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# Mapping green infrastructure based on ecosystem services and ecological networks: A Pan-European case study

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ABSTRACT

Identifying, promoting and preserving a strategically planned green infrastructure (GI) network can provide ecological, economic and social benefits. It has also become a priority for the planning and decision-making process in sectors such as conservation, (land) resource efficiency, agriculture, forestry or urban development.

In this paper we propose a methodology that can be used to identify and map GI elements at landscape level based on the notions of ecological connectivity, multi-functionality of ecosystems and maximisation of benefits both for humans and for natural conservation. Our approach implies, first, the quantification and mapping of the natural capacity to deliver ecosystem services and, secondly, the identification of core habitats and wildlife corridors for biota. All this information is integrated and finally classified in a two-level GI network. The methodology is replicable and flexible (it can be tailored to the objectives and priorities of the practitioners); and it can be used at different spatial scales for research, planning or policy implementation.

The method is applied in a continental scale analysis covering the EU-27 territory, taking into account the delivery of eight regulating and maintenance ecosystem services and the requirements of large mammals' populations. The best performing areas for ecosystem services and/or natural habitat provision cover 23% of Europe and are classified as the core GI network. Another 16% of the study area with relatively good ecological performance is classified as the subsidiary GI network. There are large differences in the coverage of the GI network among countries ranging from 73% of the territory in Estonia to 6% in Cyprus. A potential application of these results is the implementation of the EU Biodiversity Strategy, assuming that the core GI network might be crucial to maintain biodiversity and natural capital and, thus, should be conserved; while the subsidiary network could be restored to increase both the ecological and social resilience. This kind of GI analysis could be also included in the negotiations of the European Regional Development Funds or the Rural Development Programmes.


## 1. Introduction

Many aspects of human wellbeing and economic activities rely on ecosystem functions and processes. For instance, our food security is based on the existence and maintenance of fertile soil; we breathe the air that plants filter; our lives and properties are protected from flooding by soil infiltration, dune systems or riparian forests; and our mental and physical health may depend

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on the accessibility to green spaces (MA, 2005; Alcock et al., 2014). Furthermore, some nature-based technical solutions (e.g. green roofs, bio-infiltration rain gardens, vegetation in street canyons) have demonstrated in several cases to be more efficient, inexpensive, adaptable and long-lasting than the so-called "grey" or conventional infrastructure (e.g. Gill et al., 2007; Pugh et al., 2012; Ellis, 2013; Flynn and Traver, 2013; Raje et al., 2013).

The European Commission communication (2013) on green infrastructure (GI) sets the ground for a tool that aims to provide ecological, economic and social benefits through natural solutions, helping us to mobilise investments that sustain and enhance those benefits. This vision pursues the use of natural solutions (considered multi-functional and more sustainable economically and socially) in contrast with grey infrastructure (that typically only fulfils single functions such as drainage or transport). In the EC communication, GI is defined as *a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services*. This definition includes three important aspects: the idea of a network of areas, the component of planning and management, and the concept of ecosystem services. In this sense, GI integrates the notions of ecological connectivity, conservation and multi-functionality of ecosystems (Mubareka et al., 2013).

In the European context, besides the abovementioned EC communication, the conservation and development of a GI is identified as one of the priorities in EU policies covering a broad range of sectors, like the EU Biodiversity Strategy to 2020,[1] the roadmap to a Resource Efficient Europe,[2] the Commission's proposals for the Cohesion Fund and the European Regional Development Fund,[3] the new Common Agricultural Policy[4] (note the change from direct payments towards the second pillar payment that can be a strong incentive for GI restoration and maintenance), the new EU Forest Strategy[5] (especially relevant since many GI elements might be forest-based), or the forthcoming communication on "land as a resource" in 2015 (which will highlight the importance of using land efficiently and as a finite resource). Within the Biodiversity Strategy, target 2 aims at maintaining and restoring ecosystems and their services by 2020, by establishing a GI and restoring at least 15% of degraded ecosystems. Action 6 is setting priorities to restore and promote the use of GI. The forthcoming land communication will focus on the value of land as a resource for crucial ecosystem services and on how to deal with synergies and trade-offs between multiple land functions. Systematically including GI considerations in the planning and decision-making process will help reduce the loss of ecosystem services associated with future land use changes (i.e. land take and land degradation) and help improve and restore soil and ecosystem functions.

To support the planning process, approaches for mapping GI are necessary. They should focus on two basic concepts. The first one is multi-functionality, ensured by quantifying and mapping a number of ecosystem services. Decision makers can then seek for areas providing multiple services. The second concept should build on connectivity analyses such as the analysis of ecological networks. Spatial delineation of GI elements has often been based on a re-classification of available land cover data combined with information on natural values of each cover class (e.g. Weber et al., 2006; Wickham et al., 2010; Mubareka et al., 2013). Recent studies have shown the relevance of including sector specific models and connectivity in the analysis of policy impacts over GI networks (Mubareka et al., 2013). In particular, these authors find particularly relevant to forecast the land claimed by the agricultural sector, population projections, forestry and industry.

The objective of this paper is to propose a feasible and replicable methodology to identify and prioritise GI elements, including the concepts of ecosystem services and ecological connectivity. This methodology can be used at different spatial scales for planning and policy implementation. The proposed approach is applied in a continental case study, covering the EU-27 territory, focusing on a landscape scale. In this case the results could be used for conservation policies since they are aligned with the EC communication and the Biodiversity Strategy. This paper is a further refinement of a study started by EEA/ETC-SIA (EEA, 2014).

## 2. The proposed methodology

### 2.1. Conceptual aspects: criteria to identify GI elements

As we anticipated in the introduction, this study is focused on the identification of GI elements at landscape level. Unlike in urban environments, in the open landscape not all green areas qualify as GI. It is not economically or technically feasible to cover the entire territory with natural ecosystems in order to secure their positive influence on natural processes on every spot. Hence, we consider as crucial criteria to identify GI elements (i) the multi-functionality linked to the provision of a variety of ecosystem services, and (ii) the connectivity associated to the protection of ecological networks.

The first criterion, ecosystem services, are the contributions of natural systems to human wellbeing. We propose that GI elements should be multi-functional zones in terms of services' delivery (EC, 2012). Moreover, we focus on the identification of GI elements for conservation purposes, in line with one of the aims of GI in the EC Communication (2013): *protecting and enhancing nature and natural processes* as a green alternative to grey infrastructure. We concentrate on the regulating and maintenance services (as defined in Table 4 of Haines-Young and Potschin, 2013), since most of the provisioning and cultural services are mainly driven by human inputs like energy (e.g. labour, fertilisers) or capital (e.g. touristic infrastructures), and do not necessarily enhance natural processes (see trade-off analysis and conclusions in Nelson et al., 2009; Maes et al., 2012). These concerns are further explained in section 5. For example, if we include food provision in the assessment and we highlight the areas with a maximum production (crop yield) we will probably spot intensive agriculture areas that are sustained more by human inputs, like fertilisers and mechanical means, than by nature, like soil organic matter. With the available knowledge and information, by concentrating on regulating and maintenance services, we can assume that an improvement on the resulting GI network will enhance the condition of the ecosystems and natural processes.

With these premises (protecting and enhancing nature and natural processes), we decide to focus on the natural capacity of landscapes to deliver services before taking into account the human demand. This natural capacity, also refer to as "ecosystem function" in the ecosystem services' cascade framework (or "pathway" in de Groot et al., 2010), depends on the biophysical

[1] COM (2011) 244 final, http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52011DC0244&from=EN.

[2] COM (2011) 571 final, http://ec.europa.eu/environment/resource_efficiency/pdf/com2011_571.pdf.

[3] COM (2011) 612 final/2, http://www.espa.gr/elibrary/Cohesion_Fund_2014_2020.pdf; COM (2011) 614 final, http://www.esparama.lt/es_parama_pletra/failai/fm/failai/ES_paramos_ateitis/20111018_ERDF_proposal_en.pdf.

[4] COM (2010) 672 final, http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2010:0672:FIN:en:PDF; Regulations 1305/2013, 1306/2013, 1307/2013 and 1308/2013.

[5] COM (2013) 659 final, http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2013:0659:FIN:en:PDF.

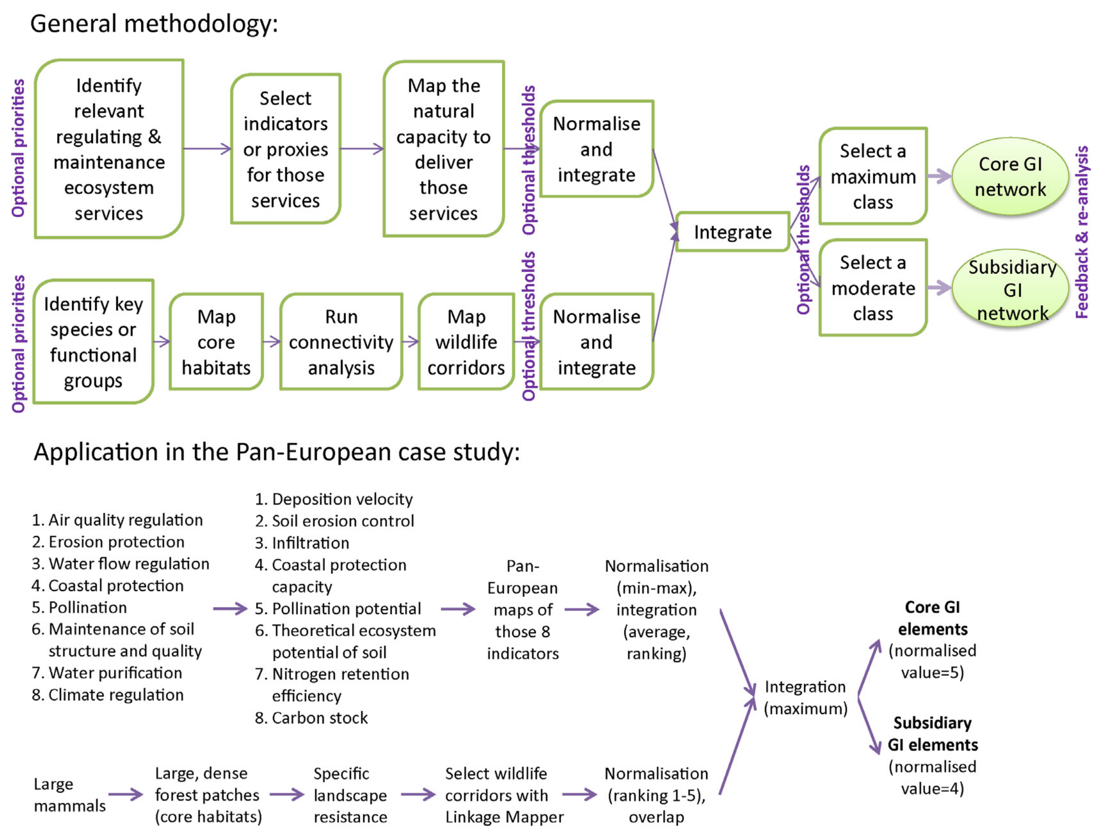


**Fig. 1.** Flowcharts representing the methodology proposed in this paper. The upper flowchart shows the general methodology followed to identify and map GI networks, while the bottom part illustrates the specific steps applied in the Pan-European case study (see Section 3).

structures and processes, ultimately linked to the ecosystems' condition (Maes et al., 2013). For example, when assessing air quality regulation for GI we measure the natural capacity as the potential of the existing vegetation to capture and remove air pollutants, instead of measuring the actual flux of pollutants coming from anthropogenic sources and being trapped by natural features (which will depend on human pressures). We insist that these choices are based on the purpose of our study (biodiversity conservation); in a different context the selection of ecosystem services or indicators may change while the application of our methodology remains invariable.

The second main criterion to design a GI is the existence and connectivity of ecological networks. All biotic functional groups need core areas where they can find living space, nourishment, nursery and breeding zones. Hence, the presence of these vital areas is crucial to maintain biodiversity. But the connectivity of those ecosystems is also a way to support genetic diversity and, thus, the viability and resilience of habitats and populations (Olds et al., 2012; Gibson et al., 2013; Ishiyama et al., 2014). Consequently, habitat modelling and ecological connectivity shall also be included in the analysis of GI.

### 2.2. Technical aspects: mapping GI

We propose a mapping methodology that focuses on (i) the capacity to deliver regulating and maintenance services, and (ii) the existence of core habitats for biota and the connectivity among those areas, as summarised in Fig. 1.

The first part of the assessment (upper branches of flowcharts in Fig. 1) starts with the identification of relevant regulating and maintenance ecosystem services for the study area. We recommend following one of the established classifications of ecosystem services such as TEEB[6] or CICES.[7]

The following and most demanding step is to assess and map the select ecosystem services. There are several approaches to do this (see review in Maes et al., 2012), from the direct conversion of land use/land cover maps (e.g. Burkhard et al., 2009), through the compilation of local primary data or statistics (e.g. Kandziora et al., 2013), to the application of dynamic process-based ecosystem models (e.g. Schröter et al., 2005). We prefer to identify and map a good proxy for the biophysical process responsible of each ecosystem service, if possible based on published scientific models and results. Each selected proxy should represent the natural capacity of ecosystems to deliver the correspondent service (service supply).

The next step is to normalise and integrate the data sets describing the ecosystem services. The selection of different normalisation methods and data thresholds will affect the final results, meaning that the user needs to consider what is the final objective before producing a map.

The second part of the assessment (lower branch of Fig. 1) is the identification of core and transitional habitats for key functional groups. As core habitats and functional connectivity are species-related, the national/local authorities should identify their most relevant species. Based on field studies and/or models, practitioners should identify (a) the core habitats for those key species, and (b) perform a habitat connectivity analysis to identify potential wildlife corridors, which requires expert knowledge

[6] The Economics of Ecosystems and Biodiversity, http://www.teebweb.org/resources/ecosystem-services/.
[7] Common International Classification of Ecosystem Services, http://cices.eu/.

**Table 1**
Selection of ecosystem services to define GI elements for the Pan-European case study. They are eight regulating and maintenance services linked to the CICES classification. We provide for each service a specific name, a short definition and a spatially explicit proxy or indicator to quantify it. The specific data sets and models used to estimate those proxies/indicators are detailed in Appendix.

| Ecosystem services classification following CICES v4.3 | | | Ecosystem services selected for this study | |
|---|---|---|---|---|
| Section | Division | Group | Selected service and short definition | Selected proxy |
| Regulation and maintenance ecosystem services | Mediation of waste, toxics and other nuisances | Mediation by ecosystems | *Air quality regulation*: Potential of ecosystems to capture and remove air pollutants in the lower atmosphere. | Deposition velocity of air pollutants on vegetation (based on Pistocchi et al., 2010) |
| | Mediation of flows | Mass flows | *Erosion protection*: Potential of ecosystems to retain soil and to prevent erosion and landslides. | Erosion control (Maes et al., 2011) |
| | | Liquid flows | *Water flow regulation*: Influence ecosystems have on the timing and magnitude of water runoff and aquifer recharge, particularly in terms of water storage potential. | Water infiltration (Wriedt and Bouraoui, 2009) |
| | | | *Coastal protection*: Natural defence of the coastal zone against inundation and erosion from waves, storms or sea level rise. | Coastal protection capacity (Liquete et al., 2013) |
| | Maintenance of physical, chemical, biological conditions | Lifecycle maintenance, habitat and gene pool protection | *Pollination*: Potential of animal vectors (bees being the dominant taxon) to transport pollen between flower parts. | Relative pollination potential (Zulian et al., 2013) |
| | | Soil formation and composition | *Maintenance of soil structure and quality*: The role ecosystems play in sustaining the soil's biological activity, physical structure, composition, diversity and productivity. | Theoretical ecosystem potential (Kleeschulte et al., 2012) |
| | | Water conditions | *Water purification*: The role of biota in biochemical and physicochemical processes involved in the removal of wastes and pollutants from the aquatic environment. | In-stream nitrogen retention efficiency (Grizzetti et al., 2012) |
| | | Atmospheric composition and climate regulation | *Climate regulation*: The influence ecosystems have on global climate by regulating greenhouse and climate active gases (notably carbon dioxide) from the atmosphere. | Carbon stocks from the carbon accounts (Simón Colina et al., 2012) |

and interpretation. We can recommend the use of tools such as Linkage Mapper Connectivity Analysis Software[8] or GuidosToolbox[9] that can be tailored for any scale and species. The spatial results of this habitat suitability analysis must be normalised and integrated in a similar way as the ecosystem services' maps.

The results of the two assessments are made comparable using a ranking normalisation method and are then integrated. In this integration the highest value (the best performing result either linked to ecological connectivity OR to ecosystem services) should supersede the other. We underline here "or" because it is not a combination of functions, but the selection of the best performing one. Thus, we propose a selection of maximum values in a pixel-by-pixel basis. Lastly, the results are classified as follows:

- The GI core network with maximum values of the integrated results (the highest capacity to provide ecosystem services and/or key habitats for biota);
- The GI subsidiary network with moderate values of the integrated results.

The thresholds used to define the maximum and moderate categories will affect the final distribution of results and the total coverage of the resulting GI network and, thus, should be linked to the management priorities and requirements of the study. Multiple outputs can be generated and compared before the final decision is taken, especially when the decision process involves several stakeholders.

This entire methodology is designed as a flexible and replicable procedure that should be tuned up for each regional study. There are three steps at which the user may decide to adjust it (Fig. 1): (1) in the initial selection of ecosystem services and functional groups to assess; (2) in the normalisation of original values, selecting the data distribution and limits between classes; and (3) in the final definition of GI categories, balancing the optional thresholds with the resulting GI network and the feedback from the interest groups.

## 3. The Pan-European case study

### 3.1. Ecosystem services in Europe

We selected eight regulating and maintenance ecosystem services and we compiled or adapted spatially explicit information about the capacity to deliver each of them in EU-27 (Croatia could not be covered due to lack of data) (Table 1). Since the formats and spatial units of each model were different, all input data were transformed into grids of 1 km spatial resolution.

In the Pan-European case study, we opt for a normalisation of each input map based on minimum and maximum values. The normalised values and maps of the eight ecosystems services are shown in Fig. 2.

A full description of the ecosystem functions and biophysical models used to map each ecosystem service is available in the

[8] http://www.circuitscape.org/linkagemapper.
[9] http://forest.jrc.ec.europa.eu/download/software/guidos/.

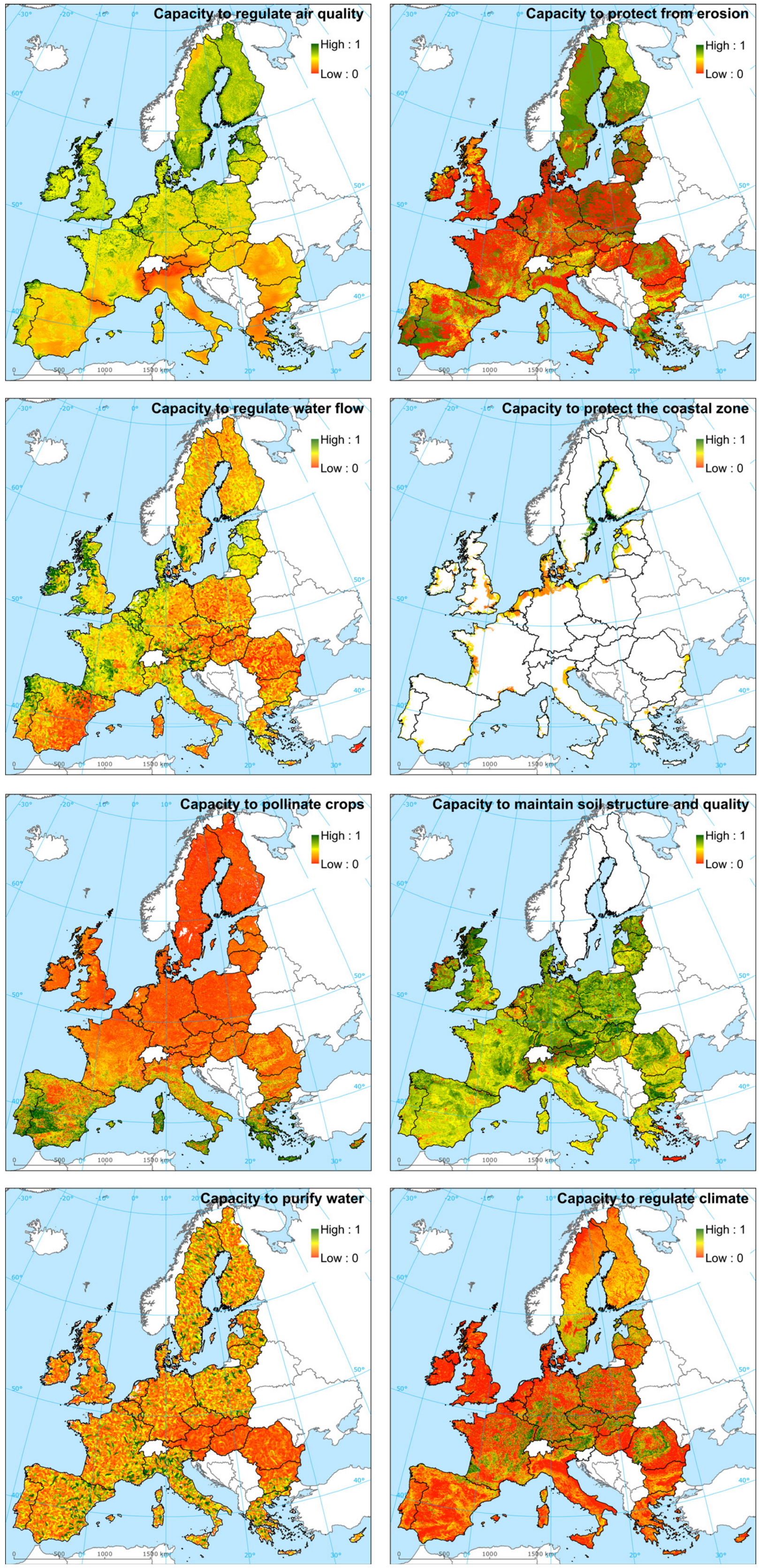


**Fig. 2.** Maps of the natural capacity to provide each of the eight selected ecosystem services across Europe. The original values (i.e. proxies listed in Table 1) come from the biophysical models and data sets described in Section 3.1 and Appendix.

Appendix. The Appendix also includes models' design, assumptions and input parameters as well as references to find the original results and interpretation. Here, we present a summary of the results at European scale.

- *Air quality regulation*: the potential dry deposition velocity of atmospheric pollutants across Europe ranges between 0.2 and 1.2 cm/s. Minimum values are concentrated in most of Romania, Bulgaria, Greece, Italy, eastern Spain and around the Alpine arc. Maximum values are scattered around the Baltic and North Sea shores, Ireland and NW Spain, W Portugal and some Greek islands. This is mostly linked to wind patterns and vegetal cover, both increasing towards the North or the Atlantic shores.
- *Erosion protection*: the erosion control indicator for Europe was a dimensionless indicator ranging from 1 to 5 before normalisation. Minimum values are widespread in continental Europe and the British Isles while maximum values are mostly found in southern Finland, southern Sweden, SW Iberia, SW France and scattered locations of Latvia, Lithuania, Poland, Germany, Denmark and Greece. The maximum values highlight areas with relatively high erosion risk and dense vegetal cover able to bind soil particles or reduce wind/water speed.
- *Water flow regulation*: the total infiltration of water in the soil fluctuates between nearly 0 and 1116 mm/yr. The lowest values are usually concentrated in eastern Spain, southern Italy and the lower Danube. The few basins with peak values (more than 700 mm/yr in the original data) are distributed in the British Islands, the NW Iberian Peninsula, and some Pyrenees and Alpine locations. Climatic factors (e.g. precipitation) play a major role on this distribution, but soil characteristics also affect the results.
- *Coastal protection (dimensionless)*: relatively low coastal protection capacity (close to 0) is present along the shores of Denmark, Germany, The Netherlands, some UK estuaries and the Gulf of Lion. Relatively high values (close to 1) are observed in Scandinavian mid-latitudes, Scotland, Ireland, NW Spain, Corsica and parts of Greece. These results are mainly driven by coastal geomorphology and topography and, to a lesser extent, by the presence of protective submarine and emerged habitats (e.g. biogenic reefs, dune systems).
- *Pollination (dimensionless)*: the indicator on pollination potential was already normalised between 0 and 1. In general, minimum values tend to accumulate in northern Europe while maximum values appear in the Mediterranean region. These differences at continental scale are driven by the location of foraging and nesting sites and the effect of the insect activity index, which depends on temperature and solar irradiance. The location of foraging and nesting sites is impacted by human activities, which are reflected in the land-use/land cover maps introduced in the model.
- *Maintenance of soil structure and quality (dimensionless)*: the original values of this indicator ranged between 2 and 12. The best soil ecosystem potential is usually present in relatively unpopulated areas across Europe with a particularly high concentration in Austria and Scotland. The worst soil characteristics are equally scattered along EU-27 but show some patches in western Ireland, northern Italy, the Danube delta, Crete and other Greek islands. Sweden and Finland are not covered by this indicator.
- *Water purification*: the in-stream retention efficiency of nitrogen (by sub-catchment) ranges from nearly 0 to 16%. The relatively low values in the Danube watershed are probably linked to the calibration of the GREEN model, which was performed by major European seas' drainage basins (Grizzetti et al., 2012), and by the fact that in-stream nitrogen tends to be conserved (not processed) in larger rivers (Alexander et al., 2000). Italy shows the largest national average retention efficiency.
- *Climate regulation*: the estimated carbon content of above-ground biomass in EU terrestrial ecosystems ranges between 0 and 20,004 tonnes of carbon per square kilometre, and it is mostly concentrated in forested areas. Countries with minimum above-ground biomass values are Ireland, Netherlands, Malta and UK. The Member States with the highest proportion of maximum carbon stock are the Czech Republic and Austria.

All these indicators (Fig. 2) are then combined through an arithmetic mean, in which the highest values represent the highest combined capacity to deliver regulating and maintenance services across EU-27. In order to combine these results with the second part of the methodology, which provides categorical data, we reclassify the average ecosystem service data into five ranks ranging from minimum (1) to maximum capacity (5), based on a natural breaks' distribution (Fig. 3A). Using the natural breaks' distribution for ranking works well for comparison purposes, like in this continental scale assessment. The areas with maximum capacity (greenish colours) are assumed to perform key ecological roles both for wildlife and for the human well-being. The areas with moderate capacity (yellowish colours) are performing important ecosystem functions but with a reduced or limited potential. In the reddish areas the ecosystem functions that support services are minor, either because their natural structure does not allow for a higher potential or because the habitats are deteriorated (for example a sparsely vegetated land when it refers to erosion protection, or an urban setting when it refers to climate regulation).

Our map 3A largely corresponds with the results from Maes et al. (2014, Fig. 3). The aggregation at regional level performed by Maes et al. obviously dilute some of the hot and coldspots found in our data, especially in the largest regions (e.g. our minimum values observed in Spain). But when we aggregate our results to the same scale, the relative distribution of values is in line with those authors. The largest regional discrepancies are found in Scandinavia. Data from Schulp et al. (2014, Fig. 3 bottom right) is more difficult to compare with our results, but the distribution of minimum and maximum values per region seems to broadly agree, again with the larger differences in Scandinavia. However, the different coverage of our maps affects the relative scale for the comparison.

### 3.2. Habitat modelling and functional connectivity

We analysed the presence of essential habitats (core and temporal habitats) for a selected functional group. In this case, being a continental scale analysis at 1 km resolution, we select the large mammals as focal group. Large mammals normally have high demands in terms of habitat area and are able to cover large migration distances. The parameters, data sources, species and models used to derive the presence of core habitats and connectivity among them are detailed in Appendix.

The main results from the habitat modelling are the identification of 67 actual core habitats (i.e. core habitats in which the presence of large mammals has been reported by EU Member States) and 53 extra potential core habitats (i.e. habitats suitable for mammalian wildlife but without reported presence). The habitat connectivity analysis highlighted 91 corridor swaths connecting the actual core areas or 156 linking all core habitats. These results refer to the European territory including all the Balkan countries (Fig. 3B).

The habitat modelling results are qualitative (i.e. presence or absence of different kinds of habitats). Using the same scale as in the ecosystem services' normalisation (ranks from 1 to 5), we assign the following categories (Fig. 3B):

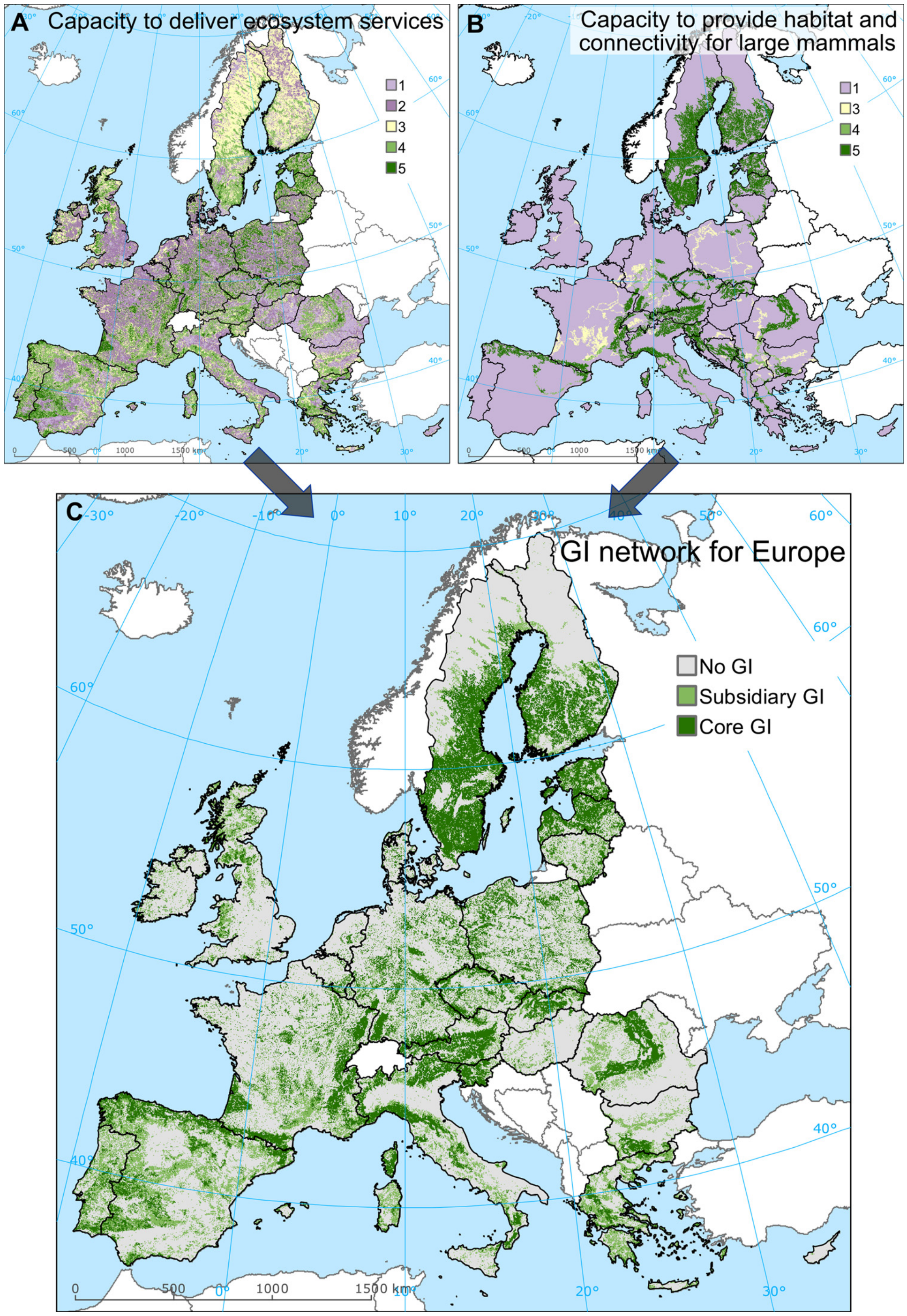


**Fig. 3.** Results of the Pan-European case study. (A) Capacity to deliver regulating and maintenance ecosystem services. (B) Key habitats to maintain healthy large-mammal populations. The results of (A) and (B) are normalised from value 1 (poor capacity for ecosystem services and habitat) to value 5 (maximum capacity for ecosystem services and habitat). (C) Proposed GI network for Europe based on the integration of (A) and (B). The GI core network comprises the best functioning ecosystems, crucial to maintain both natural life and natural capital. The GI subsidiary network covers other relevant areas sustaining ecosystem services and wildlife. The spatial data shown in these three maps will be available through the Ecosystem Services Partnership visualisation tool (www.esp-mapping.net).

- maximum value (5) to the actual core habitats
- high value (4) to wildlife corridors or transitional habitats among actual core habitats
- moderate value (3) to other potential core areas and to the potential wildlife corridors among them
- minimum value (1) to the rest of the territory.

Following this method, countries like Estonia, Slovenia, Latvia and Austria have approximately half of their territory under core habitats for large mammals, while others like Cyprus or Denmark have none. The analysis shows that 29 of the 91 actual core habitats linkages are shorter than 10 km, that is, more than 30% of the wildlife corridors are relatively feasible to be implemented and protected. The connectivity among core habitats for large mammals may enhance the genetic flow across Europe, which is particularly important to help species to adapt to climate change and other environmental alterations.

### 3.3. Towards a Pan-European GI network

The normalised results from ecosystem services (Fig. 3A) and habitat modelling (Fig. 3B) are finally integrated based on a selection of maximum values, i.e. each square kilometre will take the value of the criteria for which it is ecologically more important. This information is then transformed into an interpreted network of GI for Europe divided in two categories (Fig. 3C).

In this case the core GI network (see explanation in Section 2.2) includes areas where the integrated ecosystem service capacity and habitat modelling take value 5, thus including all the core habitats together with the areas with maximum potential to deliver ecosystem services for humans. The subsidiary GI network corresponds to value 4, where we find the wildlife corridors and where the provision of ecosystem services is above the average but not optimal. The thresholds fixed to define these two types of GI network are variable and adaptable to the case study objectives; they should take into account the ecological and land-use planning requirements.

The results illustrated in Fig. 3C indicate that 23% of the EU-27 territory might be part of the core GI network and 16% of the subsidiary GI network. The rest of the European territory did not qualify to form part of these GI networks. With the specific thresholds and criteria used in this case study, Estonia, Slovenia and Latvia show the maximum coverage of core GI network (between 56 and 63% of their territory) while Malta, Cyprus and Hungary show the minimum one (less than 2%). Regarding the subsidiary GI network, Portugal and Greece have the largest coverage (38 and 34% respectively). All in all Cyprus has proportionally the smallest GI network (6% of its territory) and Estonia the largest one (73%).

The extension of the GI network proposed in map 3C is significantly smaller than the results from Mubareka et al. (2013, Fig. 5) but the distribution is quite similar. Those authors follow a fix land cover typology to identify GI elements. Both approaches highlight mainly the large mountain ranges and densely forested areas, while Mubareka and colleagues include also more sparse forests and even semi-natural vegetation, magnifying the differences in the British-Irish Isles, Corsica, the Netherlands and Scandinavia.

## 4. Applications of the identification of GI networks

An environmental focus of GI is fundamental to secure its objectives (Wright, 2011) but it is not enough. What defines GI is the inclusion of goals for protecting ecological functions alongside goals for providing benefits to humans (McDonald et al., 2005). One of the strong points of the methodology proposed in this paper, and applied across Europe, is the prioritisation of the "green" (rural) spaces to form part of a coherent, multi-functional GI network that maximises the potential benefits both for humans and for natural conservation. The design of this methodology (Fig. 1) follows some crucial GI principles such as contribute to biodiversity conservation and enhance ecosystem services (Naumann et al., 2011). This thoroughly designed GI network could serve as an ecological backbone of the landscape supporting natural processes and ecosystem services in its surroundings.

Our proposal of two levels of GI networks (core and subsidiary) implies that biodiversity and the provision of ecosystem services can be not only protected but also improved (Rey-Benayas et al., 2009):

- The core GI network is generally related to the best condition of the ecosystems' structures and processes. These areas are crucial to maintain biodiversity and natural capital and, thus, should be preserved.
- In the subsidiary GI network ecosystem functions are still important, but they could be probably upgraded to the core GI network to increase both the ecological and social resilience. Hence, they represent areas with a potential for restoration.

Some of the grey areas in Fig. 3 may be other kind of "green" landscape providing other kind of services, like for instance agricultural or semi-natural areas. They may have a high demand of GI and, thus, be candidates for building new GI elements (i.e. restoration), but this should be analysed individually in more detail. The selection of priority areas should be settled at the appropriate management scale.

The European Biodiversity Strategy calls for establishing and promoting the use of GI, and restoring at least 15% of degraded ecosystems. Our results could be translated into the framework recently proposed by the Working Group on a Restoration Prioritisation Framework, which looks into the best ways to implement action 6 target 2 of the Biodiversity Strategy. This framework divides the continuum of ecosystem conditions from poor to excellent into four distinct levels (Lammerant et al., 2013). After an appropriate ecological analysis of thresholds in the study area, the core GI network could be ascribed to level 1 of the 4-level concept for restoration, where ecosystems' condition and their functions are in good to excellent condition. The protection or conservation of these zones may guarantee the delivery of ecosystem services and the maintenance of species and populations. It should be taken into account that, with our methodology, the proposed European GI network is not including all the protected areas and natural parks that should probably take part of level 1, like the Lemmenjoki or the Pallas-Yllästunturi National Parks in northern Finland for example. The subsidiary GI network could correspond to level 2, where abiotic conditions are satisfactory but some ecological processes and functions are disrupted with negative consequences in diversity. The ecosystem functions and, thus, the benefits from these zones could be boosted by some restoration actions. Hence, our methodology, once adapted to the regional characteristics, can serve countries and local agencies to set priority areas for GI and to identify potential areas for conservation and restoration. This can contribute to set up compatible Pan-European and national approaches to GI, both conceptually and spatially. Future applications of this approach shall serve to aggregate and compare national GI delineations with European ones (through up-scaling or nesting scales approaches).

This ecologically based GI network could be combined with other EU social and economic policies since they all address territorially dependent issues as human well-being, nature conservation or territorial cohesion and are based on a geographical concept. Hence the GI network could be included in the negotiations of the European Regional Development Funds under the EU cohesion policy 2014–2020 or the regional Rural Development Programmes under the EU rural development policy 2014–2020.

The methodology proposed in this article can be applied for other scientific uses such as the identification of data and knowledge gaps (e.g. unknown quantification of certain ecosystem services), the establishment of specific local/regional thresholds and criteria (e.g. definition of maximum capacity, or limits for habitat suitability for local species), the valuation of alternative options (e.g. cost-benefit or cost-efficiency analysis of a certain GI element), or the suitability of conservation zones linked to GI in terms of protected areas or endangered species (e.g. comparison of Natura 2000 zones with different GI networks), among others.

## 5. Main challenges and future steps

There are several sources of uncertainty that may affect the results of the Pan-European case study. Based on the suggestions of

Hou et al. (2013) and Schulp et al. (2014), the main sources of uncertainty of our analysis can be summarised as follows:

- Natural supply uncertainty, linked to the complexity and variability of ecosystem functions and species. In this case potential sources of uncertainty are:
  - The (lack of) knowledge and understanding of the biophysical processes and species behaviour
  - The selection and definition of the ecosystem services indicators
  - Uncertain information related to land-use and land cover data, dynamics and scale issues
- Technical uncertainty, linked to the tools and methods applied like:
  - Uncertainties based on model structure (assumptions, simplifications and formulations) or input parameters
  - Inaccuracy of spatial data, mapping limitations, availability of robust indicators, integration of data of varying quantity and quality
  - Selection of mapping and integrating methods

As we explained in Section 2.2 and illustrated in Fig. 1, our methodology can and should accommodate some case-specific options, from the selection of relevant ecosystem services and functional groups, to the establishment of thresholds. The unevenly distributed results obtained across EU-27 Member States highlight the need for such adaptation at national or regional level. The Pan-European case study was not designed to support the management of individual local sites, but the individual sites can benefit from landscape approaches since they take into account the site's relationship and functional connectivity with wider habitat networks (Kettunen et al., 2007). The development of multi-scale approaches may enable up- or down-scaling and maintaining compatibility with national or Pan-European approaches in relation with biodiversity policies.

The proposed methodology could be enriched in various ways. For instance, in order to support decision-making, it is highly recommended to include the stakeholders' involvement and feedback in the first steps of GI design (McDonald et al., 2005; Hostetler et al., 2011). In this environmental approach we did not attempt to include participatory processes, socio-economic aspects or human population dynamics. All these factors could be considered for the design of GI. In particular, a step further on this research should involve the integration of human demand for ecosystem services as well as the delivery of provisioning and cultural services in a nature-protection GI network. When doing so, topics such as the sustainable flow of each service (e.g. the maximum level of delivery at which ecosystems are not degraded), the geographical and temporal distribution of demand (e.g. where the services and mainly produced and where are they consumed), the energy and capital inputs, or the conflicts and trade-offs between different ecosystem functions and human uses (Horwood, 2011) should be taken into account.

Temporal variability could not be covered in this paper. However, ecosystems are not stable entities but continuously developing dynamic systems that provide services depending on their condition during each period. Also, transformations such as land use/land cover changes or climate change may have severe effects in the distribution of suitable habitats for biota and GI elements. A temporal assessment of ecosystems and ecosystem services could help understand, analyse and even predict the GI evolution.

## 6. Conclusions

GI is evaluated in this paper as an ecological and spatial concept that has the aim to promote ecosystems' health and resilience, contribute to biodiversity conservation and, at the same time, provide benefits to humans promoting the multiple delivery of ecosystem services. The multi-functionality of GI is the backbone of this analysis and is addressed by considering ecosystems services, provision of core habitats to biota and ecological connectivity.

In this paper we propose a methodology to identify and map GI networks at landscape level and we apply it in a Pan-European case study. This approach is based on (1) the quantification of the natural capacity to deliver ecosystem services, (2) the identification of essential core habitats and corridors for wildlife, and (3) the integration of all that information into a meaningful network of GI. That GI network is divided in two categories that can help identifying potential areas for conservation and/or restoration.

The methodology can be replicated at any other location and scale. One of its main advantages is its flexibility to adjust the selection criteria and data distribution (i.e. what is more important in a particular setting), which will obviously affect the final results.

Numerous policies, particularly those related to biological conservation, environment, cohesion and territory, may benefit from the definition and implementation of GI networks.

## Conflict of interest

The authors declare no conflict of interest.


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## Appendix. Data sources and models for the Pan-European case study

### A.1. Air quality regulation

Forests, parks and other GI features can reduce pollution by absorbing and filtering pollutants such as particulate matter. Pollutants can be removed from the atmosphere through deposition or by conversion to other forms. The deposition of pollutants on the earth surface can be linked to dry deposition (mainly gaseous sulphur and nitrogen compounds) and wet deposition processes (namely aerosols and soluble gases). Direct deposition to vegetation (dry deposition) is an important pathway for cleaning the lower atmosphere.

We select the dry deposition velocity on leaves as an indicator of the capacity of ecosystems to capture and remove air pollutants, as proposed in previous studies (Escobedo and Nowak, 2009; Karl et al., 2010). Data for the year 2006 are re-estimated for this paper based on the atmospheric particle deposition velocity of the MAPPE model (Multimedia Assessment of Pollutant Pathways in Europe, Pistocchi et al., 2010). MAPPE consists in a series of spatially explicit models that simulate the pollutant pathways in air, soil and surface and sea water at the European scale (now included in the FATE initiative http://fate.jrc.ec.europa.eu/rational/home).

The deposition velocity (DV) is a linear function of wind speed at 10 m height (w) and land cover type that can be noted as:

$$DV_i = \alpha_j + \beta_j \times w_i \tag{1}$$

where $\alpha$ and $\beta$ are, respectively, the intercept and slope coefficients corresponding to each broad land cover type $j$ (namely forest, bare soil, water or a combination of the previous) and $i$ denotes the calculation in each pixel.

### A.2. Erosion protection

Bare soils offer no protection against wind and rain and represent a high risk of soil erosion, landslides, sedimentation in streams and rivers, clogging of waterways and land degradation. Accelerated soil erosion by water as a result of changed patterns in land use is a widespread problem in Europe. By removing the most fertile topsoil, erosion reduces soil productivity and, where soils are shallow, may lead to an irreversible loss of natural farmland. The capacity of natural ecosystems to control soil erosion is based on the ability of vegetation (i.e. the root systems) to bind soil particles and to reduce wind/water speed, thus preventing the fertile topsoil from being blown or washed away by water or wind.

The procedure to map erosion control is based on Maes et al. (2011). The MESALES model from the European Soil Data Centre[10] analyses data on land use, slope, soil properties and climate (wind and precipitation) to predict the seasonal and annual averaged soil erosion risk. The resulting map on soil erodibility categorises soils into five risk classes (very low, low, medium, high and very high sensitivity to erosion). Next, the soil erodibility map was intersected with a map of natural vegetation based on the Corine Land Cover (CLC2000) dataset. Polygons resulting from the intersection between natural vegetation and soil erodibility received a score between 1 and 5 depending on the soil erosion risk class. This procedure gives more weight to natural vegetation in areas where erosion risk is high. The final indicator is the weighted surface area share of natural vegetation in each 1 km grid cell. This procedure uses the CLC data twice, but as a consequence of the scale difference between the erodibilty map and the map of natural vegetation, it results in a map of protective vegetation on soils with a high risk for erosion.

### A.3. Water flow regulation

The flow regulation function refers to the ability of watersheds to capture and store water from rainfall events (Le Maitre et al., 2014) in contrast with artificial impervious surfaces. The influence of soil and vegetation on the timing and magnitude of water runoff has numerous positive effects such as regulate flood events, and recharge slowly the groundwater, maintaining the baseflow of streams. Water infiltration and percolation through the soil are key processes in flow regulation.

We use the annual infiltration (F, mm/yr) estimated by Wriedt and Bouraoui (2009) based on the modelling approach suggested by Pistocchi et al. (2008), as an indicator of the capacity of terrestrial ecosystems to temporarily store and regulate water flow. This annual infiltration represents the water available for slow and fast subsurface runoff ($Q_{SSF}$), i.e. for groundwater and interflow, and it is estimated as difference between (effective) precipitation ($P$) and actual evapotranspiration (ETA) and runoff (RO).

$$F(t) = P(t) - \mathrm{ETA}(t) - \mathrm{RO}(t) \tag{2}$$

$$Q_{SSF}(t) = (F(t)|F(t) \geq 0) \tag{3}$$

where runoff (RO) is represented as a function of precipitation, in the form of a combination of an SCS (Soil Conservation Service) curve number model and a linear runoff model with runoff coefficient, and the actual evapotranspiration (ETA) is estimated by the formula proposed by Turc (1955) (Pistocchi et al., 2008).

Thus, the estimation of annual infiltration used in this study is based on climatic factors (precipitation and evapotranspiration) and the soil water holding potential (the soil type and texture affecting RO).

### A.4. Coastal protection

Coastal protection can be defined as the natural defence of the coastal zone against inundation and erosion from waves, storms or sea level rise. Habitats (e.g. wetlands, dunes, seagrass meadows) and other environmental features (e.g. cliffs, enclosed bays) act as physical barriers protecting any asset or population present in the coastal zone. In many locations, these ecosystems suffer from increasing pressure from expanding human populations and from a lack of long-term coastal management. This ecosystem service includes several processes like attenuation of wave energy, flood regulation, erosion control or sediment retention. The consequence of natural hazards on the coastal zone and their impacts on humans (usually referred to as coastal vulnerability) is a topic of high interest for science, society and policy-making alike (e.g. Adger et al., 2005).

Liquete et al. (2013) developed a specific indicator for coastal protection capacity (CP) defined as the natural potential that coastal ecosystems possess to protect the coast against inundation or erosion. We use the distribution of this indicator across Europe to represent the potential to deliver coastal protection as an ecosystem service. CP integrates geological and ecological characteristics likely to mitigate extreme oceanographic conditions, as follows:

$$\mathrm{CP}_i = 0.33 \times G_i + 0.25 \times S_i + 0.21 \times \mathrm{MH}_i + 0.21 \times \mathrm{LH}_i \tag{4}$$

where

- $G$ is the coastal geomorphology ranked in a meaningful sequence corresponding to its influence on coastal protection from minimum (e.g. polders) to maximum (e.g. rocky cliffs).
- $S$ is the average slope of each emerged coastal unit (the vulnerable area).
- MH represents the marine (seabed) habitats ranked in a meaningful sequence corresponding to its influence on coastal protection from minimum (e.g. shallow muds) to maximum (e.g. shelf rock or biogenic reef).
- LH represents the land habitats or land cover ranked in a meaningful sequence corresponding to its influence on coastal protection from minimum (e.g. sparsely vegetated areas) to maximum (e.g. dune systems).

The study area in Liquete et al. (2013) was the European coastal zone potentially affected by extreme hydrodynamic conditions, delimited in general by the 50 m depth isobath and the 50 m height contour line. Within that study area, 1414 coastal units of a length of approximately 30 km were

[10] http://eusoils.jrc.ec.europa.eu/ESDB_Archive/serae/GRIMM/erosion/inra/europe/analysis/maps_and_listings/web_erosion/index.html.

delineated perpendicular to the coast and the main topographic and bathymetry trends. All data (i.e. variables $G$, $S$, MH and LH) were extracted and aggregated for each coastal unit $i$. Finally, all the results were normalised from 0 to 1 based on minimum and maximum values.

### A.5. Pollination

Many wild and agricultural crops depend on pollinating insects, including most fruits, many vegetables and some biofuel crops. Pollinators play also an important role in maintaining plant diversity. The productivity of approximately 75% of the global crops that are used as human food benefits from the presence of pollinating insects (Klein et al., 2007). In Europe, crop production is argued to be highly dependent on insect pollination, with about 84% of all crops depending to some extent on it (Williams, 1994). However, wild pollinators face numerous threats, such as intensive farming, climate change or land use changes that disturb suitable habitats (e.g. wildflower meadows, mixed grasslands, hedgerows).

The model developed by Zulian et al. (2013) provides an index of relative pollination potential, which is defined as the relative capacity of ecosystems to support crop pollination. The authors designed a spatially explicit model that can be expressed as follows:

$$\mathrm{RPP}_i = \sum_{r=0}^{R}\left[\sum_{r=0}^{R}(F_{ir} \times K_r) \times N_i \times A_i\right]_r \times K_r \tag{5}$$

where

- RPP$_i$ is the relative pollination potential in each pixel $i$.
- $F$ is the floral availability index. It applies different weighting factors for the availability of floral resources in a composite of land-use/land cover classes, detailed agricultural land uses, high resolution (HR) forest cover, HR riparian zones and HR roadsides.
- $K$ is a weighted kernel representing the flight range of a specific guild of pollinators, following an inverse distance function (distance decay). This model uses the specific parameters of solitary wild bees.
- $R$ is threshold distance or maximum flight range for the specific guild of pollinators.
- $N$ is the nesting suitability index. It applies different weighting factors for the capacity to host pollinators' nests in a composite of land-use/land cover classes, detailed agricultural land uses, high resolution (HR) forest cover, HR riparian zones and HR roadsides.
- A is the pollinator activity index, a species-specific correction for the effect of climatic conditions on pollinators. In particular in this model:
- $A = -39.3 + 4.01 \times (-0.62 + 1.027 \times T + 0.006 \times R)$, with $T$ = temperature and $R$ = solar irradiance.
- Maps of relative pollination potential can be produced for each pollinator species provided that parameters about flight distance and activity are available. This study used a relatively short flight distance using solitary bees as model.

The EU map of relative pollination potential (RPP) is used a proxy of the capacity to pollinate crop. RPP depicts the potential of land cover cells to provide crop pollination by short-flight distance pollinators on a relative scale between 0 (minimum) and 1 (maximum).

### A.6. Maintenance of soil structure and quality

Fertile and healthy soils are a prerequisite for the sustainable and long term production of food and feed. Vegetation increases the soil's ability to absorb and retain water, produce nutrients for plants, maintain high levels of organic matter, and moderate its temperatures. Hence, soils are crucial for the conservation of biological diversity, carbon storage, water management and landscape management.

We compiled spatially explicit data about the maintenance of good soil structure and function from Kleeschulte et al. (2012). Their methodology compared two soil threats – soil compaction (SC) and soil erosion (SE) – with good soil preservation measures – top soil organic carbon (SOC) – following the ideas of Jones et al. (2012). These three parameters described the main characteristics of soil structure. They were ranked into four classes from 1 (very high susceptibility to compaction, >50 t/ha/yr of erosion, 0–2% of organic carbon content) to 4 (low susceptibility to compaction, null erosion, >8% organic carbon). These data were used to create an integrative indicator about the theoretical ecosystem potential of soil (TEP) for each pixel $i$:

$$\mathrm{TEP}_i = \mathrm{SC}_i + \mathrm{SE}_i + \mathrm{SOC}_i \tag{6}$$

where the three parameters are reclassified as explained above, and TEP gets values between 3 (minimum) and 12 (maximum). Regions with high TEP scores are considered to provide good ecosystem functions for maintaining good soil structure and quality (i.e. areas with low risk for soil erosion and compaction in combination with good organic matter content). Hence, TEP is used in this study as a proxy of the capacity of natural systems to maintain soil structure and quality.

### A.7. Water purification

Water purification relates to the role ecosystems play in the filtration and decomposition of organic wastes and pollutants in water, averting the need of further waste-water treatment plants to maintain clean water flows. Water quality is one of the most critical aspects for human populations, animals and plants. The natural supply of drinking water and water for domestic and industrial usage from ground and surface water bodies depends on the filtering potential of microorganisms, vegetation and sediments.

As a proxy of the capacity of freshwater ecosystems to remove organic wastes and pollutants from water we used the in-stream nitrogen retention efficiency, which explains what portion of the nitrogen entering rivers is naturally retained. Nutrient removal is determined by the strength of biological processes relative to hydrological conditions (residence time, discharge, width, volume). In this study we use the results from the GREEN model (Geospatial Regression Equation for European Nutrient losses), a conceptual statistical regression model developed to estimate nitrogen and phosphorus fluxes to surface water (Grizzetti et al., 2008) that has been applied at the European scale (Bouraoui et al., 2011; Grizzetti et al., 2012). The model estimates the nitrogen transported and removed per sub-basin, which in the application at the European scale have an average area of 170 km$^2$. The annual nitrogen load estimated at the outlet of each sub-basin $i$ ($L_i$, tonne N/yr) is expressed as:

$$L_i = (\mathrm{DS}_i \times (1 - \mathrm{BR}_i) + \mathrm{PS}_i + U_i) \times (1 - \mathrm{RR}_i) \tag{7}$$

where DS$_i$ (tonne N/yr) is the sum of nitrogen diffuse sources, PS$_i$ (tonne N/yr) is the sum of nitrogen point sources, $U_i$ (tonne N/yr) is the nitrogen load received from upstream sub-basins, and BR$_i$ and RR$_i$ (fraction, dimensionless) are the estimated nitrogen Basin Retention and River Retention, respectively. In the model, BR$_i$ is estimated as a function of rainfall while RR$_i$ depends on the river length, which is used as a proxy for the residence time.

In this study we used the River Retention ($RR_i$) estimated by the GREEN model as the efficiency to retain nitrogen. To ease processing and integration with the other data sets, we extrapolate the in-stream values to the corresponding sub-catchment area.

### A.8. Climate regulation

A stable and predictive climate is essential for the living conditions of humans and for the use of natural resources. The continuous sequestration of carbon dioxide by plants, algae, soils and marine sediments is a key factor contributing to stable climatic conditions, specially under the present global warming scenario. Climate regulation as an ecosystem service is usually estimated through carbon storage and sequestration processes. The maintenance of existing carbon reservoirs is among the highest priorities in striving for climate change mitigation.

We assume that carbon stocks are a proxy of the capacity of ecosystems to contribute to climate regulation. Information about terrestrial biomass, in particular above-ground carbon stocks in forests and other vegetation (e.g. shrubs, wetlands), at the European scale has been estimated by the Carbon Accounting model (Simón et al., 2011; Simón Colina et al., 2012). Carbon accounting is based on stocks (soil, forests, crops and other vegetation) and flows (felling, grazing, fodder, food, activate sludge, dead biomass, organic fertilisation).

Forest carbon estimations are based on the statistical disaggregation/downscaling of European forest data from different sources like the European Forest Information Scenario Model (EFISCEN), National Forest Inventories, or the Mediterranean Regional Office of the European Forest Institute (EFIMED). This is weighed with the mean NDVI (Normalised Difference Vegetation Index) signal and the output is used to calculate a volume of biomass per reference year. The results are converted into carbon content using carbon conversion factors derived from FAO statistics (Simón et al., 2011). The carbon content in other vegetation classes is calculated from land cover data using Corilis[11] and conversion factors derived from the literature (Simón et al., 2011).

We extracted from the carbon accounting exercise the data sets containing forest stock carbon content and carbon in other vegetation (from a total of 14 different vegetation types) for the year 2006 and summed them to derive a proxy of the above-ground total carbon content (stock in tonnes of carbon).

### A.9. Habitat modelling

We applied a habitat model taking into account minimum habitat sizes for individuals and populations of large mammals (Birngruber et al., 2012) and studies on wild animal corridors in Austria (Birngruber et al., 2012), Germany (Hänel and Reck, 2011) and the Czech Republic (Anděl et al., 2010). In particular, we based the habitat model on the following parameters:

(i) Core habitats of at least 50% forest density and 500 $km^2$ size. Information on forest density was obtained from the global Landsat Vegetation Continuous Fields tree cover layer provided by the Global Land Cover Facility (Sexton et al., 2013). From this data set we extracted the potential core habitats that satisfied the two density and area requirements (more details in EEA, 2014).

(ii) Actual presence of large mammals in those potential core habitats based on the reporting of EU Member States for the Habitats Directive (HD), in particular on the distribution maps of 8 species of large mammals (*Alopex lagopus*, *Canis lupus*, *Cervus elaphus corsicanus*, *Gulo gulo*, *Lynx lynx*, *Lynx pardinus*, *Rangifer tarandus fennicus* and *Ursus arctos*) present in Annex II of the HD (more details in EEA, 2014). These species do not cover the entire range of large mammals present in Europe, and are not equally distributed across Member States. However, this data set is the only spatially explicit, continental information available to ''ground truth' the presence of biota.

(iii) Habitat permeability and landscape resistance for the transit of large mammals derived from Beier et al. (2011) and Birngruber et al. (2012) and mapped based on CLC 2006 data (merged with the only available CLC 2000 information for Greece) (specific scoring and other details in EEA, 2014). The landscape resistance represents the degree to which the landscape facilitates or impedes movement among different patches as a combined product of structural and functional connectivity (i.e. the effect of physical structures and the actual species use of the landscape).

For the habitat connectivity analysis we used the Linkage Mapper v1.0.3 tool (McRae and Kavanagh, 2011). This tool automates mapping of wildlife corridors using core habitat areas and maps of resistance such as the ones described above and further parameters described in EEA (2014). The results identify least-cost linkages between core areas which represent paths of minimum energetic cost, difficulty, or mortality risk for animal migration. To define wildlife corridors we use not only least-cost paths (i.e. single pixel lines) but also corridor swaths (natural patches of up to 10 km cost distance width) that take into account if the habitats surrounding the least-cost paths are appropriate for migration and, hence, if they represent functional connectivity (i.e. if they are biologically relevant and likely to be used by biota).

[11] CORILIS, from CORIne and LISsage (smoothing in French) purpose is to calculate "intensities" of a given theme in each point of a territory. See http://www.eea.europa.eu/data-and-maps/data/corilis-2000-2 and http://goo.gl/biKcQ.

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