	0.000
Croplands	1.60	0.032	3.33	0.067	4.06	0.081
Dunes, beaches and sand	0.39	0.008	0.27	0.005	0.51	0.010
Forest	0.31	0.006	0.48	0.010	0.67	0.013
Grassland/Rangeland	0.10	0.002	0.36	0.007	0.07	0.001
Inland wetlands	0.13	0.003	0.27	0.005	0.35	0.007
Rivers/Lakes	0.44	0.009	0.66	0.013	1.39	0.028
Urban	0.00	0.000	0.00	0.000	0.00	0.000
Urban green	0.00	0.000	0.03	0.001	0.07	0.001
Non vegetated coastal wetlands and marine	0.33	0.007	0.52	0.010	0.71	0.014

Discussion

The lack of current or historical LULC and wetland inventory data is one of the most significant problems for a) tracking wetland and other LULC changes, and b) the improvement and application of evidence-based policies and decision making (Clare and Creed, 2014). Additionally, results from other studies have shown that over 80% of wetland area losses have occurred without government permission, highlighting an important governance issue regarding public compliance and government enforcement of existing wetland regulation (Clare et al. 2011; Clare and Creed, 2014). Thus, in most of the cases there is a lack of information regarding the legal status under which these landscape changes were performed. The development of inventory data for VMG succeeded in providing the evolution of the system for three different dates 1954, 1976 and 2008, which cover a period of more than half century. These data can be used as a reference for future studies providing that one (i.e. inventory data) of the two basic elements for the total control of landscape changes. The second element, which is related to the information about the legal status and regime of compliance for these changes, is missing and efforts should be made towards this direction.

Taking into account the activation of landscape conservations actions during the period 1976-2008, it was observed that the direct human intervention (croplands and urban expansion) was significantly reduced. This reduction was not followed by an analogous reduction of vegetated wetlands' losses probably due to natural factors and indirect human interventions, which favored conversions to unvegetated coastal wetlands and marine habitats. Such conversions have also been reported in other studies from other regions and they were associated to the coastal wetland dilution phenomena leading to conversion to open water (Craig et al. 1979; Childers and Day 1991). The indirect human interventions in the VMG is difficult to be identified because the final outflow is regulated by the Po basin which is extremely large. This also makes difficult the evaluation of conservation actions. Some detected indirect interventions which were responsible for aquatic vegetation loss are reported by Fogli et al. (2002) and Viaroli et al. (1992, 1996, 2006). For example, part of the observed wetland losses during the second period were mainly caused by the loss of coastal wetlands vegetation in the eastern part of the Sacca di Goro Lagoon (called Valle di Gorino). An increase of salinity in the Valle di Gorino, as a consequence from channelization interventions for the amelioration of hydrodynamic conditions, has progressively led to complete extinction of the extended and dense reed stand (Fogli et al. 2002). Other reported case was the extensive loss of submerged vegetation in Sacca di Goro caused by the increase of nutrient run-off from the surrounding agricultural land, which led to intense pleustophytic algal blooms (Viaroli et al. 1992, 1996, 2006). The shading effect of pleustophytic algae, such as *Ulva rigida*, inhibited the growth of submerged plants (Viaroli et al. 2006), and shifted the coastal habitat from a dense macrophytes meadows to an algal dominated ecosystem.

Regarding the ESs analysis, the most important component in ES supply through wetlands was the presence of aquatic vegetation. In fact, all the ESs considered by Costanza et al. (2014) for coastal wetlands face strong reductions or total losses when aquatic vegetation is absent. This occurs because aquatic vegetation is crucial to guarantee climate regulation at different scales by sequestering carbon, regulating evapotranspiration and albedo (Windam 2001; Windam et al. 2001; Gissi et al. 2014). Hydrological impacts are mitigated both by emergent and submerged vegetation, preventing erosion phenomena and mitigating storm surges and wave mechanical action. Additionally, aquatic vegetation stabilizes soil substrate, increases soil organic matter and contributes to nutrients removal and regulation of their biogeochemical cycles (Gedan et al. 2011). These functions are particularly relevant in the Po river delta, which receives a great amount of nutrients originating from the whole Po river basin (Castaldelli et al. 2013). Although nutrients assimilation by plants represents only a small fraction of their abatement (Pierobon et al. 2013), the presence of aquatic rooted vegetation boosts nutrient processing by sustaining microbial processes and uptake by micro-phytobenthos (Reddy et al.1989; Toet et al. 2003; Castaldelli et al. 2015). The lack of aquatic vegetation leads to a loss of waste treatment function, which is the most valuable service supplied by wetlands according to Costanza et al (2014) conversion parameters and the results of this study. The capacity of wetland ecosystems to remove pollutants is provided by vegetation which uptake, translocate, sequestrate or degrade contaminants (Lee 2013). Moreover, aquatic vegetation supports habitat complexity providing ecological niches for aquatic and bird species with market and recreational value (Kiviat 2013; Ludovisi et al. 2013). Thus, when no aquatic rooted vegetation is present, the ESs provided by coastal wetlands and lagoons are heavily affected by lack of environmental heterogeneity and biomass production, restricting their suitability to support aquaculture activities and to mitigate the impact of hydrologic phenomena. The above highlight the need for proper management of coastal wetlands for preserving high ecological standards, rather than the mere maintenance of coastal wetland areas through PAs establishment. Thus, the provision of ESs in transitional environments, such as deltaic areas, is strongly dependent on the maintenance of wetlands' ecological functions, which in turn need the support of both biotic and abiotic components. It should be mentioned that the role of the different observations/information related to the different types of wetlands wouldn't be available without the use of TMs provided in Table 3. Thus, TMs are important in order to describe adequately the specific LULC changes and to evaluate properly the implementation of environmental protection measures in transitional environments (e.g. river deltas), where it is difficult to separate the impact between human and natural factors. Towards this direction, some significant observations after comparing the results of TMs from the two periods 1954-1976 and 1976-2008 were obtained:

a) The vegetated wetlands losses purely associated to anthropogenic impact (expansion of croplands and urban areas) were reduced from 1480.6 ha (1954-1976) to 26.4 ha (1976-2008) suggesting a high efficacy of the landscape conservation plans which were activated during the second period.
b) The vegetated wetlands losses associated to other factors (natural plus indirect human interventions) were increased from 216.7 ha (1954-1976) to 606 ha (1976-2008). Their increase during the second period can be explained by two justifications: i) during the first period many vegetated wetlands, which would anyway be lost by natural factors or indirect human interventions, were converted to croplands and urban areas reducing the final potential effect of these factors, and ii) the probable intensification of hydro-climatic phenomena in the upstream watersheds which contribute to the final downstream flow and sediment transport.

The ESs approach used in this study, provided by Costanza et al. (2014) (Table. S.1 in supplementary material), is a ranking method that evaluates different ecosystem types using their mean global attributes. This ranking scheme is reliable when the attributes of investigated ecosystem types approximate the ones, which were used to develop the method of ES ranking. In our case, wetlands are considered the most important LULC types of the study area regulating significantly the total value of ESs. On the other hand, a more detailed investigation should also be performed on their qualitative characteristics. For example, polluted wetlands, which have disturbed characteristics, are not capable of adequately supporting ecosystem functions and thus their contribution is expected to be much lower or in other cases much higher. Thus, the qualitative data of ecosystem types involved in ESs analysis should also be included in data inventories. Other limitation of the specific ESs approach is that some ESs are not properly evaluated for some LULC types due to the lack of data. For example, the ES of nutrient cycling in coastal wetlands with submerged vegetation is not considered, despite the fact that the presence of *Ruppia cirrhosa* meadows in our wetlands may influence nitrogen cycling (Welsh 2000; Gennaro et al. 2004; Bartoli et al. 2008). However, though sacrificing precision, the global averaged values supply a wide applicable dataset for the estimation of spatio-temporal flow of ESs which is extensively used in ESs studies (e.g. Kreuter et al. 2001; Zhao et al. 2004; Petrosillo et al. 2009, 2010; Zang et al. 2011; Aretano et al. 2013; Ayanlade and Proske 2015; Crespin and Simonetti 2016).

An additional issue, which requires special attention, is the validity of ES monetary values. These values should not be interpreted as real market values directly applied for payment of ESs schemes or compensation actions. Their use aims to support future governance and to raise awareness about the importance and the magnitude of loss of natural capital. According to Clare and Creed (2014), natural resources policies should be designed with quantifiable metrics of success for facilitating the reflexive and adaptive management of natural resources. In this study, which focused on river deltas associated with coastal wetlands, the use of ESs was an advantageous metric providing:

a) an indexing for estimating the overall intensity/rate of changes,

b) information about the impact of specific changes on specific ecosystem functions associated with different types of provided services, and

c) the overall economic impact as a function of the dominant LULC type of interest (wetlands).

Conclusions

An extensive analysis of the LULC changes and associated ESs of a protected wetland area in the Po river delta was conducted in this study. A combination of methodologies which take into account both the assessment of socio-economic benefits (approach of ESs) and the monitoring of ecosystems attributes (LULC change analysis using transition matrices TMs) were used by comparing the changes observed during two periods (1954-1976 and 1976-2008) described by different environmental protection governance. TMs identified the exact LULC conversions with special attention to the direct human impact (croplands and urban areas expansion) while the ESs approach adequately described the overall economic impact through the *NCV* assessment. This impact was mostly associated to the significant loss of specific ESs such as the waste treatment, which was the most important ES of the specific system. Despite the high magnitude of ESs losses, the VMG area is still described by a high economic value while the total *NCV* of 1954 provides an aspect of the economic potential, which could be achieved by the initial form of the system providing a strong basis for future evaluation of the environmental protection measures. The major part of the *NCV* was preserved by the closed valley while the significant losses of *NCV* were mostly associated to the reduction of coastal wetlands with emergent and submerged vegetation. These observations can define clear priorities for their protection in the conservation plans, while more robust methods for ESs assessment are required using real economic data in order to improve the estimation of economic impact. The study also addressed the fact that probable effects from indirect human effects may be involved in the loss of vegetated wetlands in the VMG. These effects should better be identified in future studies in order to be able to evaluate the local conservation actions. The proposed combination of the selected methods (TMs and ESs) provides a detailed description of landscape changes and their economic impact, which can be used as decision support tool for landscape conservation policies.

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3.2. "Changes in land use and ecosystem services in tropical forest areas: a case study in Andes mountains of Ecuador"

This study provides a spatio-temporal analysis on a protected area of tropical Andes in Central Ecuador, where extremely rapid landscape changes occur. The ESs mapping was carried out at landscape scale through a benefit transfer method, based on bio-physical values.

The analysis captured an expansion of agricultural land, followed by a conversion to pastures.

Different patterns of change were observed for forest ecosystems, along different altitude belts.

The impacts of LULC changes on ESs were caused by the transition from one ecosystem type to another.

The results of environmental protection were discussed.

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Abstract

Tropical Andes are subjected to severe land use/land cover (LULC) changes that significantly alter the capacity of the landscape to provide ecological functions for supporting human well-being. The aim of the study is a) to investigate the LULC changes in the Ecological Corridor Llaganantes-Sangay (CELS) (Central Ecuador), a buffer semi-protected area, during the period 2000-2014 and b) to analyze their possible consequences on Ecosystem Services (ESs) provision. The analysis was performed using LULC maps of 2000, 2008 and 2014. Ecosystem services were analyzed using the "landscape capacity" index, which is based on a multi-criteria assessment framework. The study captured an extremely rapid LULC transition from croplands to pastures during 2008-2014 below the 2000 m altitude, which was followed by a respective rapid socio-economic change of the local society. The landscape index changes were insignificant showing a slight decrease (-1.92%) during 2000-2014. Although, the overall coverage of natural ecosystems slightly increased during 2000-2014, it was found that the passive landscape conservation might not be sufficient to maintain ESs provision. This was justified by the different ESs contribution

1. Introduction

The provision of Ecosystem Services (ESs) is strictly related to Land use/Land cover (LULC) (Costanza et al. 1997; Metzger et al. 2006). ESs can be strongly affected by changes in LULC patterns, practices, intensity and trade-offs (Fu et al. 2015; Gissi et al. 2016; Gaglio et al. 2017). Despite the fact that LULC changes are ruled by drivers acting at regional or continental extent, the provision of ESs is relevant at different smaller scales (Hein et al. 2006). This scale mismatch results in a process of change that does not pay the proper attention to ecosystem conversions and their consequences. Moreover, ecological structures and functions vary along altitudinal gradients together with the variation of ecosystems and environmental conditions (Coûteaux et al. 2002; Kitayama & Aiba 2002; Moser et al. 2011), introducing an additional dimension to the ESs assessment framework.

The contribution of the majority of ESs to human well-being is not often considered or is underestimated, while humans are prone to exert pressures and changes in LULC with the aim to maximize the provision of one or few ESs, leading to a decline or loss of many others. This phenomenon is widespread around the globe (Millennium Ecosystem Assessment 2005; Ellis et al. 2010), but is particularly severe in tropical regions of developing countries under the pressure of strong socio-economic changes (Lambin et al. 2003; Curatola Fernandez et al. 2015). Among these, the tropical Andes of Ecuador are characterized by landscapes with peculiar climatic and topographic conditions where human settlements both affect and depend on natural ecosystems. This region is an extraordinary biodiversity hotspot (Jørgensen et al. 2011; Bendix et al. 2013) that experience forest clearance and land degradation since centuries (Valencia et al. 1999; Etter et al. 2008; Bare & Ashton, 2016).

For the mitigation of the dramatic deforestation rate of the country (Mosandl et al. 2008), Ecuadorian government promoted incentive-based policies for the conservation of native forests, such as the Socio-Bosque program (Bertzky et al. 2010), as well as the establishment of several protected areas (Keating 2007; Cuenca et al. 2016). The establishment of several protected areas is designated to conserve natural values and processes and can significantly support numerous ESs (Willemen et al. 2013) but such conservation activities do not always guarantee the livelihood of local populations which is mainly supported by food production from croplands, raw materials from forests and pastures/grasslands for livestock production (Kovacs et al. 2015). Despite the fact that protected areas seem to be effective for reducing deforestation in Ecuadorian Tropical Andean forests (Cuenca et al. 2016), the outcome of conservation efforts on the capacity of these areas to support human well-being need to be investigated. Non-natural ecosystems contribute to the provision of ESs (Jose 2009; Porter et al. 2009; Breuste et al. 2013; Rodríguez-Ortega et al. 2014) for which landscapes designed for conservation should also consider these. Overall, the positive correlation between nature conservation and ESs provision is not always observed and should be assessed based on the contribution of ecological functions of both natural and non-natural ecosystems.

Moreover, when analysing the role of environmental protection in maintaining ESs provision, ecosystems variability within the landscape should be also be considered. Altitudinal gradients determine high levels of environmental heterogeneity, which, in turn, was described as conditioning factor of LULC transitions in Latin American countries (Redo et al. 2012). In fact, ecological heterogeneity seems associated to socio-economic and demographic variables (Redo et al. 2012; Aide et al. 2013), which are the main drivers for LULC changes (Sanchez-Cuervo & Aide 2013; Nanni & Grau 2014).

The aim of this study is to examine the temporal LULC changes during a 14-years period (2000-2014) considering the altitudinal dimension in the Ecological Corridor Llaganantes-Sangay (CELS), a buffer area between two national parks in the tropical Andes of central Ecuador, and to assess their consequences on the ESs provision at landscape scale considering both natural and anthropic ecosystems.

2. Material and Methods

2.1 Study area

The study area is the Ecological Corridor Llaganantes-Sangay (Corredor Ecológico Llanganates-Sangay - CELS), which is a transitional area in the Central Ecuador between the Eastern Cordillera of the Andes and the western Amazon forest covering about ~42,850 ha. The study area is a buffer zone between two national parks (the Llaganantes National Park at North and the Sangay National Park at South) (Fig.1a) and it is shared by five municipalities (*parroquias*): Rio Verde (8%) and Rio Negro (47%), Cumandà (23%), Mera (19%) and La Shell (3%). The altitude ranges between 960 and 3756 m above the sea level (Fig.1b) and the climate belongs to the Af class (Tropical Rainforest) according to Köppen classification (Peel et al. 2007). The mean annual precipitation and temperature show a very steep transition to higher values towards East with ranges 2,500-5,500 mm/year and 9-22 ºC, respectively (Fig.1c,d). The strong relief and steep slopes favour the occurrence of highly differentiated habitats with very distinguishable zonation that results in an extraordinary animal and plant biodiversity (Viteri et al. 2002). Animal biodiversity in CELS accounts for 101 mammals, 242 birds, 49 amphibians and 30 reptiles species, whereas plant endemism accounts for 195 species endemic for Pastaza watershed, of which 181 limited to the area between Baños and Puyo (Yaguache 2014), with a perspective of increasing continuously the record in the next years.

The CELS was established in 2002 with the support of World Wildlife Fund. Nevertheless, this area is not under a true coordinated protection as it happens in the cases of Llaganantes and Sangay National Parks. The EcoMinga and Socio-Bosque foundations established additional conservation areas within the CELS that cover only 8,000 ha (19% of total area). The inclusion of additional areas in the future will depend on stakeholder awareness for implementing the development and application of proper incentives. The economy of CELS is mainly based on agricultural activities (mainly orchards and annual crops), tourism and timber production (Yaguache 2014), which support a population of about 13,000 people (INEC 2010). Puyo and Shell are the larger urban systems located at the south-east edge of the territory and they are partly expanded inside CELS with a current population of ~37 thousand people. Puyo was outside CELS territory until 2002 but a clear expansion of city boundaries inside CELS is evident the last years.

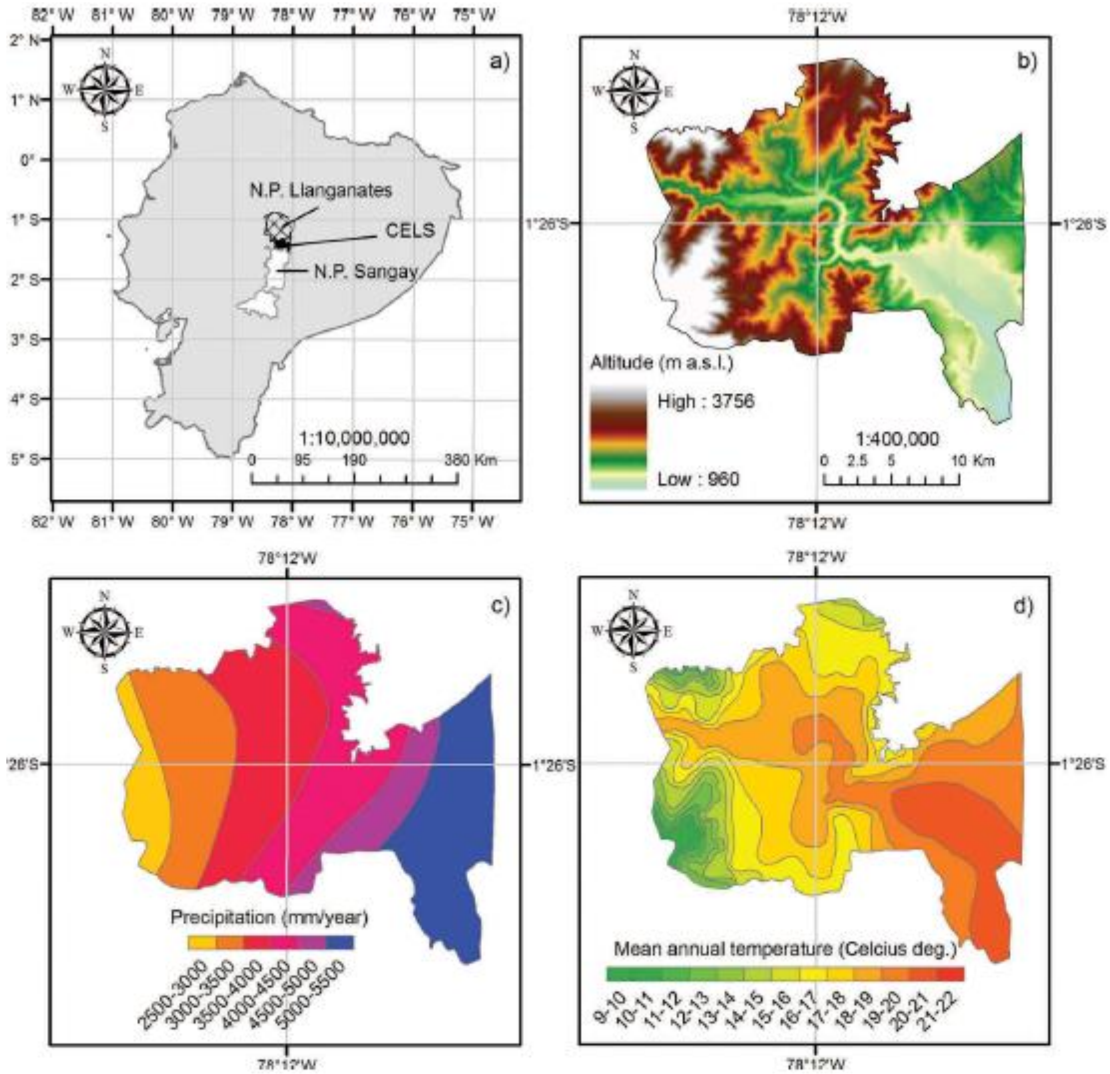

Fig.1: (a) Location of CELS area in Ecuador, (b) altitude, (c) mean annual precipitation and (d) and mean annual temperature

2.2 CELS Ecosystems along the altitudinal gradient

Distinct spatial changes in natural ecosystems occurrence and structure appear following the increase of elevation in tropical mountains (Bruijnzeel et al. 2011). Ecosystems and related functions respond to changes in environmental gradients related to altitude, such as the decrease in temperature in higher altitudes. Lower temperatures and consequent weaker microbial activity, nutrient limitations and decrease of primary decomposers limit decomposition rate at increasing altitude (Coûteaux et al. 2002; Wilcke et al. 2002), and therefore promoting soil organic carbon accumulation (Maraun et al. 2008). Above ground biomass, leaf area index and canopy height decrease with altitude while the restricted nutrient uptake lead to an increase in root production (Kottke et al. 2008; Unger et al. 2013). However, even if the above general patterns are widely documented, local conditions (e.g. slopes) can affect soil properties and their role on biomass production (Moser et al. 2011). In general, the environmental conditions of CELS promote a high natural ecosystems diversity that follows altitudinal patterns, with consequent variations in ecological functions provided at landscape scale. The forests of these ecological zones are also divided in four main categories based on altitude as follows: Foothill forest – FF (<1300 m), Lower mountain forest – LMF (1300-2000 m), Cloud forest – CF (2000-2900 m) and Higher mountain forest – HMF (>2900 m) (Vargas et al. 2000; Muriel et al. 2008). A general description of main CELS ecosystems is provided in Table 1 and Fig.2. Urban centres within CELS are limited to few villages, where inhabitants have a rural lifestyle. More complex urban zones and infrastructures are located at the eastern part of CELS, in the municipality of Shell and Puyo, in proximity of Rio Amazonas airport. Water environments are mainly represented by the river Pastaza and by very few scattered water bodies. The river Pastaza flows from the Andes to Amazonian lowlands, crossing the CELS from West to East. A significant feature of CELS is that LULC changes are regulated by a traditional type of LULC rotation of anthropic ecosystems (croplands to pastures rotation), which serves the provision of different food products depending on the needs of local population.

Table 1: Main LULC types and their ecological functions inside CELS region.

LULC type	**Description**	**Typical vegetation species**	**Main ecological functions**
Agricultural land	Mainly orchards, located along water courses. Monocultures with fertilizers and pesticides application.	*Solanum quitoense*, *Solanum betaceum*	Food provision
Pastures	Both cultivated and natural grasslands for feeding livestock. Stabling of animals is not performed while animal grazing is free following a rotating system by moving the animals from one to another area.	*Pennisetum clandestinum*, *Lolium perenne*	Livestock supply for meat and milk production
Paramo	Typical ecosystem of tropical Andes, located above 3400 m a.s.l. Vegetation can reach 50 cm height. Deep A-soil horizon where organic matter accumulation is favoured by the cold and wet climate and low atmospheric pressure (Buytaert et al. 2007; Hofstede et al. 2002). The humic and dark soils have excellent water infiltration and retention capacity (Buytaert et al. 2007, 2005).	Perennial erbaceous plants (e.g. Poacee)	Water regulation, medicinal resources
Higher Mountain Forest (HMF)	Trees can reach 10-15 m of height with thick and sometimes gnarled trunks, with adventitious roots occupying up to 70 m^2. Very steep slopes (> 15°) affect soil organic carbon content.	*Clusia spp*. in lower part (3200-3330m). Sclerophyllus in upper part	Erosion prevention
Cloud Mountain Forest (CF)	Trees reach a height of 15-25 m. The underwood is very rich and epiphytes and mosses are very abundant. Persistent presence of fog at the vegetation level, which significantly reduces incident solar radiation and evapotranspiration. The frequent contact between canopy and clouds increases water interception (i.e. horizontal rain) and water input to the system (Bendix et al. 2004; Célleri & Feyen 2009).	Melastomataceae, Solanaceae, Myrsinaceae, Aquifoliaceae, Araliaceae, Rubiaceae and several fern families	Water regulation, erosion prevention, biodiversity
Lower Mountain Forest (LMF)	The canopy height can reach 20-35 m tall with sporadic tree of 40 m. Composed by different layers such as canopy, sub-canopy, shrub and herbaceous species. Epiphytes are more abundant than in lower altitudes, while lianas decrease in abundance and diversity (Valencia 1995).	Lauraceae, Rubiaceae, Melastomataceae and occasionally Moraceae	Aboveground biomass (carbon storage, charcoal and timber production)
Foothill Forest (FF)	Forest transition between the foothills of Eastern Cordillera and Amazon forest. Substrate mainly composed by volcanic rocks and sediments of recent origins. The canopy height reaches 30 m and subcanopy and undergrowth are very dense (Vargas et al. 2000). The flatter zones near the River Pastaza are characterized by alluvial and terraced sediment deposits newly formed with high percentages of soil organic carbon. It presents extremely high biodiversity (Reyes-Puig et al. 2013; Titira 1999; Yánez-Muñoz et al. 2010).	Saurauia, Hedyosmum, Brunellia, Weinmannia	Aboveground biomass (carbon storage, charcoal and timber production), biodiversity, soil regulation

Fig.3: CELS ecosystems considered in the analysis: (a) foothill forest (FF), (b) lower mountain forest (LMF), (c) cloud forest (CF), (d) higher mountain forest (HMF), (e) paramo, (f) pastures and (g) agricultural land.

2.3 LULC maps and LULC change analysis

LULC change analysis was based on LULC maps of 2000, 2008, 2014. The maps were produced by the Ministry of Environment and the Ministry of Agriculture, Livestock, Aquaculture and Fisheries of Ecuador by using LANDSAT ETM+ for 2000 (MAE 2012), LANDSAT ETM+ and ASTER for 2008 (MAE 2014) and LANDSAT 8 and RapidEye satellite images for 2014 (MAE 2015). All maps were calibrated using data from field surveys (at least 30 positions were monitored for each land use type) (MAE 2015). Seven LULC types were considered in the LULC change analysis according to the three maps: urban, bare soil, agricultural land, water bodies, páramo, pastures and native forests. The latter was further classified in four classes (FF, LMF, CC, HMF) according an altitudinal gradient, as described above. Pastures include also a small portion of grasslands-shrublands, which are also used as areas for livestock grazing.

The analysis of LULC changes was performed using LULC transition matrices (TMs). In our study, TMs were developed directly by the LULC changes between 2000, 2008 and 2014 without using probabilistic approaches in order to show the exact change from one LULC type to another (Wang et al. 2014; Gaglio et al. 2016). TMs compare the extent of LULC types between two time intervals (e.g. t_1 and t_2) providing the area of each LULC type that remained intact and the specific changes to other LULC types during t_1-t_2. The LULC maps of 2000, 2008 and 2014 correspond to three time intervals and for this reason, three TMs were built that correspond to the periods 2000-2008, 2008-2014 and 2000-2014.

Altitudinal patterns of LULC transitions were also investigated, according to four altitudinal zones (960-1300, 1300-2000, 2000-2900 and 2900-3756 m) which were delineated using a 30 m resolution Digital Elevation Model (DEM). These zones were based on the altitudinal zonation between the forest classes FF (<1300 m), LMF (1300-2000 m), CF (2000-2900 m) and HMF (>2900 m) (Vargas et al. 2000; Muriel et al. 2008) (Table 1). Since the provision of forest ESs significantly varies along altitudinal zones (Becker et al. 2007; Leuschner et al. 2013), forested areas were further classified in four forest ecosystem classes according to specific altitudinal zones reported by Muriel (2008) and Vargas et al. (2000) for the study area. In this case, the specific altitudinal zones were used not only as a proxy to identify the different forest ecosystems but also to better describe the related services involved in the specific landscape transitions.

The significance of LULC changes was investigated through the comparison of proportion with χ^2 test for P-value≤0.01, using StatGraphics Centurion XV (StatPoint Inc.). For each altitudinal range, the comparison was performed between the proportion of each LULC type of the three dates 2000, 2008 and 20014 versus the proportion of the remaining LULC types (e.g. agricultural land vs. non-agricultural land). The null hypothesis was that the extension of the two classes did not change over the three dates. Also an Analysis of Means (ANOM) plot with 99% confidence was applied. This procedure was not used to denote strict statistical differences between the years (e.g. as in the case of LSD test in ANOVA) but to provide indications about the direction of the significant changes based on the deviation from the grand mean of the ANOM plots. Thus, the three codes a, b and c were used to denote the location of the proportion values from the three dates: above, inside and below the 99% confidence limits of ANOM plots (Fedrigotti et al. 2016).

Additionally, the annual rate of change for each LULC type was calculated by using the following equation (Puyravaud 2003):

$$r = \left({}^{1}/_{(t_2 - t_1)}\right) \times \ln\left({}^{A_2}/_{A_1}\right) \tag{1}$$

where r is the annual rate of change of a given ecosystem, A_1 and A_2 the area extension of a given ecosystem at the time t_1 and t_2, respectively.

2.4 ESs change assessment

According to the cascade-model (Haines-Young & Potschin 2010), the provision of ESs depends on ecological functions that are intercepted by humans to support their own well-being. Despite the so-called "ES delivery chain" includes potential ESs stock (capacity), actual supply (flow) and beneficiaries (users demand) (Egarter-Vigl et al. 2017), different mapping methods use proxies to assess the ecological function of ecosystems (ESs capacity) assuming that they are directly or indirectly exploited by humans. For example, the "benefit transfer" method is based on the assumption that a given spatial unit provides a set of ecological functions (e.g. Costanza et al. 1997; 2014; de Groot et al. 2012). The method proposed by Balthazar et al. (2015) is an adaptation of benefit transfer derived by the framework proposed by Koscke et al. (2012), where a set of ecological functions is used to assess the ESs provision (Kremen et al. 2005). This method allows combining qualitative and semi-quantitative indicators to obtain a comprehensive index, sensitive to LULC changes, which expresses the overall capacity to a given landscape to sustain the human well-being. Thus, the ESs analysis was performed at landscape level and the consequences of LULC change on ESs provided by CELS were assessed through the concept of "landscape capacity" index (Burkhard et al. 2009; Koschke et al. 2012; Balthazar et al. 2015). This index uses a multi-criteria assessment framework, which is based on the most important biophysical parameters of ecological functions related to specific ESs. The use of specific biophysical parameters were used to develop a normalized scoring that avoids subjectivity due to qualitative expert judgment (Balthazar et al. 2015).

A scoring matrix of 11 indicators was developed for 7 LULC types: 2 non-natural (agricultural land and pastures) and 5 natural (foothill mountain forest, lower mountain forest, cloud mountain forest, higher mountain forest and paramo grassland) (Table 2). Other LULC types observed in CELS such as urban sites, bare soils and water environments were included in the maps but they were not considered in the ESs assessment. The rivers were not included in the ES assessment due to the lack of data for biophysical indicators. A main problem for rivers ES assessment of the study area is that the main courses have intermitted flow regulated by upstream dams while the small streams have very small area coverage and high discharge acting as intermediate links for ESs transfer among other land uses. In general, the riverbeds are mainly composed by large stones and when the discharge is low, large stony surfaces appear mainly in the west lowland part.

The 11 ecological functions (Table 2) were selected according to their importance for the human well-being and data availability (Millennium Ecosystem Assessment 2005). Only peer-reviewed studies, technical reports and documents were considered in order to assign the bio-physical values to each indicator. Field surveys and personal communication from official sources were used to assess the number of touristic sites, the number of plant species used for medicinal resources and livestock supply capacity (see supplementary material, Sources and details about indicators presented in Table 2). When no local studies were present, we considered studies performed at national scale or studies carried out in similar environments (Andean regions).

The calculation of the landscape capacity index according to Balthazar et al. (2015) is performed by the following steps. In order to allow merging of indicators of different nature, the values of each indicator are standardized between 0 (no relevant capacity) and 5 (very high relevant capacity):

$$I_{norm} = \left({(I - I_{min})}/{(I_{max} - I_{min})} \right) \times 5 \tag{2}$$

where I_{norm} is the standardized value from 0 to 5, I is the indicator value for a given ecosystem, I_{max} and I_{min} are the maximum and minimum values observed for the indicator, respectively. The overall potential of each LULC type is calculated as the sum of the standardized values of each indicator:

$$P_i = \sum I_{norm\ ij} \tag{3}$$

where P_i is the potential of an i LULC type to provide the considered indicator, and $I_{norm\ ij}$ is the standardized indicator value (Eq.2) of an i LULC type for a j ecosystem service. Then, the landscape capacity index is calculated for each LULC type as follows:

$$L_i = A_i \times P_i \tag{4}$$

where L_i is the landscape capacity of i LULC type, A_i the area coverage of the i ecosystem (ha) and P_i (Eq.3) is the potential of the i LULC type. It has to be noted that all the ecosystem services were equally weighted to calculate the index. Finally, the total landscape capacity index is calculated as follows:

$$L = \sum L_i \tag{5}$$

where L is the total landscape capacity index and L_i the landscape capacity for the i LULC type (Eq.4). The landscape capacity index is calculated for each of the three considered dates (2000, 2008, 2014), in order to assess the temporal variation of the ESs provided at landscape scale as consequence of the LULC changes occurred in the CELS.

Finally, the contribution of each LULC type to the total landscape capacity (L) was calculated as follows:

$$R_i = \frac{P_i \times A_i}{L} \tag{6}$$

where R_i is a ranking index which expresses the contribution of the *i* LULC type to the total landscape capacity (*L*). The use of the specific index is based on the simplified version of elasticity coefficient or coefficient of sensitivity provided by Aschonitis et al. (2016) after recalculation of the terms in the original function provided by Kreuter et al. (2001). Aschonitis et al. (2016) found that the initial form of elasticity-sensitivity coefficient could be simplified because the ESs prices are considered always stable without being affected by changes in the demand.

Table 2: Indicators used for the estimation of landscape capacity index and their relations to specific ESs in the study area.

Ecosystem service	Agricultural land	Pastures	Paramo	HMF	CF	LMF	FF	Indicators (unit)	References
Food production	1808	97	64	67	67	122	122	Monetary prices (US$2016 ha^{-1} yr^{-1})	Guayasamín Guanga 2015; Grimes et al. 1994; Kocian et al. 2011
Medicinal resources	0	0	14	9	8	4	6	No. of suitable species	Local interview; de la Torre et al.2008
Livestock supply (bovines)	0	4	0	0	0	0	0	Cattle density (cattle ha^{-1})	Ministero de Agricultura, Ganaderia, Acuacultura y Pesca (pers.comm.)
Water regulation	646	648	933	741	837	748	748	Discharge (mm $year^{-1}$)	Balthazar et al. 2015; Crespo et al. 2010; Fleischbein et al. 2006
Erosion prevention	43.0	20.0	100.0	97.6	97.6	98.3	98.2	Vegetation cover (C-Factor) (%)	NREFD, 2015; Molina et al. 2008; Ochoa-Cueva et al. 2013
Soil structure	17.4	16.0	20.8	15.8	15.8	16.6	27.6	Organic matter (%)	WWF Ecuador 2014; NREFD 2015; Potthast et al. 2009. Hoffstede et al 2002
Soil carbon storage	77	80	204	121	160	112	106	Organic matter (Mg C ha^{-1})	Moser et al., 2011; Hall et al. 2012; Lopez-Ulluloa et al. 2005
Above ground biomass	102.0	41.0	54.1	105.1	105.1	123.1	122.8	Biomass (Mg ha^{-1})	NREFD, 2015; McGroddy et al. 2015
Biodiversity (Vascular plants)	39	15	2000	2800	3000	2700	2500	Vascular plant richness (No. of species ha^{-1})	Jorgensen 2011; Ministero de Agricultura, Ganaderia, Acuacultura y Pesca online database
Scenic quality	2	1	4	5	5	5	5	Relative scale	Burkhard et al. 2009
Recreation/education (Turism)	0	0	5	6	10	15	10	No. of turistic sites	Interviews of local stakeholders

HMF: Higher Mountain Forest; CF: Cloud Mountain Forest; LMF: Lower Mountain Forest; FF: Foothill Forest

3. Results

3.1 LULC changes

The maps of LULC for 2000, 2008 and 2014 are given in Fig.3. The transition matrices of LULC changes are given in Table 3 while the absolute, relative and annual rate of LULC changes are given in Table 4. Table 4 also includes the respective changes in the different forest classes (FF, LMF, CF and HMF). The most important changes during the whole period 2000-2014 were related to a) agricultural land and pasture coverage rotations and b) urban areas expansion (Table 3, 4). During 2000-2008 agricultural land gained 1138.64 ha, mainly from pastures and forest conversion. This trend was completely inverted during 2008-2014 when the 92.65% of the agricultural land of 2008 was lost. After 2008, pastures showed the higher relative gain (208.92%) among all LULC types. During 2000-2008, deforestation occurred with an annual rate of 0.16%, while during 2008-2014 afforestation processes where observed with an annual forest gain of 0.28%. New fragmented urban zones were settled along the Pastaza river during 2000-2008, while the intense urbanization during 2008-2014 was due to the expansion of Mera, Shell and Puyo towns in the south-eastern part of CELS. Urban areas showed the most important relative increase in the total period (239.71%) during 2000-2014 (Table 3 and 4).

The aforementioned general changes were not evenly distributed along altitudinal ranges (Table S.1 in the supplementary material). The results of ANOM analysis based on the altitudinal zonation are given in Table 5. Human activities related to LULC typologies, such as agricultural land, pastures and urban areas, are mainly located in 960-1300 m and 1300-2000 m zones. Therefore, these two altitudinal zones were mostly affected by LULC changes. Agricultural land ~~was~~ significantly expanded during 2000-2008 versus forested areas within the 960-1300 m zone and versus pastures within 1300-2000m zone (Table 3, 4, 5), while during 2008-2014 an extensive decrease occurred at all altitudinal levels. Loss of agricultural land at 960-1300 m was due to a shift of land use activity towards pastures, while afforestation phenomena were detected only at 1300-2000 m. In fact, pastures expansion within 960-1300 m zone affected the foothill forest causing further deforestation process during 2008-2014. Significant increase of forested areas was also observed in the 2000-2900 m zone, where cloud forest colonized previously cultivated land, pastures and bare soil. The LULC type of pastures is the only one showing statistically significant changes in the upper altitudinal zone (2900-3756 m) (Table 5) because their coverage in this zone during 2000 was zero (Table S.1). Higher altitudes are less accessible for human activities, which are the main drivers of changes.

During the time span considered, and particularly during the period 2008-2014, LULC changes analysis showed a general migration of human activities to the lower altitudes, resulting in the re-naturalization of upper lands.

Regarding the water bodies, the only significant change was a decrease within the lower altitudinal belt during 2000-2008 (Table 5) probably caused by the establishment of upstream dams for hydroelectric power generation.

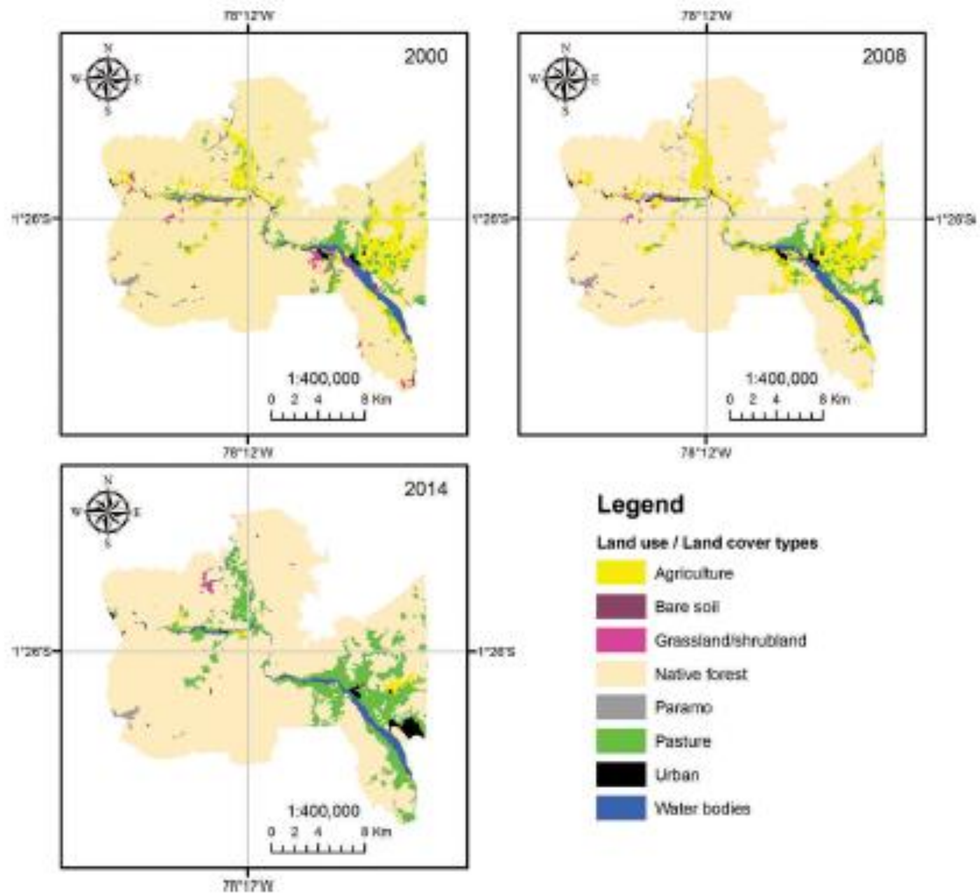

Fig.3: LULC maps of for the CELS region for the years 2000, 2008 and 2014.

Table 3: LULC transition matrices for a) 2000-2008, b) 2008-2014 and c) (total study period) (values expressed in ha).

a)	Period 2000-2008								
LULC type	Agricultural lands	Bare soils	Native forests	Paramo	Pastures	Urban	Water bodies	Loss	Total 2000
Agricultural lands	**3320.57**[a]	8.36[b]	203.31[b]	0.00[b]	234.91[b]	17.99[b]	13.97[b]	478.54[c]	3799.10[d]
Bare soils	4.93[e]	**49.55**[a]	29.98	0.00	1.92	6.44	9.56	52.83	102.38
Native forests	649.38[e]	4.51	**34124.69**[a]	3.78	397.45	7.47	16.24	1078.83	35203.52
Paramo	0.00[e]	0.00	3.93	**97.47**[a]	7.15	0.00	0.00	11.08	108.55
Pastures*	916.55[e]	2.44	381.67	0.00	**1175.81**[a]	30.98	29.90	1361.55	2537.35
Urban	1.65[e]	0.00	0.30	0.00	0.73	**103.24**[a]	0.55	3.23	106.47
Water bodies	44.67[e]	37.83	21.83	0.00	31.18	7.05	**804.68**[a]	142.56	947.25
Gain	1617.18[f]	53.14	641.02	3.78	673.34	69.94	70.22		
Total 2008	4937.74[g]	102.70	34765.71	101.25	1849.15	173.18	874.90		42804.63

b)	Period 2008-2014								
LULC type	Agricultural lands	Bare soils	Native forests	Paramo	Pastures	Urban	Water bodies	Loss	Total 2008
Agricultural lands	**270.47**	7.09	1494.40	0.00	3021.82	53.55	90.42	4667.27	4937.74
Bare soils	0.00	**1.12**	33.32	0.00	19.78	0.01	48.47	101.57	102.70
Native forests	23.01	18.61	**33502.70**	2.77	1095.14	59.83	63.65	1263.01	34765.71
Paramo	0.00	0.00	3.56	**90.66**	7.02	0.00	0.00	10.59	101.25
Pastures*	65.30	0.33	246.60	13.73	**1342.22**	164.37	16.59	506.93	1849.15
Urban	2.92	0.00	11.31	0.00	73.13	**81.98**	3.84	91.20	173.18
Water bodies	1.11	12.25	70.65	0.00	153.30	1.96	**635.63**	239.27	874.90
Gain	92.34	38.28	1859.86	16.50	4370.20	279.71	222.97		
Total 2014	362.81	39.41	35362.55	107.16	5712.41	361.69	858.60		42804.63

c)	Period 2000-2014								
LULC type	Agricultural lands	Bare soils	Native forests	Paramo	Pastures	Urban	Water bodies	Loss	Total 2000
Agricultural lands	**238.21**	2.89	1145.86	0.00	2314.62	40.67	56.86	3560.90	3799.10
Bare soils	0.00	**0.00**	44.97	0.00	31.72	5.97	19.72	102.38	102.38
Native forests	27.79	19.76	**33383.31**	9.39	1531.29	160.37	71.62	1820.21	35203.52
Paramo	0.00	0.00	3.71	**97.77**	7.08	0.00	0.00	10.78	108.55
Pastures*	89.72	3.47	699.29	0.00	**1594.30**	103.76	46.81	943.05	2537.35
Urban	0.00	0.05	4.74	0.00	53.43	**45.56**	2.70	60.91	106.47
Water bodies	7.10	13.24	80.67	0.00	179.97	5.37	**660.90**	286.35	947.25
Gain	124.61	39.41	1979.24	9.39	4118.11	316.13	197.70		
Total 2014	362.81	39.41	35362.55	107.16	5712.41	361.69	858.60		42804.63

[a]The diagonal bold values show the area coverage of a LULC type, which remained intact during each period.
[b]The values of each row, except the bold ones, show how many hectares of a specific LULC type were converted to another LULC type (for example: 234.91ha of agricultural lands were converted to pastures during 2000-2008).
[c]Loss: The total sum of the values of each row, except the bold ones, which provides the total area of a specific LULC which was converted to another LULC types (for example: 478.54 ha of agricultural lands were converted to other LULC types during 2000-2008).

[d]Total area of a LULC type at the beginning of the study period (for example: the total coverage of agricultural lands was 3799.10 ha in 2000).
[e]The values of each column, except the bold ones, show how many hectares of a specific LULC type were gained (for example: agricultural lands gained 916.55 ha after conversion of pastures to the specific LULC during 2000-2008).
[f]Gain: The total sum of the values of each column, except the bold values, which provides the total area which was gained for a specific LULC (for example: agricultural lands gained a total area of 1617.18 ha during 2000-2008).
[g]Total area of a LULC type at the end of the study period (for example: the total coverage of agricultural lands was 4937.74 ha in 2008).

Table 4: LULC changes observed in CELS during 2000-2008, 2008-2014, and 2000-2014 (total period).

	Absolute changes (ha)			**Relative changes %**			**Annual rate of change (Eq.1)**		
LULC type	2000-2008	2008-2014	2000-2014	2000-2008	2008-2014	2000-2014	2000-2008	2008-2014	2000-2014
Urban	66.71	188.51	255.22	62.66%	108.85%	239.71%	6.08%	12.27%	8.74%
Bare soil	0.32	-63.29	-62.97	0.31%	-61.63%	-61.51%	0.04%	-15.96%	-6.82%
Agricultural land	1138.64	-4574.93	-3436.29	29.97%	-92.65%	-90.45%	3.28%	-43.51%	-16.78%
Water bodies	-72.34	-16.31	-88.65	-7.64%	-1.86%	-9.36%	-0.99%	-0.31%	-0.70%
Paramo	-7.30	5.91	-1.39	-6.73%	5.84%	-1.28%	-0.87%	0.95%	-0.09%
Pastures	-688.21	3863.27	3175.06	-27.12%	208.92%	125.13%	-3.95%	18.80%	5.80%
Forest	-437.81	596.84	159.03	-1.24%	1.72%	0.45%	-0.16%	0.28%	0.03%
HMF	-9.92	2.54	-7.38	-1.11%	0.29%	-0.83%	-0.14%	0.05%	-0.06%
CF	-4.56	71.30	66.75	-0.06%	1.01%	0.95%	-0.01%	0.17%	0.07%
LMF	128.83	551.95	680.78	0.59%	2.49%	3.09%	0.07%	0.41%	0.22%
FF	-552.17	-28.95	-581.12	-10.53%	-0.62%	-11.09%	-1.39%	-0.10%	-0.84%

Absolute changes are expressed in ha, relative and annual changes in percentages. Annual change rates were calculated according to Eq.1. The forest area extension is given by the sum of the four forest ecosystems in which was further classified (see Table 2). HMF =Higher Mountain Forest; CF=Cloud Forest; LMF=Lower Mountain Forest; FF=Foothill forest.

Table 5: LULC changes analysis using Analysis of Means at 99% confidence level.

	Urban					Bare soil				
Altitude (m a.s.l.)	χ^2 (df=2)	P-value	2000	2008	2014	χ^2 (df=2)	P-value	2000	2008	2014
2,900 – 3,756	-	-	-	-	-	2.63	0.2687	b	b	b
2,000 – 2,900	-	-	-	-	-	6.76	0.034	b	b	c
1,300 – 2,000	10.97	0.0041	c	a	b	11.34	0.0034	b	b	a
960 – 1,300	178.9	<0.0001	c	c	a	45.91	<0.0001	a	a	c

	Forest					Agricultural land				
Altitude (m a.s.l.)	χ^2 (df=2)	P-value	2000	2008	2014	χ^2 (df=2)	P-value	2000	2008	2014
2,900 – 3,756	0.51	0.7744	b	b	b	2.51	0.2851	b	b	b
2,000 – 2,900	54.85	<0.0001	c	c	a	33.53	<0.0001	b	a	c
1,300 – 2,000	113.12	<0.0001	c	c	a	2073.69	<0.0001	a	a	c
960 – 1,300	87.52	<0.0001	a	c	c	2132.4	<0.0001	a	a	c

	Water bodies					Paramo				
Altitude (m a.s.l.)	χ^2 (df=2)	P-value	2000	2008	2014	χ^2 (df=2)	P-value	2000	2008	2014
2,900 – 3,756	0	0.9985	b	b	b	0.32	0.8529	b	b	b
2,000 – 2,900	4.1	0.1287	b	b	b	-	-	-	-	-
1,300 – 2,000	3.65	0.1608	b	b	b	-	-	-	-	-
960 – 1,300	7.48	0.0237	a	b	b	-	-	-	-	-

	Pastures				
Altitude (m a.s.l.)	χ^2 (df=2)	P-value	2000	2008	2014
2,900 – 3,756	15.03	0.0005	c	a	b
2,000 – 2,900	28.48	<0.0001	a	b	c
1,300 – 2,000	1415.04	<0.0001	c	c	a
960 – 1,300	1785.98	<0.0001	c	c	a

The three codes a, b and c were used to denote the location of the proportion values from the three dates: above, inside and below the upper and lower 99% confidence limits.

3.2 Changes in ecosystem services

The consequences of LULC changes in ESs provision were assessed through the quantification of a set of indicators to calculate the landscape capacity *L* index.

Table 6 presents the standardized values (Eq.2) used for the calculation of the *L* index. The capacity to support ecological functions in CELS expressed by P_i (Eq.3) for each LULC type is also given in Table 6. The natural ecosystems show higher P_i in comparison to the anthropic ones. Foothill forests present the larger potential to support human well-being, followed by the other forest types and paramo grassland. Pastures have the lower potential, mainly related to livestock supply, while their potential for other indicators is limited. Agricultural lands present more than double P_i value in comparison to pastures but less than half value if compared with natural ecosystems (Table 6).

The difference between the P_i values for agricultural land and pastures is mainly due to the gaps concerning soil-related functions and above ground biomass. The intensive grazing activity of cattle causes the decrease of soil coverage and organic matter content with detrimental effects on erosion prevention, soil structure and soil carbon storage. Marked differences in above ground biomass can easily be identified because of the intensive characteristics of grazing management adopted by breeders, which do not allow the growing of trees and shrubs. Contrary, agricultural land in CELS are characterized by a considerable extension of orchards, which provide a good amount of above ground biomass. Different values on scenic quality are due to the different scores proposed by (Burkhard et al. 2009) for these two ecosystems.

Table 6: I_{norm} values (Eq.2) for the ecosystem functions and total potential P_i of each ecosystem to provide ecological functions (Eq.3).

Ecosystem service	Agricultural land	Pastures	Paramo	HMF	CF	LMF	FF
Food production	5.00	0.09	0.00	0.01	0.01	0.17	0.17
Medicinal resources	0.00	0.00	5.00	3.21	2.86	1.43	2.14
Livestock supply (bovine)	0.00	5.00	0.00	0.00	0.00	0.00	0.00
Water regulation	0.00	0.03	5.00	1.66	3.33	1.78	1.78
Erosion prevention	1.44	0.00	5.00	4.85	4.85	4.89	4.89
Soil structure	0.67	0.08	2.11	0.00	0.00	0.33	5.00
Soil carbon storage	0.00	0.12	5.00	1.73	3.27	1.38	1.14
Above ground biomass	3.71	0.00	0.80	3.90	3.90	5.00	4.98
Biodiversity (Vascular plants)	0.04	0.00	3.32	4.66	5.00	4.50	4.16
Scenic quality	1.25	0.00	3.75	5.00	5.00	5.00	5.00
Recreation/education (Turism)	0.00	0.00	1.67	2.00	3.33	5.00	3.33
Pi	12.11	5.33	31.65	27.03	31.55	29.47	32.59

HMF =Higher Mountain Forest; CF=Cloud Forest; LMF=Lower Mountain Forest; FF=Foothill forest.

The landscape capacity indexes for each LULC type L_i (Eq.4) and its total value L (Eq.5) for 2000, 2008 and 2014 are given in Table 7. The total landscape capacity L decreased by 0.42% during 2000-2008, by 1.51% during 2008-2014 and by 1.92% during 2000-2014. These L changes were quite small and mainly regulated by the transitions between agricultural lands and pastures, and urban areas expansion. The high and almost constant coverage of natural LULC types during 2000-2014 (84.71% for 2000, 83.50% for

2008 and 84.87% for 2014) was the main reason of the *L* insignificant changes. The % contribution of each ecosystem type R_i (Table 7) showed the importance of lower mountain forests (LMF) with a contribution ranging between 57-60% during the period 2000-2014.

From a qualitative point of view, even when the total landscape capacity (*L*) does not suffer any significant changes, the LULC transitions determine qualitative changes in ESs provision. For example, the transition from agricultural land to pastures implies the change in provisioning services, with a decrease in crop-derived food and an increase in meat and milk production. Moreover, this transition causes a decrease in erosion prevention and soil structure maintenance, since croplands guarantee a good and constant soil coverage compared to pastures subjected to intensive grazing.

No significant total *L* change could be detected also when the loss of forest at lower altitudes is offset by forest gain at higher altitudes. Nonetheless, a qualitative change in the indicators set, and therefore in ES provision capacity, occur, since different functions are carried out by different forest ecosystems. Forest expansion at upper altitudes (HMF and CF) offers higher protection against soil erosion and better regulation of runoff, while the decrease of forested habitat at lower altitude (LMF and FF) results in loss of biodiversity, carbon storage (i.e. climate change mitigation) and potential for recreational services. The latter is higher for natural LULC types at lower altitudes, whose touristic sites are more accessible if compared with those located at higher and stepped zones.

Table 7: Landscape capacity index for each ecosystem type Li (Eq.4) and its total value L (Eq.5) for 2000, 2008 and 2014.

Ecosystem type	*Li*			*Ri*		
	2000	2008	2014	2000 (%)	2008 (%)	2014 (%)
Agricultural land	46008.94	59798.44	4393.84	4.07	5.32	0.40
Pastures	13529.41	9859.82	30459.13	1.20	0.88	2.75
Paramo	3435.95	3204.74	3391.89	0.30	0.28	0.31
HMF	24094.21	23826.09	23894.69	2.13	2.12	2.16
CF	222530.54	222386.70	224636.23	19.71	19.78	20.28
LMF	648853.07	652649.79	668916.22	57.46	58.04	60.39
FF	170827.45	152831.79	151888.18	15.13	13.59	13.71
Total *L*	1129279.58	1124557.37	1107580.18	100.00	100.00	100.00

Also the Ranking index R_i is reported (Eq. 6).

3.3 Transitions of agricultural land to pasture

One of the most interesting issues of this research study is that LULC changes in CELS were regulated by an extremely high transition between different types of anthropic ecosystems. LULC transitions where croplands, pastures and secondary vegetation replace each other are commonly observed in the Andean region (Rodríguez Eraso et al. 2013), as well as in all the tropical part of South America (Wassenaar et al. 2007). The transition from agricultural land to pasture was the most relevant LULC change and mainly occurred during 2008-2014 covering an area of 3,004.8 ha, equal to 7.02% of the total area (Fig.4). From the 3,004.8 ha, the 69% was already agricultural land, 20% was covered by pastures and 11% was forest during 2000. This indicates that 31% of this area experienced a double conversion (pastures-agricultural-pastures or forest-agricultural-pastures) during 2000-2014. The forest loss during 2000-2008 was mainly observed in the altitudinal zone of 960-1300 m (Foothill forest).

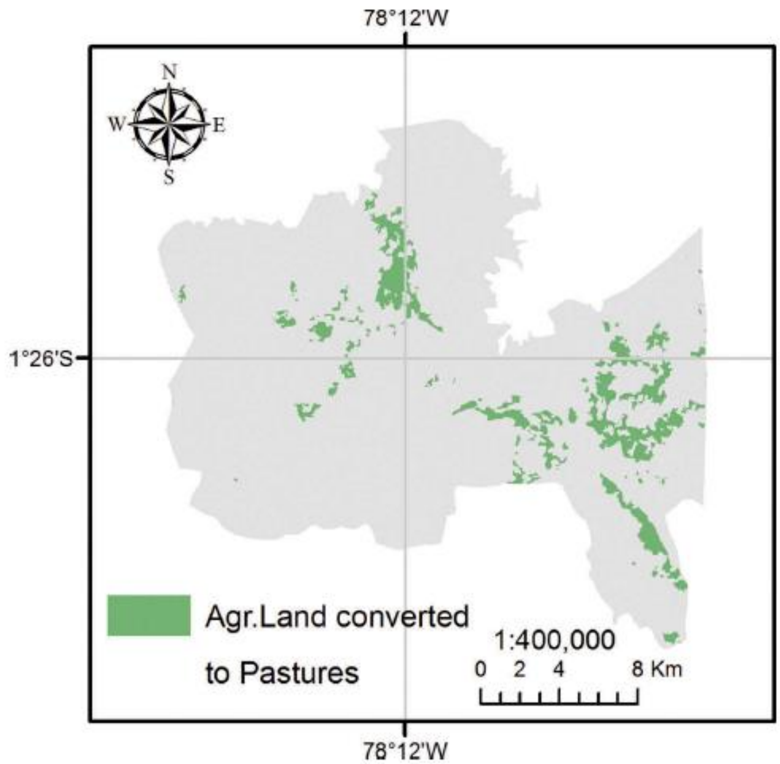

Fig.4: Agricultural land converted to pastures during 2008–2014.

4. Discussion

The LULC changes observed in CELS highlight the typical pathway of changes in Ecuadorian Andean mountains. Deforestation typically occurs for wood or charcoal extraction (for one or two years), then the land parcel is converted to agriculture (two-five years) and then to pasture (seven-ten years), before returning the land to fallow for another one-five years (Luoma 2004). Rodríguez Eraso et al. (2013) described general patterns of change for Colombian Andes where abandoned agricultural areas evolve to secondary vegetation, where the latter is converted to pastures. The scarce amount of secondary vegetation observed in all the three LULC maps suggests that the conversion from agricultural land to pastures during 2008-2014 occurred very fast. In general, even when short time intervals were considered for the comparison of LULC, some intermediate stages between LULC changes could not be detected due to the very fast regeneration capacity of CELS ecosystems. Natural regeneration in tropical

Andes is influenced by several factors related to the previous land use and management, such as seed availability and dispersion, presence of remnant vegetation, soil structure, light and water availability (Guariguata & Ostertag, 2001; Günter et al. 2007; Lozada et al. 2007). In CELS, the natural regeneration in a native forest dominated landscape is fostered by the proximity of natural environment to cropland and pastures, the favourable temperatures and the constant precipitation throughout the year, resulting in up to two meters of pioneer species growing after only two years (Yaguache 2014). Even though transitions in both directions between pastures and croplands are common in tropical landscapes (Rodríguez Eraso et al. 2013; Wassenaar et al. 2007), the massive conversion of agricultural land into pastures during 2008-2014 highlights the important role of this transition to respective changes in socio-economic conditions of the local population. In the case of agricultural land, the cultivation of Naranjilla, the most widespread cultivation on CELS, provides good yields between the second and fourth year but falls markedly after, forcing producers to abandon the plantation for about ten years (Bajaña & Viteri 2002). Moreover, Naranjilla crops require the application of agro-chemicals in order to cope pests and fungal attacks (Ochoa & Ellis 2005), which affect economic profits. Conversely in the case of pastures, the cattle production offers economic flexibility and lower financial risks (Wassenaar et al. 2007), even if pasture degradation may occur in time as well (Fearnside 1989). Thus, conversion to pastures seems to be more economically sustainable in comparison to agricultural land since the economic contribution of the latter is reduced due to overexploitation and unsustainable practices, which decrease the soil fertility very fast.

Since anthropic environments are focused on the exploitation of one market-oriented ecosystem service (e.g. food production), the provision of other functions is just a “side effect” that, in case of cultivated areas and pastures, depends on management practices, which are not considered by farmers and breeders. The protection of mountain natural ecosystems and biodiversity seems to be already effective in CELS, despite the high deforestation rate detected at the country scale in Ecuador (Mosandl et al. 2008). Thus, strategies for supporting farmers towards more sustainable practices are needed, with the aim to avoid agricultural land abandonment and to manage the croplands capacity to support a wider set of ecological functions. These targets are partially discussed in the Landscape Restauration Plan of CELS (Yaguache 2014), which shapes the objective of improving the supply of ecosystem goods and services as well as to strengthen ecosystems resilience and adaptation capacity to climate change. The Plan also suggests the development of better productive practices and restoration priorities for their implementation on the 37% of croplands and pastures during the next years, with the goal to increase productivity and to improve hydrological and biodiversity conditions in croplands and pastures. Namely, it suggests the use of mix permanent crops (e.g. mandarins) with annual ones, the maintenance of orchards multicultures and dispersed trees in pastures (Yaguache 2014). If these measures would be respected and extended to all agriculture and pasture fields, both market and non-market ESs provided by CELS could be significantly increased with noticeable benefits for the local population. Mixed crop systems, such as those with fruit trees and annual crops, can improve the ecological functions and sustain farmer's economic profits by reducing the need of chemicals applications. Unlike annual monocultures, the mix of permanent and annual crops provides a continuous vegetation coverage, which is found to exert a fundamental role in preventing soil erosion in Ecuadorian Andes (Molina et al. 2008). Permanent crops lead to an increase on soil organic matter content (Blanco-Canqui 2010) which significantly boosts soil fertility and regulates soil-water dynamics and microbial activities (Lal 2004). The maintenance of species diversity in orchards systems could avoid land degradation caused by monocultures of Naranjilla. Diverse and multi-strata orchards can provide additional benefits for biodiversity and biological control (Simon et al. 2009). In general, mixed crops show a better capacity to capture and use biophysical resources (Jahansooz et al. 2007) and to limit disease and pest organism (Perrin 1977; Sapoukhina et al. 2010), thus concurring to a decrease of chemicals need.

Regarding natural ecosystems, the analysis showed that foothill forest has the larger potential for ESs provision, while the lower mountain forest exhibits the greater contribution in ESs due to its large coverage. Taking into account these observations, forest management should consider these attributes in order to maintain a high contribution of non-market ESs, which sustain ecological quality. Fuelwood and charcoal are important forest products in Ecuador (Luoma 2004), and for this reason, alternative approaches to mitigate deforestation for such purposes are needed. An alternative approach for obtaining such products could be the use of trees in pastures. This practice was also found to be effective at reducing soil erosion (White & Maldonado 1991), improves biodiversity, and provides shadow and protection to livestock (Luoma 2004).

The uneven LULC changes of CELS determined by altitude are in line with the situation of other mountainous regions of Latin America (e.g. Redo et al. 2012; Nanni & Grau 2014). From an ESs perspective, this has great relevance since the capacity of forests of providing ESs and biodiversity is different, due to ecological factors. This highlights the fundamental role of ecological management in conserving natural environments. The maintenance of agricultural land at higher altitudes could significantly decrease human pressures on lower forests, with the consequent conservation of their ecological role.

However, some limitations concerning the method should be considered. The benefit transfer approach does not consider the spatial position of ecosystems, approximating the landscape to a simple sum of ecosystems. Moreover, possible bias can occur when data from different sources are collected. Nonetheless, the landscape index minimizes the error due to non-local data application, as well as intra and inter-site variability, by normalizing the indicators (eq.2), thus reducing the area effect. At the same time, when altitudinal variability is considered, it provides an acceptable approximation of ESs variation.

5. Conclusions

This study provided a description of LULC and ESs changes in the CELS region. Although, a 14-years study period may seem a relative short timespan for LULC change analysis, the study captured an extremely rapid LULC transition from croplands to pastures followed by a respective rapid socio-economic change of the local society, suggesting also its high degree of adaptability.

Although the overall coverage of natural ecosystems slightly increased during 2000-2014, confirming the effectiveness of forest protection in Ecuador. It was found that the passive landscape conservation focused on natural ecosystems and biodiversity may not be sufficient to maintain ESs. Urbanization, agriculture abandonment and pasture expansion under unsustainable practices are the main threat to the maintainance of ESs provision in CELS. Governance plans of CELS, such as the Landscape Restoration Plan, should focus more on management practices for croplands and pastures, including also organic cropping and more sustainable alternatives to chemicals applications, with the aim to guarantee both monetary incomes and high environmental standards to CELS population. The role of specific forest types on ESs provision was also highlighted providing significant information about forest conservation based on different altitudinal zones.

The framework applied by this study could represent an effective monitor strategy for environmental and ESs governance. Moreover, more detailed future field studies are required in order to improve the knowledge of how the different ecosystems of CELS support ecological functions, according to environmental gradients (e.g. altitude). Finally, different weights could be assigned to each function according to stakeholder preferences and/or political addresses including also stakeholders' perception and preferences, which play a crucial role in environmental planning.

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3.3. "Past, present and future Ecosystem Services supply by protected floodplains under land use and climate changes"

This study assessed the effects of floodplain protection initiatives on climate regulation and water-related services. A model tool was used to map carbon storage and sequestration and water budget in different dates at landscape scale. Moreover, future LULC scenarios to 2050 were projected for both no climate change effects and climate change effects. Environmental conservation led to trade-offs among ESs, which were confirmed also for future projections. Predicted climatic changes are expected to exacerbate such conflicts under the different management options. Carbon storage increased as a consequence of increase of biomass (i.e. changes due to alteration of ecological attributes), while changes of water budget components were caused by transition to one ecosystem type to another.
Implications on environmental management were discussed.

Abstract

The understanding of protection initiative effects on the delivery Ecosystem Services (ESs) is of paramount importance to attain sustainable management in Protected Areas (PAs). Protected floodplains provide important ESs to local populations such as water regulation and climate regulation through carbon storage. This study analyses the spatio-temporal variation of land use/land cover (LULC) changes in the Nature Reserve of Paul do Boquilobo (Central Portugal), with the aim to describe the effects of management policies, since PA declaration in 1980, PA until 2015 using climate mitigation (i.e. carbon storage and sequestration) and water-related services (flood mitigation, water regulation and supply), by means of InVEST modelling. Three dates of LULC conditions were considered in the analysis (1967, 1990 and 2015). Moreover, two future alternative LULC scenarios for 2050 were designed (a

"Business"- BUS and a "Naturalization"- NAT scenario). The BUS scenario considers a LULC distribution towards high productive agricultural systems (irrigated croplands), considering only a strict natural area central core, while the NAT scenario considers full coverage of natural areas. The two future extreme LULC scenarios were analyzed with InVEST model for both no climate change effects and climate change effects based on the pessimistic rcp8.5 climatic scenario (climate derivatives of HadGEM2-ES climate model). The results brought to evidence that PA declaration and conservation efforts during 1965-2015 increased carbon storage-sequestration and flood mitigation (higher water storage, less recharge and runoff). The analysis of future LULC scenarios demonstrated that the complete renaturing in combination with climate change (reduction of precipitation, increase of temperature) can lead to severe reduction of recharge and runoff. Such effects may be desirable in floodplains and flood vulnerable areas, however conflict may appear between specific water regulation services by the application of PA initiatives in places where groundwater resources are limited or minimum ecological flows in surface waters are impaired. Thus, the use of modeling approaches to assess complex ESs interactions, considering the possible effects of climate change can be a valuable tool for the description of ecological mechanisms that underlie trade-offs and synergies among ESs. This is particularly important when considering protected areas, where the assessment of delivered services is urgently needed in order to provide arguments for biodiversity conservation.

Introduction

The declaration of Protected Areas (PAs) is one of the most important instruments to control the loss of biological diversity and to safeguard threatened species and ecosystems all over the world. PAs are designed to protect local and regional biodiversity from anthropogenic pressures and impacts (Margules and Pressey 2000). According to the Protected Planet Report (UNEP-WCMC and IUCN 2016), PAs cover almost 15% of the world's terrestrial and inland waters, nevertheless their relevant global extension is likely to increase in the future, due to the increased awareness about their importance in developing countries. When properly managed, PAs may play an important role on poverty alleviation and sustainable development of local populations (Naughton-Treves et al. 2005; Andam et al. 2010; Canavire-Bacarreza and Hanauer 2013). Moreover, the management of PAs requires a long-term political view and financial commitments that are often not attained (Bruner et al. 2004; Hockings et al. 2006).

Since biodiversity and ecosystem services (ESs) are intrinsically related, PAs deliver a wide set of valuable services, particularly those that are not directly marketable, consequently PAs are fundamental assets, whose maintenance can be justified also by their functional values (Millennium Ecosystem Assessment 2005). Regulating services are often involved in relationships (trade-off and/or synergies) with other ESs (Bennett et al. 2009) and for this reason can be considered as suitable indicators for assessing ecological resilience (Bennett et al. 2005). However, their environmental value is frequently ignored, as it is not incorporated in the market dynamics (Turner and Daily 2008). For this reason, severe losses of regulating services were observed when provisioning services were boosted, particularly on floodplain environments (Jia et al 2014; Palomo et al 2014; Gaglio et al 2017).

Floodplains host ecotones with an exceptional biological diversity, particularly when high-levels of spatial and temporal heterogeneity is present (Ward et al. 1999; Ward and Tockner 2001). The water-terrestrial interface supports a large number of species, which are benefited from intermediate disturbance conditions (Crandall et al. 2003) regulated by flood dynamics (Tockner and Ward 1999; Thomaz et al. 2007). Floodplains also perform regulating functions of great importance for the surrounding environment. They can significantly regulate floods' impact through water absorption and nutrients retention (Weng et al. 2003; Forshay and Stanley 2005; Hoffmann et al. 2009), avoiding damages to human settlements and protecting groundwater quality.

Due to their high productivity (Güneralp et al. 2014), floodplain ecosystems can store significant amounts of carbon in soil and living biomass, playing a role in the climate change mitigation and being an important carbon sink for other terrestrial ecosystems (Robertson et al. 1999; Cartisano et al. 2013).

Climate change is expected to exacerbate biodiversity changes (Sala et al. 2000; Araújo and Rahbek 2006) and the capacity of ecosystems to provide goods and services (Pecl et al. 2017), especially on Mediterranean climate (Sala et al. 2000). Climate change is expected to potentially accelerate changes in species distribution (Araújo and Rahbek 2006), reshuffling the geographic distributions of plant and animal species world-wide (Parmesan and Yohe 2003). However, even if current estimates in biodiversity changes are very variable (Bellard et al. 2012), there is the need to improve knowledge about potential changes in biodiversity (Sala et al. 2010) and ES provision (Pecl et al. 2017) in order to be prepared for adapting the management of PAs to address those changes.

The aim of this study is to assess the effects of PA designation on spatial and temporal changes of regulating ESs (climate regulation based on carbon storage, water supply and water regulation) in a protected floodplain of Central Portugal: the Nature Reserve of Paul do Boquilobo (NRPB). In order to describe the ES flow before and after the establishment of PA in 1980 in this study area, three dates with their respective land use/land cover (LULC) maps were considered: 1960, 1990, and 2015. Moreover, future projections of land use and climate change to 2050 were combined with the aims a) to simulate the response of ESs to such modifications using the InVEST (Integrated Valuation of Ecosystem Services and Trade-offs) (Sharp et al. 2016) and b) to use the simulation results to support future management measures for the study area.

Data and methods

Study Area

The Nature Reserve of Paul do Boquilobo (NRPB) (39°23'N and 8°32'W) is a natural floodplain of 817 ha, with an altitude between +15 m and +32 m a.s.l. It is located in the catchment area of Almonda River, a tributary of Tagus River, in the municipality of Torres Novas and Golegã (district of Santarem, central Portugal) (Fig.1). The annual precipitation is about 713 mm yr-1, and the area can be classified within the Dry Sub-humid (C1) category according to the Thornthwaite moisture classification (Feddema 2005). The Tagus valley has been subjected to land reclamation works during the last decades, with the aim to gain new arable land. To counterbalance the impact of human pressures on these environments, the NRPB was declared PA based on the National Law No. 198/80 of July 24 (green area in Fig.1). This area hosts one of the most important communities of herons in Portugal, and is a popular spot for wintering water birds such as ducks, coots and waders. It is the only national reproductive site for Aythya ferina, one of the main sites for refuge species such as Anas penelope and Anas clypeata and one of the few sites with the potential for nesting of Chlidonias hybrid. In addition, several fish species use the reserve for reproduction like the endemics Chondrostoma oligolepis and Iberochondrostoma lusitanicum (ICNF - Instituto da Conservação da Natureza e das Florestas 2005). The NRPB is currently characterized by riparian corridors, with predominance of Salix, Populus and Fraxinus species, particularly in the core area. In favor of protecting such biodiversity assets, the Forestry and Nature Conservation Institute (ICNF) implemented a Management Plan following the IUCN directives, where a core area of total protection was established (core PA in Fig.1), while the part of NRPB not belonging to the core PA was considered as areas of partial or complementary protection. NRPB plays an important hydrological role for all the surrounding ecosystems. During the wet periods, the area functions as an area of water retention during the frequent floods of the Almonda river, protecting the surrounding croplands. During dry periods, the NRPB provides valuable water supply for irrigation by recharging local aquifers (Baptista and Santos 2016).

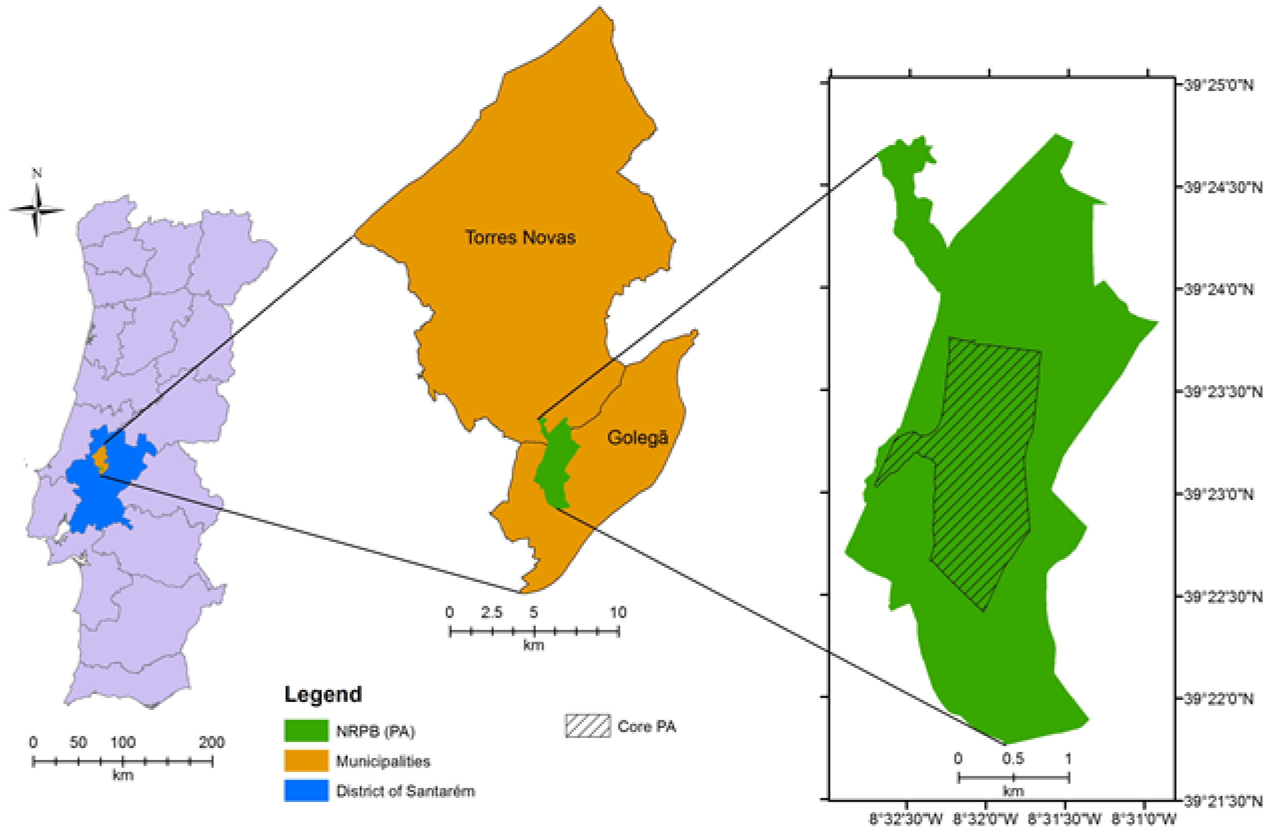


Fig.1 Location of Natural Reserve of Paul do Boquilobo (NRPB) in Portugal and boundaries of PA.

Methods and Data

LULC maps and future LULC scenarios

The LULC maps for 1967 and 1990 derived from the Map of agricultural and forested land of Portugal, elaborated by the Recognition and Planning Service of the Ministry of Economy of Portugal for the respective year. The map of 2015 was generated from the Habitat map (drawn according to the Habitat Directive of the European Union 92/43/EEC (European Commission 1992), after correction using data from field surveys. All the maps were harmonized according to a common LULC classification, as reported in Table 1. A further classification for riparian forest was also carried out since fast growth rates are reported for these ecosystems (or for the species that compose them) (Giese et al 2000; Rheinhardt et al 2012), and thus the riparian forest ecosystems were further classified in “young” and “mature” (see definitions in Table 1). Moreover, two future alternative LULC scenarios were developed: a “business scenario - BUS” and a “naturalization scenario - NAT”.

For the BUS scenario it was assumed that all the private lands inside RNPB will be exploited in such way to maximize market-ESs (i.e. for food and fiber provision) and consequently the income of the owners (i.e. though conversion to irrigated crops, olive groves and poplar plantations), while the natural evolution of ecosystems is guaranteed inside the core protected area (as proposed in the Management Plan of ICNF). Only the minimal prerequisites of the management Plan would be respected. The recommendations of the management Plan would guarantee the protection of the ecological evolution process towards climax conditions inside the total protected area. Furthermore, the presence of a cork oak forest area would be also promoted on the lands belonging to the state outside the core area, as traditional landscape feature supported by the management Plan (Santos et al. 2016).

For the NAT scenario, the projection assumes that all the private lands will be purchased by the RNPB under the premises of the Management Plan provisions currently in place (ICNF - Instituto da Conservação da Natureza e das Florestas 2005). Thus, a process of landscape naturalization in most of the Nature Reserve will occur as it will be owned by the government. Larger areas would undergo naturalization processes, except for olive groves in the Northern part of the Reserve. Croplands and poplar plantations would be replaced by Mediterranean grasslands, and oak forests would be maintained in the western part to preserve traditional landscape. Riparian forest would expand from buffer zones nearby the watercourses and the existing young riparian forest would evolve to climax stage (i.e. mature riparian forest).

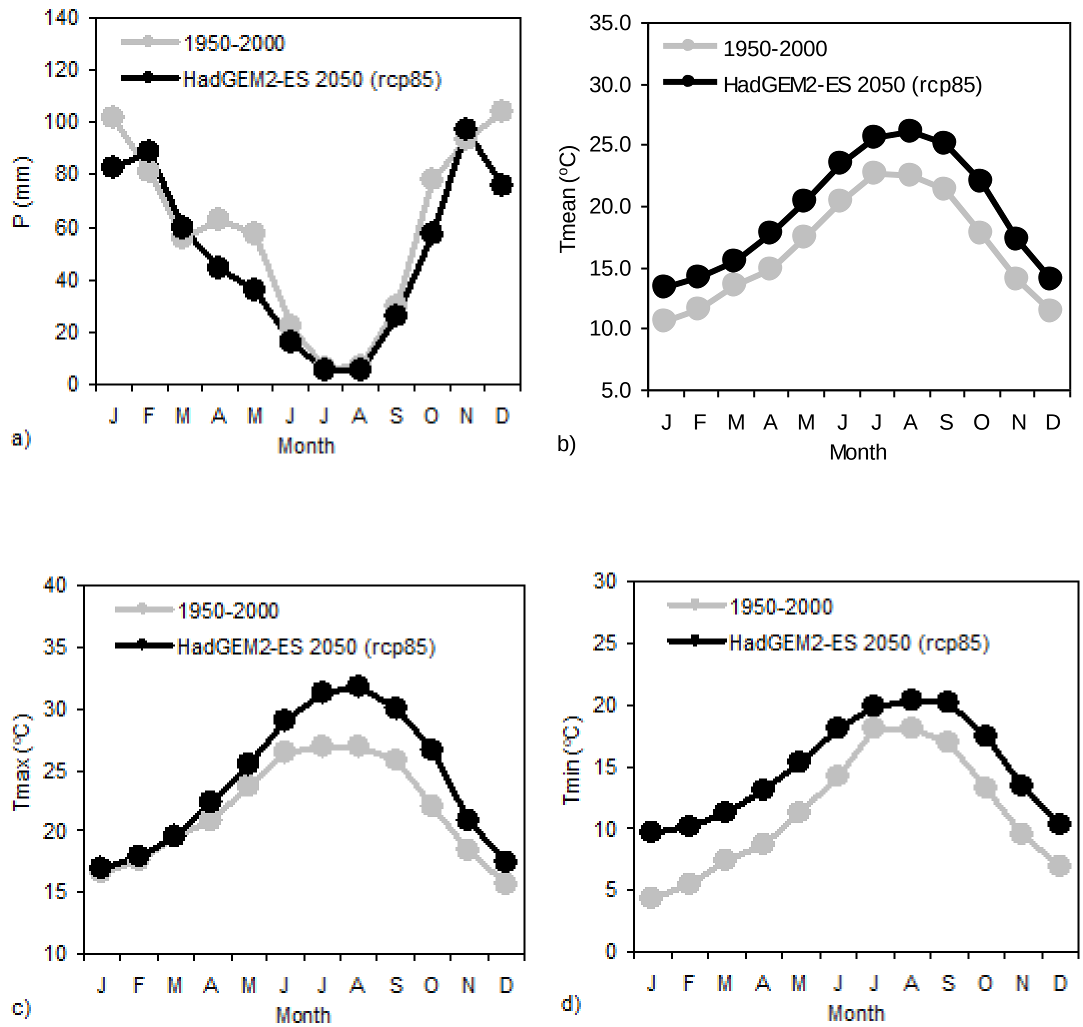
In order to get two further realistic future scenarios, it was assumed that they could occur in ~30 years (~2050). Thus, some "young riparian forest" areas from the 2015-scenario will evolve to "mature" stage in future-scenarios considering that no human impacts or intervention affects them. The two future LULC scenarios were developed, allowing for two extreme situations, providing the two opposite extreme gradients that define the maximum range of LULC changes in the study area and consequently the respective two extreme possible trends of ESs changes.

Table 1. LULC classification of LULC maps for 1967, 1990, 2015 and 2050.

Land cover type	**Description**
Irrigated crops	Land cultivated with annual crops that is under irrigation (mainly corn)
Aquatic	Permanent wetlands and Almonda river
Eucalyptus plantation	Plantations of *Eucalyptus globulus*
Oak forest	Forest dominated by *Quercus suber*
Pasture/grassland	Lands covered by permanent herbaceous species
Poplar plantation	Plantations of *Populus sp.*
Young riparian forest	Established riparian forest with dominant trees age between 25 and 50 years old. Predominance of *Populus. sp., Salix sp., Fraxinus sp.*
Urban	Railroads and farm buildings
Olive groves	Plantations of *Olea europea*
Rice cultivation	*Land cultivated with Oryza sativa*
Orange cultivation	Plantations of *Citrus sinensis*
Non-irrigated crops	Land cultivated with annual crops without irrigation (mainly wheat)
Figs cultivation	Plantations of *Ficus carica*
Vineyards	Plantation of *Vitis vinifera*
Oak and Pine forest	Forest dominated by *Quercus suber* and *Pinus pinaster*
Oak and young riparian forest	Forest dominated by *Quercus suber* and riparian species (25-50 years old)
Oak and holm oak forest	Forest dominated by *Quercus suber* and *Quercus ilex*
Mature riparian forest	Riparian forest with dominant trees older than 50 yr. Trees increased in canopy, basal area and stem density. Tree height and canopy show higher degree of differentiation
Sclerophyllous vegetation	Land covered by sclerophyllous woody vegetation that do not meet the forest or permanent crop definitions

Climate data

For the past scenarios, the climate raster data of the period ~1950-2000 of mean monthly precipitation (P), minimum, mean and maximum temperature (Tmin, Tmean, Tmax) from WorldClim database (http://worldclim.org) (Hijmans et al. 2005) and reference evapotranspiration ETo of ASCE-standardized method (Allen et al., 2005) for short reference crop (clipped grass) from Aschonitis et al. (2017) were used (data of ~1 km resolution). The mean monthly rasters data of ETo, Tmin, Tmean, Tmax and P were used to build a simplified ETo model for the study area {ETo = 6.3837Tmean - 0.6466P + 2.5109(Tmax-Tmin), $R2=0.99$, $p<0.001$}. This model was used to build rasters of future ETo conditions considering the Tmin, Tmean, Tmax and P raster values of 2050 estimated by the HadGEM2-ES model (Collins et al., 2008) for the rcp85 scenario (worst scenario of greenhouse gas emissions). The difference between the past and the future conditions for the P, Tmean, Tmax, Tmin and ETo are given in Fig.2a,b,c,d,e, respectively.



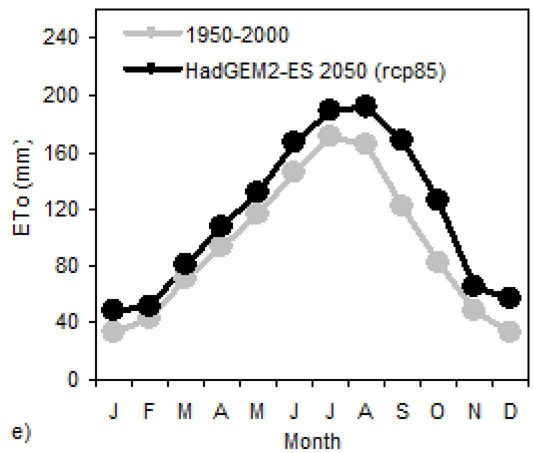


Fig.2 a) Mean monthly precipitation *P*, b) mean monthly temperature T_{mean}, c) mean monthly maximum temperature T_{max}, d) mean monthly minimum temperature T_{min} and e) mean monthly evapotranspiration ET_o, of 1950-2000 and 2050 for the rcp85(HadGEM2-ES) climate change scenario

InVEST model

The response of protection initiatives on regulating ESs was analyzed using InVEST (Integrated Valuation of Ecosystem Services and Trade-offs) Version 3.3.3 package (Sharp et al. 2016). The model simulated a) climate regulation based on carbon storage and sequestration (model of Carbon storage and sequestration: Climate regulation), and b) water supply and water regulation (model of Seasonal water yield), using the three LULC maps of NPRB that correspond to the dates 1967, 1990 and 2015 plus the “business” and “naturalization” LULC future scenarios for 2015 and 2050. The models were run for each past date and future projections, to allow a comparison of the outputs. Since the NRPB was declared PA in 1980, the changes in LULC and regulating ESs occurred between 1967 and 1990 period can assess the first consequences of the protection initiative, while those occurred during between 1990 and 2015 can estimate the effects of its management. Similarly, the comparisons of the future LULC scenarios (BUS and NAT) with or without climate change versus the past scenarios can assess the effects of future LULC designation and climate change on ESs. The main data sources used for the simulations are presented in Table 2.

Table 2. Data sources used for the simulations of InVEST model scenarios.

Data	Source
LULC map of 1967	Map of agricultural and forest lands of Portugal. 1967. Recognition and Planning Service. Ministry of Economy of Portugal
LULC map of 1990	Map of agricultural and forested land of Portugal. 1990. Recognition and Planning Service. Ministry of Economy of Portugal
LULC map of 2015	Habitat map corrected by the authors based on field surveys
LULC maps of 2050, BUS and NAT scenarios	Management Plan of ICNF (2005)
Carbon content, MAI, shoot-to-root ratio	Portuguese national inventory report on greenhouse gases, 1990-2012 (Pereira et al. 2014) (Table S.1)
Maps of monthly reference crop evapotranspiration (mm)	Aschonitis et al. (2017a) (https://doi.pangaea.de/10.1594/PANGAEA.868808, http://esrn-database.org/gis-data.html)
Maps of monthly precipitation (mm) and temperature (ºC) for current and future scenarios	(Hijmans et al., 2005; Fick and Hijmans, 2017) WorldClim (http://www.worldclim.org/). Future scenarios of HadGEM2-ES for rcp8.5 are also provided by WorldClim database as elaborated rasters.
Digital Elevation Model (DEM) (m)	SRTM of 90 m resolution (http://srtm.csi.cgiar.org/)
Map of SCS soil hydrologic groups	(Atlas of soil water) European Data Portal (https://www.europeandataportal.eu/data/it/dataset/e66eced9-8319-4584-8038-cf9b05e0e207)
Curve Number (CN)	(NRCS, 1986)
Crop Factor (Kc)	InVEST database, FAO (http://www.fao.org/docrep/X0490E/x0490e0b.htm)
Average numbers of events per month (Santarém weather station, 1961–1990)	https://www.yr.no/place/Portugal/Santar%C3%A9m/Vale_de_Estacas/statistics.html

Climate regulation

Carbon sequestration is the mechanism through which natural ecosystems withdraw carbon from the atmosphere that is stored in their living and dead biomass of the soil. The amount of carbon stored in natural ecosystems can significantly mitigate climate change due to the excess of carbon dioxide concentration in the atmosphere. The amount of carbon stored and sequestered based on the different LULC and climate scenarios was estimated using the tool "Carbon Storage and Sequestration: Climate Regulation" of InVEST biophysical model (Sharp et al. 2016). The tool estimates the amount of carbon currently stored in a landscape and the amount of carbon sequestered over time, by integrating four carbon pools (aboveground biomass, belowground biomass, soil, dead organic matter) with land cover maps. Input data for carbon pools is provided in Table.S1 (Pereira et al., 2014). Optionally, the model can provide monetary values of sequestered carbon for the period 2015-2050. The computation of monetary value (Net Present Value) considered a social cost per metric ton of carbon of 220US$ (Moore and Diaz 2015) and a market discount rate in price of carbon of 7% (as suggested by the model). Values for terrestrial carbon pools were obtained from the Portuguese national inventory (Pereira et al. 2014), except for soil carbon content in grasslands, which was derived from local survey data (Santos et al. unpublished data). No value for carbon storage and sequestration was attributed to water ecosystems, despite the important role of the marsh in storing such a resource. When no direct correspondence was evident between own and national inventory LULC classifications, the values were selected from the most coherent class. Carbon storage values for "mixed" LULC classes were calculated by combining the most representative habitats. For example, the carbon storage of "young riparian forest" was calculated as the weighted average of "Quercus spp." (50%) and "Other broadleaf" (25% Poplar + 25% Willow). Aboveground and belowground carbon content of "mature riparian forest" were calculated by considering the Mean Annual Increment (MAI) of total living biomass, expressed as m3 ha-1. These values were converted in Mg C ha-1 considering the gravity weight (t m3) of the respective species (Quercus spp. = 0.85, Poplar spp. = 0.85 and Willow spp. = 0.75) and a default carbon fraction of 47% of biomass (as suggested by IPCC 2006 guidelines). Carbon content was therefore allocated in above and below ground biomass pools, according to the shoot-to-root ratio of the respective LULC types. Both MAI and shoot-to-root ratio for forest type were also provided by Pereira et al (2014).

Water regulation and supply

Natural systems can regulate hydrological flows by retaining water amounts in soil and groundwater, affecting evapotranspiration and infiltration. The water regulation services refer to the regulation of water flows on earth surface for maintaining the normal conditions in the watershed, while the water supply service is related to the infiltration, retention and storage of water in streams, lakes and aquifers (De Groot et al. 2002). The "Seasonal Water Yield" tool of InVEST model was used to analyze the water budget of RNPB. This model computes spatial indices that quantify the relative contribution of a land parcel to the generation of both base flow and quick flow, integrating the calculation of punctual water yield with the position of each pixel on the landscape (Sharp et al. 2016). The model requires a set of spatially explicit inputs, describing climate, soil, topography and biophysical factors (Table 1). It provides outputs for local recharge, actual evapotranspiration and run-off, computed on an annual time scale but using monthly values. Moreover, the map of "SCS runoff Curve Number" (CN) was processed in ArcGIS 10.3 (ESRI) to obtain the contribution of soil to absorb excess water, calculated as following (NRCS 1986):

$$SWP = \frac{25400}{CN} - 25.4$$

where SWP is the potential maximum soil moisture retention after runoff begins (mm) and CN is the curve number (dimensionless) for the specific for hydrological soil group and LULC type. The SWP can be a suitable proxy for water flood mitigation (e.g. Chen et al 2014).

Results

LULC changes

The LULC composition of the reserve suffered significant changes during time even after the PA classification decree (Table 3). During 1967, the landscape of NRPB was predominantly devoted to agricultural activities, where natural environments were almost absent. Pastures were located mainly in the central areas of the reserve and accounted for 18.9% of the total area. Rice fields were present on a large part of the naturally flooded areas, and irrigated crops were the most abundant LULC class.

After the PA classification decree, a permanent wetland was created to provide new habitats for aquatic and terrestrial species. This led also to the development of riparian forest ecosystems surrounding the wetland and along the course of the Almonda river, as evident in 1990 map (Fig.3). However, agricultural activities maintained with the growing interest and spread of maize crops.

In 2015, due to the management decision of acquiring property, permanent aquatic habitats covered the 8.2% of the total area as results of protection initiative aiming to the conservation of aquatic species. Many of these ecosystems were recorded as a valuable habitat according to 92/43/EEC Directive, including those corresponding to the habitat codes 3130, 3150, 3260, 3280, 3290 (Table S2).

Riparian forests increased up to cover the 17.3% of the NRPB in 2015, leading to the classification of three riparian habitats (no. 91B0, 92A0 and 9240). Mediterranean tall humid grasslands were also restored (no. 6420) together with pastures activities. Agricultural activities suffered a strong simplification with the complete loss of rice fields, non-irrigated crops, orchards and vineyards. Corn monocultures (irrigated crops) currently cover more than half of the NRPB.

Concerning the BUS scenario of LULC changes in 2050, the irrigated crops were hypothesized to spread again up to 65.2% of the total area. Inside the area under total protection, permanent wetlands would be maintained, and young riparian forests would evolve to mature stages. Poplar stands would be planted over all the north-west part of the NRPB.

The projection performed according to the NAT scenario of LULC changes in 2050 would lead to complete agricultural abandonment except for few hectares covered by olive groves. The abandoned cropland would evolve to Mediterranean grasslands, which would cover the larger part or the Reserve. This scenario would lead to the maximum coverage with previously mentioned habitats included in 92/43/ECC Directive.

Table 3. LULC patterns for 1967, 1990, 2015 and the two LULC scenarios to 2050. Values are expressed in both absolute (ha) and relative (%) values.

	1967		**1990**		**2015**		**2040 BUS**		**2040 NAT**	
LULC type	**Area**	**%**	**Area**	**%**	**Area**	**%**	**Area**	**%**	**Area**	**%**
Irrigated crops	310.25	38.0%	347.25	42.5%	417.74	51.2%	532.56	65.2%	0.00	0.0%
Aquatic	3.48	0.4%	52.55	6.4%	66.83	8.2%	66.83	8.2%	66.83	8.2%
Eucalyptus plantation	0.00	0.0%	4.24	0.5%	1.10	0.1%	1.10	0.1%	0.00	0.0%
Oak forest	31.27	3.8%	27.80	3.4%	30.99	3.8%	26.86	3.3%	20.64	2.5%
Pasture/grassland Past	154.15	18.9%	5.29	0.6%	124.00	15.2%	34.83	4.3%	423.83	51.9%
Poplar plantation	11.50	1.4%	0.00	0.0%	14.76	1.8%	40.11	4.9%	0.00	0.0%
Young riparian forest	0.00	0.0%	86.09	10.5%	140.88	17.3%	10.39	1.3%	145.69	17.8%
Urban	9.92	1.2%	9.32	1.1%	8.49	1.0%	8.49	1.0%	8.49	1.0%
Olive groves	80.56	9.9%	12.19	1.5%	11.77	1.4%	11.77	1.4%	10.16	1.2%
Rice cultivation	173.87	21.3%	179.80	22.0%	0.00	0.0%	0.00	0.0%	0.00	0.0%
Orange cultivation	9.99	1.2%	3.21	0.4%	0.00	0.0%	0.00	0.0%	0.00	0.0%
Non-irrigated crops	20.06	2.5%	1.78	0.2%	0.00	0.0%	0.00	0.0%	0.00	0.0%
Figs cultivation	11.49	1.4%	0.16	0.0%	0.00	0.0%	0.00	0.0%	0.00	0.0%
Vineyards	0.00	0.0%	39.10	4.8%	0.00	0.0%	0.00	0.0%	0.00	0.0%
Oak and Pine forest	0.00	0.0%	0.44	0.1%	0.00	0.0%	0.00	0.0%	0.00	0.0%
Oak and young riparian forest	0.00	0.0%	12.88	1.6%	0.00	0.0%	0.00	0.0%	0.00	0.0%
Oak and holm oak forest	0.00	0.0%	0.38	0.0%	0.00	0.0%	0.00	0.0%	0.00	0.0%
Mature riparian forest	0.00	0.0%	0.00	0.0%	0.00	0.0%	83.60	10.2%	140.91	17.3%
Sclerophyllous vegetation	0.00	0.0%	34.04	4.2%	0.00	0.0%	0.00	0.0%	0.00	0.0%
	816.5	100%	816.5	100%	816.5	100%	816.5	100%	816.5	100%

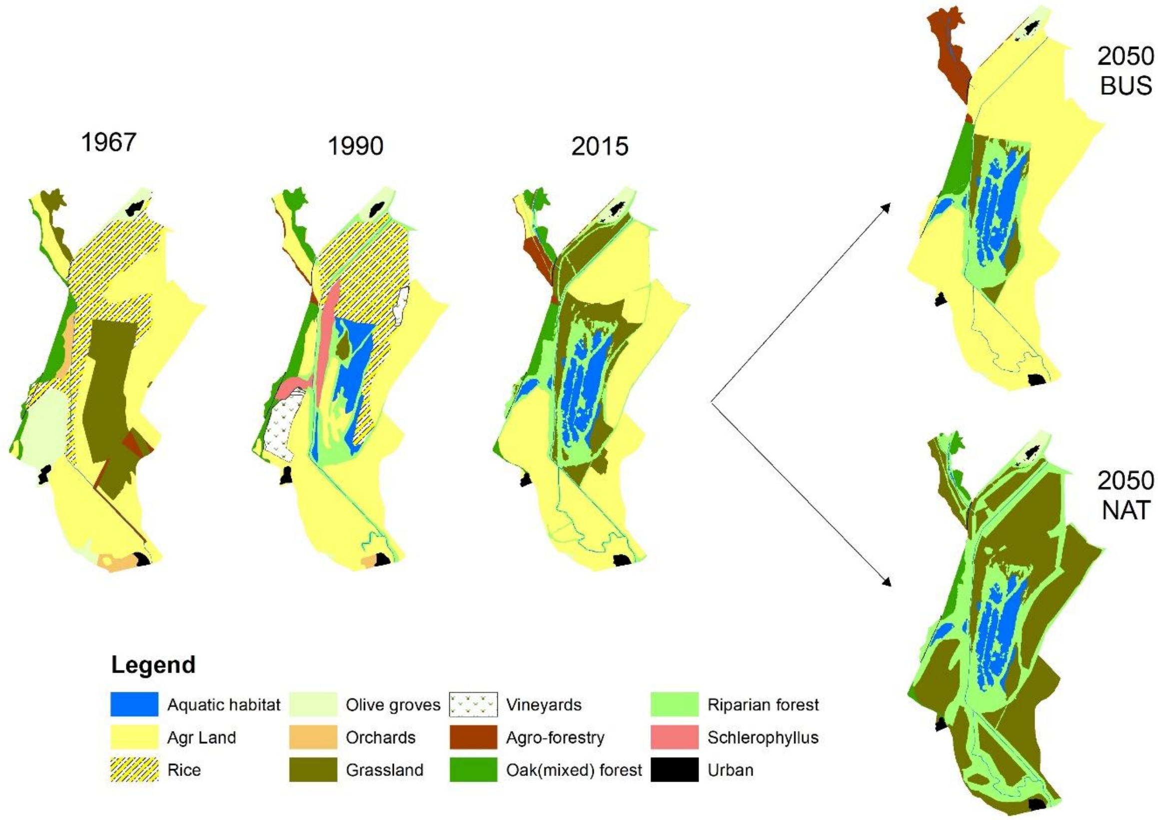


Fig.3. LULC maps for 1967, 1990, 2015 and for two future LULC (BUS and NAT) scenarios to 2050.

Climate regulation services

The amount of carbon increased in time as a consequence of LULC changes occurred between 1965-2015 (Table 4) (Fig.4). The period 1967-1990, during which the protection initiative started, unfold a large gain (+10.5%) of carbon stored into the ecosystems, despite the creation of wetlands in the core area. During 1990-2015, an additional lower gain was estimated (+2.9%). In this period, the further increase of wetlands and irrigated crop areas was overcompensated by expansion of riparian habitats. Among the LULC classes, irrigated crops were the larger contributor to carbon sequestration for all the three considered dates.

Gains of C amount were also predicted in both future LULC scenarios (BUS and NAT) without climate change effects. Surprisingly, even the transition from 2015 to 2050 according to the "Business" scenario" would lead to a considerable C increase (+6.9%). Despite the expansion of irrigated crops and the loss of riparian forests, the latter would switch to a mature stage (i.e. transition from "young riparian forest" to "mature riparian forest"), sequestering high amounts of C from the atmosphere. On the other hand, the transition according to the NAT scenario would lead to even larger carbon stock gain (+18.6%), with riparian forests and grasslands being the largest contributors.

The hypothesized net present values of future carbon sequestration (2015-2050), based on social cost of carbon emission, were assessed to be 313,845 US$ and 848,577 US$ for the BUS and NAT scenarios in total, respectively. This means that a value range of 534,732 US$ will depend on the selection of future LULC management plan (BUS or NAT) of the area.

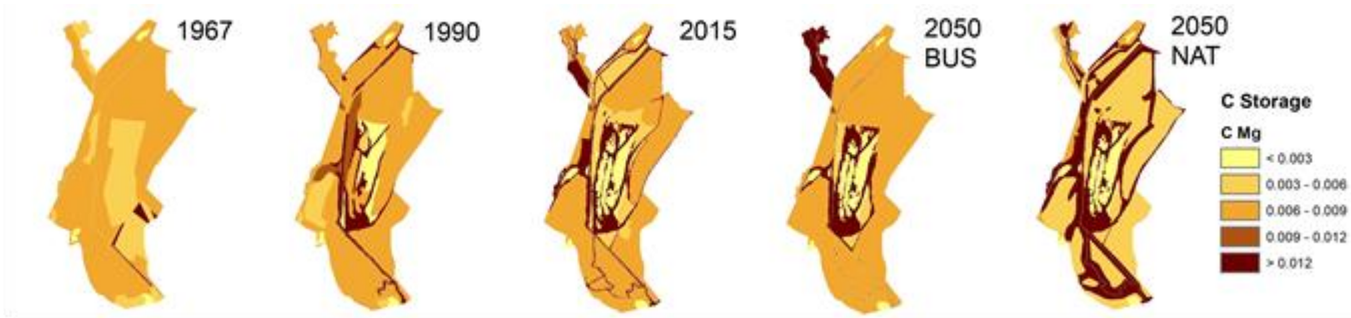


Fig.4. Carbon Storage modeled in NRPB for the different dates and scenarios.

Table 4. Carbon storage and sequestration in NRPB in the dates considered. The Net Present Value was calculated only for future scenarios as described in Section 2.3.1.

	Tot C (Mg)	**[2]Sequestered C (Mg)**	**[3]Mean Annual Seq C (Mg yr^{-1})**	**[4]Net Present Value (US$)**
1967	51239.0			
1990	56602.4	5363.4	233.2	
2015	58244.3	1641.9	65.7	
[1]2050 BUS	62244.5	4000.2	160.0	313845.4
[1]2050 NAT	69060.1	10815.8	432.6	848577.6

[1]Without considering climate change.
[2]Sequestered carbon in comparison to the previous date. 2050 scenarios are compared with the 2015 scenario.
[3]Sequestered carbon between two dates divided by the number of years.
[4]Net Present Value is estimated only for 2050 scenarios in comparison to 2015.

Water regulation and supply

The effects of LULC changes on water budget distribution was evaluated for each date (1965, 1990, 2015) of the period 1965-2015 (Fig.5). Actual evapotranspiration AET and runoff were observed to be the most and less contributing water budget components, respectively, as expected in a flat and dry environment. However, the LULC changes that occurred over time slightly affected the water budget (Table 5) but with a decreasing trend of recharge and runoff accompanied by an increasing trend of AET in time, due the respective increase of SWP indicating a better water retention and a better exploitation of water resources for covering evapotranspiration requirements. In fact, despite the loss of terrestrial areas, estimated values of total SWP increased, as a consequence of the larger contributions of forests rather than arable land.

Considering the two LULC scenarios of 2050 (BUS and NAT) without climate change (Table 5, Fig.5), it is observed that BUS scenario presents similar hydrologic responses (recharge, AET, runoff, SWP) with the LULC responses of 1990 and 2015 LULC conditions. On the other hand, the NAT scenario shows an evident decrease of recharge and runoff and increase of AET due the evident increase of SWP indicating that the NAT LULC conditions lead to even higher water retention and much better exploitation of water resources for covering evapotranspiration requirements.

Considering the two LULC scenarios of 2050 (BUS and NAT) with climate change (Table 5, Fig.6), it is observed reduction of -75.5% in recharge, -0.8% in AET and -35% in runoff for the BUS scenario and reduction of -98% in recharge, -2.2% in AET and -37.6% in runoff for the NAT scenario due to climate change. The projected -16.1% decrease of annual precipitation for 2050 according to rcp85 is the reason for the aforementioned reductions and especially for recharge and runoff. In the case of NAT scenario, the reduction of recharge and runoff is even higher compared to BUS due to the higher SWP.

Table 5. Modeled water budget and *SWP* in NRPB. Values are expressed in mm.

	Recharge	**Actual ET**	**Runoff**	**SWP**	**Precipitation**
1967	145.0	527.5	40.1	278.9	712.6
1990	139.5	537.8	35.3	287.0	712.6
2015	122.7	557.0	32.9	290.6	712.6
[1]2050 BUS	129.1	546.8	36.6	292.4	712.6
[1]2050 NAT	97.3	598.0	17.3	317.3	712.6
[2]2050 BUS	31.6	542.3	23.8	292.4	597.7
[2]2050 NAT	2.0	584.9	10.8	317.3	597.7

[1]Without considering climate change.
[2]Considering climate change.

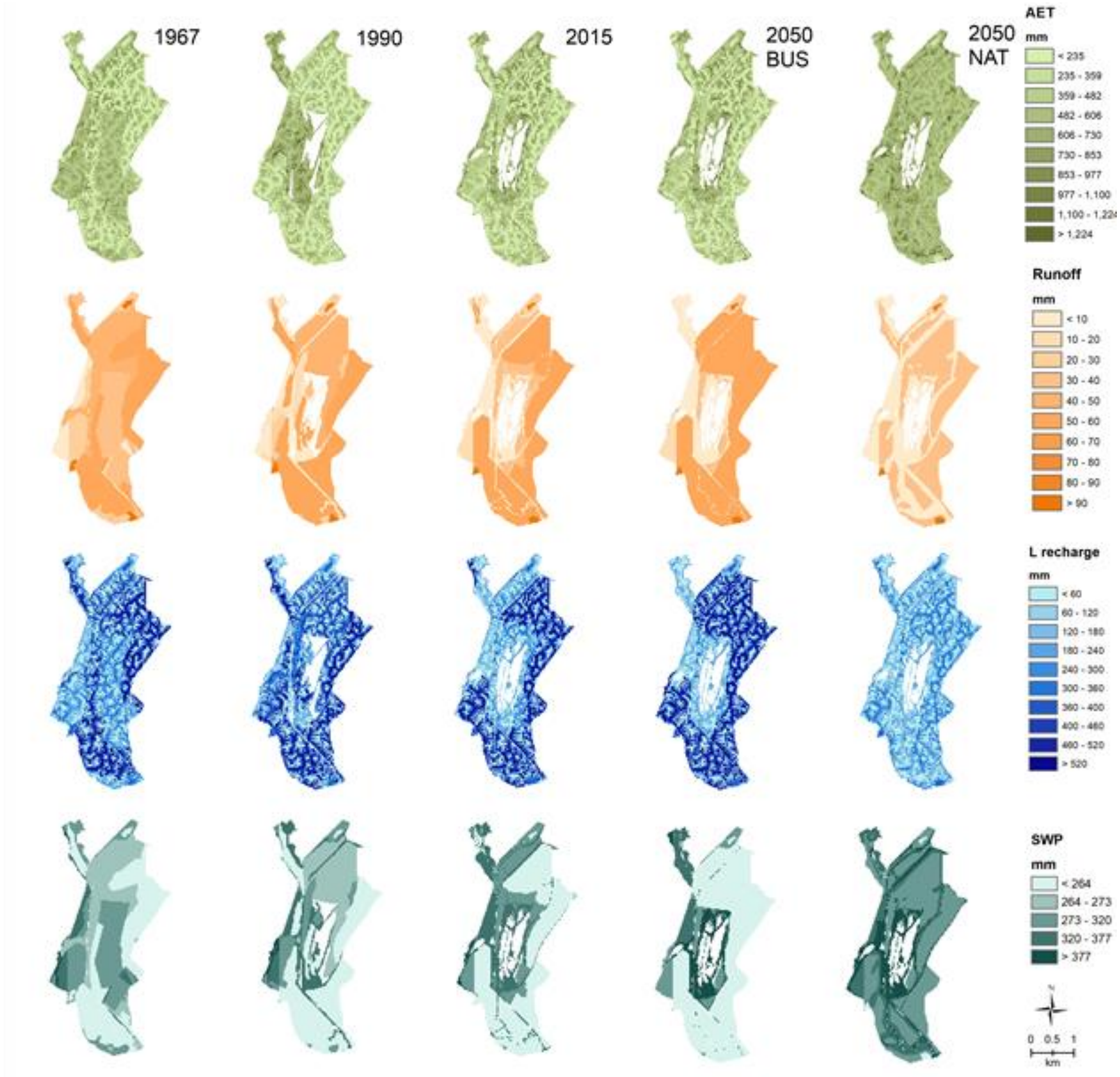


Fig.5. Water budget components and SWP modeled in NRPB for the different dates and scenarios (climate change is not considered).

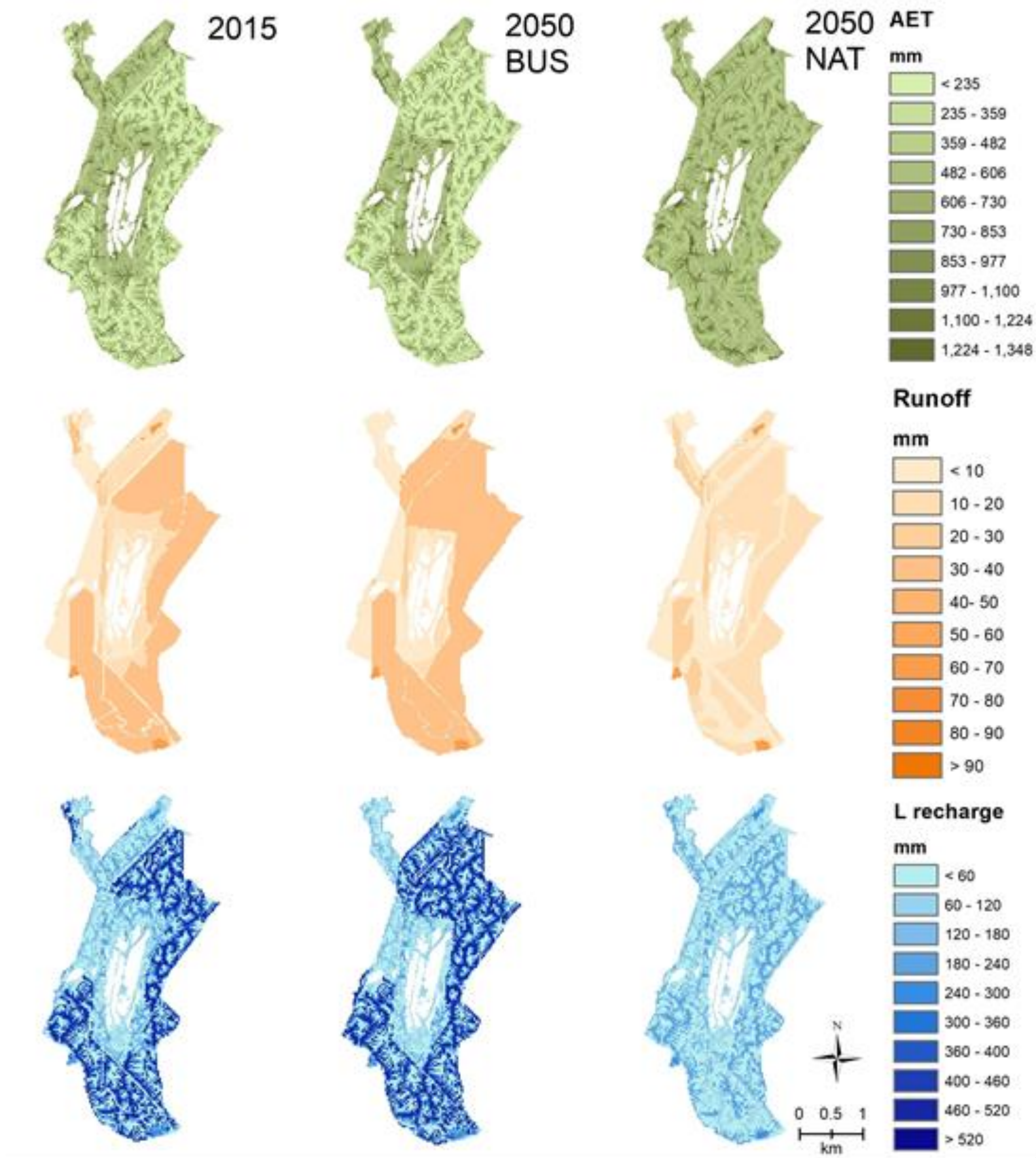


Fig.6. Water budget components modeled in NRPB for the 2015, Business and Naturalization scenarios to 2050 considering climate change effects.

Discussion

Possible conflicts of ESs supply

The identification of conflicts and synergies among ESs gained an increasing interest in literature during recent years, under the lens of environmental and PAs management. For example, Palomo et al (2014) highlighted a positive contribution of Doñana National Park on maintaining ESs delivery, because of the isolation of PA with respect to the LULC changes which occurred outside. Onaindia et al (2013) described the spatial congruence of biodiversity, carbon storage and water flow regulation, arguing that biodiversity conservation would ensure the provision of regulation services by the maintenance of native forest ecosystems. Synergy between biodiversity and carbon storage was confirmed in many other studies (e.g. Steinbeiss et al 2008; Williams et al 2008; Hall et al 2012; Peh et al 2016; Vergílio et al 2016), to such an extent that some high-biodiversity areas could be protected by carbon-based conservation strategies (Strassburg et al. 2010). Moreover, a relatively high overlap between carbon and soil-ESs was detected in literature (Izquierdo and Clark 2012; Gissi et al. 2014; Gissi et al. 2017). For example, the carbon stored in aboveground biomass was positively correlated with both water supply and water regulation in Rodríguez et al (2015). Flood prevention seems positively correlated with biodiversity conservation (Egoh et al. 2009; Newton et al. 2012) and carbon storage (Chan et al. 2006), even though correlations values are generally low. On the other hand, conservation efforts may lead to controversial outcomes. Castro et al (2015) found that the higher levels of protection do not ensure the provision of multiple ESs. Negative outcomes were detected in deltaic environment (Gaglio et al. 2017a) and tropical mountain areas (García Márquez et al. 2017; Gaglio et al. 2017b). Izquierdo and Clark (2012) observed slight negative correlation between water yield and biodiversity, as well as with carbon storage.

This study demonstrated that conservation actions affected the ESs provision in the NRPB over time during 1967-2015, while the future designed LULC scenarios for 2050 with and without climate change effects revealed that management decisions have important effects on ESs supply. The ESs analysis highlighted that the efforts for biodiversity conservation through PAs can lead to parallel positive responses in terms of climate mitigation and flood prevention in floodplains such as NRPB. However, some conflicts may arise in water regulation services (Carvalho-Santos et al. 2016a; Carvalho-Santos et al. 2016b). As indicated by Table 5, all the LULC conditions after 1965, including the future scenarios, led to reduction of recharge and runoff and increase of water retention, which is desirable for floodplains and vulnerable environments to floods in general. On the other hand, respective LULC changes may have a negative impact in places where groundwater resources are limited or minimum ecological flows in surface waters are difficult to guarantee. These observations are in line with the findings of Quintas-Soriano et al (2014), who described that non-protected areas are more relevant providers of groundwater recharge service in comparison to the protected landscapes. Conversely, the increase of real evapotranspiration due to the increase of water retention may boost the biomass production and consequently the provisioning services (e.g. food and timber production) counterbalancing the economic loss from water regulation services. On the other hand, even this pathway of ESs exploitation is not a secure plan since AET was slightly reduced in the of BUS and NAT scenarios considering the climate change effects.

Overall, the analysis of ESs trends in NRPB revealed that the mere biodiversity conservation may not be effective in maintaining some ESs and that an active management is required. Future management of NRPB should consider the relationships among ESs raised in this analysis and the role of conservation strategies. If only biodiversity conservation is fostered (e.g. NAT scenario), the contribution of the Reserve to climate change mitigation and the capacity to buffer flooding will increase, though this would cause a decrease of water supply (especially if rcp85 will occur) with unpredictable consequences on the preservation of the aquatic environment. Contrarily, the maintenance of a limited extension of agricultural land and pastures on the NRPB, managed with non-intensive practices, could be an ecological solution to maintain a balanced water budget. This result is in line with the findings of Felipe-Lucia and Comin (2015), who demonstrated that a mosaic of different LULC types can support ESs and biodiversity in agricultural floodplains.

Special consideration should be given to the riparian ecosystems of NRPB. Particularly, mature riparian forest provide high values of evapotranspiration and SWP, as well as carbon storage, soil conservation and nutrient retention (Dindaroğlu et al. 2015). On the other hand, such forest transitions in lowland areas may cause a dramatic decrease of local recharge capacity. Therefore, the management and regulation strategies concerning these ecosystems are fundamental for the provision of ESs at regional scale (Garrastazú et al. 2015; Sáenz et al. 2016; Vidal-Abarca et al. 2016).

As far as climate change is concerned, it was observed that decision making in land use will be challenging when considering future climate projections. When conservation measures include the total renaturing of the Reserve, the trade-off between water infiltration and flood prevention capacity results were exacerbated. These findings are confirmed also by other studies. In fact, increase of flood attenuation potential of floodplains under climate change was predicted also by Moor et al (2015). Other studies carried out in Mediterranean basins evaluated that water supply service may be reduced up to 49% (Bangash et al. 2013) or almost 100% (Terrado et al. 2014).

Limitations of the modelling approach

Even though spatially explicit models are probably among the finest tools to map ESs provision, their application presents some limitations, generally related to both the reliability of input data and to the simplifications that lie under the assumptions on which models are based on (Bagstad et al. 2013; Hou et al. 2013). Moreover, when models are based on a low number of input parameters, their sensitivity to these factors can be high, especially when data is obtained from national inventories, which represent desirable sources when no field data is available. On the other hand, national inventory datasets may not capture adequately the variability in fine spatial scales.

Additionally, some model simplifications are necessary to be accepted when describing very complex phenomena, such as those that regulate ecological functioning. For instance, the "Carbon Storage and

Sequestration" model assumes that none of the LULC types gains or loses carbon over time or under the effects of climate change. The only changes in carbon storage captured by the model over time are those due to changes from one LULC type to another. As suggested by InVEST 3.3.3 users'guide, this problem can be at least partially addressed classifying some LULC types into age classes, as it was made in this work. Similar problems have been observed in other studies (Bottalico et al. 2016).
Similar limitations appear also in the case of hydrological parameters (e.g. crop factors, soil hydrologic groups), which are necessary in the "Seasonal Water Yield" tool. Crop factors of the same crops/land uses, which are given in various databases (e.g. database of InVEST model or FAO), may differ significantly from place to place either due to local soil-climate conditions or due to different cultivation practices (Aschonitis et al. 2017a). Additionally, in the case of soil hydrologic groups, it is not taken into account the evolution in soil conditions from the LULC changes.

Conclusions

The present study highlighted the relevance of conservation initiatives in ESs provision and the significance of modeling approaches for their explanation. The spatially explicit assessment of ESs using modeling approaches like InVEST can inform and support managers on environmental decisions considering trade-off between ESs that are strongly associated to the human well-being of local populations. From the results of the study, it was concluded that floodplain riparian habitats like NRPB display a high capacity for the delivery of multiple ESs. The limitation of human activities, such as agriculture and grazing, could be desirable to optimize the delivery of ESs, which are related to carbon and water retention. On the other hand, significant conflicts between specific water regulation services may appear by the application of PA initiatives. Possible reductions of water recharge to groundwaters and runoff, such as those observed in the specific study, may not be desirable in places where groundwater resources are limited or minimum ecological flows in surface waters are difficult to be maintained. Thus, the use of modeling approaches to assess complex ESs interactions taking also into account the possible effects of climate change can be a valuable tool for the description of ecological mechanisms that underlie trade-offs and synergies among ESs. This is particularly important when considering protected areas, where the assessment of the services delivered is urgently needed in order to provide arguments for biodiversity conservation.

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3.4. "Soil-related ecosystem services trade-off analysis for sustainable biodiesel production"

This section presents an application of ES mapping to improve the sustainability of biofuel supply chain in agricultural land of Veneto region (Italy), with the aim to fill the current gap between certification schemes and environmental assessment. ES mapping was carried out using logical indicators at regional scale. The proposed instrument estimated that the achievement of regional target would lead to significant impacts on ES supply. This case study provides example of how change in land use intensity can harm ESs. The presented tool can be applied to support decision-makers towards a more sustainable management of bioenergy provision.

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Abstract

Certification schemes are meant to verify that biofuel production from renewable sources is sustainable, especially for land-related sources such as oilseeds. Nevertheless, current schemes are not able to depict the locally based negative effects of intensive feedstock cropping, especially those affecting soil properties and functions. The aim of this study is to quantify and map the amount of potential sustainable biodiesel production from oilseed feedstocks (soybean, sunflower and rapeseed stocks), which is intended to avoid harming the provisioning of other soil-related Ecosystem Services (ESs). The methodology is based on the analysis of trade-offs between the current provision of soil-related ESs (habitat for soil organisms, soil carbon storage, groundwater quality protection and food provisioning) with feedstock production for biodiesel. This method is tested on current land uses that are devoted to intensive agriculture in the Veneto region plain (Northern Italy). The results for the potential sustainable biodiesel production are 20.7 dam^3 per year, which would be able to cover only 52% of the regional target forecasted for the year 2020. The areas that are currently exploited for other annual crops (primarily cereals and maize) exhibited a significant potential contribution that would greatly exceed the regional target (if exploited). This finding indicates that the achievement of the regional target will be

impossible without producing significant impacts on the ES provisioning capacity or causing indirect land use changes. The proposed methodology could provide a tool to improve biodiesel certification schemes and improve the effectiveness of strategic environmental assessments of regional energy.

1. Introduction

Soil is considered an important natural resource that, because of its properties and multiple functions, contributes to the provision of several Ecosystem Services (ESs), such as food, erosion protection, and carbon storage [1–5]. Soil biodiversity is a key aspect that influences processes and functions necessary for the provision of many ESs [6]. Increasing attention has been devoted to soil management, because it can affect the provision of multiple ESs in agro-ecosystems [7]. Soil management practices include the combination of tillage, fertilization and farming practices [7–9]. In Europe, soil is considered as a non-renewable natural resource, and has become the subject of protection according to the Soil Thematic Strategy [10]. Moreover, the Seventh Environment Action Programme, which has been in force since January 17th, 2014, implies that Member States should increase efforts (i) to reduce soil erosion, (ii) to increase soil organic matter, and (iii) to remediate contaminated sites.

As a non-renewable resource, soil is increasingly under stress because of urbanization and agricultural intensification [5,11,12], climate change and other external drivers [4,7]. Specifically, a source of prominent pressure to soil is the increasing production of biomass for bioenergy, which may conflict with the provision of other ESs delivered with the contribution of well-maintained soils [13,14]. Possible effects of the biofuels supply chain may relate deforestation [15], competition with food production [19] greenhouse gas emissions and soil carbon storage [18–20], land use change [13,15,21], soil degradation [22], and biodiversity loss [15,23,24]. Moreover, increased water consumption [25] and air/water pollution [13,15] can have indirect effects on soil characteristics and its qualities can collectively affect “the capacity of a soil to function, within ecosystem and land use boundaries, to sustain productivity, maintain environmental quality, and promote plant and animal health” [4,26].

According to the EU Renewable Energy Directive (EU RED) 2009/28/EC [27] on the promotion of the use of energy from renewable sources, biofuels are considered a valuable fuel option that can support Member States in meeting the 10% renewable sources target for transport fuels by 2020. The primary sustainability criteria concerning biofuels are related to (i) greenhouse gas emission savings related to the entire lifecycle (from feedstock production to biofuel consumption); (ii) the maintenance of land with a high carbon stock; and (iii) the preservation of land with high biodiversity [27].

With respect to certifying the sustainability of feedstock production, as reported in COM 2010/C 160/02 [28], voluntary schemes and bilateral/multilateral agreements are valuable tools to support local supply and rural development. Such schemes can have positive implications in supporting the sustainability of biofuels [29–31], connecting the feedstock supply to local production and supporting the innovation and development of the agro-food industry in Europe [32,33].

Under the EU RED [27], the European Commission has established some minimum requirements with respect to the sustainability of biofuel feedstock production. Biofuels and bioliquids used in the EU must conform to specific sustainability criteria if they are to be counted towards the national renewable energy targets established by EU RED [27] and to access supporting policies (and related funds) [31,33].

Soil quality receives very different recognition among the 19 certification schemes approved by the European Commission (source: https://ec.europa.eu/energy/en/topics/renewable-energy/biofuels/voluntary-schemes). Table A1, in the Supplementary Electronic Material, shows how soil protection is considered in the certification schemes that are actually approved by the European Commission. All certification schemes account for soil contributions in GHG emissions because they are called to comply explicitly with the methodology for GHG evaluation given in Annex V.C of the EU RED [27], and its follow-ups [28,34]. In the same way, cross-compliance with good agricultural practices is considered in every certification scheme.

However, the detailed monitoring and related audit of soil protection and erosion control, soil organic matter, and soil biological, chemical and physical conditions are mentioned only in two schemes, the International Sustainability and Carbon Certification (ISCC) and the Roundtable on Sustainable Biomaterials (RSB), in which a soil management plan is considered as valuable but not compulsory. RED CERT and the Round Table on Responsible Soy EU RED (RTSR) consider soil quality indicators. Solomon and Bailis [35] acknowledge that, when it comes to soil, the standards of certification schemes vary in scope, ranging from general principles to specifications in land management and tillage practices. From their review, cross-compliance and certification as a formal procedure emerge as the primary approaches to assessing sustainability in feedstock production.

In general, certification schemes suffer from the unresolved challenge to having to apply common and harmonized standards to respond to local environmental characteristics [31,36], while at the same time reconizing that it is necessary to consider local practices and physical environments [37,38]. For example, the RSB certification scheme foresees a possible adaptation to specific "political, legal, customary and/or technical social, environmental, cultural, ethical and/or economic conditions in a particular geographic region" [33]. As a result the effectiveness of certification schemes often depends on the local environmental and sectorial characteristics [37,38], considering that sustainable bioenergy systems are embedded in unique social, economic, and environmental contexts [39]. Moreover, cross-compliance with environmental sustainability criteria (exclusively applied to biomass produced in the EU) is accounted for only through the formal verification of meeting pre-established regulations. There is no on-site verification of impacts related to feedstock production [36,40], in relation to the preservation, maintenance and enhancement of soil properties and quality. Moreover, certification schemes appoint feedstock producers individually, at farm level [41,42]. In this respects they cannot account for the possible cumulative effects of feedstock production, or even exclude “considerations of indirect land use change and social and environmental impacts above farm or plantation level” [31,36].

This study applies an ecosystem services-related approach to quantify the biodiesel potential of biomass production. This analysis hypothesizes that the current oilseed production is entirely destined for biodiesel production without any land use change. It is assumed that feedstock for biodiesel production (oilseeds) is one of the soil-related ESs because it depends on primary productivity, as stated by the Common International Classification of Ecosystem Services [43].

These main objectives of the study include:

i) quantify the fraction of current oilseed production that can be considered as environmentally sustainable for biodiesel production with respect to soil-related ESs; and

ii) identify potential areas among those currently used for annual crops (e.g., cereals, maize, beetroot, tobacco and others) that might be converted for biofuel feedstock production (oilseeds), while avoiding significant trade-offs with soil-related ES.

The analysis is performed in different steps. Firstly, the potential biodiesel production is calculated from the current oilseed production. Areas of current feedstock production and related theoretical potential are compared with their current capacity to provide other soil-related ESs such as i) habitat for soil organisms (supporting services), ii) soil carbon storage (regulating service), iii) groundwater quality protection (regulating service) and iv) food crop (provisioning service). After these soil-related ES are mapped individually, a pairwise trade-off comparison is performed between soil-related ES and biodiesel feedstock production. These trade-offs are identified by considering the current agricultural practices for feedstock cultivation as identified in literature. Subsequently, pairwise trade-offs is undertaken to identify the areas characterized by different trade-off severities with biodiesel feedstock production.

The results are discussed with respect to the gaps in biodiesel sustainability certification, and the framework of energy planning for the Veneto region. Regional renewable energy targets for the year 2020 are derived from the burden sharing of national targets, which is in line with the EU RED [27]. These targets are verified with respect to the biodiesel energy potential deriving from current sustainable oilseed production.

2. Materials and Methods

2.1 Study site

The study site is the Veneto plains region (Fig. 1), which has a surface area of 10,311.91 km^2 (56% of the regional total surface) and is part of the soil region of the “Po plain and moraine hills” [44]. The Veneto region contains intensively managed agricultural areas and dense urbanized areas. The main oilseed produced are soybeans (*Glycine max* L.), sunflowers (*Helianthus annuus* L.) and rapeseed (*Brassica napus* L.). It is characterized by quaternary alluvial and glaciofluvial deposits, with an average slope of 1% and an altitude that varies from 70 m on the mean sea level, down to 0 m at the coast of Adriatic Sea in the southeast. This soil classification is relevant when considering the biomass yield conversion parameters, which are homogeneous in areas with the same climatic conditions. In this study, yield conversion parameters are calculated for territories overlapping with the administrative boundaries of the Provinces delineated by ISTAT [45]. These provinces are intended to represent areas with the same climatic conditions, which is consistent with the study of bioenergy potential by Motola et al. [46].

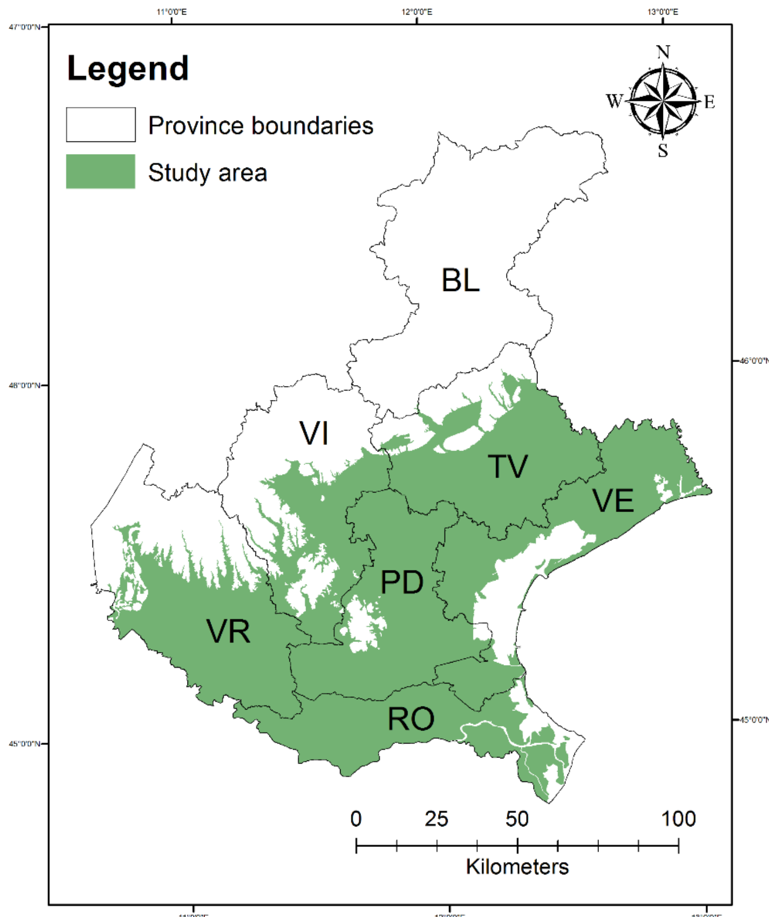

Fig. 1: Location of the study area within the Veneto Region. Province boundaries are highlighted: BL = Belluno, TV = Treviso, VI = Vicenza, VR = Verona, VE = Venezia, PD = Padova, RO = Rovigo

The major soil degradation processes in the Veneto plain region are related to urbanization and intensive agriculture due to the high potential agricultural productivity (Fig. A1). Of the total area of the region (10,311.91 km^2), only 7,847.35 km^2 (76%) were considered in the analysis (Table 1). Artificial surfaces, natural areas, wetlands and water bodies (i.e., land classes 1, 3, 4, and 5 in the first level of the CORINE

Land Cover Classification) were not considered in the analysis. Thus, the potential trade-offs due to indirect land use change [47–49], were excluded from this study. The crop types and areas under oilseed crops (i.e. soybeans, sunflowers and rapeseed) and of other arable land were derived from the Land Cover Map provided by Veneto Region. The map outlines the land cover on five levels, adopting the CORINE Land Cover nomenclature, at scale 1:10.000.

With respect to biofuels, Italy produced approximately 1.2 x 10^3 dam^3 of biofuels in 2014 (99% was biodiesel, and 99.8% was certified as sustainable), primarily for domestic consumption, with approximately 20.68% of the Italian production capacity being located in the study site of the Veneto region [50]. It is worth noting that only 8% of the sustainable biodiesel consumed in Italy in 2014 was produced with domestically produced feedstock. Feedstock was primarily imported from other European countries (47%), with the remaining 53% coming from developing countries outside of the EU (of which 46% from Indonesia) [51]. Palm oil, largely from Indonesia and Malaysia, is the primary raw material for biodiesel production in Italy (47%), followed by rapeseed oil (27%) and soybean oil (6%) [51].

Table 1: Land use classes and their extent.

Land use	Crop types	Surface (km^2)
Soybean	Soybean	910.15
Sunflower	Sunflower	37.86
Rapeseed	Rapeseed	254.81
Other arable land	Cereals, maize, beetroot, tobacco and other arable land in general	5052.78
Not available	Greenhouses, horticulture, orchards, nurseries, complex cultural systems established by law, perennial crops in general, rice, vegetable gardens	1591.74
Total		7847.35

2.2 Trade-off mechanisms between soil-related ES

This study assesses the impact of biofuel feedstock production on soil-related ESs depending on ecological variables such as soil characteristics and soil hydrological conditions. It expands the framework proposed by Gissi et al. [40], that defines the sustainable potential as "the fraction of energy potential whose exploitation causes no harms to other ES delivered by the sources of renewable energy" (p. 2). In the present analysis, the aforementioned definition is applied to the production of biodiesel feedstock from rapeseed, soybeans and sunflowers.

The Common International Classification of Ecosystem Services (CICES) [43] has recently appointed Biomass-Based Energy Sources (BBES) as a provisioning ES, given that agro-ecosystems can provide suitable biomass for bioenergy production. In this sense, oilseed production can be interpreted as part of the BBES, since it represents the feedstock that is transformed into biodiesel.

Our analysis starts from the assumption that trade-offs may occur between ecosystems services when the provision of one or more services inhibits the provision of others [52,53]. Competing services can be provided by common drivers or biological dynamics that are somehow interrelated [52]. Agro-ecosystems can provide more ESs that are synergistic or complementary [54,55]. For example, some agricultural products involves raw materials that can be used for food (i.e., food crops); or feedstock for biofuel production (i.e., BBES).

Feedstock production for biodiesel is an ecosystem service deriving from primary production in agro-ecosystems and can therefore result in trade-offs with other services (e.g., [56]). These trade-offs can occur according to the following mechanisms: i) competition for land use, e.g., land diversion from food production and/or other uses to BBES feedstock production; ii) competition for the end-product, e.g. use of raw materials that are initially devoted to human consumption (as food) or breeding (as animal feed); iii) interference in biological dynamics providing other ESs, e.g., depletion of soil functions, as habitat biological characteristics, due to the intensive use of pesticides and fertilizers to support intensive feedstock production, or by the uptake of in-field residues that nourish the biological cycles of soils [57–59]; and iv) indirect effects due to the impacts of growing biofuel crops, e.g., the use of chemicals that threaten the groundwater quality in areas where soil characteristics make it vulnerable to nitrates.

Moreover, the severity of the trade-off can be classified according to the different types of relations between ESs as follows: i) the severity of the potential land use change (which is higher in the case of higher natural levels); ii) severity related to the capacity of potential substitute production (which is higher when there is a higher provisioning capacity for food, or when the production related to end-products is higher in terms of quality and revenues); iii) severity in the depletion of other services (which is higher when the ES provisioning is higher and the potential loss in ES provision is more severe); and iv) the intensity of the potential threat in the case of the negative interaction between ES, assuming that the trade-off is higher when the potential provisioning of mitigating effect from ES is lower).

2.3 Step-by-step methodology for the analysis of soil-related ESs

To assess the sustainable amount of biofuels production by BBES feedstock that does not affect other soil-related ESs, we follow seven methodological steps (Fig. 2):

- Step 1) select crop types and related areas within the study site to perform the analysis;
- Step 2) calculate the potential biodiesel production from BBES feedstock derived from current oilseed production, as if it was all devoted to biofuel production (under the hypothesis of different uses of the same end-product without any land cover change);
- Step 3) select and map of other soil-related ESs, which can potentially compete with BBES feedstock production;

Step 4) identify and map of pair-wise trade-offs between the production of BBES feedstock and other soil-related ESs, according to the three types of relationship explained in section 2.2;

Step 5) analize the combinations of tradeoffs between BBES and the other soil-related ESs, according to the combined severity of pair-wise relationships;

Step 6) calculate the sustainable potential production of biofuels derived from current BBES production that is not competing, interfering or interacting negatively with the other ESs currently provided; and

Step 7) identify areas for potential oilseed production, among the ones that are currently devoted to the intensive production of annual crops (primarily cereals and maize).

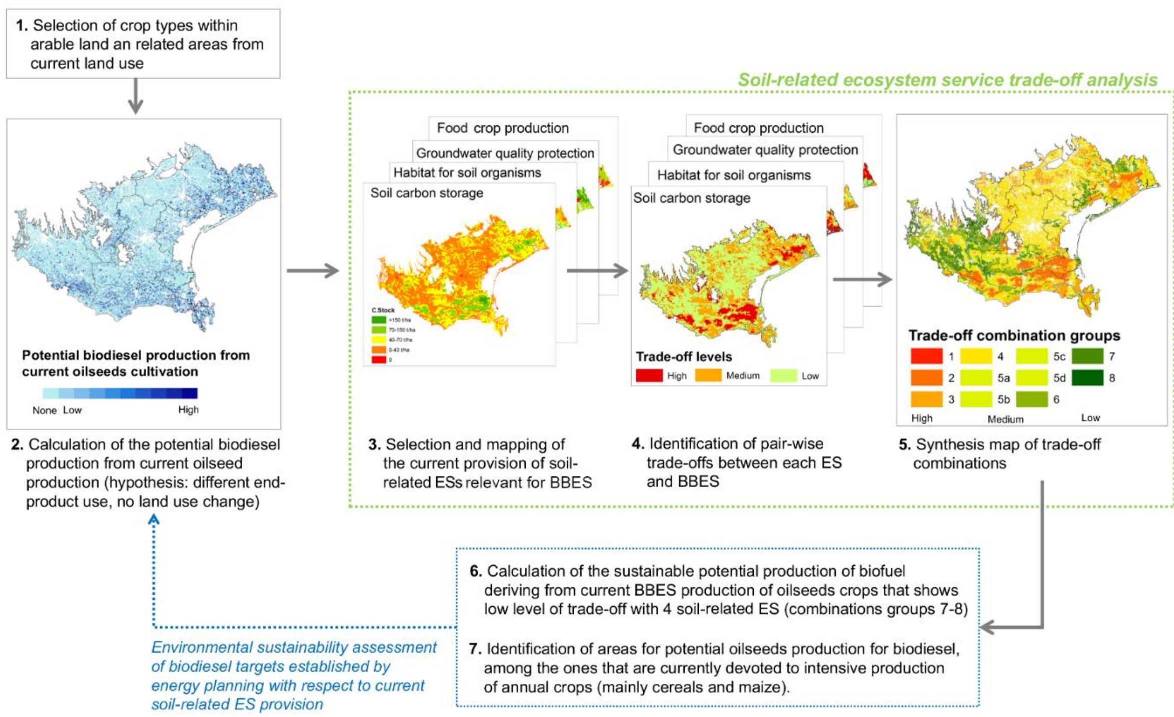

Fig. 2: Conceptual framework of the step-by-step analysis

In this study we consider that the change in the use of the final product (oilseeds) implies a change in agricultural practices to produce oilseed feedstock for bioenergy, instead that for food/feed. Specifically, “land cover” is defined as “the observed (bio)physical cover on the earth's surface”, while “land use” is defined “by the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it", in line with the definitions given by United Nations Food and Agriculture Organization (FAO) [60].

Based on the above definitions for Steps 1-6, we consider that no land cover change occurred. In other words, we assume that there was not any change from natural land cover to agriculture, nor any change in crop type. Instead, we consider that only land use change occurred. This is because we assume that oilseed feedstock production for bioenergy entails an intensification of agricultural practices when compared to oilseed production for food/feed. This is demonstrated by the fact that oilseed feedstock production for bioenergy implies i) shifting from traditional crop rotations to a continuous oilseed production [61,62] and ii) increasing fertilizers application to boost oilseed yields [63]. Even though farmers attempt to boost yields when growing oilseeds for food, oilseed production for bioenergy purposes requires their continuous and stable production to supply feedstock-processing facilities. This makes crop rotations and the possibility to switch towards organic farming practices practically impossible.

Considering the above, the intensification of agricultural practices for the production of biofuel feedstock can alter the provision of ESs, as has been observed in different food production systems around the world [64–66]. Thus to calculate the oilseed potential production (i.e. BBES) we only consider areas that are currently devoted to oilseed production for food, but under the assumption that agricultural practices in these areas will change in order to allow for the change in the final use of the oilseed (i.e. from food to bioenergy).

Step 1 extracts from the CORINE land cover map (i.e., an arable land class at the first level of the CORINE Land Cover classification) the crop types and related production areas within the case study site. This analysis was performed only for current land uses related to annual oilseed crops (i.e. rapeseed, sunflowers, soybeans) and other annual crops (e.g. maize, cereals), and excluded perennial crops and other types of cultivation. These were classified into five land use groups (Table 1). Artificial surfaces, natural areas, wetlands and water bodies (i.e., land classes such as 1, 3, 4, and 5 at the first level of the CORINE Land Cover Classification) were not considered. This excluded potential trade-offs from this study due to the competition between land uses.

Step 2 quantifies the potential biodiesel production in the study site. The potential feedstock production is defined as the fraction of the gross energy that can be harvested by the energy conversion system, assuming that all suitable crops are destined for oilseed production in accordance with legal and technological limitations [40,67]. Thus, the potential biodiesel production was calculated only from the current land use destined to oilseeds, accounting for the potential oilseed production by rapeseed, soybeans, and sunflowers in the Veneto region. The algorithm for calculating the potential biodiesel production (Eq. 1) is modified from [68] as follows:

$$BP_{ij} = Y_{ij} \cdot A_{ij} \cdot E_i \qquad \text{(Eq.1)}$$

where, BP_{ij} is the maximum energy per hectare that can be achieved for each oilseed crop type (i) (i.e. rapeseed, soybeans, sunflowers) within province j, Y_{ij} is the average yield of each crop type in each province (see Table A2 in Supplmentary Electronic Material), A_{ij} is the area of each crop type in each province as obtained by the Land Cover map (Veneto Region 2013) and E_i is the specific energy provision capacity, which is considered as the biodiesel yield, with specific values for each crop type (see Table A3 in Supplementary Electronic Material). Average yields for each crop type have been calculated using the ISTAT database for the time period between 2006 and 2015 [69]. Unlike Gissi et al. [40], this paper maintains the administrative domains of provinces to calculate the yields, and as a result crop yields vary between provinces due to local climatic and ecological conditions (Table A2 in Supplementary Eletronic Material).

Step 3 maps the main soil-related ESs in the study area. Initially, through a literature review, we identify those ESs that are most affected by the cultivation of biofuel feedstock (Table 2). In total four ESs are selected and mapped individually for the case study area, namely carbon storage (a regulating service), habitat for soil organisms (a supporting service), groundwater quality protection (a regulating service), and ood production (a provisioning service).

Table 2: Relationship between oilseed production for biofuel (BBES) and other soil-related ESs.

ES class	Ecosystem functions and processes	Ecosystem services	Trade-off type	Mechanism	Map resolution	Map sources	References
Supporting	Habitat provision	Habitat for soil organisms	Interference	Cropping systems affect soil biota communities	500m	[79] for Organic Carbon fraction, [80] for Bulk Density	[57-59]
Regulating	Soil buffering	Groundwater quality protection	Indirect impact	Increased tillage and agro-chemical application affect the quality of water bodies by increasing nitrogen leaching	1:250,000 (scale)	[94]	[89,90]
Regulating	Organic matter accumulation	Soil carbon storage	Interference	Annual crop production decrease the accumulation of soil organic carbon, particularly when crop residues are not retained in fields	1 km	[71]	[87,88]
Provisioning	Primary production	Food crop production	Competition in end-product use	Trade-offs between biofuel and food provision occur both through land conversion and competition for the use of final products	1:250,000 (scale)	[105]	[99] [102-104]

The indicators selected for to map the soil-related ESs reflect the context of the study and the geographical scale of policy questions. In more detail biofuel production (and its related targets) are managed at the regional scale (through the Regional Energy Plan of Veneto Region [70]), and operationally implemented at the provincial administrative level. Soil-related ESs are mapped in the case study area by considering the climatic and environmental characteristics of each province. The selected indicators for carbon storage, habitat for soil organisms, groundwater quality protection, and food production are extensively explained in Section 2.4.

Step 4 analyzes the potential conflicts between ESs and BBES across three trade-off levels (i.e. low, medium and high). The type of each trade-off level is identified for each area. To rank the trade-offs across the three levels, appropriate thresholds were defined (Table 3). These thresholds discriminate among a positive (low), medium (medium) or negative (high) pair-wise trade-off relationship between the BBES and each ES. For example, according to the Regional Agency for Environmental Protection of the Veneto Region (ARPAV) [71], areas that have soil carbon contents of >70 t ha^{-1} are recognized as having a high current capacity to deliver carbon storage ES. If such areas are used to produce biodiesel feedstock, then the loss of regulating ESs related to carbon storage will be more severe when compared to areas with lower soil carbon content (e.g. 40 t ha^{-1}).

Table 3: Thresholds for trade-off levels for each soil-related ES.

Ecosystem Services	Trade-off levels			References
	High	Medium	Low	
Habitat for soil organisms	>0.235	0.188-0.235	<0.188	[4]
Carbon storage	> 70 t ha^{-1}	40-70 t ha^{-1}	< 40 t ha^{-1}	[71]
Groundwater quality protection	"Low"	"Moderatly low" "Moderatly high"	"High"	[94,97]
Food crop production	"Intensive", "Intensive/Moderate"	"Moderate"	"Limited" and "Moderate/Limited"	[4,40,105,107]

The quantities in Table 3 represent the thresholds used to identify the level of ES provision. The level of ES provision is related to the current state of the soils, according to current land use (obtained from Step 1) and other characteristics such as soil texture and organic matter content. According to the capacity level to provide ESs (i.e. high, medium and low capacity to provide), the actual trade-off level with BBES is identified.

The underlying hypothesis here is that soils can deliver directly (or contribute to the delivery of) some ES (i.e. habitat for soil organisms, carbon storage, food production). Thus they should be conserved or utilized sustainably, because once this capacity is lost, it is difficult and costly to be restored artificially [72]. However, the trade-off mechanism is a bit different for groundwater quality protection. Trade-offs between feedstock production and water regulating ES are expected to be low in areas with soils that have a high capacity to buffer nitrate. This is because the increased fertilizer use for the production of oilseeds for bioenergy purposes at an industrial scale will be mitigated by the natural filtering capacity of the soils. Conversely, the trade-offs are expected to be high in areas of low groundwater quality regulation potential. Table 3 represents the thresholds related to the current capacity of soils to provide

ESs under prevailing environmental and agricultural management conditions (e.g. fertilizer use, tillage). These thresholds were identified according to a literature review of peer-reviewed papers and policy documents that define acceptable trade-off levels for the maintenance of ES provision. Thresholds for indicators that were not already ranked in classes were fixed by ranking the range of values into three quantiles; the higher quantile has been associated with a high potential trade-off with feedstock production.

Step 5 obtains various combinations of trade-off levels by overlapping ESs maps with ArcGis 10.3 (ESRI). These combinations were classified into eleven groups (Table 4) according to the severity of pair-wise trade-offs. They represent the strength of the association between the distribution of BBES potential feedstock and trade-off levels. Moreover, the correlation between BBES feedstock production and other soil-related ESs was statistically tested with a Spearman's Rank Coefficient test, both at the regional and provincial levels. The ranking scores were used as input values, since the different ESs are originally measured through different variables.

Table 4: Trade-off groups ranked from the most severe (Group 1) to the least severe (Group 8).

Trade-off group description	No.
High trade-off levels for all 4 ESs	Group 1
High trade-off levels for 3 ESs	Group 2
High trade-off levels for 2 ESs	Group 3
High trade-off levels for 1 ES plus at least one ES at medium trade-off level	Group 4
High trade-off level for Habitat for soil organisms, and low trade-off level for the other ESs	Group 5a
High trade-off level for Soil Carbon Storage, low trade-off levels for other ESs	Group 5b
High trade-off level for Groundwater Quality Protection, low trade-off level for other ESs	Group 5c
High trade-off level for Food Crop Production, low trade-off level for other ESs	Group 5d
Medium trade-off level for at least 3 ESs, no High trade-off level for other ESs	Group 6
Medium trade-off level for at least 1 ES, no High trade-off levels for others ESs	Group 7
Low trade-off levels for all ESs	Group 8

Step 6 calculates the distribution of potential oilseed production from BBES feedstock within trade-off combination groups. The aim here is to assess both the amount of oilseed production that can be produced sustainably and its spatial distribution, when considering the current availability of rapeseed, soybeans and sunflowers.

Step 7 identifies and maps other potential suitable areas for sustainable feedstock production. With respect to “other arable land” (mainly under maize and other cereals) we only analyze its capacity to provide soil-related ESs other than feedstock, and capture potential trade-offs with BBES. Subsequently, we identify among these areas those that can have low trade-offs with other soil-related ESs, and we designate them as areas of potential land use change (i.e. from maize and other cereals to oilseeds for bioenergy). In particular the potentially available areas for feedstock expansion were extracted from the land use group “other arable land” in Table 1, and especially those areas that have low expected trade-offs with soil-related ESs. This calculation is only meant to verify how much area would be needed to achieve the Energy Plan targets in the Veneto Region (and if this amount is actually available in the region). The estimated gap is then calculated and discussed in relation to the potential conversion of land use among the provinces.

2.4. Soil-related ES

2.4.1. Habitat for soil organisms

Organisms living in soils can sustain most ecosystem processes and are a key resource for maintaining above and below-ground functions [73,74]. Soil organisms are crucial for nutrient [75] and carbon cycling [76], pathogen control [77], and the degradation of synthetic pesticides or industrial contaminants [78]. However, the cultivation of biofuel feedstock can strongly alter soil biota communities [57,58], for example, by decreasing the microbial processing potential [61] and arthropod abundance [11,59].

To map the capacity of soils to support biodiversity (a supporting ecosystem service), we apply the indicator proposed by Calzolari et al. [4] as follows (Eq. 2):

$$BIO_{0-1} = (\log OC_{0-1} - \log BD_{0-1}) + QBS_{ar\ 0-1} \qquad \text{(Eq. 2)}$$

where BIO is the potential of soil to preserve biodiversity, OC is the organic mass fraction of the soil (%) in the first 30 cm of soil (derived from [79]), BD is the bulk density (t m^{-3}) (derived from [80]), and QBSar is an index for assessing the biological quality of the soil, based on the abundance of microarthropod groups (ar) [81,82]. According to Calzolari et al. [4] (and given the lack of spatially explicit information on agricultural practices and intensive management for the entire Veneto plain), the QBSar was set at a low level (=0.25) for the entire study area. All variables were standardized between 0 and 1.

2.4.2 Soil carbon storage

Soil organic carbon (SOC) plays a crucial role in agro-ecosystems by performing several ecological functions related to soil structure, water cycling, and nutrient availability among others [83]. Moreover, the soil has a great potential for mitigating climate change since it is the most important terrestrial carbon pool [84,85]. In this respect SOC provides important regulating ecosystem services related to carbon sequestration and climate change regulation [86].

It is widely recognized that the cultivation of annual crops such as oilseeds for bioenergy feedstock can lead to the decrease of soil carbon content when no organic agriculture is performed [22,87,88]. This can raise a conflict between SOC preservation (and the climate regulation services it offers) with the cultivation of annual crops for bioenergy purposes [68].

The SOC content in the study area was mapped based on the regional carbon stock map developed by the Regional Agency for the Environment of Veneto [71]. The indicator used was the amount of SOC (in t ha^{-1}) in the first 30 cm of soil. The map was developed at a 1 km-pixel resolution by cross-mapping the Veneto region map of soil types with data from field-measurements. However, it should be mentioned that this indicator does not account for the superficial humic layer, which is an important component of soil in mountains of the region [89]. However, because the study area includes only the Veneto region plains, this effect is expected to be negligible.

2.4.3 Groundwater quality protection

Soil is a natural filter that protects groundwater from the leaching of chemicals, such as fertilizers and pesticides. The soil attenuation capacity is due to the vertical retention of water-soluble pollutants, which, in turn, depends on soil characteristics, climatic and hydrological conditions, and agronomic practices [90–92]. However, as discussed in Section 2.3 in the case of biofuel crops, agricultural management practices are strongly oriented towards intensive tillage and the massive application of agro-chemicals to maximize biomass production [63,93]. This means that extensive use of these chamicals, combined with soil disturbances from tillage, could negatively affect the provision of water quality protection ES offered by soil (a regulating ecosystem service).

Nitrogen retention capacity was mapped by the Regional Agency for Environmental Protection of the Veneto Region (Agenzia Regionale per la Protezione Ambientale Regione Veneto - ARPAV) [94], by applying the MACRO model [95] for the simulation of the hydrological balance, and the SOIL-N model [96] for the simulation of the nitrogen balance. The MACRO model was applied for 31 different soil-climate-aquifer conditions, considering the same cropping system (maize monoculture) for a period of 10 years (1993-2002). Agricultural practices were considered the same in all areas except for the use of irrigation. The SOIL-N model was applied to simulate the relation between hydrological fluxes and nutrient leaching.

The output of this analysis was ranked into four categories representing the protective capacity of the soil according to the leaching fluxes and nitric oxide loss. According to Villamagna et al [97] because the index represents the potential for nitrate to leach, we used inverse values to denote the soil-related groundwater quality protection potential (i.e., the nitrogen retention capacity). In other words, the higher the soil capacity to buffer nutrient leaching is, the lower the threat to the groundwater quality, and the higher the level of the water quality protection ES.

2.4.4 Food production

The direct use of crops and/or agricultural land for biofuel production has raised important concerns as exemplified with the “fuel vs food” controversy, e.g. [98]. The actual trade-off between feedstock and food provisioning ESs is mainly due to the final use of crop production [99] (i.e. energy conversion vs food consumption) and the direct and indirect land use change effects, e.g. [100]). While such trade-offs can have important effects on food security [13], there is conflicting evidence about the actual effect of biofuel production on food prices [101–104].

In this study, potential food production in the study area was mapped using the Land Capability Classification (LCC) index [105]. This indicator has already been applied for ES mapping [4,40], based on the principle that the most productive agricultural land should not be targeted for bioenergy production [40,106]. The LCC map is available at a 1:50,000 scale for the Veneto region [107] and classifies areas according to their potential to support agricultural production considering 13 characteristics related to soil, water, erosion risk and climate.

3. Results

3.1 Potential biodiesel production and provision of soil-related ESs

Table 5 shows the maximum potential biodiesel production that is achievable under the hypothesis that all current oilseed feedstocks would be destined for biodiesel production. Total biodiesel potential in the region was estimated at approximately 97.6 dam^3. At the regional scale, the potential biodiesel production from current land use was primarily attributed to soybeans (65.85%), followed by rapeseed (29.26%) and sunflowers (4.89%). Sunflower production was shown to be slightly more efficient in terms of the potential biodiesel production per unit area (175 $m^3\ km^{-2}$), followed by rapeseed (109 $m^3\ km^{-2}$). Soybeans, on the other hand, were largely inefficient (65.7 $m^3\ km^{-2}$).

Table 5: Annual potential biodiesel production from current levels of feedstock production.

			Not available	Other arable land	Rapeseed	Soybean	Sunflower	Tot
Belluno	Area	km^2	260.00	60.00	0.00	0.00	0.00	320.00
		%	82.10%	17.69%	0.00%	0.00%	0.00%	100.00%
	Biodiesel	m^3	0.00	0.00	0.00	0.00	0.00	0.00
	Potential	%	0.00	0.00	0.00	0.00	0.00	0.00
Padova	Area	km^2	228.31	1 124.10	34.50	122.52	1.91	1 511.34
		%	15.11%	74.38%	2.28%	8.11%	0.13%	100.00%
	Biodiesel	m^3	0.00	0.00	3 563.16	7 903.85	213.16	11 680.17
	Potential	%	0.00%	0.00%	30.91%	67.19%	1.90%	100.00%
Rovigo	Area	km^2	136.68	904.57	57.60	245.53	9.03	1 353.42
		%	10.10%	66.84%	4.26%	18.14%	0.67%	100.00%
	Biodiesel	m^3	0.00	0.00	6 463.29	17 784.69	991.33	25 239.31
	Potential	%	0.00%	0.00%	25.64%	70.25%	4.11%	100.00%
Treviso	Area	km^2	382.09	746.40	46.35	119.59	0.00	1 294.43
		%	29.52%	57.66%	3.58%	9.24%	0.00%	100.00%
	Biodiesel	m^3	0.00	0.00	5 061.88	8 919.89	0.00	13 981.78
	Potential	%	0.00%	0.00%	37.32%	62.68%	0.00%	100.00%
Venezia	Area	km^2	216.18	958.07	23.97	282.51	2.15	1 482.88
		%	14.58%	64.61%	1.62%	19.05%	0.15%	100.00%
	Biodiesel	m^3	0.00	0.00	2 881.27	20 625.44	282.79	23 789.50
	Potential	%	0.00%	0.00%	12.65%	86.06%	1.29%	100.00%
Vicenza	Area	km^2	168.32	496.89	55.95	43.55	0.73	765.44
		%	21.99%	64.92%	7.31%	5.69%	0.09%	100.00%
	Biodiesel	m^3	0.00	0.00	5 812.39	3 038.30	87.23	8 937.91
	Potential	%	0.00%	0.00%	64.75%	34.21%	1.04%	100.00%
Verona	Area	km^2	459.89	822.70	36.44	96.46	24.04	1 439.53
		%	31.95%	57.15%	2.53%	6.70%	1.67%	100.00%
	Biodiesel	m^3	0.00	0.00	4 261.17	6 746.09	2 979.93	13 987.19
	Potential	%	0.00%	0.00%	31.30%	46.72%	21.99%	100.00%
Veneto Region	Area	km^2	1 591.74	5 052.78	254.81	910.16	37.86	7 847.35
		%	20.28%	64.39%	3.25%	11.60%	0.48%	100.00%
	Biodiesel	dam^3	0.00	0.00	28.04	65.02	4.55	97.62
	Potential	%	0.00%	0.00%	29.26%	65.85%	4.89%	100.00%
Regional conversion parameter	BP/A	m^3km^{-2}	0.00	0.00	110.06	71.44	120.30	81.16

The provinces with the highest biodiesel potential from soybeans were Padova (67%), Rovigo (70%), Treviso (62.6%) and Venezia (86%). Rapeseed was the most dominant in Vicenza (64.7%), while the three biodiesel potential from the three feedstocks was more balanced in Verona (31.3% from rapeseed, 46.7% from soybeans, and 22% from sunflowers).

Fig. 3 shows the spatial distribution of the potential biodiesel feedstock production according to the current land use and in relation to different conversion parameters, which were specific for each feedstock and province (see Table A2 in Supplementary Electronic Material).

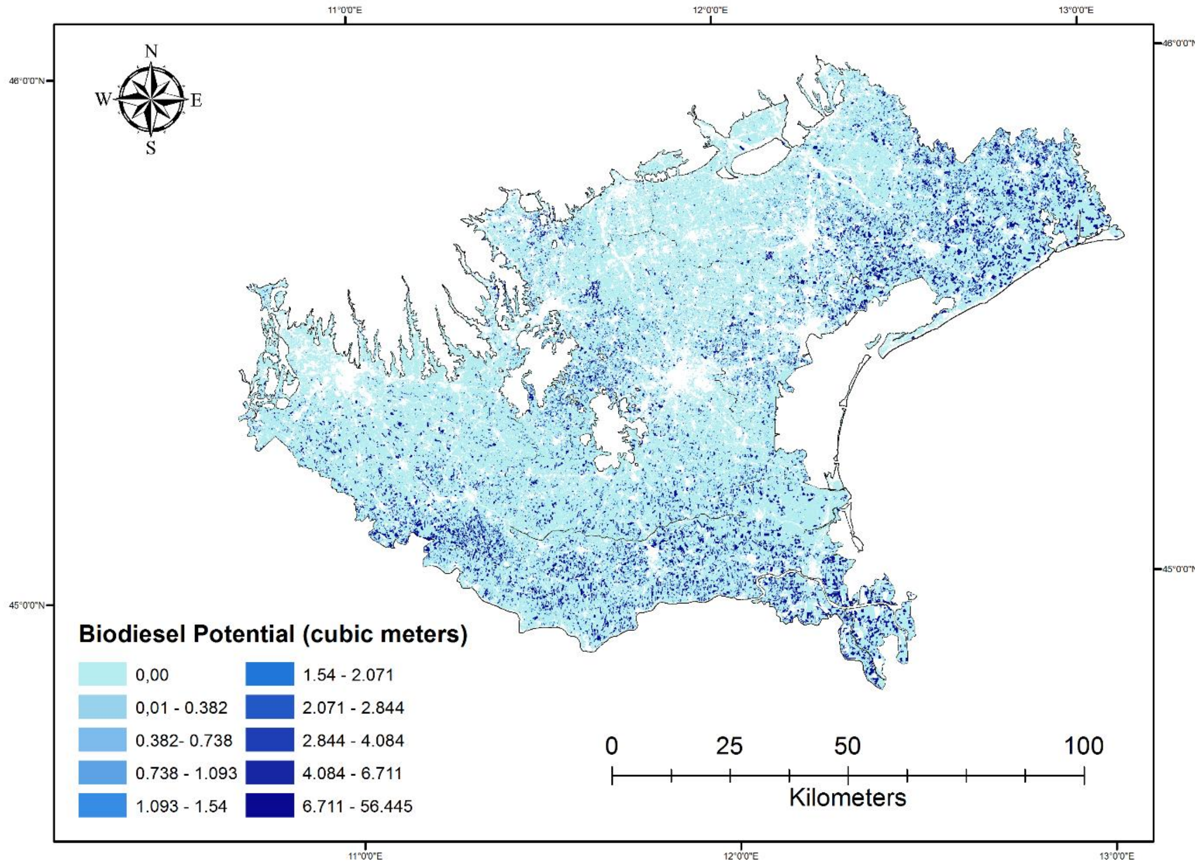

Fig.3: Spatial distribution of the annual potential biodiesel production in the Veneto plain region.

Fig. 4-5 map individually for the plains of the Veneto Region the different soil-related ESs such as carbon storage, habitat for soil organisms, groundwater quality protection, and food production. These ESs are distributed differently between the provinces according to the distinct environmental characteristics and soil properties. For example, higher values for "habitat for soil organisms" are mapped in proximity of rivers, where coarse texture soils are present, as the indicator is a function of bulk density (Eq.2). Groundwater quality protection is low on some zones in the southern part of the region, where higher levels of soil organic matter are responsible for nitrogen mineralization, and on higher plains, where more coarse soil texture classes are present.

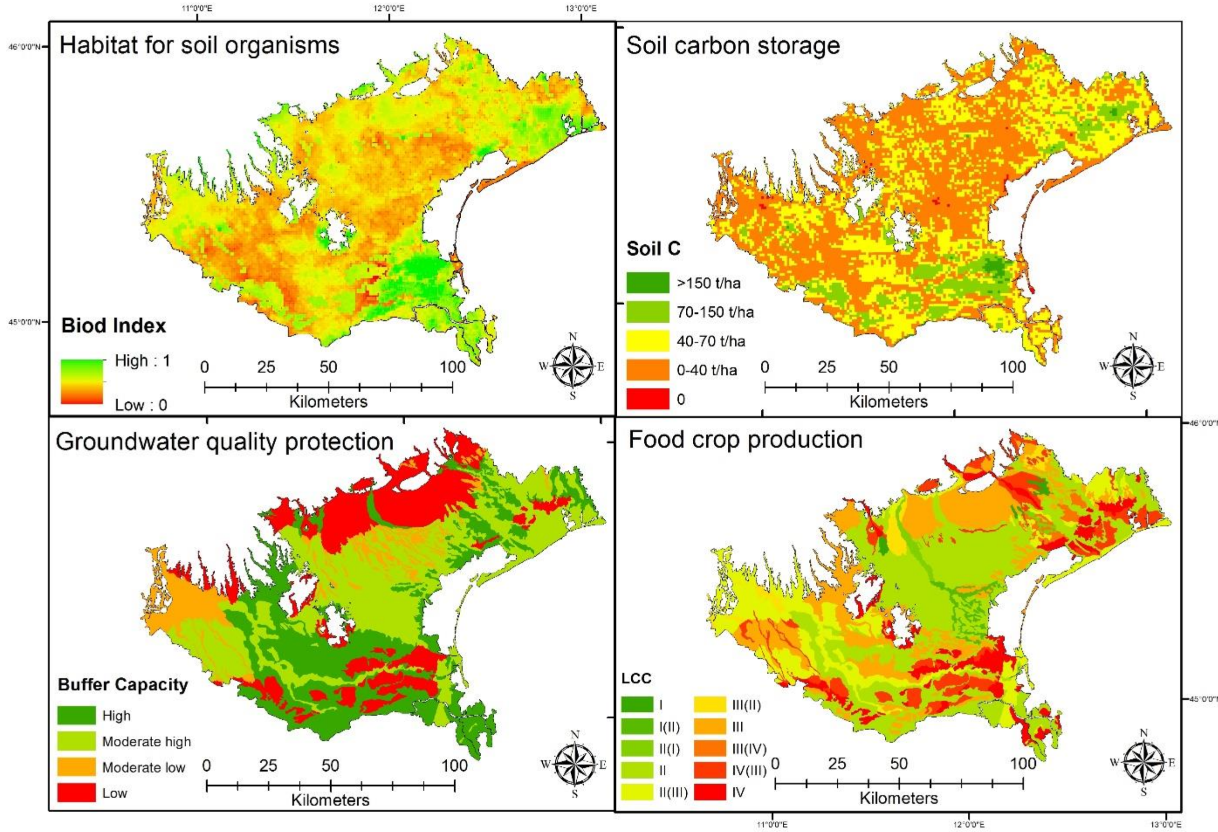

Fig. 4: Spatial distribution of the four soil-related ESs.

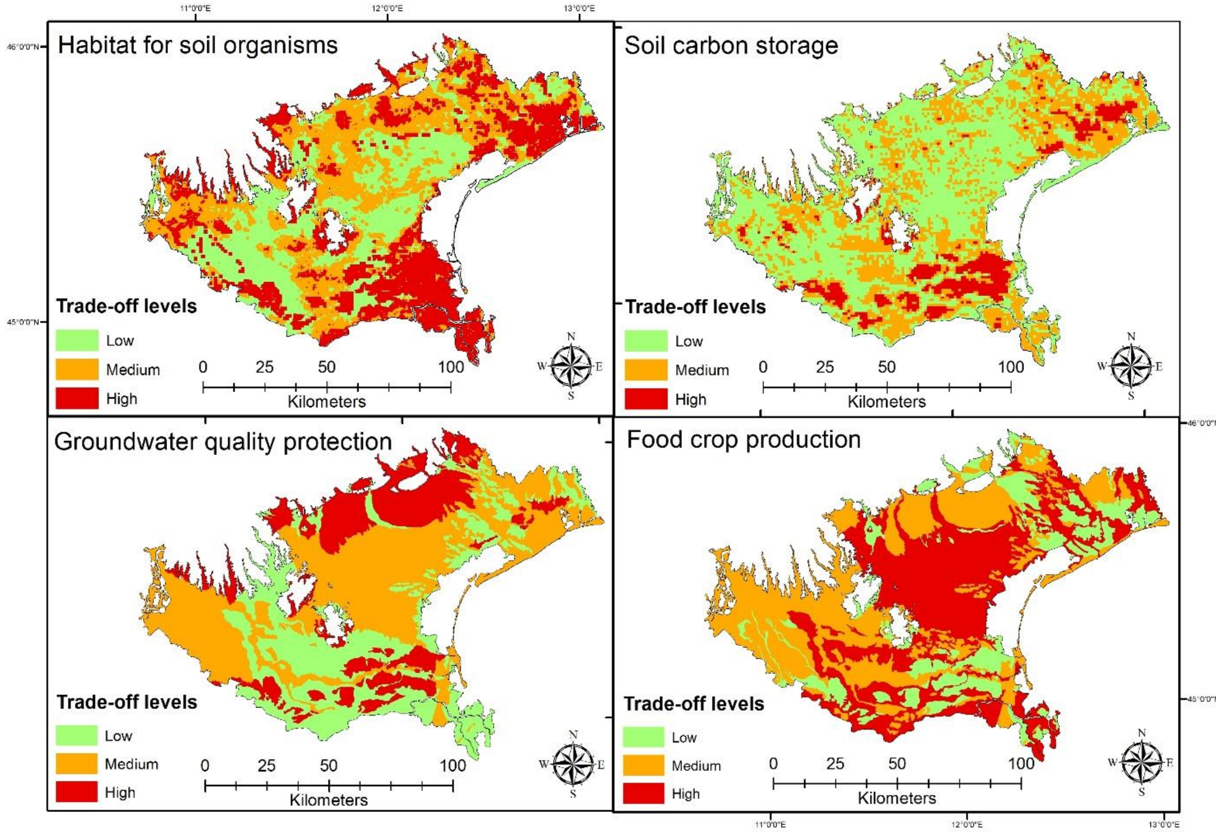

Fig.5: Spatial distribution of the trade-off levels of the four soil-related ESs with BBES.

3.2 Relationship between soil-related ESs and potential biodiesel production

Table 6 summarises over the whole study area (i.e., the regional level) the trade-offs between potential biodiesel production with each soil-related ES. Rapeseed showed a higher potential conflict with food production as about 43.7% of rapeseed biodiesel potential is located in areas with high levels of food production. The lowest potential trade-offs of rapeseed production were with soil carbon storage, as only 6.8% of rapeseed biodiesel potential is located in areas with high levels of soil carbon storage. On the other hand large areas of soybean biodiesel potential had high trade-offs with habitat for soil organisms (46.7%) and food production (43.4%). Trade-offs of sunflower biodiesel potential were the highest with carbon storage (49.4%) and habitat for soil organisms (46.7%).

Table 6: Distribution of potential Biodiesel Production (BP) and Areas (A) per land use class, with respect to the trade-off levels of the four soil-related ESs.

	Soil-related ESs							
	Carbon storage		Habitat for soil organisms		Groundwater quality protection		Food crop production	
Trade-off level	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)
Other arable land								
High	-	13.22	-	34.92	-	22.03	-	42.64
Medium	-	41.88	-	32.26	-	44.60	-	37.54
Low	-	44.90	-	32.81	-	33.36	-	19.81
Tot		100.00		100.00		100.00		100.00
Rapeseed								
High	6.82	6.63	36.69	36.45	22.59	23.02	43.66	44.43
Medium	43.88	44.01	36.04	36.44	45.08	44.50	38.36	37.78
Low	49.30	49.36	27.27	27.11	32.32	32.48	17.98	17.80
Tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Soybean								
High	31.46	20.23	46.86	46.55	18.98	18.89	43.37	43.35
Medium	48.25	48.17	26.14	26.21	40.97	41.14	26.33	26.51
Low	20.29	31.60	27.00	27.23	40.05	39.97	30.30	30.15
Tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Sunflower								
High	49.44	48.86	39.99	40.45	16.08	16.39	24.74	25.39
Medium	32.41	32.85	12.29	12.55	56.38	54.90	51.74	51.00
Low	18.15	18.29	47.72	47.00	27.54	28.71	23.52	23.60
Tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Veneto region								
High	16.24	14.01	43.55	36.71	19.90	21.58	42.55	42.71
Medium	46.20	42.82	28.36	31.44	42.93	44.16	31.09	36.03
Low	37.56	43.17	28.09	31.85	37.18	34.26	26.36	21.26
Tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00

Table 7 considers the distribution of trade-off between provinces. Higher trade-off levels were found with habitat for soil organisms in Rovigo (for 70.5% of the potential biodiesel production in the region) and with food production in Padova (60.4%).

Table 7: Distribution of potential Biodiesel Production (BP) and Areas (A) per province, with respect to the trade-off levels of the four soil-related ESs.

	Soil-based ESs							
	Soil carbon storage		Habitat for soil organisms		Groundwater quality protection		Food crop production	
Trade-off level	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)
Belluno								
High	0.00	0.00	0.00	44.72	0.00	100.00	0.00	0.00
Medium	0.00	59.95	0.00	0.90	0.00	0.00	0.00	0.00
Low	0.00	40.05	0.00	54.38	0.00	0.00	0.00	100.00
tot	0.00	100.00	0.00	100.00	0.00	100.00	0.00	100.00
Padova								
High	9.79	8.10	21.91	21.12	12.76	14.71	60.45	59.86
Medium	47.65	45.12	41.05	41.04	49.86	49.17	27.63	30.16
Low	42.56	46.78	37.04	37.83	37.38	36.12	11.91	9.98
tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Rovigo								
High	22.20	21.61	70.53	59.44	23.96	23.24	52.22	52.38
Medium	51.22	45.94	14.44	20.19	6.90	10.24	15.17	18.95
Low	26.58	32.45	15.03	20.37	69.14	66.52	32.61	28.68
tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Treviso								
High	6.13	4.77	29.16	30.95	31.25	48.39	38.79	32.38
Medium	52.23	49.35	45.80	46.63	46.55	33.94	38.58	45.17
Low	41.64	45.88	25.05	22.42	22.20	17.67	22.63	22.45
tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Venezia								
High	23.98	24.44	51.46	49.85	14.20	15.45	39.20	41.22
Medium	47.95	42.55	23.45	21.42	58.24	58.54	22.64	22.82
Low	28.06	33.02	25.09	28.74	27.56	26.01	38.16	35.97
tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Verona								
High	16.13	7.89	29.15	22.28	13.64	9.59	14.57	14.26
Medium	28.42	33.72	20.06	28.77	67.57	67.71	63.56	75.18
Low	55.45	58.39	50.79	48.95	18.79	22.70	21.88	10.56
tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Vicenza								
High	3.79	2.83	20.83	22.04	25.05	29.19	50.01	40.77
Medium	44.18	41.82	49.47	45.69	50.95	44.25	39.74	50.50
Low	52.03	55.35	29.70	32.27	24.00	26.56	10.25	8.73
tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Veneto region								
High	16.24	12.42	43.55	35.08	19.90	22.35	42.55	40.28
Medium	46.20	43.06	28.36	32.86	42.93	44.63	31.09	39.56
Low	37.56	44.53	28.09	32.06	37.18	33.01	26.36	20.16
tot	100.00	100.00	100.00	100.00	100.00	100.00	100.00	100.00

Spearman's correlation values (r_s) are shown in Table 8. Some ES relationships were significant (p-value<0.05) in some provinces and not significant at the regional level (or vice versa).

For example, habitat for soil organisms, soil carbon storage and groundwater quality protection showed significant correlation with potential biodiesel production at the regional scale, while they were not significant in 4 out of 6 provinces. Conversely, the trade-off between potential biodiesel production and food production was significant in 3 out of the 6 provinces, while it was not significant at the regional scale. Overall, negative correlations were observed for trade-offs with soil carbon storage and groundwater quality protection, and positive with habitat for soil organisms and food production. This means that oilseed crops, suitable for biofuel production, are currently distributed on areas whit already low levels of soil carbon storage (i.e. avoided trade-off with carbon storage). By other hands, these zones are very productive (i.e. trade-off with food production) and present poor capacity to buffer nutrient leaching to groundwater (i.e. trade-off with groundwater quality) and high capacity to support soil biodiversity (i.e. trade-off with habitat for soil organisms). Overall, this framework shows that oilseed crops are currently distributed and managed according industrial scopes.

When significant, the r_s ranged between -0.408 and +0.3705.

Table 8: Statistical correlations between biodiesel potential and soil-related ESs at regional (Veneto) and at provincial level.

Ecosystem services	Region	Provinces					
	Veneto	Rovigo	Padova	Treviso	Venezia	Vicenza	Verona
Habitat for soil organisms							
r_s	0.222	0.3705	0.0708	0.1455	0.31	-0.0108	0.0634
n°	80	42	74	71	63	65	51
p-value	**0.0484**	**0.0177**	0.5452	0.2234	**0.0146**	0.9309	0.6538
Soil carbon storage							
r_s	-0.3349	-0.2438	-0.2207	-0.3997	-0.1711	-0.408	-0.2725
n°	80	42	74	71	63	65	51
p-value	**0.0029**	0.1185	0.0593	**0.0008**	0.178	**0.0011**	0.054
Groundwater quality protection							
r_s	-0.3299	-0.2657	-0.2368	0.1252	-0.0929	-0.2761	-0.1526
n°	80	42	74	71	63	65	51
p-value	**0.0034**	0.0889	**0.0431**	0.2948	0.4646	**0.0272**	0.2806
Food crop production							
r_s	0.0308	0.3489	0.3538	0.1229	0.306	0.0495	-0.0551
n°	80	42	74	71	63	65	51
p-value	0.7846	**0.0255**	**0.0025**	0.3037	**0.016**	0.6923	0.6967

Note: r_s = Spearman's rank correlation; n° = number of trade-off combinations. Significant correlations (p-value < 0.05) are highlighted in bold.

3.3 Sustainable potential biodiesel production

The sustainable potential biodiesel production was calculated through a trade-off analysis for different types of trade-offs (trade-off groups). The trade-off groups are defined and listed in Table 4, and range from the most severe (Group 1, all trade-offs are at a high level) to the least severe (Group 8, all trade-offs are at a low level).

We consider the “sustainable biodiesel potential” in the Veneto Region as the amount of biodiesel in areas where trade-offs fall under trade-off Groups 6-8 (i.e., the groups that are not involved in trade-offs at a high level) (Section 2.2). This sustainable biodiesel potential corresponds to 20.7 dam^3 per year, which is equal to 21.2% of the total current biodiesel potential of the study area (Table 9). The remaining fraction of biodiesel potential falls into groups with at least one trade-off at a high level. No areas have been mapped for trade-off Group 1, meaning that there are no areas where all four soil-related ESs have high trade-offs with potential BBES feedstock production at the same time.

Table 9: Distribution of potential Biodiesel Production (BP) and Areas (A) among trade-off groups.

	Belluno		Padova		Rovigo		Treviso		Venezia		Verona		Vicenza		Veneto Region			
Trade-off Group	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)	BP (%)	A (%)	BP (dam^3)	BP (%)	A (km^2)	A (%)
1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2	0.00	0.00	7.60	5.97	20.64	19.14	1.48	1.30	15.02	15.88	11.74	3.92	2.88	1.96	11.77	12.06	672.99	8.58
3	0.00	44.72	8.47	9.62	36.28	29.93	19.59	23.71	19.50	17.83	6.83	6.36	12.75	15.36	19.62	20.10	1,331.01	16.96
4	0.00	44.01	60.16	61.61	26.02	27.05	59.20	62.45	43.58	44.55	19.46	22.32	62.70	56.01	40.56	41.55	3,516.27	44.81
5a	0.00	0.00	0.05	0.04	0.56	0.86	0.13	0.10	0.25	0.38	0.00	0.00	0.00	0.00	0.22	0.23	19.17	0.24
5b	0.00	0.00	0.00	0.01	0.00	0.00	0.24	0.05	0.06	0.07	0.00	0.00	0.00	0.00	0.05	0.05	1.80	0.02
5c	0.00	11.27	0.01	0.04	0.02	0.05	1.22	2.19	0.00	0.00	0.15	0.04	1.11	0.92	0.30	0.31	37.23	0.47
5d	0.00	0.00	5.11	4.95	7.88	11.43	0.44	0.41	2.18	2.67	7.57	7.17	1.43	1.31	4.35	4.46	387.53	4.94
6	0.00	0.00	12.21	10.99	5.06	6.95	7.18	4.33	9.24	7.48	21.95	28.13	8.00	8.96	9.69	9.93	900.81	11.48
7	0.00	0.00	6.37	6.74	2.93	4.21	10.26	5.37	10.00	10.85	32.30	32.06	11.12	15.48	10.81	11.08	969.31	12.35
8	0.00	0.00	0.02	0.03	0.61	0.37	0.27	0.11	0.18	0.30	0.00	0.00	0.00	0.00	0.24	0.24	11.22	0.14
Tot	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	97.62	100%	7,847.35	100%
Sust. BP (dam^3)	0.00		2.17		2.17		2.48		4.62		7.59		1.71		20.74			
Sust. BP (%)	0.00		10.48		10.48		11.94		22.27		36.59		8.24		100.00			

Note: Sustainable BP is given by the sum of groups 6,7 and 8.

If we limit the definition of “sustainabile biodiesel” only to areas of biodiesel production that have Group 8 trade-offs, then the sustainable potential biodiesel production from the Veneto Region would only be 0.24 dam^3 per year (0.24% of the total potential). The highest biodiesel potential (41.5%) is in areas with one with high level ES trade-off and at least one medium ES trade-off (Group 4). Fig. 6 provides spatially explicit information about the distribution of the trade-off groups, allowing for the mapping of those areas where sustainable oilseed production is possible.

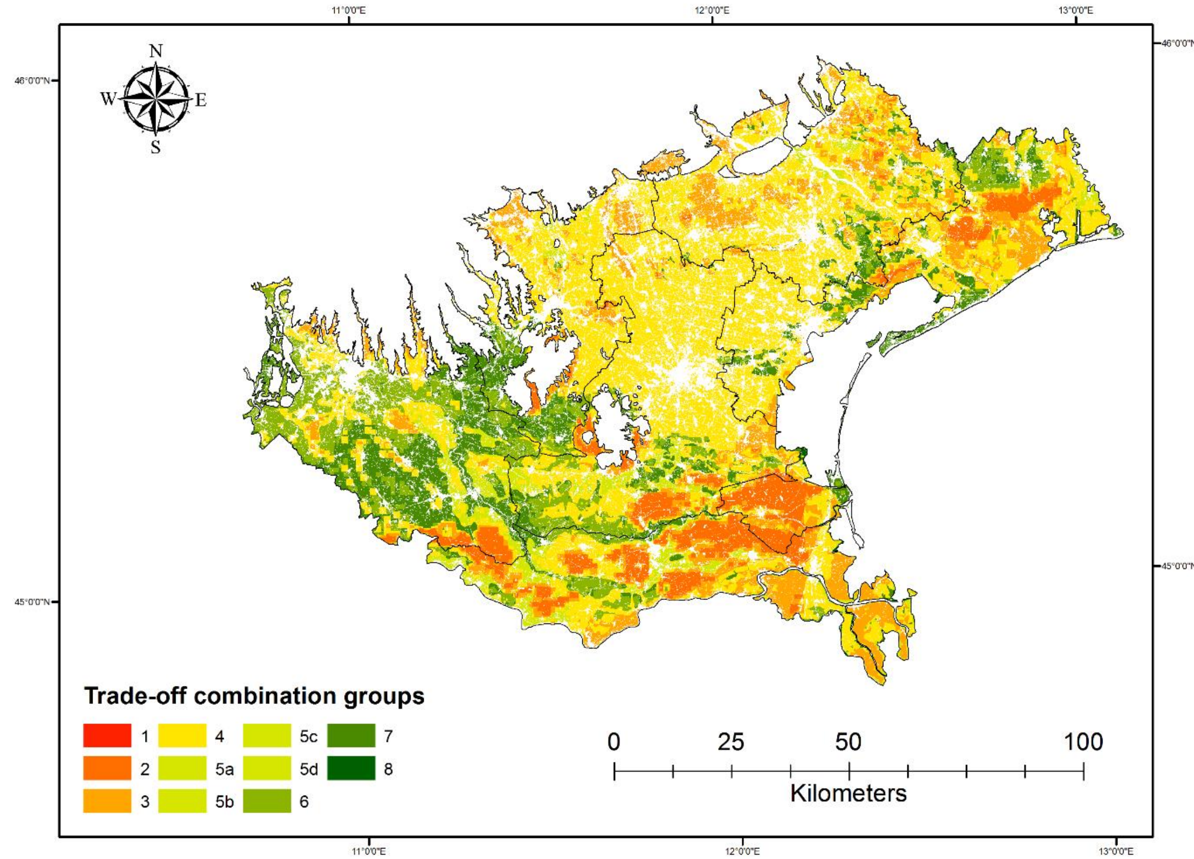

Fig.6: Spatial distribution of trade-off combination groups.

Table 9 also provides information about the contribution of each province to the regional biodiesel potential. The highest sustainable potential is in the province of Verona (37% of the total), while Rovigo has the lowest (only 8.6% of the total).

3.4 Conversion of other arable land to increase sustainable biodiesel potential

"Other arable land" in the Veneto plain accounts for an area of 5,052.78 km^2 (Table 10). If the portion of other arable land that falls into trade-off Groups 6-8 (1,131.33 km^2) is converted into sunflower crops (the most efficient oilseed crop in the region, Section 3.1), the sustainable biodiesel potential would increase by 133.8 dam^3 per year. The province of Verona would contain 61.6% of the total are with these characteristics in the Venero Region, confirming its potential capacity to produce a significant amount of oilseed without affecting the provisioning of other ESs.

Table 10: Area distribution of trade-off groups for “Other arable land”.

	Area distribution among provinces								
	Belluno	Padova	Rovigo	Treviso	Venezia	Verona	Vicenza	Veneto region	
Trade-off Group	km^2	km^2	km^2	km^2	km^2	km^2	km^2	km^2	%
1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00%
2	0.00	66.18	179.79	8.10	175.32	31.38	10.00	470.78	12.01%
3	0.03	106.54	262.45	178.29	174.04	42.06	74.36	837.77	21.36%
4	0.01	698.39	246.09	484.85	433.00	174.02	277.10	2,313.46	58.99%
5a	0.00	0.41	7.11	0.68	3.68	0.00	0.00	11.88	0.30%
5b	0.00	0.18	0.00	0.20	0.73	0.00	0.00	1.11	0.03%
5c	0.02	0.35	0.60	13.44	0.00	0.18	3.84	18.44	0.47%
5d	0.00	56.90	107.91	3.18	26.25	68.29	5.49	268.02	6.83%
6	0.00	123.15	63.94	24.51	52.02	223.33	49.44	536.38	13.68%
7	0.00	71.72	34.46	32.37	90.11	283.44	76.67	588.76	15.01%
8	0.00	0.28	2.22	0.78	2.92	0.00	0.00	6.20	0.16%
Tot other arable land	0.06	1,124.10	904.57	746.40	958.07	822.70	496.89	5,052.78	100%
Unsustainable	0.06	928.96	803.96	688.74	813.02	315.93	370.78	3 921.45	
Sustainable	0.00	195.14	100.61	57.65	145.04	506.77	126.11	1 131.33	

Note: Sustainable biodiesel potential area is given by the sum of trade-off group 6, 7 and 8.

4. Discussion

4.1. Importance of trade-offs for sustainable feedstock production

The present study outlines and tests a methodology to evaluate the trade-offs of potential biodiesel production from oilseeds, with soil-related ESs. The Regional Energy Plan of the Veneto Region (REP) [70] proposed a biodiesel production target for the year 2020 in which the annual biodiesel production from oilseeds in the region should be 39.6 dam^3 per year. When comparing the potential sustainable biodiesel production that was quantified by our analysis (Section 3.3) and the REP target, it becomes obvious that only 52% of the REP target could be achieved without leading to a high-level trade-off with at least one soil-related ES. This suggests that achieving the REP target by 2020 would be impossible without significantly impacting the current capacity of the region to provide soil-related ESs, or without causing indirect land use change.

Converting portions of “other arable land” (i.e., land under other annual crops) can allow to an extent the “sustainable expansion” of oilseeds production for biodesel (Section 3.4) preventing at the same high-level trade-offs with soil-related ESs.However, such a conversion might not be desirable, as such land use change could lead to the loss of landscape diversity (i.e., due to monoculture expansion) [108,109], with possibly extensive impacts to supporting and cultural ecosystem services [40,110–114] not quantified in this paper.

The trade-off analysis between soil-related ESs and feedstock for potential biodiesel production demonstrates the complex spatial nature of the relationships between ESs. First, it is evident from our analysis that the sustainability and trade-off levels of potential biodiesel production vary across the case study area largely due to the differences in soil characteristics and land cover.

Second, the statistical correlation between soil-related ESs and potential feedstock production varies significantly in the study region (e.g. between the entire study region and the single provinces, as well as between the provinces) (Table 8). This can have important implications for energy planning. As energy planning requires the adoption of both regional and provincial plans, the quantified differences in trade-off patterns in the present analysis should be considered. For instance, the trade-off with food production is not significant at the regional level, while it is significant for the provinces of Rovigo, Padova and Venezia. In other words, in these provinces, there is a considerable biodiesel feedstock potential located in zones characterized by high levels of food productivity. However, the spatial conflict between these two ecosystem services could not be detected in regional energy plans as shown by our analysis at the regional level. This means that regional energy plans for the Veneto Region need to take into consideration such variations between provinces.

4.2. Indicators and certification

As already discussed, our methodology can identify areas where the trade-offs between bioenergy feedstock production and the provisioning of other soil-related ES are high. Such spatially-disaggregated information is essential for assessing the territorial and cumulative effects of biofuel production when considering local environmental conditions, as well as to model the effects of large-scale feedstock introduction in specific contexts [115,116].

While ecosystem services have not been properly integrated into biofuel-related certifications schemes [42], the ecosystem services narrative has began featuring in some certification schemes such as Bonsucro and the Roundtable for Sustainable Biomaterials (RSB) [117,118]. Besides the conceptual issues of integrating meaningfully ecosystem services in such schemes (e.g. [13]), there is a lack of proper on-field impact assessment guidelines in existing certifications schemes [37,38].

However the potential biofuel trade-offs with other ESs can be a key element of certification standards and can be included in feedstock certification documentation. In particular, tools that can develop ES trade-off maps can be very useful to decision-makers and certification agencies as this allows they can visualize the impact/trade-offs of feedstock production with other ES rather than simply focus on compliance with good agricultural practice.

The analysis presented in this paper can also improve the set of indicators under the principle of "protection of soil, water and air and the application of Good Agricultural Practices" under the EU-RED [27]. Furthermore, the analysis of trade-offs between soil-related ESs can improve the contents of soil management plans, which are actually considered only within two certification schemes (ISCC, RSB, see Table A1 in Supplementary Electronic Material).

However, the primary gap in the current implementation of soil management plans at the farm level still remains a barrier for achieving feedstock sustainability through certification schemes in the EU. In the UK, for example, DEFRA [119] proposed guidelines to compile soil management plans as a means of cross-compliance with Good Agricultural and Environmental Conditions (GAEC) for environmental stewardship. However, these guidelines required farmers to be supported by experts during the preparation of the soil management plan.

4.3. Strategic environmental assessment

ES trade-off maps could become part of the knowledge frameworks developed for regional energy plans in the EU. Such energy plans usually identify strategic objectives and other related targets with respect to the implementation of EU RED at the national level, and then attribute them at the regional level through burden sharing. Energy plans are subjected to sustainability compliance assessment under the provisions of the Strategic Environmental Assessment (SEA) Directive 2001/42/EC [120]. Recently, many authors have suggested that the integration of the ES approach into SEAs could be beneficial [121–125]. However, Baker et al. [125] note that such an integration "requires a pragmatic, context specific consideration of how ecosystem services can be used to help addressing some of the common problems with current environmental assessment practice" (p. 3). Among others, a limitation of SEAs is their lack of analytical methods [111], especially when dealing with renewable energy [121] and when considering "genuine, reasonable alternatives" [125].

Among the full range of environmental issues addressed in SEAs [126], our methodology can allow the identification of potential areas for energy crops that in order to achieve biofuel targets. It is characterized for the ES trade-offs that are potentially associated because of soil characteristics. The methodology discussed in this paper can be used to evaluate the sustainability of the production targets forecasted for biodiesel production in the Veneto region, especially related to soil-related impacts.

Considering our results (Section 3.3) the biodiesel production targets in the Veneto region can only be met by i) producing oilseed in areas with low trade-offs, ii) affecting other soil-related ES or inducing land use change, or iii) importing feedstock or vegetable oil from outside the case study area (which can potentially shift environmental burdens to other areas of energy crop production). All these solutions imply different potential impacts on soil characteristics, as well as involving other environmental receptors identified in the SEA Directive [120].

Moreover, the proposed methodology can be used to develop different scenarios to explore development alternatives to meet the objective of the energy plan. The different parameters derived from our analysis such as biodiesel production potential (Section 3.1), biodiesel potential not competing with soil-related ES (Section 3.2), and sustainable biodiesel potential (Section 3.3-3.4) can represent the baseline information for evaluating potential alternatives to achieve the targets of the Energy Plan of Veneto Region.

Finally, as already discussed our methodology can identify areas where trade-offs between oilseed production for energy purposes and ESs are high. These areas can be devoted to other types of agricultural production in order to minimize impacts on soil-related ES. Alternatively if feedstock production is located in areas with high trade-offs the impacts can be detected and monitored if the oilseed crops are devoted to biodiesel feedstock production in areas with high trade-offs. In fact, the SEA Directive implies the monitoring of significant environmental effects while the plan is implemented to identify adverse effects and then remediate them [121].

4.4. Challenges and limitations

One of the main limitations of this study is that we assumed no land cover change (Section 2.3). However, land cover change has been considered as one of the most important impacts in bioenergy cropping [11,16,17], which can affect carbon stocks and biodiversity loss. This means that our study provides a static analysis of biofuel potential, which does not take into account impacts due to the conversion from one land coverage to another. By other hand, the quantification and localization of the "sustainable biofuel potential" is a suitable tool for both mitigate ES trade-offs and prevent land cover changes.

Another fundamental aspect is related to the selection of ES related to biofuel production. For instance, while erosion regulation is an important soil-related ES that might be affected by intensive agricultural practices for BBES feedstock production [22,35], it has not been included in this analysis. This is because the case study area is a plain, and the potential rates of soil erosion are very limited [127]. Furthermore, although biofuel crops have important impacts in terms of water supply [100], their effect on water availability has not been assessed in this study. Since the Veneto plain is characterized by a well-managed irrigation system, water availability is unlikely to be a limiting factor. Finally, as the study does not assume any land cover change in the case study area (as required by [27][128]), cultural ESs were not included. Trade-offs between BBES and cultural ESs were associated with landscape diversity loss due to potential land cover change [40,129,130].

5. Conclusions

The results demonstrated the significant potential production of agro-environments in the Veneto plain in terms of biodiesel. Unfortunately, only a limited fraction of this production can be exploited without affecting the provisioning of soil-related ESs, which could cover approximately half of the REP target for the year 2020. Large amounts of additional biodiesel production could be obtained without significant impacts on soil-related ESs, but with a strong risk of harming the landscape spatial composition, which is important for the provisioning of supporting and cultural ecosystem services.

The approach presented in this study allows for an effective assessment of the sustainability of oilseed crops for biodiesel conversion, providing a tool to mitigate the controversial impacts of biofuels. We have demonstrated that the integration of ES trade-off analysis can be beneficial when assessing the sustainability and multiple impacts of energy crop production at a local scale with respect to soil-related ES. However, the management of the baseline information and the construction of related knowledge frameworks would be more effective at the regional scale, where the analysis can be sensitive to local environmental conditions and dynamics as well as connected with energy planning at the operative level. Nonetheless, the feedstock market for biodiesel and biofuels is a global one, and the effectiveness of routinizing ES trade-off approaches in assessing the sustainability of energy crops in certifications schemes when applied between countries and continents might be challenged by operational issues such as the ease of application in local environmental contexts above the company level. Furthermore, we demonstrated that both local (on provinces) and crop type-related approaches need to be considered in order to be effective in preserving soil characteristics for multiple ES provisioning. The provincial extension was observed as the most effective scale for trade-off analysis in RES planning.

The aim of limiting the use of first-generation biofuels is meant to stimulate a more rapid development and deployment of second-generation and third-generation biofuels. However, these latter two classes are not free of impacts, and biodiesel production from land-related biofuels will remain limited. Urgent action is needed to counteract possible negative impacts effectively, besides transport to other countries. Mapping the spatial distribution of trade-offs as well as studying the relations between feedstock-for-biodiesel provisioning and other ESs is a key challenge in guaranteeing the sustainable development and exploitation of biomass-based energy sources.

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3.5. "Fine-scale analysis of urban flooding reduction from green infrastructures: an ecosystem services approach for urban planning"

This analysis presents the contribution of green infrastructures in an urban center to mitigate flood risk. Green spaces were detected using remote sensing technics at high-resolution. The rainfall runoff was estimated using the SCS model. The capacity of green space to mitigate flooding is affected by fragmentation and stressed conditions (i.e. alteration of ecological functions/attributes). Moreover, a vulnerability index was calculated for urban zones (sections) in order to rank them according the priority of intervention for the maintenance and increase of green spaces. The analysis is an example of how ES mapping can be applied to manage flood risk in densely populated urban ecosystems.

Abstract

Climate change is expected to exacerbate precipitation patterns in future, increasing the demand for water-related service. These changes require effective adaptation strategies at local scale in order to mitigate the consequences of extreme events, which are predicted to increase in frequency and magnitude. Green infrastructures can contribute to adaptation while providing ecosystem service of regulation of water fluxes, mainly related to rainfall-generated runoff and flooding.

This study provides a suitable tool for urban planning by modelling the supply capacity and the demand for urban flooding reduction, by integrating remote sensing technique and statistical census data. We used a high-resolution urban digital model to distinguish between permeable and impermeable areas at fine scale. We calculated the flooding reduction capacity through two indexes: i) the amount of runoff reduced by green spaces, and ii) the runoff reduction coefficient. We also analyze the flooding reduction demand through a vulnerability index. The method was applied to different scenarios in a historical urban center of the Northern Italy. We finally contrast the supply and demand to identify priority areas of intervention.

Results show that the flooding reduction capacity is unevenly distributed in the study area. Public and private surfaces contribute differently to the total runoff and showed different performances of flooding reduction. In eight urban sections out of nine, private properties generate larger amount of rainfall runoff than public ones under the worst scenario conditions. The study suggested two urban sections as priority areas of intervention, because of the ES demand and supply mismatch.

1. Introduction

Urban areas have been appointed to be extremely vulnerable to the effects of climate change as urban ecosystems are generally less resilient than natural ones, as mainly covered by impervious surfaces (Ashley et al., 2005; Huong and Pathirana, 2013). Moreover, given that the impacts of climate change are experienced locally (Carter et al., 2015) several cities have already started in elaborating mitigation and adaptation strategies to climate change in order to reduce their vulnerability (Rosenzweig et al., 2011).

Climatic changes are expected to exacerbate precipitation patterns in future (Schröter et al., 2005), increasing the demand for water-related services (Zheng et al., 2016). These changes require effective adaptation strategies at local scale in order to mitigate the consequences of extreme events, which are predicted to increase in frequency and magnitude (IPCC, 2014). Adaptation and mitigation strategies for the cities involve a paradigm shift, from a current approach based on the resistance of human settlements (e.g. through building new infrastructures) to an ecosystem-based approach (Ojea, 2015). The latter requires the conservation and restoration of ecosystems for the benefits they provide to humans in terms of Ecosystem services (ESs) (Temmerman et al., 2013), allowing local communities to adapt to climate change.

Green infrastructures are defined as 'all natural, semi-natural and artificial networks of multifunctional ecological systems within around and between urban areas, at all spatial scales' (Tzoulas et al., 2007). This definition includes a wide range of vegetation types that are characterized by different structures, providing a bundle of different ESs. Among these, several regulating services are particularly relevant for the human well-being in urban ecosystems, and include, among others, climate regulation, air quality regulation, water flow regulation, water depuration (Haase et al., 2014). The water regulation service refers to the regulation of water flows on earth surface to maintain the normal levels in the watershed (De Groot et al., 2002). When this definition is applied to urban contexts, regulation of water fluxes mainly concern rainfall-generated runoff and flooding (Gómez-Baggethun et al., 2010; McPhearson et al., 2014), which can cause severe damages to public and private assets and affect life quality and safety of citizens (Hammond et al., 2015). Conventional sewer systems are often ineffective to manage the storm water amount during peak events, as they are not designed considering climate change projections (Ashley et al., 2005). For this reason, the role of green infrastructures in delivering services in urban contexts is gaining particularly attention in scientific research and urban planning (Haase et al., 2014). The increasing of urban population and urban areas led to a decrease of ecological quality of the cities (Haaland and van den Bosch, 2015), pushing local governance to adopt measure for improving urban environment and the potential set of benefits deriving from urban green infrastructures.

An increasing number of studies was carried out to describe the role of urban green infrastructures in the delivery or potential delivery of ESs (Endreny et al., 2017; Pappalardo et al., 2017; Pulighe et al., 2016;

Wang et al., 2014). However, very few studies attempted to propose tools to implement such information in the decision-making processes (e.g. Kabisch, 2015; Nikodinoska et al., 2018). Burkhard et al. (2012) appointed that the joint analysis of the ES demand and the supply can effectively been used to inform decision-making processes around ES delivery. The supply of ES is defined as the capacity of a certain area or ecosystem to provide ESs (Burkhard et al., 2012), while the demand of ES is defined as the ES that are currently recognized or required by beneficiaries or end user (Wolff et al., 2015).

Generally, the transferring of ES concept from theory to practice is one of the most important challenges in ES science (Gissi et al., 2015). Different methods for mapping ES provision capacity (i.e. ecological functions) are presented in literature, including direct measures, proxy indicators and models (Egoh et al., 2012), but they are rarely combined with the analysis of ES demand (Wolff et al., 2015). For instance, the targets fixed by energy plans can be used to quantify the demand for bioenergy provision (Gissi et al., 2017). The assessment of ES demand differs according the purpose of the analysis. Wolff et al. (2015) classify demand types in four typologies: risk reduction, preferences and values, direct use or consumption of goods and services. Among these, the need for risk reduction can be used for assess demand of flood mitigation (Liquete et al., 2013; Nedkov and Burkhard, 2012).

The identification of priority areas for ES management and protection can be an effective approach to implement ES into local decision-making processes to adapt to climate change. For example, Snäll et al. (2016) demonstrated that spatial conservation prioritization could represent a suitable tool for green infrastructures design, allowing cost-effective allocation of conservation efforts. Verhagen et al. (2017) mapped capacity and demand for five ES at European level. They found that ignoring ES demand led to the localization of priority areas in remote regions where benefits from ES capacity to society were small. Thus, the implementation of ESs into urban planning requires the spatial assessment of the supply and the demand of the ESs.

This study aims to provide a suitable tool for urban planning by modelling the provision capacity and demand for urban flooding reduction, by integrating remote sensing techniques and statistical census data, under the framework of mitigation measures to climate change effects. Our study focuses on the ecosystem service of urban flooding reduction, defined as the capacity of urban green infrastructures to absorb urban stormwater runoff produced during rainfall events, in order to mitigate potential flooding events in urban ecosystems. Among the different typologies of green infrastructures, this study considers the green spaces lying within the urban area, intended as all the impervious surfaces characterized by vegetated soils, including both private and public properties.

In the following section, we firstly introduce the method for mapping the assessing flooding reduction supply of urban green infrastructures. A high-resolution urban digital model was used to distinguish between permeable and impermeable areas at fine scale, with a precision of 25 cm. Then, we classified the not-built areas for their soil coverage with respect to the vegetation, materials of construction and ground coverage. Different hydrological models, including SCS method, are applied to assess the runoff generation at catchment scale. When the application of SCS method is limited to rainfall-generated flooding in an urban context, the analysis can be also carried out in a limited part of the watershed, without considering three-dimensional effects (i.e. slopes). Moreover, for this reason, in this study runoff and flooding generations were considered as synonyms. We applied the SCS method, based on Curve Number (CN) (USDA - Soil Conservation Service, 1972), in order to quantify the urban runoff, and then we calculate the flooding reduction capacity ES through two indexes: i) the amount of runoff reduced by green spaces (Δv) (Zhang et al., 2012), and ii) the runoff reduction coefficient (Cr). Secondly, we analyze the flooding reduction demand through a Vulnerability Index (VI), which represents the vulnerability of local population and buildings to urban flooding. The method was applied to the historical urban center of Dolo, a highly urbanized area in Northern Italy. The analysis was replicated for 24 scenarios of rain events, emerging from the combination of three factors: i) precipitation depth, ii) antecedent moisture condition of soils and iii) conditions of initial abstraction. We finally contrast the flooding reduction ES supply of urban green infrastructures and the respective ES demand in order to provide insight to local decision makers to identify priorities of intervention at the scale of urban design to mitigate potential flooding events in the historical urban center of Dolo.

2. Methods

2.1 Study area

The area under analysis is the historical urban center of the Municipality of Dolo (coordinates 45°25′29.57″N 12°04′32.92″E), located inside the Metropolitan Area of Venice, Italy. Dolo Municipality covers an area of 24.8 km2 and account for about 15,000 inhabitants, of which 4,226 live inside the historical urban center (1.67 km2).

The climate of the study area is classified as B1 (Humid) according to the Thornthwaite classification (Feddema, 2005), the average temperature is 13.2°C. The mean annual rainfall ranges within 600 and 1100 mm yr-1, with an annual average of 912 mm. Considering average monthly values, the rainiest month is May (94.4 mm), while the driest is January (49.9 mm). The hydraulic soil group of this area is classified as B type according to the USDA-NRCS classification (NRCS, 1986). Soils belonging to this category typically has between 10% and 20% clay and 50% to 90% sand with a loamy sand or sandy loam texture (NEH, 2009).

The historical urban center of Dolo is frequently subjected to urban flooding events because of insufficient urban drainage network and large amount of impervious surfaces (Municipality of Dolo, 2012). Evidence of the effects of climate change has been detected and studied by Bixio (2009). The analysis of the rainfall dynamics in the Venice lagoon drainage basin on the last 60 years (1956-2010) have put in evidence the rapid intensification of pick events concentrated in time and space around the Venice lagoon inshore, where the study area lies, on a yearly decrease of rainfall level. Several recurrent peak events were recorded yearly from 2007 around the month of September (Municipality of Dolo, 2012). Several damages were produced, as the urban drainage network is not designed to evacuate high amount of rainfall in a short period.

2.2 High-resolution urban digital model

The soil coverage of the study area was mapped by processing spatial data obtained from LiDAR (Light Detention and Ranging) survey with ArcGis 10.3 (ESRI).

An aerial survey commissioned by the Metropolitan City of Venice Administration in 2014 produced 4,000 high-resolution images. Then, a 3D digital model of the area was created with the Dense Image Matching technique (Hirschmüller, 2008). Raster images -DSM (Digital Surface Model) and DTM (Digital Terrain Model) were generated with a precision of 25 cm (Pixel 0.25 m). The DSM reports the altimetric data of all natural and anthropogenic elements (namely impervious) in a specific area, while the DTM reports the morphology of the territory without anthropogenic elements and vegetation (Maragno et al., 2015). Finally, a digital atlas was created in order to distinguish between permeable and impermeable areas every 25 cm. The Atlas reports also the height of each element, from which it was possible to calculate the volumes of natural and anthropogenic elements.

This information was used to depict and map pervious and impervious elements, both public and private, within the historical urban center of Dolo and their relative height.

2.3 Green infrastructures and soil coverage classification

Since Remote Sensing surveys provided tridimensional datasets, the urban vegetation was firstly classified in two classes according to the height: tall vegetation (>1.5m) and short vegetation (<1.5m). Moreover, a field survey was performed in order to improve the digital atlas and correct bias due to the presence of roof vegetation. The areas detected as tall vegetation (i.e. urban trees) were reclassified depending on the observed underneath ground coverage. This correction is fundamental because soil physical properties are the most important factor affecting urban flooding and runoff (Holman-Dodds et al., 2003). The impervious surfaces were further classified by the materials type for their capacity to retain water and promote infiltration through field survey. Descriptions for each ground coverage type are presented in Tab.1.

The analysis was carried out at i) property unit and ii) urban section levels. Firstly, the study area was classified in private and public properties. Subsequently, private areas were mapped in polygons representing single property units. Public areas were subdivided according to zone destinations adapted from Panduro & Veie (2013). Urban sections boundaries were derived from the Italian National

Institute of Statistic (ISTAT) and represent the minimum territorial units inside municipalities where census records are collected.

Tab.1: Land cover classes and respective Curve Number values

Ground cover class	Description	CN	References
Gravel		85	USDA – NRSC, 1989
Semipermeable blocks		97	USDA – NRSC, 1989
Concrete		98	USDA – NRSC, 1989
Asphalt		98	USDA – NRSC, 1989
Rubberway pervious pavement		97	Shirini and Imaninasab (2016)
Other impervious		98	USDA – NRSC, 1989
Bare soil		86	USDA – NRSC, 1989
Agricultural land	as Straight row at poor condition	81	USDA – NRSC, 1989
Short vegetation		61	USDA – NRSC, 1989
Open space with tall vegetation	as Open space at poor condition (grass cover >50%)	79	USDA – NRSC, 1989
Water		0	USDA – NRSC, 1989

2.4 Runoff calculation

The water regulation service was modeled using the SCS method, based on Curve Number (CN) (SCS 1972). This model estimates rainfall-runoff based on land coverage, soil type and precipitation. The SCS model can be successfully applied to urban contexts with acceptable results. The model calculation is based on three parameters (precipitation, initial abstraction and potential maximum storage of soil) and it is based on the following equations:

$$Q = \begin{cases} (P - I_a)^2/(P - I_a + S), & P \geq I_a \\ 0, & P \leq I_a \end{cases} \qquad \text{eq.(1)}$$

$$S = \frac{25400}{CN} - 254 \qquad \text{eq.(2)}$$

$$I_a = \lambda \cdot S \qquad \text{eq.(3)}$$

Where Q is the rainfall runoff depth (mm), P is the precipitation depth (mm), S is the potential maximum water storage in soil (mm), CN is the tabulated value of Curve Number (dimensionless) ranging from 0 to 100, Ia is the initial abstraction of rainfall (mm) and λ is the initial abstraction coefficient (constant).

The CN values was derived from (SCS 2009), and mainly depends on corresponding soil coverage type, Hydrological Soil Group (HSG) and Antecedent Moisture Condition (AMC). The higher is the CN value the higher is the runoff generate by a rain event. The HSG of Dolo municipality soils was classified by ARPAV (2009) within the category B. Soils belonging to this category typically has between 10% and 20% clay and 50% to 90% sand with a loamy sand or sandy loam texture (NEH-630, 2009). NRCS classifies three AMC classes, representing the relative moisture before the rainfall event: “dry” (AMC-I), “moderate/normal” (AMC-II) and “wet” (AMC-III) conditions. Since soil absorption capacity is lower in wet soils, CN values attributed to a specific soil coverage with a specific HSG should be corrected for the antecedent conditions (CN for AMC-III>AMC-II>AMC-I).
The initial abstraction coefficient (λ) is usually defined equal to 0.2 (SCS 1985). However, some studies proposed different values, particularly for urban areas (Lim et al 2006; Ling et al. 2014).
The contribution of urban ecosystems to runoff generation and mitigation was calculated both at property and at urban section levels. At property level, the overall CN for each unit was calculated as weighted average of values for respective areas, as follow:

$$CN_p = \frac{\sum_i CN_i \cdot A_i}{\sum_i A_i} \qquad \text{eq.(4)}$$

Where the CN_p is the curve number of the property *p*, CN_i and Ai are the curve number and the area of the *i* soil coverage type, respectively.
At urban section level, the total runoff values were computed considering the values of each polygon:

$$Q_s = \sum_p Q_p \qquad \text{eq.(5)}$$

Where Q_s and Q_p are the runoff depth (mm) of the *s* urban section and of *p* property, respectively.

2.5 Mapping runoff mitigation capacity

In order to estimate the contribution of each spatial unit to runoff reduction, two indexes were calculated: the amount of runoff reduced by green spaces (Δv) (Zhang et al. 2012) and the runoff reduction coefficient (Cr) (Yao et al. 2015). These indexes express the reduction of surface runoff provided by the presence of urban green spaces and therefore are proposed in this analysis as a proxy for the capacity to deliver the ESs of flood prevention. Δv quantifies the general benefit provided by green spaces, in terms of runoff volume reduction, calculated as follow:

$$\Delta_v = \sum_i 0.001 \cdot (Q_b - Q_i) \cdot A_i \qquad \text{eq.(6)}$$

Where Δv is the runoff reduction (m^3), Q_b is the runoff depth (mm) generated by an hypothetical scenario where green spaces are replaced by 100% impervious surfaces (CN=98), Q_i is the runoff depth (mm) generated by the *i* urban green space type and A_i is the area of urban green space *i* within a spatial unit.
Cr represent the efficiency in runoff reduction. A higher Δv means greater potential hydrologic benefits provided by urban green space, whereas a higher Cr indicates less need to improve future urban rainwater management in a specific area. The index was calculated as follow:

$$C_r = \Delta_v \cdot (0.001 \cdot P \cdot A)^{-1} \qquad \text{eq.(7)}$$

Where Cr is runoff reduction coefficient, Δv is the runoff reduction (m^3), P is the daily rainfall depth (mm) and A the area (m^2) of the study unit.

2.6 Scenario analysis

The runoff mitigation capacity was evaluated for different scenarios, which simulate the potential conditions of rainfall and runoffs generation depending on three factors: i) rainfall depth (Pi) (four precipitation values); ii) the AWC (AWC-I-II-III), and iii) the Ia values (0.02 and 0.005). The analysis of Δv and Cr was performed for the 24 scenarios deriving from the combination of the three factors (Pi, AWC, I_a).

The four precipitation depths considered in the analysis were 10, 45, 90 and 160mm (Tab.2). These values were chosen according their relevance on planning decisions. In fact, the 10 mm and 45mm precipitation values correspond to the mean and high average daily values, respectively. A three-hourly precipitation depth of 90mm was calculated by the Municipal Water Plan as the capacity limit of the urban drainage network. Finally, the rainfall depth of 160mm was the most extreme event recorded (2009) by the Regional Agency for Environmental Protection (ARPAV).

Tab.2: Precipitation scenarios considered for simulations

Precipitation depth (mm)	Description	Frequency	Source
10	Mean daily value (per hour)	69 per year	ARPAV
45	High daily value (per hour)	32 per year	ARPAV
90	Capacity limit of the urban drainage network (on 3 hours)	20 years	Municipal Water Plan
168	Extreme event (16/09/2009) (per hour)	unknown	ARPAV

2.7 Assessing runoff mitigation demand

Since the flow of ESs is defined as the intersection between the provision capacity (described by the eq.7) and the demand, the latter is fundamental to inform urban planning. The demand for the flood mitigation service was calculated as a function of people and buildings vulnerability:

$$VI_i = X_{1i} + X_{2i} + .. X_{ni} \qquad \text{eq.(8)}$$

Where VI is Vulnerability Index of the i section, X_{ni} are the single vulnerability parameters (adjusted for 0 to 1) of the i section describing local population and buildings vulnerability. The index is based on the expectation that some population (e.g. children, elderly people, etc.) and building categories (e.g. buildings with poor or bad status conditions) are more susceptible to the consequences of urban flooding. All the parameters computed in eq.9 are assumed to have the same importance (i.e. weight).

2.8 Priority areas of intervention

With the aim to address the decision makers, a ranking of the priority area of intervention (i.e. section ranking) was provided:

$$PRI = \frac{VI \cdot Q}{Cr} \qquad \text{eq.(9)}$$

Where VI is the Vulnerability Index, Q is the generated runoff (mm) and Cr is the Reduction Coefficient. VI was calculated per urban section.

We assume that: i) the lower is the Cr (ES supply) the higher is the potential benefit that could be reached by the inclusion of new green infrastructures in the related urban section, ii) the higher are the vulnerability of local population and buildings (ES demand) and the urban runoff (Q) the higher is priority for intervention.

3. Results

3.1. Spatial analysis of green infrastructures

As emerging from the analysis of the high-resolution urban digital model, the urban green spaces are unevenly distributed in the historical urban center of Dolo (Fig.1). The pervious areas cover the 43.6% of the total study area. Small patches of tall vegetation mainly characterize the pervious areas within Dolo urban center, except for an urban park at the east side of the study area with tallest vegetation (up to 29m). Tab.3 shows the ground cover distribution for the study area, subdivided in public and private properties.

Overall, the green urban spaces are fragmented and scattered within the high-density urban fabric.

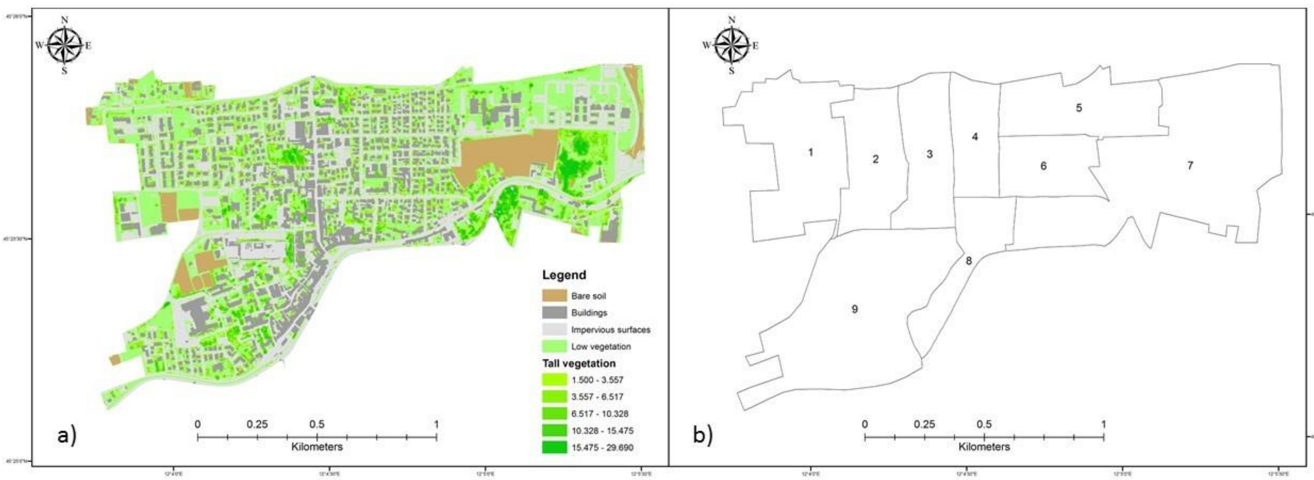

Figure.1: Ground cover and vegetation height of the study area (a) and urban section boundaries (b) of the study area.

Tab.3: Public, private and total areas for each land cover class

	public areas		private areas		Total area	
	m^2	%	m^2	%	m^2	%
Gravel	33172.6	5.39%	34199.4	3.27%	67371.9	4.05%
Semipermeable blocks	13689.5	2.22%	34215.8	3.27%	47905.4	2.88%
Rubberway pervious pavement	2477.9	0.40%	0.0	0.00%	2477.9	0.15%
Concrete	50876.7	8.27%	70171.7	6.71%	121048.4	7.28%
Asphalt	78434.0	12.74%	24791.1	2.37%	103225.1	6.21%
Bare soil	12283.7	2.00%	108248.3	10.35%	120532.1	7.25%
Agricultural land	0.0	0.00%	1741.7	0.17%	1741.7	0.10%
Open space with tall vegetation	52470.2	8.53%	160629.6	15.35%	213099.8	12.82%
Water	32706.8	5.31%	0.0	0.00%	32706.8	1.97%
Other impervious	11880.7	1.93%	1930.4	0.18%	13811.2	0.83%
Build-up	81419.1	13.23%	287476.7	27.48%	368895.9	22.20%
Short vegetation	246067.2	39.98%	322749.4	30.85%	568816.5	34.23%
Total	615478.5	100%	1046154.1	100%	1661632.6	100%

3.2. ES provision capacity

Figures 2 shows the spatial distribution of the flood (runoff) depth generated under the 24 different scenarios, considering the combinations of AWC, Ia and precipitations. The results at section level are presented in Tab.4. The relative contributions of public and private areas to the total runoff roughly follow those of the respective total areas.

The capacity of urban green spaces to mitigate rainfall runoff was modeled calculating the Cr (eq.7) under different values of AWC and Ia (Fig.3). Public and private surfaces contribute differently to the total runoff and showed different performances of flooding reduction. In eight urban sections out of nine, private properties generate larger amount of rainfall runoff than public ones under the worst scenario conditions (Tab.4). The same trend can be observed also for runoff depth and for Cr (i.e. the contribution of green spaces to reduce rainfall runoff), with higher ES performances for public areas than private proprieties.

Since Cr depends on precipitation, as well as on soil conditions and initial abstractions, the index was modeled under a rainfall gradient for the different AWC and Ia values (Fig.4a,b). This analysis allows identifying the optimal precipitation value to which the ES provision capacity of the study area is maximum. The maximum Cr values range within 0.345 and 0.389 (observed at the rainfall values of 13 and 103mm, respectively), according to the different combinations of biophysical factors.

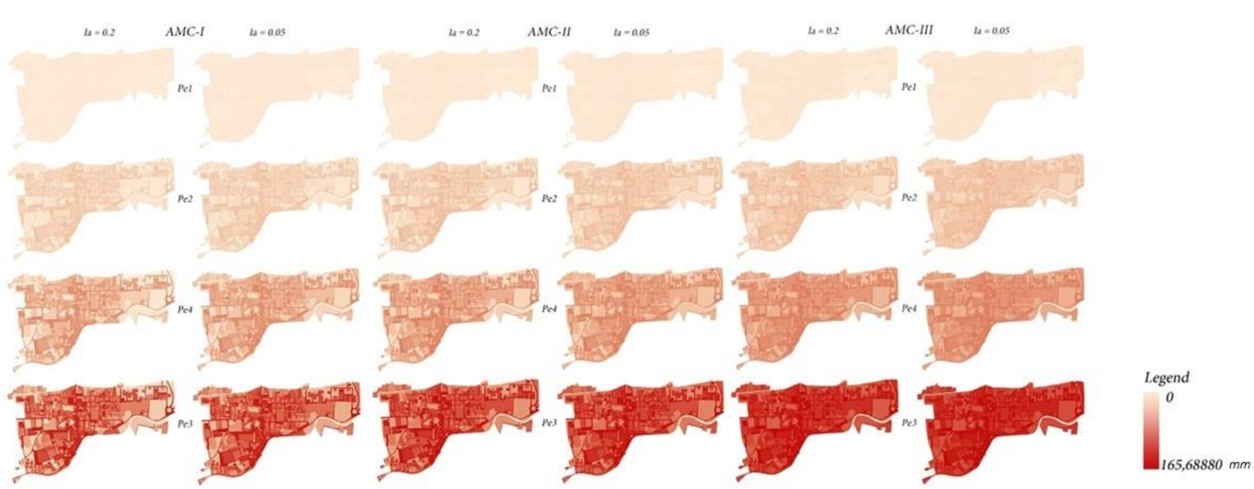

Figure.2: Modelled runoff (mm) for each of the 24 considered scenarios.

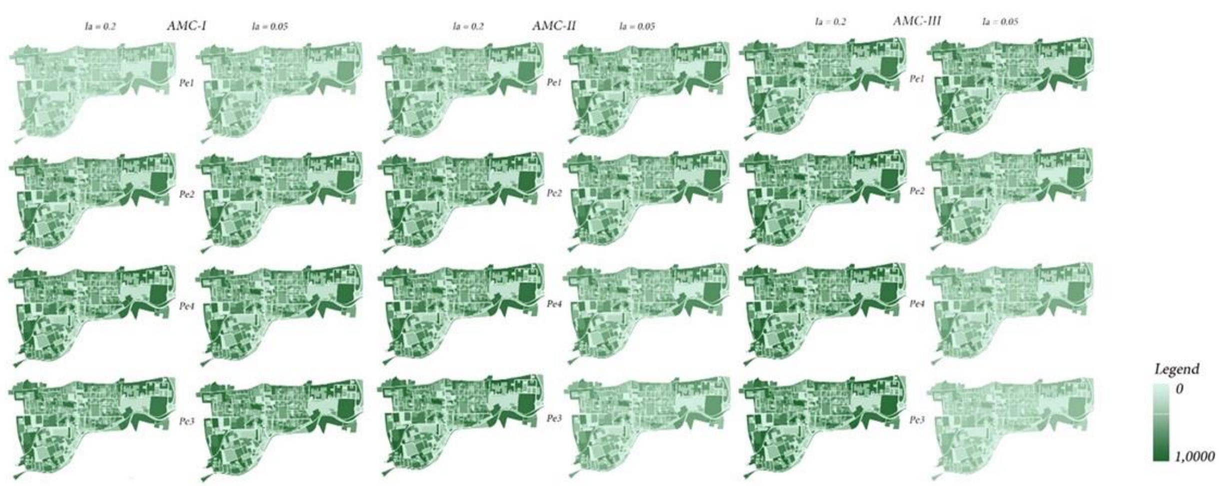

Figure.3: Coefficient of Reduction (Cr) for each of the 24 considered scenarios.

Tab.4: Areas, runoff (both in mm and m3) and Cr for each urban section. Contributions of public and private areas are reported.

Section number	area (m^2)			runoff (mm)			runoff (m^3)			Cr		
	pub	prv	tot	pub	prv	tot	pub (%)	prv (%)	tot	pub	prv	tot
1	50.90%	49.10%	275976.2	57.78	64.58	61.12	48.12%	51.88%	16868.13	0.27	0.19	0.23
2	23.71%	76.29%	128358.1	63.11	74.73	71.98	20.79%	79.21%	9238.66	0.20	0.14	0.16
3	20.21%	79.79%	116444.5	59.91	72.86	70.24	17.24%	82.76%	8179.07	0.15	0.08	0.10
4	14.16%	85.84%	84137.8	68.39	70.97	70.61	13.71%	86.29%	5940.74	0.15	0.10	0.11
5	44.07%	55.93%	138182.3	64.48	66.72	65.73	43.23%	56.77%	9083.14	0.19	0.17	0.18
6	21.55%	78.45%	89968.7	72.25	70.10	70.56	22.06%	77.94%	6348.13	0.13	0.12	0.12
7	25.00%	75.00%	433922.8	52.96	61.99	59.73	22.17%	77.83%	25919.47	0.20	0.13	0.15
8	31.43%	68.57%	85285.1	59.83	76.96	71.58	26.27%	73.73%	6104.60	0.07	0.06	0.06
9	62.56%	37.44%	309357.0	67.03	68.44	67.54	62.08%	37.94%	20894.95	0.11	0.12	0.12

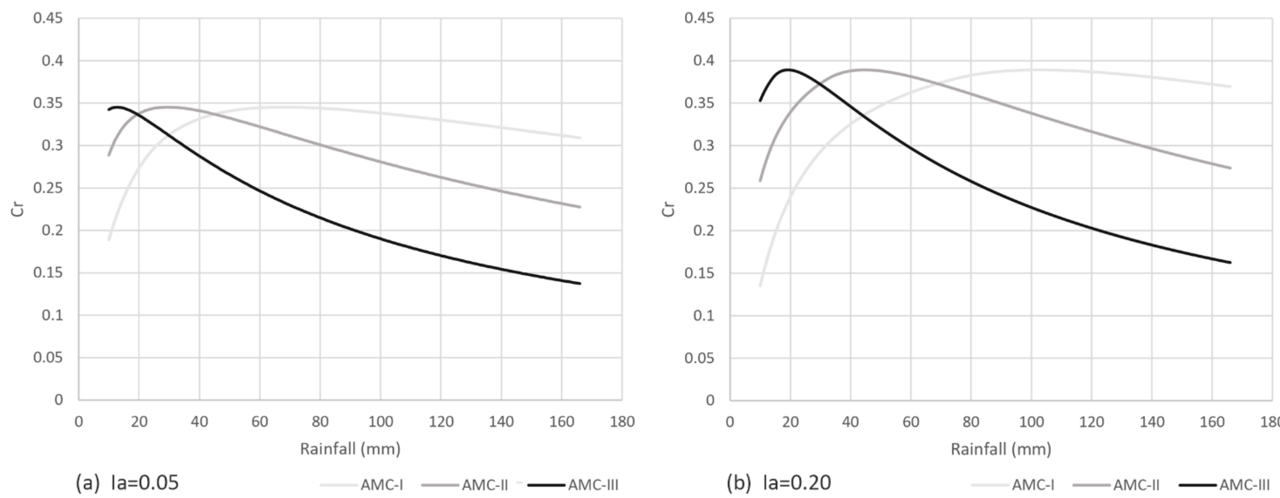

Figure.4: Variation of the Coefficient of Reduction (Cr) along a rainfall gradient for the different values of AMC and with Ia = 0.2 (a) and Ia = 0.05 (b).

3.3. Flooding reduction demand

The vulnerability index (VI) was calculated for all the urban sections of the municipality of Dolo (Fig.5, Tab.5). The sections with the highest VI values are the no.5, which lie in the northeastern part of the study area. This is due particularly to the scores related to population and, just in lesser extent, to buildings' conditions. The center population is densely inhabited by more vulnerable categories of people, such as children and elderly. The sections with a higher demand of the flooding reduction service are those with the larger percentages of public properties (sections 5 and 9) (Tab.5).

The capacity of green infrastructure to mitigate rainfall runoff was modeled calculating the Cr (eq.7) under different values of AWC and I_a. This index expresses the capacity of pervious surfaces of each area (i.e. property unit or section) to avoid flooding events and, therefore, can be considered as an indicator for the ES of flood prevention. Public and private surfaces contribute differently to the total runoff and showed different performances of runoff mitigation. In eight sections out of nine, private owned areas generate larger amount of rainfall runoff than public ones (Tab.4). The same trend can be observed also for runoff depth and for Cr (i.e. the contribution of green spaces to mitigate rainfall runoff), with higher ES performance for public than private owned zones.

Since Cr depends on precipitation, as well as on soil conditions and initial abstractions, the index was modeled under a rainfall gradient for the different AWC and I_a values (Fig.3 a,b). This analysis allows identifying the optimum precipitation value to which the ES provision capacity of the study area is maximum. The maximum Cr values range within 0.345 and 0.389 (observed at the rainfall values of 13 and 103mm, respectively), according to the different combinations of biophysical factors.

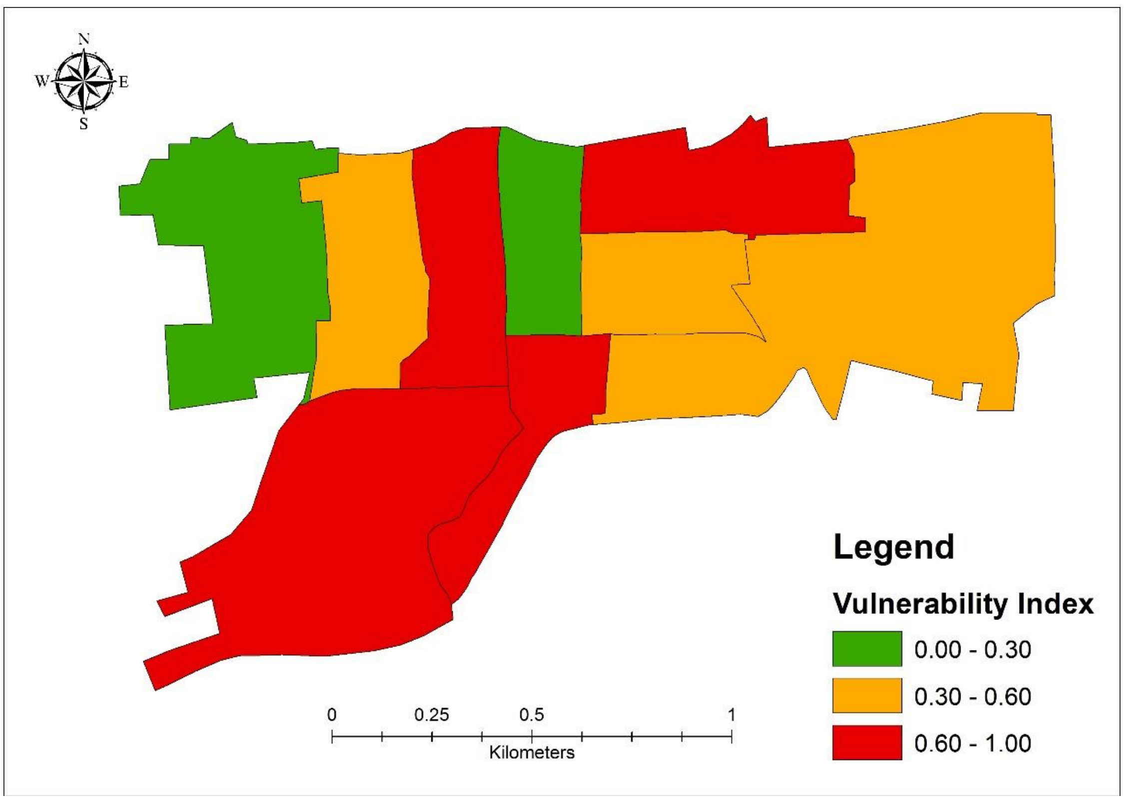

Figure.5: Vulnerability Index (VI) values obtained for each urban section.

3.3. ES demand

The vulnerability index was calculated for all the urban sections of the municipality of Dolo (Fig.4). Large part of the sections with the highest VI values lie in the urban center, confirming that this zone is the one with the highest ES demand. This is due particularly to the scores related to population and, just in lesser extent, to buildings' conditions. The center population is dense and includes vulnerable people, such as children and elderly persons. Tab.5 shows the PRI values calculated for the four rainfall depths considered. The sections with higher demand are those with the larger percentages of public properties (sections 5 and 9) (Tab.5).

3.4. Priority rank of intervention

The priority rank of intervention (PRI) represent the urgency to improve green infrastructure, as a function of ES provision capacity and demand, calculated at urban section level (eq.9). Tab.5 shows the PRI values calculated for the four rainfall depths considered in the scenarios, considering with AMCIII and Ia=0.05. The runoff and Cr are those calculated at the rainfall depth of 90mm (corresponding to the critical threshold for drainage network according to the Municipality Water Plan (Municipality of Dolo, 2012), with AMCIII and Ia=0.05. These values for Antecedent Moisture Condition and initial abstraction were chosen for the priority rank calculation according a conservative approach. In fact, they represent the worst possible conditions for runoff mitigation. The values for PRI, as well as for the single components (VI, runoff and Cr), are shown in Tab.5. According to the results, the urgency of interventions for improving the green infrastructures in the study area should be focused on sections 8 and 3. These sections show higher values of generated runoff and the lower for Cr. Despite section 5 has the highest VI value, its low ranking is due to a good level of Cr performed, meaning that even though this area is vulnerable, the urban green spaces can mitigate flooding events. Section 1 and 7 are those with the lower ranking score, meaning lowest priority of intervention.

Tab.5: Vulnerability index (VI), runoff and Cr values for each urban section

Section number	VI	P = 10mm Runoff	Cr	PRI	P = 45mm Runoff	Cr	PRI	P = 168mm Runoff	Cr	PRI	P = 90mm Runoff	Cr	PRI
1	1.00	1.12	0.39	**2.88**	1.15	0.31	**3.66**	1.13	0.16	**7.30**	1.11	0.23	**4.84**
2	1.46	1.58	0.27	**8.68**	1.00	0.21	**6.82**	2.00	0.11	**27.68**	2.00	0.16	**18.68**
3	1.64	1.77	0.17	**17.50**	1.80	0.13	**22.00**	1.69	0.07	**41.93**	1.86	0.10	**31.14**
4	1.24	1.78	0.18	**12.40**	1.83	0.14	**15.82**	1.69	0.07	**29.68**	1.89	0.11	**22.32**
5	2.00	1.45	0.30	**9.55**	1.48	0.25	**12.09**	1.41	0.12	**23.35**	1.49	0.18	**16.64**
6	1.49	1.73	0.21	**12.18**	1.80	0.17	**15.69**	1.75	0.08	**30.97**	1.88	0.12	**22.55**
7	1.41	1.00	0.26	**5.53**	1.07	0.21	**7.34**	1.00	0.10	**13.91**	1.00	0.15	**9.39**
8	1.61	2.00	0.11	**29.30**	2.00	0.09	**36.31**	1.60	0.04	**59.10**	1.97	0.06	**48.93**
9	1.69	1.52	0.20	**13.12**	1.61	0.16	**17.21**	1.50	0.08	**32.62**	1.64	0.12	**24.01**

4. Discussion

The study proposes a methodology to inform urban planning through the results of ES assessment, by integrating the analysis of ES supply and demand for the case of the urban flooding reduction capacity. The method aims contributing to fill the gap between ES theory and practice in urban areas.

The analysis of ES provision capacity spatially maps the potential of urban green spaces to mitigate urban runoff. These values were combined with vulnerability index to obtain the priority rank of intervention (PRI) for urban sections. Based on the results, several addresses for urban planning may be outlined. The sections 8 and 3 are those with higher PRI values. High PRI scores for these sections are due to the lowest Cr and high runoff values, rather than high levels of vulnerability. This means that interventions in these zones should be aimed to increase the extension and quality of green spaces. The lower scores were observed for the section 1 and 7. Section 1 showed the lower level of vulnerability and the higher performance of Cr, suggesting that the conservation of current level of ES may be obtained by the maintenance of current green infrastructures, while low priority for the section 7 was mainly due to the lower value in runoff. Specific attention should be payed to section 5, observed as the most vulnerable. In fact, even though the analysis did not highlighted high priority of intervention, the higher observed level of vulnerability suggests monitoring this area with particularly attention in case of rainfall peak events.

Since the higher runoff amount is generated by private surfaces, an incentive-based mechanism for private owners may be effective in promoting the increase of green spaces and pervious surfaces in the study area in general and in the urban areas potentially more prone to runoff generation. For example, the Biotope Area Factor (BAF) (Bauen & Becker, 1990) is an urban greening policy tools designed to improve the ecosystem's functionality and improve the development of biotopes in city centers. Measures can be applied also to public areas to improve their contribution to urban runoff mitigation.

The installation of rain gardens is demonstrated to increase water infiltration and absorption by collecting water form parking, roofs and other pervious surfaces (Davis et al., 2009).

The increase and maintenance of green spaces would also lead to multiple benefits provided by urban vegetation, as air depuration from chemical and particulate pollution (Grote et al., 2016), microclimate regulation by mitigating heat waves (Gillner et al., 2015), carbon storage (Fares et al., 2017) and improving of urban landscape aesthetic (Southon et al., 2017). In addition, efforts for increasing people awareness to the role of green infrastructures to support human well-being may be effective to orient citizens' choices in their private proprieties.

Due to the extreme heterogeneity of build-up areas, high-resolution data are needed to capture different components of urban green spaces at fine scale, such as urban vegetation of trees, grass and bushes, which are providers of different ecological functions (Davies et al., 2011; Jim and Chen, 2008). The use of LIDAR-derived information allowed the mapping of vegetation structure at fine scale. When these data are modelled and integrated with information concerning local ES demand, the analysis can inform urban planning by prioritizing measures and actions at urban section scale where flooding reduction is more urgently needed, as resulting from the ES demand and supply analyses.

In general, the use of remote sensing technics to map ESs is a key challenge for the ES science (Dawson et al., 2016). Many studies process images derived by passive sensors (e.g. satellite scenes) to obtain land cover maps of vegetation indexes, used to model ES in space and time (De Araujo Barbosa et al., 2015). The use of products derived by the use of active sensors (Lefsky et al., 2002), such as LIDAR, allows the access of three-dimensional information on vegetation and, for this reason, has greater potential in mapping vegetation structure at finer scale. Detecting spatial variation of vegetation structure is fundamental for mapping ESs in urban ecosystems (Lehmann et al., 2014). In this study, information on vegetation height was used to discriminate areas covered by tall vegetation from grass and bushes (grouped as "short vegetation"). However, field surveys were necessary to avoid erroneous interpretations for tall vegetation, due, for example, by the presence of single trees growing on small flowerbeds surrounded by impervious surfaces, rather than by permeable soil (e.g. grass).

Some limitations related to the SCS method for runoff modelling should be considered. For example, specific values for CN and initial abstraction should be calibrated on measured data in the study area to obtain reliable results. Since no empirical data were available for the study area, the uncertainties related to the AWC and I_a parameters were managed by combining all the possible values in the 24 scenarios, to obtain a final range of values. AWC varies according to the previous climatic conditions and can be adjusted to describe the moisture conditions according to Ward & Trimble (2004). Furthermore, values for initial abstraction (I_a) can differ largely as well. The Soil Conservation Service defines Ia equal to 0.02. Nonetheless, some studies highlighted that this value could not be reliable when applied to urban landscapes (Lim et al., 2006). Ling & Yusop (2014) showed that the most adopted value for this parameter in urban contexts is 0.05. Because of the above-mentioned uncertainties, the analysis was performed accounting a set of the combinations of AWC (AWC-I-II-III) and I_a values (0.02 and 0.005). Additional limitations concern the intrinsic simplification of the model. For instance, the slope and barriers to water flows were not considered. Hydrological behavior of impervious surfaces is complex to predict, as preferential runoff paths are the result of interactions with drainage systems, as well as presence of temporary and permanent barriers (Fletcher et al., 2013). In this study, the SCS model was applied on the urban historical center of Dolo, which represents only a portion of the urban catchment area. In this case, the runoff depends only on the excess of rainfall in the area, and not to the contribution of rainfall coming from other areas of the catchment as for Dolo. Moreover, urban soils are difficult to be studied and sampled, because they are usually severely disturbed and show highly variable infiltration rates (Pitt et al., 1999). Our method overcomes the difficulties in sampling disturbed urban soils by considering pervious and impervious surface, which influence runoff formation (Yang et al., 2015). In any case, the output of the proposed method provides a qualitative ranking of the different zones (i.e. urban sections) for priority of intervention, rather than a quantitative assessment for drainage infrastructure calculation. Moreover, the use of normalized values, as in the case of PRI computation, can mitigate the bias due to the use of indirect measures in ES studies (e.g. Gaglio et al., 2017).

The elaboration of the results obtained from the analysis performed at private property scale leads to issues related to the use and publication of data concerning private properties. For this reason, the outcomes of this analysis should be strictly managed by public bodies under protocols dealing with confidentiality and privacy regulation, in order to guarantee the correct use as well as the privacy of citizens' data.

It has to be mentioned that the PRI considers all the parameters as equally important (see eq.9). However, the index can be corrected according stakeholder perception and/or local conditions through attributing different weight at the single component.

Finally, the Cr response along a rainfall gradient was studied under the different combinations of AMC and I_a (Fig.4a and b). Since climate change are expected to increase the frequency and the magnitude of rainfall extreme events, the response of Cr along a precipitation gradient can be used to project the response of runoff mitigation service to future climate change. Fig.4a and b suggest that the capacity of green infrastructures of the study area to regulate extreme events will decrease as a consequence of climate change. In fact, Cr values tend to decrease together with the increase of rainfall depth.

The effects of AMC seems more sensible to the rainfall variation, while those due to I_a values are more relevant for determining the maximum value of Cr. For this reason, direct measures of both soil condition and initial abstraction are important for the accuracy of runoff mitigation. Maximum Cr values correspond to the breaking point of the curve, after which the Cr performance declines (Fig.4a and b). The saturation of water retention capacity of urban green spaces during severe rainfall events and the consequent release of the excess surface water more quickly mainly causes this decrease (Yao et al., 2015). Information on the rainfall depths corresponding to the maximum Cr values represent the threshold, over which the ES declines. These thresholds may be used to activate specific early-warning measures to protect the most vulnerable population from flooding.

The different performances of rainwater retention capacity under the three different soil moisture conditions suggest that ES provision depends on the climatic conditions occurring before the extreme event. Practically, urban planning decisions should carefully consider precipitation pattern variations within the context of climate change, in order to improve adaptation strategies.

5. Conclusions

Climate change force decision-makers to adaptation governance measures to mitigate risks related to the increased frequency of extreme events. Transferring ES assessment from theory to practice has great potential to support and inform such decisions, especially in urban areas. The development of tools that integrate ES supply and demand is a key challenge for the future planning and management of the most vulnerable areas, such urban ecosystems. This study elaborated a method for the identification of priority areas of intervention for the management of urban green infrastructures, which could be applied even without the availability of runoff field measurement. The qualitative approach can assist cost-effective measures to prioritize the need for green infrastructure management in different zones. The analysis also provides the estimation of ES demand under climate change. The projected increase of precipitation intensity is likely to overcome the green infrastructure capacities to mitigate rainfall runoff. Information on ES responses to climate change are fundamental to inform environmental managers towards more sustainable governance.

Remote sensing techniques gave a great potential to support ES supply mapping in urban centers, by providing high-resolution maps and information that are fundamental in heterogeneous and complex ecosystems as urban areas.

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4. Discussion

Mapping ESs provides scientific arguments to inform decision-makers and support policy actions. For this reason, ES theory can be implemented in different applications. Through different case studies (presented in section 3), located in different environments and socio-economic contexts, the present research attempted to answer to the research questions stated in Section 1.6.

Mapping ES in space and time provides outcomes concerning the effects of conservation efforts on human well-being. As discussed in Section 1.4, this is fundamental to guarantee sustainable development of PAs and surroundings. Moreover, this is also required by the paradigm shift on PAs and conservation (Watson et al. 2014).

Based on the three case studies proposed (Sections 3.1, 3.2, 3.3.), a new hypothesis can be stated: passive protection is not effective to maintain ecological functions and therefore to support human well-being. The efforts for remaining biodiversity in the respective areas do not result in maintaining ESs supply or can result in significant trade-offs among ESs.

In the case of Volano-Mesola-Goro Station in the Po river delta (Section 3.1), the establishment of PA during the second period (1976-2008) failed to maintain the ecological functions of coastal wetlands. Unlike the first period (1954-1976), the loss in ecological function was caused by the indirect impacts of human activities. This constitutes an evidence that conservation strategies should consider and face impacts deriving from nearby and remote zones. The loss of aquatic vegetation observed in the area was likely to lead to negative impacts on biodiversity, as vegetated wetlands support a valuable number of species (Alistock et al. 2001; Kiviat 2013). Therefore, protection efforts more likely failed to conserve both biodiversity and ESs, suggesting a general decrease of ES supply when aquatic biodiversity is lost.

Spatio-temporal changes in tropical mountain environment are the focus of Section 3.2. These zones are subjected to rapid socio-economic transitions, as a common phenomenon in developing countries. At the same time, vegetation recovery rates in these zones are significantly faster than those in temperate climates. For these reasons, significant landscape changes were observed even if the two considered periods (2000-2008 and 2008-2014) were relatively short. Socio-economic drivers led to agricultural land expansion and, after few years, to an almost complete conversion of these areas to pastures (at lower altitude) and forest (at upper altitude). Overall, the analysis captured an increase of natural ecosystems (i.e. forest) during the period with environmental protection (2008-2014), with possible positive effects on biodiversity. In this case, a slight decrease of ESs capacity was estimated even with positive outcomes in terms of biodiversity protection. Since the altitudinal gradient is responsible for the different ESs provided by forested areas, LULC transitions had different impacts at different altitudes.

The study case of National Reserve Paul do Boquilobo (Section 3.3) describes the effects of protection initiatives on specific ESs (carbon storage and water-related services). Increase of stored and sequestered carbon were estimated after the establishment of the Reserve. On the other hand, trade-offs concerning water-related services were observed. Water infiltration, important for groundwater supply recharge, is favored by the presence of arable lands, while natural ecosystems as grasslands and forests increase the capacity to mitigate flood risk at the cost of decreasing water infiltration. These trends are also confirmed by the two alternative projections to 2050, highlighting the importance of future management strategies on the delivery of ESs. Climate change is expected to exacerbate such trade-offs. The predicted decrease of precipitation and increase of temperature may dramatically reduce water availability for groundwater recharge and surface waters.

The different case studies also highlighted other aspects related to biodiversity-ESs relationship and ES trade-offs. The relation between biodiversity and ES supply confirmed to be complex. Increase of biodiversity can be assumed when LULC transitions lead to an increase of natural ecosystems. The findings provided by the three case studies showed that these changes do not necessary correspond to an increase of ESs at landscape level. In the case presented in section 3.2, the overall gain in forested areas were not sufficient to increase ESs. In section 3.3, the natural ecological successions led to a gain in carbon storage and flood risk mitigation and a loss in water infiltration. This means that conservation initiatives generally cause trade-offs among ESs.

Overall, the three study cases confirmed the need for an active management of PAs to maintain and protect ES supply. Active management measures include:

- the maintenance of non-natural ecosystems (e.g. agricultural land) together with natural habitats to guarantee the provision of those services that do not seem to have direct dependence on biodiversity;
- the environmental management of surrounding areas;
- the consideration of socio-economic dynamics in and outside the PAs
- the involvement of stakeholders and their information on the outcomes of ES assessment and mapping to share management decisions.

Even when landscape conservation may be effective to protect biodiversity and ESs, such result is dependent on financial investments and costs. These can be justified by the direct and indirect value of ES provided by areas under protection, in order to guarantee the conservation of biodiversity and ESs themselves. Assessing and mapping ESs provide information for such sustainable processes, for example as a basis for Payment for Ecosystem Services (PES) or restoration efforts. PES schemes are suitable mechanisms for implement such support, involving ES providers and beneficiaries. There is a growing interest in PES scheme applications and some implementations are described in literature, particularly on developing countries. Nonetheless, application of PES schemes suffers for the lack of a clear legal framework that discipline transaction costs, fiscal arrangement, and property rights issues (Fauzy and Anna 2013).

Maintaining a given amount of agricultural land within PAs provides the opportunity to mitigate some ES trade-offs (as demonstrated in the case presented in Section 3.3) and generate incomes for farmers providing goods and services with direct market value (Section 3.2). On the other hand, agricultural activities are envisaged as sources of pollution and this is particularly undesirable in PAs. The harmonization of agricultural activities and conservation targets can be possible with the adoption of proper management practices. In this respect, the EU Common Agricultural Policy (CAP) sustains “greening” actions (i.e. green direct payments) to adopt and maintain farming practices that help meet environment and climate goals. Such practices involve diversifying crops, maintaining permanent grassland and dedicating a proper amount of arable land (at least 5% of farmer properties) to 'ecologically beneficial elements' (i.e. 'ecological focus areas'). Among the latter, landscape features (e.g. hedge and trees) are those providing the best results in terms of potential impact on ESs, as observed by EU report on the implementation of the ecological focus area (EU 2017). Landscape features are defined as “elements subject to cross-compliance like hedgerows, single trees, rows or groups of trees, boundary ridge, ditches, and other landscape elements” (Zinngrebe et al 2017). Future studies should investigate the optimal amount and spatial location of agricultural land within PAs and the effects of extensive agricultural practices on ESs.

The riparian vegetation can significantly contributes to the enhancement of ESs at landscape level, as demonstrated in Section 3.3. Despite the role of riparian habitats in supporting biodiversity and ESs is widely described in literature (González et al 2017), their restoration and management are not currently regulated by specific policy at European level. Rather, it is directly or indirectly included in legal acts and initiatives from the environmental legislation, such as EU Biodiversity 2020, Flood Directive, CAP, Water Framework Directive (WFD) and Habitat and Bird Directives. However, such documents may be not sufficient to support the enhancement of the ESs provided by these habitats. For instance, Vidal-Abarca et al (2016) showed that the use of biological and hydro-morphological indices proposed in the WFD allowed the evaluation of only a limited set of ESs. Moreover, the monitoring of restoration outcomes of riparian vegetation suffers for a lack of long-term studies and needs for incorporating functional approaches (e.g. assess of functional traits), more rigorous experimental designs, enhanced comparisons among projects and reporting failure (González et al 2015). Therefore, some management measures provided in scientific literature could be better implemented at EU level. Kuglerová et al 2014 proposed a site-specific riparian management allowing wider buffers at groundwater discharge areas and more narrow buffers on sites of lower ecological significance (i.e. riparian sites without groundwater flow paths). Specific attention should also be paid to connectivity of riparian habitats (Gray et al 2016) and soil properties of surrounding land (de Sosa et al 2018).

The governance of PAs usually includes prescriptions and limitations of land use to minimize negative effects of human activities on biodiversity. However, negative impacts on both biodiversity and ecological functions may be due to the land use intensity on surrounding areas, as flows of materials and energy between PAs and surrounding occur. In the case presented in Section 3.1, the loss of submerged aquatic vegetation was caused by the increase of upstream land use intensity, both inside and outside the PA.

De Fries et al (2007) introduced the concept of "greater ecosystem", as "ranges of particular species, hydrologic boundaries, or other ecological attributes, characterize very large areas well outside the boundaries of existing protected areas". These authors argued that PA boundaries should be delineated within the biophysical gradients of the greater ecosystem, to identify movements of species, critical habitats, and other ecological interactions. When this is not possible, a cross-boundary governance may consider the greater ecosystem to manage the impacts generating outside PAs.

The socio-economic dimension significantly drives stakeholder choices and consequently LULC transitions (Lambin, & Meyfroidt 2010), as observed in Section 3.2. For this reason, the governance of PAs needs to involve stakeholders to drive community choices towards sustainable development. The conflicts between the use of natural resources and conservation could be addressed through enhancement of local knowledge of the economic potential of biodiversity and the rejuvenation of the traditional involvement of the whole village community in decision-making (Maikhuri et al 2000). Web-based Public Participation GIS (PPGIS) seems to be a promising tool to inform and include communities in decision-making processes concerning the harmonization of conservation initiatives and human activities (Engen et al 2017).

The cases presented in Sections 3.4 and 3.5 describe tools to support spatial planning thought ES mapping applications. Both cases concern non-natural ecosystems (agricultural land and urban area), where the impacts of human activities on ecosystems are more relevant. The studies aimed to provide spatially explicit information for governance. In order to provide applicable instruments, the study cases included the assessment of both ES supply capacity and ES demand. In the first case study (Section 3.4), the ES demand was represented by the target for biofuel production fixed by the Regional Energy Plan. The analysis demonstrated that this target cannot be reached without significantly harm to soil-related ESs and/or causing LULC changes. The proposed tool also map the location of area that may be destined to oilseed production and quantify the biodiesel amount that could be produced without harming ESs. This instrument aim to support future energy plans, by avoiding fixing targets that could be not be achieved in sustainable ways.

The second case study (section 3.5) includes the use of a priority index to rank urban zones according to their need for improvement of green infrastructures. The ES demand is expressed as a function of vulnerability of population and buildings to urban flooding. This instrument can be applied to perform a cost-effective analysis for specific urban interventions, addressing investments allocation in zones with higher vulnerability (ES demand) and higher potential for green infrastructure improvements (potential increase of ES supply capacity).

Overall, the described case studies demonstrated that ES mapping could be implemented to provide instruments suitable for spatial planning. Such tools should:

- be designed according the local contexts;
- consider both ES supply capacity and demand;
- use assessment and mapping methods tailored to address the policy goal for which they are used;
- be directly applicable in plans and/or decision-making processes.

The studies presented in this research allow some additional considerations concerning mapping methods and application scales. The lack of historical data is the main limitation for mapping the ES changes in space and time (Eigenbrod et al. 2010). Given this limitation, data availability and the specific context are the most important variables that drive the choice of mapping methods. Benefit transfer based on global average values, as applied in the case of section 3.1, is a suitable solution for transitional environments. These environments are characterized by dynamic interactions with high variability in space and time (e.g., Shen et al 2016), which are difficult to model even with complex methods. However, representativeness of benefit transfer analysis can be improved in mountain environments by considering altitudinal effects on ecosystems (see Section 3.2). Indicators were used to map ESs in section 3.4. This method can be used when spatial biophysical information are available. As more complex level of ESs mapping (see Section 2), this method can be applied to design instrument for spatial planning. An indicator is also applied in section 3.3 (i.e. the Coefficient of reduction - Cr) to estimate the relative contribution to urban green infrastructures to mitigate urban flooding. Model tools was applied in section 3.3 and 3.5: InVEST (Sharp et al. 2016) and the Curve Number method (SCS USDA 1986), respectively.

The scale at which the mapping is performed is another important issue in ES mapping. PAs analysis requires mapping at landscape scale, since they often are designed for the conservation of environmental units. When ESs mapping is targeted to inform spatial planning in non-natural ecosystems, as in the case of Sections 3.4 and 3.5, the analysis may have to respond to specific governance problems at administrative unit scale. Case study of Section 3.4 is performed at regional scale, as the research problem concerned Regional Energy Plan, whereas, analysis of Section 3.5 is carried out at municipal level. The latter was possible thanks to high-resolution data derived by remote sensing technics.

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5. Conclusions

Overall, the study provided the following responses:

- The passive management of PAs does not seem to be effective in maintaining ESs. The simultaneous conservation of biodiversity and ESs may be achieved with an active management that support PA resilience, which: i) considers socio-economic drivers and stakeholder involvement, ii) regulate human activities in and outside the PAs and iii) maintain a proper amount of non-natural ecosystems inside PAs to support human well-being.
- ES assessment and mapping can be implemented in decision support-tools. Such instruments have to: i) consider both ES supply capacity and demand and ii) be tailored to a specific problem and context.

These findings can be used to address decision makers towards more sustainable management strategies regarding nature conservation and spatial planning.

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