Universitat Autònoma de Barcelona

URBAN GREEN INFRASTRUCTURE
Modeling and mapping ecosystem services for sustainable planning and management in and around cities

Francesc Baró

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Ph.D. Dissertation

Supervisor: Dr. Erik Gómez-Baggethun

Co-supervisor: Dr. Dagmar Haase

Academic tutor: Dr. Victoria Reyes-García

Ph.D. Program in Environmental Science and Technology

Institut de Ciència i Tecnologia Ambientals (ICTA)

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Note: U.S. spelling is used in this work

Als meus pares, al meu germà, a la Marie i al Gael

"By leaves we live."

Sir Patrick Geddes, pioneering town planner

(Geddes' final lecture to his students at University College Dundee, 1918)

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MAIN ABBREVIATIONS AND ACRONYMS

ANGSt	Accessible Natural Greenspace Standard
ASPB	Public Health Agency of Barcelona
BMR	Barcelona metropolitan region
BVOC	Biogenic volatile organic compounds
CBO	Cities and Biodiversity Outlook
CICES	Common International Classification of Ecosystem Services
CO	Carbon monoxide
CO_2	Carbon dioxide
CO_2eq	Carbon dioxide equivalent
DBH	Diameter at breast height (forestry)
EC	European Commission
EEA	European Environment Agency
EQS	Environmental Quality Standard
ES	Ecosystem services
ESTIMAP	Ecosystem Services Mapping tool
EU	European Union
GHG	Greenhouse gas
GI	Green infrastructure
GIS	Geographic information system
IPBES	Intergovernmental Panel on Biodiversity and Ecosystem Services
LAI	Leaf area index
LUR	Land use regression
MAES	Mapping and Assessment of Ecosystems and their Services
MCSC	High resolution land cover map of Catalonia
MEA	Millenium Ecosystem Assessment
NbS	Nature-based solutions
NO_2	Nitrogen dioxide
O_3	Ground-level ozone
OpenNESS	Operationalization of Natural Capital and Ecosystem Services (project)
PCA	Principal component analysis
PECQ	Energy, Climate Change and Air Quality Plan of Barcelona (2011–2020)
PM_{10}	Particulate matter with a mean aerodynamic diameter of 10 μm

PTGC	General Territorial Plan of Catalonia
PTMB	Territorial Metropolitan Plan of Barcelona
SE	Standard error
SEAP	Local Sustainable Energy Action Plan
SITxell	Territorial Information System for the Network of Open Areas in the province of Barcelona
SO_2	Sulfur dioxide
TEEB	The Economics of Ecosystems and Biodiversity
UHI	Urban Heat Island
UN	United Nations
URBES	Urban Biodiversity and Ecosystem Services (project)
US	United States of America
USD	United States dollar
WHO	World Health Organization

PREFACE

This dissertation is submitted in fulfilment of the requirements for the doctoral degree in Environmental Science and Technology, organized by the Institute of Environmental Science and Technology (ICTA) of the *Universitat Autònoma de Barcelona* (UAB), Spain. The Ph.D. Program in Environmental Science and Technology is under the legal framework of the Spanish Royal Decree 99/2011, of 28 January, by means of which official Ph.D. studies are regulated in Spain. The dissertation is the main result of a three-year Ph.D. project (2013-2016) supervised by Dr. Erik Gómez-Baggethun and co-supervised by Dr. Dagmar Haase. During this period I have been member of the ICTA research group 'LASEG' (Laboratory for the Analysis of Socio-Ecological Systems in a Global World), coordinated by Dr. Victoria Reyes-García (also my Ph.D. academic tutor) and Dr. Esteve Corbera.

Following UAB requirements, the dissertation includes six chapters: a general introduction and research objectives; four original research chapters (each of them including introduction, material and methods, results, discussion and conclusion); and a general discussion and conclusions chapter. Three of the research chapters are already published as articles in peer-reviewed scientific journals and the last one is submitted for publication.

This dissertation is an individual work, but it has been carried out under the framework of various research projects and collaborations. Specifically, the Ph.D. project was strongly supported by my participation in the European research projects 'URBES' (Urban Biodiversity and Ecosystem Services, ERA-Net BiodivERsA, 2011-2014) and 'OpenNESS' (Operationalization of Natural Capital and Ecosystem Services, EU FP7, 2012-2017). Further, the Ph.D. project has benefited from my collaboration with the following institutions: Barcelona Regional Council (*Diputació de Barcelona*) through the initiative "Development and mapping of a system of ecosystem services indicators in the province of Barcelona within the framework of the Territorial Information System for the open spaces network (SITxell)"; Barcelona City Council (*Ajuntament de Barcelona*); and CREAF (Centre for Ecological Research and Forestry Applications).

My research achievements during the Ph.D. period go beyond the completion of this dissertation. During this period, I had the opportunity to participate in other scientific publications and reports, and also to present my research in various conferences, meetings and seminars. Moreover, I have reviewed for several scientific journals, participated in the development of research project proposals and carried out other research activities. All these achievements are listed in Appendix D.

ACKNOWLEDGEMENTS

The completion of this Ph.D. dissertation is the end of a long journey which I have fortunately made with the support of many people and various institutions. In this section I would like express my gratitude to all of them. Besides, all contributors and funding organizations of the individual research papers included in this dissertation are also acknowledged at the end of each chapter.

First, I would like to acknowledge the different funding organizations which have made the realization of this work possible. This research was partially funded by the ERA-Net BiodivERsA through the Spanish Ministry of Economy and Competitiveness project 'URBES' (code PRI-PIMBDV-2011-1179) and also by the 7th Framework Program of the European Commission project 'OpenNESS' (code 308428). Besides, I received funding from an agreement of collaboration between ICTA and the Barcelona Regional Council (*Diputació de Barcelona*) and during the last Ph.D. year from a scholarship by the *Fundación Iberdrola España* (*Ayudas a la Investigación en Energía y Medio Ambiente 2015*).

Second, I am grateful to all of my colleagues in the URBES and OpenNESS projects. It has been (and still is) an honor for me to work with and learn from some of the most influential researchers on the fields of ecosystem services and urban ecology. My special thanks go to Niki Frantzeskaki, Thomas Elmqvist, Neele Larondelle, David Barton, Leena Kopperoinen, Grazia Zulian, Pilar Vizcaino, Ignacio Palomo and Berta Martín-López with whom I have worked more closely.

Third, I am thankful to all of the members participating in the case study advisory board (CAB) on ecosystem services and green infrastructure in the Barcelona province. The interest and support shown by all the institutions and organizations has been extremely stimulating for my research. Particularly, I would like to thank the fruitful collaboration with Marga Parés, Coloma Rull and Montse Rivero (*Ajuntament de Barcelona,* Barcelona City Council), with Carles Castell and all his team (*Diputació de Barcelona*, Barcelona Regional Council) and with Joan Pino and Corina Basnou (CREAF). I am also grateful to Jaume Terradas (CREAF) and Lídia Chaparro for their generosity during the first stage of my PhD period.

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A very special and warm thanks goes to my Ph.D. supervisors, Erik Gómez-Baggethun and Dagmar Haase. Despite your hectic agendas, you have always found the necessary time to support and revise my work and provide advice if needed. Dagmar, thanks very much for your always positive, encouraging and quick feedbacks and for supervising my stay in Berlin. Erik, siempre te estaré agradecido por haber confiado en mí y haberme dado la oportunidad de realizar este proyecto de investigación y participar en otros, especialmente en un tiempo de crisis donde este tipo de oportunidades eran prácticamente ausentes en España.

Another warm thanks to my colleague and friend Johannes. We started this journey together and we have successfully ended it together. Gracias por todo tu apoyo, ayuda, consejos y actitud positiva durante estos años, especialmente en los momentos más complicados. Ha sido un privilegio trabajar a tu lado y espero que esto sea solamente el principio de una larga colaboración y amistad.

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Last but not least, my greatest debt of gratitude goes to Marie. Gracias por tu paciencia y comprensión durante todos estos años, especialmente los últimos meses, por traer al mundo a nuestro hijo Gael, lo mejor que me ha pasado en esta vida, por estar siempre a mi lado en los momentos más difíciles. Sin tu apoyo incondicional, esta tesis no hubiera sido posible y por esto va dedicada especialmente a ti. Je t'aime.

SUMMARY

In an increasingly urban planet, many cities and their inhabitants are facing multiple pressing threats within their borders, including heat stress, pollution and growing disconnection with the biosphere. Improving sustainability, resilience and livability in urban areas should be thus a major goal on the policy agenda, from local to global authorities. The operationalization of the ecosystem services framework, building on the concepts of 'green infrastructure' and 'nature-based solutions', is claimed by a mounting number of policy-makers, practitioners and scientists as the way forward to address many of these urban challenges. However, the extent to which urban green infrastructure can offer relevant solutions to these challenges is rarely considered in ecosystem service assessments, and therefore unknown to decision-makers.

This dissertation critically examines the role and contribution of green infrastructure to cope with diverse urban challenges (with a focus on air pollution, greenhouse emissions, heat stress and opportunities for outdoor recreation) at different spatial scales. Building on the ecosystem services cascade model, an operational framework is proposed and applied across four original research chapters to inform planning and management decisions on the basis of the relationships between the green infrastructure's capacity to deliver ecosystem services, the actual provision or use of these services (flow), and the amount of services demanded by the urban population. Identification of unsatisfied demand, i.e., the mismatch between ecosystem service flow and demand, is a main focus of the assessments since it expresses the limits of urban green infrastructure in relation to the considered challenges. The dissertation uses and refines a variety of methodological approaches for modeling and mapping the capacity, flow and demand of urban ecosystem services (e.g., i-Tree and ESTIMAP tools). The spatial scope of the research carried out within the assessment framework of this dissertation principally encompasses the urban area of Barcelona, Spain, considering both the local or city scale (Barcelona municipality) and the metropolitan or regional scale (Barcelona metropolitan region).

Results from the research indicate that the contribution of ecosystem services provided by urban green infrastructure to cope with urban problems is often limited (e.g., its impact on air quality or carbon offsetting was lower than 3% considering total carbon emissions and air pollution in all case studies) and/or uncertain at the city and metropolitan scales. In addition, the positive impact of green infrastructure on environmental quality and human wellbeing is usually challenged by ecosystem disservices (e.g., biogenic emissions), trade-offs (e.g., provisioning versus regulating services) or spatial mismatches between service supply and demand (e.g., air purification and outdoor recreation capacities of large metropolitan green infrastructure blocks are too far from demand sites).

On the basis of these findings, several implications for urban/landscape planning, management and decision-making are drawn, including: (1) the prioritization of abatement policies on the pressures generating a demand for certain ecosystem services (e.g., air purification and carbon sequestration); (2) combining land sharing strategies in urban and agricultural land in order to increase their multifunctionality and resilience and, concurrently, assure the conservation of large patches of multifunctional periurban forest areas; (3) development of new green spaces in compact urban cores using innovative strategies (e.g., rooftop gardens); and (4) consideration of ecosystem services trade-offs and disservices in planning and management. Finally, I contend that urban green infrastructure planning and management requires a holistic approach, considering the whole range of ecosystem services potentially provided by different types of green infrastructure and the interactions between them, together with the different spatial scales at which these ecosystem services can be relevant for the resilience, sustainability and livability of urban areas. This calls for a strong multi-scale and multi-disciplinary institutional coordination between all the authorities dealing with urban and environmental policy and for the harmonization of planning and management instruments in a multi-level governance approach.

RESUM (SUMMARY IN CATALAN)

En un planeta cada vegada més urbà, moltes ciutats i els seus habitants s'enfronten a múltiples i urgents amenaces dins de les seves fronteres, incloent l'estrès per excés de calor, la contaminació i la creixent desconnexió amb la biosfera. Millorar la sostenibilitat, la resiliència i l'habitabilitat de les àrees urbanes ha de ser per tant un objectiu de importància primordial en l'agenda política, des de les autoritats locals a les globals. L'aplicació del marc de serveis dels ecosistemes, a partir dels conceptes de 'infraestructura verda' i 'solucions basades en la naturalesa', es considera per un creixent nombre de responsables polítics, professionals i científics com el camí a seguir per fer front a molts d'aquests desafiaments urbans. No obstant això, el grau en què la infraestructura verda urbana pot oferir solucions adequades a aquests reptes és rarament considerat en les avaluacions de serveis dels ecosistemes, i per tant la seva potencial contribució és sovint desconeguda per als prenedors de decisions.

Aquesta tesi examina de manera crítica el paper i la contribució de la infraestructura verda per fer front a diversos reptes urbans (amb especial atenció a la contaminació de l'aire, les emissions d'efecte hivernacle, l'estrès per excés de calor i les oportunitats per al lleure a l'aire lliure) a diferents escales territorials. Partint del model de cascada de serveis dels ecosistemes, es proposa i s'aplica un marc operacional a través de quatre capítols d'investigació originals per informar les decisions de planificació i gestió sobre la base de les relacions entre la capacitat de la infraestructura verda per proporcionar serveis dels ecosistemes, la prestació efectiva o l'ús d'aquests serveis (flux), i la quantitat de serveis que demanda la població urbana. La identificació de la demanda insatisfeta, és a dir, el desajust entre el flux de serveis dels ecosistemes i la seva demanda, és un objectiu principal de les avaluacions ja que expressa els límits de la infraestructura verda urbana en relació als reptes considerats. La tesi utilitza i refina una varietat d'enfocaments metodològics per a la modelització i la cartografia de la capacitat, el flux i la demanda de serveis dels ecosistemes urbans (per exemple, les eines ESTIMAP i i-Tree). L'àmbit territorial de la investigació duta a terme dins el marc d'avaluació de la tesi doctoral abasta principalment l'àrea urbana de Barcelona, Espanya, tenint en compte tant l'escala local o de ciutat (municipi de Barcelona) i l'escala metropolitana o regional (regió metropolitana de Barcelona).

Els resultats de la investigació indiquen que la contribució dels serveis ambientals proporcionats per la infraestructura verda urbana per fer front als problemes urbans sovint és limitada (per exemple, el seu impacte sobre la qualitat de l'aire o la mitigació del canvi climàtic és inferior al 3% tenint en compte les emissions totals de carboni i la contaminació de l'aire en tots els estudis de cas) i/o incerta a les escales de ciutat o metropolitana. A més, l'impacte positiu de la infraestructura verda en la qualitat ambiental i el benestar humà es troba generalment limitat per 'perjudicis' ambientals (per exemple, les emissions biogèniques), *trade-offs* (per exemple, la provisió enfront de la regulació dels serveis) o desajustos espacials entre la provisió i la demanda de serveis (per exemple, les capacitats de purificació de l'aire i de recreació a l'aire lliure de grans blocs d'infraestructura verda metropolitanes estan massa lluny dels llocs de demanda).

Sobre la base d'aquests resultats, s'identifiquen diverses implicacions per a la planificació i gestió urbana/territorial, incloent: (1) la priorització de les polítiques de reducció de la pressions que generen una demanda per determinats serveis dels ecosistemes (per exemple, la purificació de l'aire i la captura de carboni); (2) la combinació d'estratègies de diversitat d'usos en sòl urbà i agrícola per tal d'augmentar la seva resiliència i multifuncionalitat i, al mateix temps, assegurar la conservació de grans àrees periurbanes forestals multifuncionals; (3) el desenvolupament de nous espais verds en els nuclis urbans compactes utilitzant estratègies innovadores (per exemple, cobertes verdes); i (4) la consideració de perjudicis i *trade-offs* en la planificació i gestió dels serveis dels ecosistemes. Finalment, sostinc que la planificació i gestió de la infraestructura verda urbana requereix un enfocament holístic, tenint en compte tota la gamma de serveis dels ecosistemes potencialment proporcionats pels diferents tipus d'infraestructura verda i les interaccions entre ells, juntament amb les diferents escales espacials a les quals aquests serveis poden ser rellevants per a la resiliència, la sostenibilitat i l'habitabilitat de les zones urbanes. Això exigeix una important coordinació institucional multi-escala i multidisciplinari entre totes les autoritats amb competències en polítiques urbanes i ambientals, així com l'harmonització dels instruments de planificació i gestió en un enfocament de governança a múltiples nivells.

RESUMEN (SUMMARY IN SPANISH)

En un planeta cada vez más urbano, muchas ciudades y sus habitantes se enfrentan a múltiples y apremiantes amenazas dentro de sus fronteras, incluyendo el estrés por exceso de calor, la contaminación y la creciente desconexión con la biosfera. Mejorar la sostenibilidad, la resiliencia y la habitabilidad de las áreas urbanas debe ser por lo tanto un objetivo de suma importancia en la agenda política, desde las autoridades locales a las globales. La aplicación del marco de servicios de los ecosistemas, a partir de los conceptos de 'infraestructura verde' y 'soluciones basadas en la naturaleza', se considera por un creciente número de responsables políticos, profesionales y científicos como el camino a seguir para hacer frente a muchos de estos desafíos urbanos. Sin embargo, el grado en que la infraestructura verde urbana puede ofrecer soluciones adecuadas a estos retos es rara vez considerado en las evaluaciones de servicios de los ecosistemas, y por lo tanto su potencial contribución es a menudo desconocida para los tomadores de decisiones.

Esta tesis examina de manera crítica el papel y la contribución de la infraestructura verde para hacer frente a diversos retos urbanos (con especial atención a la contaminación del aire, las emisiones de efecto invernadero, el estrés por exceso de calor y las oportunidades para la recreación al aire libre) a diferentes escalas espaciales. Partiendo del modelo de cascada de servicios de los ecosistemas, se propone y se aplica un marco operacional a través de cuatro capítulos de investigación originales para informar las decisiones de planificación y gestión sobre la base de las relaciones entre la capacidad de la infraestructura verde para proporcionar servicios de los ecosistemas, la prestación efectiva o el uso de estos servicios (flujo), y la cantidad de servicios que demanda la población urbana. La identificación de la demanda insatisfecha, es decir, el desajuste entre el flujo de servicios de los ecosistemas y su demanda, es un objetivo principal de las evaluaciones ya que expresa los límites de la infraestructura verde urbana en relación a los desafíos considerados. La tesis utiliza y refina una variedad de enfoques metodológicos para la modelización y la cartografía de la capacidad, el flujo y la demanda de servicios de los ecosistemas urbanos (por ejemplo, las herramientas ESTIMAP y i-Tree). El ámbito espacial de la investigación llevada a cabo dentro del marco de evaluación de la tesis doctoral abarca principalmente el área urbana de Barcelona, España, teniendo en cuenta tanto la escala local o de ciudad (municipio de Barcelona) y la escala metropolitana o regional (región metropolitana de Barcelona).

Los resultados de la investigación indican que la contribución de los servicios ambientales proporcionados por la infraestructura verde urbana para hacer frente a los problemas urbanos a menudo es limitada (por ejemplo, su impacto sobre la calidad del aire o la mitigación del cambio climático es inferior al 3% teniendo en cuenta las emisiones totales de carbono y la contaminación del aire en todos los estudios de caso) y/o incierta en las escalas de ciudad metropolitana. Además, el impacto positivo de la infraestructura verde en la calidad ambiental y el bienestar humano se encuentra generalmente limitado por 'perjuicios' ambientales (por ejemplo, las emisiones biogénicas), *trade-offs* (por ejemplo, la provisión frente a la regulación de los servicios) o desajustes espaciales entre la provisión y la demanda de servicios (por ejemplo, las capacidades de purificación del aire y de recreación al aire libre de grandes bloques de infraestructura verde metropolitanas están demasiado lejos de los sitios de demanda).

Sobre la base de estos resultados, se identifican varias implicaciones para la planificación y gestión urbana/territorial, incluyendo: (1) la priorización de las políticas de reducción de la presiones que generan una demanda por determinados servicios de los ecosistemas (por ejemplo, la purificación del aire y el secuestro de carbono); (2) la combinación de estrategias de diversidad de usos en suelo urbano y agrícola con el fin de aumentar su resiliencia y multifuncionalidad y, al mismo tiempo, asegurar la conservación de grandes áreas periurbanas forestales multifuncionales; (3) el desarrollo de nuevos espacios verdes en los núcleos urbanos compactos utilizando estrategias innovadoras (por ejemplo, cubiertas verdes); y (4) la consideración de perjuicios y *trade-offs* en la planificación y gestión de los servicios de los ecosistemas. Por último, sostengo que la planificación y gestión de la infraestructura verde urbana requiere un enfoque holístico, teniendo en cuenta toda la gama de servicios de los ecosistemas potencialmente proporcionados por los distintos tipos de infraestructura verde y las interacciones entre ellos, junto con las diferentes escalas espaciales a las que estos servicios pueden ser relevantes para la resiliencia, la sostenibilidad y la habitabilidad de las zonas urbanas. Esto exige una importante coordinación institucional multi-escalar y multidisciplinar entre todas las autoridades con competencias en políticas urbanas y ambientales, así como la armonización de los instrumentos de planificación y gestión en un enfoque de gobernanza a múltiples niveles.

Rambla del Raval, Barcelona, Spain (illustration by Kayla, published with kind permission of the author and the series of seminars on city, environment, health and drawing "Ciutat Verda" – www.ciutatverda.info)

CHAPTER I

Introduction and research objectives

1.1. Background and motivation

Our planet is increasingly urban: over half of world's population now lives in cities, and by 2050 that fraction will have increased to 66% according to United Nations prospects (UN, 2015). These prospects estimate that continuing population growth and urbanization will add 2.5 billion people to world's urban population by 2050, an increase mostly concentrated in Asia and Africa (see **Fig. 1.1**). Causes and effects of urbanization are manifold. Generally, cities are major hubs for economic and job opportunities and centralize many basic services such as healthcare or education. Although urban areas still cover a relatively small proportion of the terrestrial land surface of the planet (estimates range from 0.2% to 2.4%[1] circa 2000, according to Potere and Schneider, 2007), they have disproportionate environmental impacts well beyond their borders, affecting ecosystems at the local, regional, and global scales (Grimm et al., 2008; Seto et al., 2012). For example, 60% of residential water use has been attributed to cities (Grimm et al., 2008) and likely 60-70% of total anthropogenic greenhouse gas (GHG) emissions could be assigned to urban-related activities (Satterthwaite, 2008). Consequently, cities and their surrounding metropolitan areas often require vast areas of functioning ecosystems in order to fulfill their consumption (e.g., food, fresh water or construction materials) and waste assimilation needs. This 'ecosystem appropriation' by cities is often assessed through the 'ecological footprint' concept (Rees, 1992; Folke et al., 1997) or the 'ecology of cities' framework (Jansson, 2013). These approaches acknowledge the major dependence of cities on their hinterland (and beyond) and the links between urban and rural, viewing the city as an ecosystem itself (Grimm et al., 2008).

[1] This substantial 2.2% variation between the lowest and the highest estimate is mainly due to the varying approaches used to define what is urban land. In this dissertation, I use a flexible approach and define urban land as those areas where environmental conditions are linked to high population density, high extent of land transformation, and a large energy flow from surrounding area (Potschin et al., 2016 after McIntrye et al., 2000).

Concurrently, urban areas are also facing pressing challenges within their borders. Many cities worldwide are increasingly vulnerable to environmental extremes such as droughts, (coastal and inland) flooding or heatwaves because their frequency and magnitude is rising due to climate change (Revi et al., 2014). Pollution and other disturbances (e.g., noise) generated in cities have also direct and sometimes dramatic health impacts on the urban population (e.g., Brunekreef and Holgate, 2002; WHO, 2014). Many urban dwellers also suffer the manifold negative effects of sedentary lifestyles, social exclusion and increasing disconnection with the biosphere's ecological dynamics (Andersson et al., 2014).

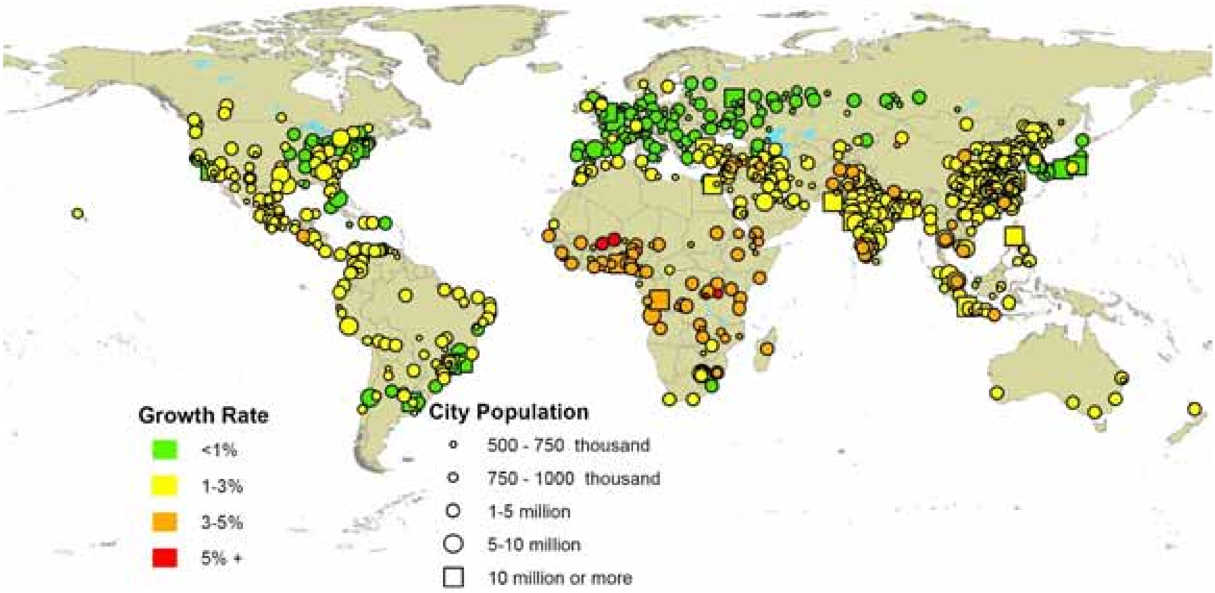

Fig 1.1. Growth rates of urban agglomerations by size class (prospect 2014 – 2030). Source: UN, 2015.

Improving sustainability, resilience and livability in cities should therefore be a major goal on any government's agenda, from local to global authorities. At a global scale, for example, one of the seventeen *United Nations Sustainable Development Goals*[2] is to "make cities inclusive, safe, resilient and sustainable". In this context, policy-makers, practitioners and scientists are paying growing attention to the sustainable planning and management of urban and periurban green spaces as a way to address many of these growing threats affecting urban areas (see some examples in McDonnell and MacGregor-Fors, 2016). In the European Union (EU), these strategies relying on urban ecosystems and their processes are mostly built on the concepts of 'green infrastructure' (GI, see EC, 2013) and, more recently, 'nature-based solutions' (NbS, see EC, 2015). Both terms are

[2] See http://www.un.org/sustainabledevelopment/sustainable-development-goals/

very much related as reflected in the EU GI strategy, which defines GI as "a successfully tested tool for providing ecological, economic and social benefits through natural solutions" and states that GI is based on the principle that "the many benefits human society gets from nature, are consciously integrated into spatial planning and territorial development" (EC, 2013:2; see also Section 1.3.2).

GI and NbS are useful notions in the context of operationalizing the ecosystem services (ES) framework which provides a powerful way of examining the interaction between ecosystems and human well-being. Since the seminal works of de Groot (1992), Daily (1997) and Costanza et al. (1997), research on ES has grown significantly. The Millennium Ecosystem Assessment (MEA, 2005), the Economics of Ecosystems and Biodiversity global initiative (TEEB, 2010) and the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES[3]) have brought the concept into broader planning and policy arenas. Generally, ES are defined as "the direct and indirect contributions of ecosystems to human well-being" (TEEB, 2010) and classified into four main categories: provisioning, regulating, cultural and supporting or habitat services (MEA, 2005; TEEB, 2010). Provisioning ES include all the material goods obtained from ecosystems, such as food, fiber, fresh water or medicinal resources. Regulating ES include all the ways in which ecosystems can mediate or moderate the ambient environment, including climate regulation, moderation of extreme events, erosion prevention or biological control. Cultural ES are the non-material outputs of ecosystems that affect physical and mental states of people, for example through spiritual experience, recreation, aesthetic appreciation or sense of place. Finally, supporting or habitat ES are defined as the ecological processes and functions that are necessary for the production of the previous 'final or end ES'[4], including habitat for species and maintenance of genetic diversity. In relation to ES classification systems is worth mentioning the initiative for a Common International Classification of Ecosystem Services (CICES[5]). CICES is complementary to MEA and TEEB classifications and aims to provide a systematic standardization of ES in the context of environmental accounting, mapping and valuation. The CICES classification follows a five level hierarchical

[3] See http://www.ipbes.net/

[4] The outcomes from ecosystems that directly lead to goods or benefits that are valued by people. This definition and the previous ES category definitions are based on the OpenNESS Glossary v3.0 (Potschin et al., 2016).

[5] See http://cices.eu/

structure (section, division, group, class and class type) to describe final ES (i.e., supporting ES are not considered in order to avoid possible 'double counting').

Attention paid to urban areas in the ES literature was initially modest as compared to other ecosystems located in more rural or natural landscapes (see MEA, 2005). This has changed over recent years. Since the seminal paper by Bolund and Hunhammar (1999), a growing body of literature has advanced our understanding of urban ES in their spatial, temporal, value or practical dimensions (Gómez-Baggethun et al. 2013; Haase et al., 2014). Gómez-Baggethun and Barton (2013) synthesized knowledge and methods to classify, assess and value urban ES for planning, management and decision-making. Urban ES such as air purification, noise reduction, urban temperature regulation or runoff mitigation, not explicitly considered in MEA (2005) and TEEB (2010) classifications, were highlighted in that work due to their expected relevance for the quality-of-life of the urban population. This dissertation largely follows the nomenclature used in this classification of urban ES (see also Gómez-Baggethun et al., 2013).

The book "Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities" (Elmqvist et al. (eds.), 2013), an output of the Cities and Biodiversity Outlook (CBO) project[6], identified at least four knowledge gaps related to urbanization and ES research. First, there is a geographical gap, since most scientific studies of urban ES are undertaken in Europe, North America and China (see also Haase et al., 2014; Luederitz et al., 2015). Second, there is also a valuation gap because non-monetary (e.g., socio-cultural) values of urban ES are still not considered on an equal basis with monetary values in decision-making processes (Gómez-Baggethun and Barton, 2013). Further, methods to approach insurance values, i.e., the value of ES and biodiversity in reducing urban vulnerability to shocks and disturbance from a resilience perspective, are still poorly developed (TEEB, 2010, but see Green et al., 2016 for recent progress in this direction). Third, despite their important role in cities (see Gómez-Baggethun et al., 2013), cultural ES provided by urban ecosystems are still under-researched relative to other categories such as regulating ES (e.g., Daniel et al., 2012; Haase et al., 2014; Langemeyer et al., 2015). Finally, there is a so-called 'supply-demand' gap because an increasing body of knowledge exists on the provision of ES (supply side) at different

[6] See http://www.cbobook.org

scales, but there is little information on needs, preferences and policy targets on ES (demand side) in urban areas, and whether these demands match or not the capacity of urban ecosystems to deliver ES (Haase et al., 2014). The main motivation of this dissertation is to bridge this last knowledge gap since the inclusion of demand in urban ES assessments is essential to evaluate the potential of GI (and NbS) strategies to cope with current and future urban challenges across different spatial scales and to identify the appropriate levels of planning, management and policy (Demuzere et al., 2014; Wolff et al., 2015)

1.2. Research objectives

Following the motivation described above, the general aim of this dissertation is to critically examine the current and potential contribution of GI to cope with diverse urban challenges (with a focus on air pollution, GHG emissions, heat stress and opportunities for outdoor recreation), both at the city (local) and metropolitan (regional) scales.

The specific objectives of this dissertation are the following:

- To advance conceptual understanding of urban GI and ES considering the GI's potential to deliver ES, the actual provision or use of the ES, and the amount of ES demanded by the urban population.
- To provide an operational framework for the application of these concepts in urban/landscape planning, management and decision-making.
- To develop and refine methodological approaches for modeling and mapping both the supply and demand of urban ES that can be used to design and inform GI strategies at the city and metropolitan scales.
- To apply the ES assessment framework and methods developed and refined to concrete place-based urban case studies (particularly the city and metropolitan region of Barcelona) and derive recommendations for planning, management and decision-making.

These objectives are addressed across the four original research chapters included in this dissertation (Chapters II-V). However, each of the Chapters considers these objectives in the context of the following specific aims:

- **Chapter II**: To quantify regulating ES provided by urban GI and discuss their potential contribution in achieving air quality and climate change mitigation policy targets in the city of Barcelona, Spain.
- **Chapter III**: To assess potential mismatches between the supply and demand of regulating ES on the basis of environmental quality standards and policy targets in five European cities.
- **Chapter IV**: To develop an operational framework for assessing and mapping ES capacity, flow and demand using the Barcelona metropolitan region as case study.
- **Chapter V**: To identify, map and assess supply of and demand for ES bundles along the urban-rural gradient in order to support GI planning and management in the Barcelona metropolitan region.

1.3. Conceptual and methodological framework

1.3.1. Ecosystem service capacity, flow and demand

This dissertation builds on the so-called 'ES cascade model' (Potschin and Haines-Young, 2011 based on previous frameworks such as de Groot et al., 2002, see **Fig. 1.2**) used widely in numerous global, national and subnational ES assessments such as TEEB (2011) or MAES (Maes et al., 2016a). This conceptual model describes key steps in the 'production chain' linking ecosystems (left-hand box) to human well-being (right-hand box) within socio-ecological systems. The framework highlights the relationships between ecosystem structures and processes, functions, services and the benefits that people gain from ecosystems which are finally valued either in monetary or non-monetary dimensions. It hence emphasizes that ES exist only in relation to people's needs and illustrates the possible implications of ecosystem degradation for human well-being if pressures driven by the socio-economic system are not limited via policy action. The main components of the ES cascade model are defined in **Box 1.1** together with other relevant concepts considered in this work. Even if other conceptual frameworks have been proposed in the literature for assessing ES (e.g., van Oudenhoven et al., 2012; Bastian et al., 2013; Villamagna et al., 2013; Schröter et al., 2014; Maes et al., 2016a), all of them are more or less rooted in variations of the ES cascade model and adapted to specific scale, policy or methodological goals.

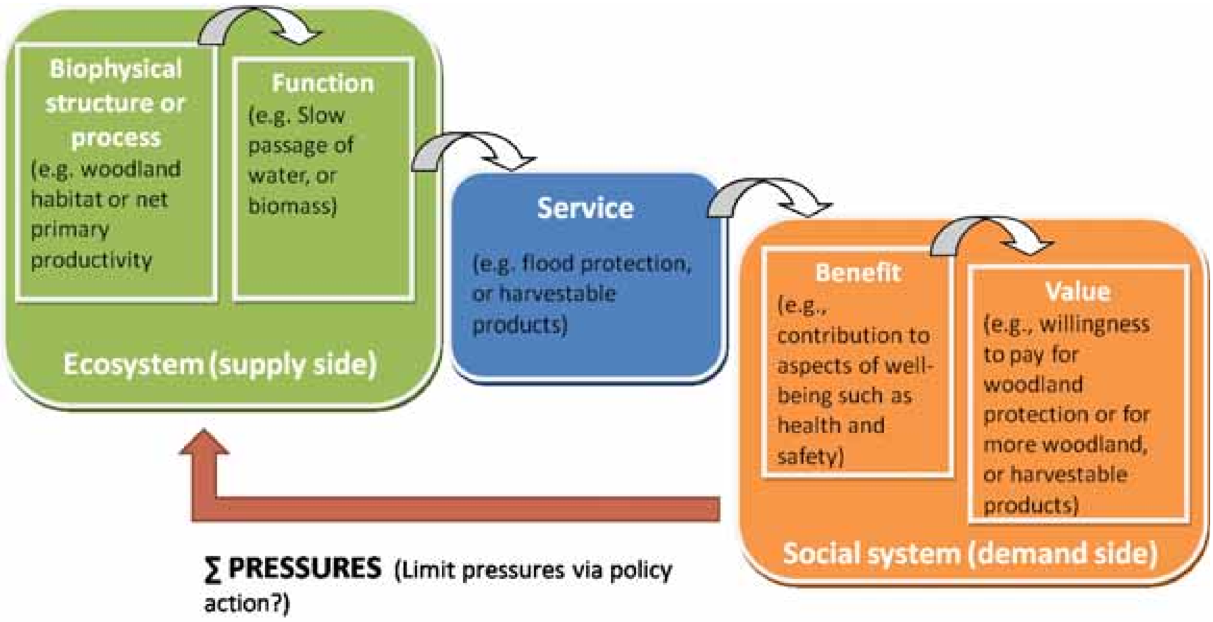

Fig. 1.2. ES cascade model. Source: own elaboration adapted from Potschin and Haines-Young (2011)

In this dissertation, I build on the ES cascade model to conceptually distinguish between ES capacity, flow and demand (see also Villamagna et al., 2013; Burkhard et al., 2014; Schröter et al., 2014). Following Villamagna et al., (2013:116), here I define ES capacity as "the ecosystem's potential to deliver ES based on its structures, processes and functions under the current management of the ecosystem", ES flow as "the ES actually received, used or experienced by people", and ES demand as "the amount of a service required or desired by society" (see also **Box 1.1.**).

It is worth noting that in the ES literature there are still different approaches and terminologies for framing these components, especially in regard to ES flow and demand (see Villamagna et al., 2013; Wolff et al., 2015). Some authors (e.g., Burkhard et al., 2012) have framed ES demand as direct use or final consumption, whereas other authors (e.g., Villamagna et al., 2013) argue that actual use or consumption of ES constitute its flow and that ES demand should be framed on the basis of societal desires and needs. The latter conceptualization of ES demand is inherently challenging at the operational level because it requires information about desired or required end conditions which can vary among different stakeholder groups. According to the review by Wolff et al. (2015), the different approaches used in the literature to operationalize the concept of ES demand are generally determined by the different ES categories (supporting or habitat services are not considered because they are not final ES and therefore they don't have a direct demand). A risk reduction approach is commonly applied to quantify demands for regulating ES. Demand indicators of regulating ES usually capture the magnitude of pressures or inputs needing regulation (e.g., air pollution levels for air purification) and the vulnerability or exposure of society to these pressures. Therefore, this approach assumes that ES demand is oriented toward a reduction of the indicator values (Burkhard et al., 2014). For most cultural ES, demand has been assessed using people's stated preferences, expectations or values, usually complemented with accessibility levels to ES providing areas such as parks or other green spaces (Wolff et al., 2015). Finally, a direct consumption approach is usually used for provisioning ES, considering indicators such as population density combined with average consumption rates or market prices (e.g., Burkhard et al., 2012; Kroll et al., 2012). This approach implicitly assumes that consumption rates (i.e., flows) of provisioning ES satisfy basic demands or needs for nutrition, medicinal resources or materials. These various approaches to ES demand are considered and discussed in this work, with a special focus on the risk reduction approach used to indicate demands for regulating ES (e.g., Chapter III). However, in all cases ES demand is framed as the desired or required level of the ES (i.e., considering the definition by Villamagna et al., 2013 mentioned above).

In this dissertation, an operational framework is proposed and applied across the different research chapters (see **Fig. 1.3**) to inform planning and management decisions on the basis of the relationships between ES capacity, flow and demand. On the one hand, the relationship between ES capacity and flow indicates ES overuse (if flow is higher than capacity) or scope for additional use (if capacity is higher than flow), when the ES is rival or congestible (i.e., the degree to which their use prevents other beneficiaries from using it, see Schröter et al., 2014). On the other hand, the relationship between ES flow and demand can indicate unsatisfied demand (if demand is higher than flow) or satisfied demand (if flow is higher or equal to demand). These two relations can be articulated in terms of ES (mis)matches, where the former expresses (un)sustainable uptake of ES and the latter expresses (un)satisfied demand for ES. Therefore an ES mismatch can be defined as the differences in quality or quantity occurring between the capacity, flow and demand of ES (Geijzendorffer et al., 2015). The relationship between capacity and demand is not explicitly considered in this framework because if demand is higher than capacity, the mismatch usually expresses an unsatisfied ES demand, unless flow is meeting the demand, in which case the mismatch would express an unsustainable ES uptake. This framework also considers that management and planning affect and are affected by the dynamics of all the three components.

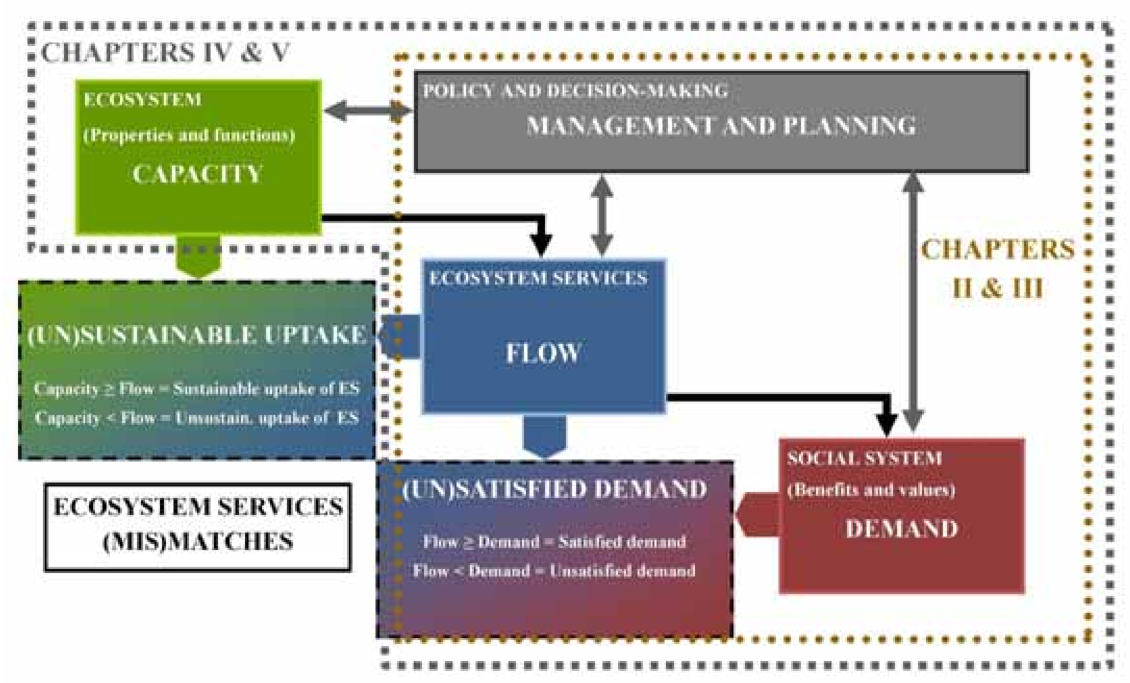

Fig 1.3. Main assessment domains of Chapters II-V in the conceptual framework of the dissertation. Source: modified from Chapter IV building on Potschin and Haines-Young (2011), Villamagna et al. (2013) and Geijzendorffer et al. (2015). Note: Chapter V does not explicitly assess ES flow and unsatisfied demand.

BOX 1.1. Definition of the main concepts discussed in this dissertation. Sources: based on OpenNESS Glossary 3.0 (Potschin et al., 2016), Villamagna et al. (2013), Geijzendorffer et al. (2015) and EC (2015). Note: for other concepts used in this dissertation it applies the OpenNESS glossary definition unless specified otherwise.

Ecosystem structure: A static characteristic of an ecosystem that is measured as a stock or volume of material or energy, or the composition and distribution of biophysical elements. Examples include standing crop, leaf area, % ground cover, species composition.

Ecosystem process: Dynamic ecosystem characteristic measured as a rate that is essential for the ecosystem to operate and develop, such as decomposition, production, nutrient cycling, and fluxes of nutrients and energy.

Ecosystem function: The subset of the interactions between biophysical structures, and ecosystem processes that underpin the capacity of an ecosystem to provide ES.

Ecosystem services: The direct and indirect contributions of ecosystems to human well-being.

Benefit: The direct and indirect outputs from ecosystems that have been turned into goods or experiences that are no longer functionally connected to the systems from which they were derived. Benefits can be valued either in monetary or social terms.

Value: The worth, usefulness, importance of something. Thus value can be measured by the size of the well-being improvement delivered to humans through the provision of goods and services. Values can be expressed in monetary or non-monetary dimensions.

ES capacity: The sustained ecosystem's potential to deliver ES based on its structures, processes and functions under the current management of the ecosystem.

ES flow: The ES actually received, used or experienced by people, which can be measured directly as the amount of a service delivered, or indirectly as the number of beneficiaries served.

ES supply: Generally used as ES capacity, flow or both. In this dissertation I consider that ES supply and ES provision are synonymous terms.

ES demand: The amount of an ES required or desired by society.

ES mismatch: The differences in quality or quantity occurring between the capacity, flow and demand of ES.

ES bundle: A set of associated ES that are supplied by or demanded from a given ecosystem or are associated with a particular place and appear together repeatedly in time and space. In a bundle, ES can be positively (**synergy**) or negatively (**trade-off**) associated.

Green infrastructure: A strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ES. It incorporates green spaces (or blue if aquatic ecosystems are concerned) and other physical features in terrestrial (including coastal) and marine areas. On land, GI is present in rural and urban settings (see also next subsection).

Nature-based solution: Nature-based solutions aim to help societies address a variety of environmental, social and economic challenges in sustainable ways. They are actions inspired by, supported by or copied from nature; both using and enhancing existing solutions to challenges, as well as exploring more novel solutions, for example, mimicking how non-human organisms and communities cope with environmental extremes (see also next subsection).

1.3.2. Green infrastructure and nature-based solutions: emerging concepts for the operationalization of ecosystem services

The concepts of green infrastructure (GI) and nature-based solutions (NbS) have recently emerged in the environmental research and policy agendas, especially in the European urban context (EC, 2013; 2015). As mentioned above, GI and NbS are related concepts since both can be considered a practical approach for the operationalization of the natural capital and ES framework (**Fig. 1.4**). Despite both GI and NbS can be framed as “boundary” or “umbrella” terms (i.e., words that function as concepts in different disciplines or perspectives, refer to the same object, phenomenon, process, or quality of these, but carry different meanings in those different disciplines or perspectives”, Mollinga, 2010:4), the inclusion of the idea of a ‘solution’ in the NbS concept explicitly recognizes (unlike GI) that there must be a problem that needs to be solved. This problem focus is, perhaps, a key characteristic that distinguishes it from more general notions of an ecosystem-based approach, or from more holistic framings of ES as sustaining or enhancing well-being. Therefore, the identification of problems or challenges that could be effectively addressed by NbS is a key aspect of this approach. This dissertation analyzes the potential of GI to provide NbS in relation to urban challenges such as air pollution, GHG emissions or heat stress (Chapters II, III and IV), but it also assesses the wider multi-functional character of GI (Chapter V). In the following lines, the ‘state-of-the-art’ of both concepts is described in more detail building on Baró et al. (2015) and Potschin et al. (2015).

Fig. 1.4. Definitions of natural capital, green infrastructure and nature-based solutions along an operationalization gradient. Own elaboration based on Potschin et al. (2016), EC (2013; 2015).

The concept of GI is gaining political momentum in both planning theory and policy, especially in US and Europe (Lennon, 2014). Yet, it does not have a single widely recognized or accepted definition (Wright, 2011). The term has been adopted by various disciplines (e.g., land conservation, urban design and landscape architecture), sometimes with substantially different conceptual meanings (see EEA, 2011 for a thorough list of GI definitions). For example, the development of GI is a strategic approach to land conservation, addressing the ecological and social impacts of consumption and fragmentation of open land (Benedict and McMahon, 2006). In urban design, the concept is mainly approached as a planned network of living systems affecting the quality of life of urban population (Defra and Natural England, 2013). Although the lack of a clear and unequivocal definition can lead to confusion and misuse among academics and practitioners, and eventually to a generalization of the term to "anything green", Wright (2011) argues that a single precise meaning of GI is problematic because the concept is still evolving and has developed in response to different needs. Still, a comprehensive but flexible definition of GI was proposed in the

European Commission (EC) communication "Green Infrastructure – Enhancing Europe's Natural Capital", commonly known as EU's GI Strategy (EC, 2013). GI is defined there as "a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ES. It incorporates green spaces (or blue if aquatic ecosystems are concerned) and other physical features in terrestrial (including coastal) and marine areas. On land, GI is present in rural and urban settings".

The physical components of GI can vary from very small local elements like urban gardens and green roofs to trans-boundary features such as mountain ranges or watersheds, and therefore, the EU's GI Strategy is rather broad and inclusive with respect to scales. For example, the Natura 2000 network can be considered a very relevant component of GI at the European scale, as the GI Strategy explicitly refers to this network of protected areas as "the backbone of the EU's GI". GI planning and implementation can be actually adapted to various scales along the urban-rural gradient. For example, Allen (2014) attempts to frame GI planning across three spatial scales – named as site, regional, and landscape - with specific implementation strategies at each scale.

Following EU's strategy GI definition, the identification and assessment of GI functions and benefits is increasingly underpinned by the conceptual framework of ES (EC, 2012; EEA, 2011; 2014; Maes et al., 2014; Kopperoinen et al., 2014; Liquete et al., 2015). The report on GI and territorial cohesion (EEA, 2011) identifies eight groups of GI benefits and the report by Mazza et al. (2011) suggests a very similar classification based on ten categories. Both proposals can be merged as follows: (1) biodiversity/species protection and conservation benefits; (2) climate and climate changed related benefits; (3) water management; (4) food production and security; (5) recreation, health and well-being; (6) land and property values; (7) education, culture and communities; (8) investment and employment; and (9) natural resources. Tzoulas et al. (2007) also proposed a conceptual framework linking GI elements, ecosystem functions and services, ecosystem health (such as habitat diversity) and four aspects of human health (physical, psychological, socio-economic and community health). Taking these multiple benefits into consideration, GI is often contrasted to 'grey' or built infrastructure. The EU, for example, argues that GI can be a cost-effective and environmentally friendly alternative, or complementary, to standard 'grey' solutions and, while the latter is normally designed as single-purpose, GI-based solutions can provide many benefits due to its multifunctionality (EC, 2012). For example, an increasing number of studies call for GI-based approaches in disaster risk management like flood protection instead of traditional grey infrastructures such as levees or dikes (e.g., Costanza et al., 2006 after Hurricane Katrina's catastrophe occurred in 2005). It is from this problem-oriented perspective that the NbS concept has emerged.

The term NbS entered the scientific literature in the early 2000s, in the context of solutions to agricultural problems – including integrated pest management, use of habitats to mitigate farm run-off etc. Blesh and Barrett (2006), for example, discussed integrating ecology into agricultural education and practice as a means of enhancing sustainability of food production. At around the same time, the idea of NbS began to appear in discussions on land-use management and planning and water resource management – including use of wetlands for waste water treatment (see Guo et al., 2000; Kayser and Kunst, 2002). From the mid-2000s, the concept also began to appear in literature on industrial design in the context of 'biomimicry'. For example, Singh et al. (2007) explored the hydrophobic and friction-reducing properties of artificial surfaces designed to mimic the topographies of water-repellent leaves as a solution to problems of wear in mechanical systems, and so promoted the search for NbS to industrial design challenges.

In recent years, the concept of NbS has been embraced by relevant international organizations. For example, the UN Secretary General has referred to NbS in the context of improving urban planning for better water resource management (UN, 2013), the World Economic Forum has highlighted the potential of NbS to support innovation in the travel and tourism sector (Marton-Lefevre and Borges, 2011), and NbS has also been a focus of World Bank investment in climate mitigation and adaptation projects (WB, 2008). The role of NbS has been actively promoted by the Nature Conservancy and the International Union for Conservation of Nature (IUCN). The Jeju Declaration arising from the IUCN World Congress in 2012 (IUCN, 2012), explored the theme of 'Nature+', which highlighted the importance of nature to enhancing societal resilience. It introduced the idea of NbS as a means of dealing with challenges linked to climate change, sustainable energy, food security, and economic and social development. The Declaration states that

“NbS build upon the proven contribution of well-managed and diverse ecosystems to enhance human resilience and to provide additional development opportunities for men and women in poor communities”. NbS has also emerged as a priority area for the EU’s Horizon 2020 Research Program. The Final Report of the Horizon 2020 Expert Group on 'Nature-Based Solutions and Re-Naturing Cities' (EC, 2015:24) defines NbS as “actions inspired by, supported by or copied from nature; both using and enhancing existing solutions to challenges, as well as exploring more novel solutions, for example, mimicking how non-human organisms and communities cope with environmental extremes”. The Report further notes that “NbS use the features and complex system processes of nature, such as its ability to store carbon and regulate water flows, in order to achieve desired outcomes, such as reduced disaster risk and an environment that improves human well-being and socially inclusive green growth. This implies that maintaining and enhancing natural capital is of crucial importance, as it forms the basis for solutions. These NbS ideally are resilient to change, as well as energy and resource efficient, but in order to achieve these criteria, they must be adapted to local conditions”. Finally, the EU BiodivERsA program (Balian et al., 2014) stresses that NbS should encompass a wider definition of how to conserve and use biodiversity in a sustainable manner. Their approach intends to additionally integrate societal factors such as poverty alleviation, socio-economic development and good governance beyond the traditional biodiversity conservation principles.

1.1.3. Modeling and mapping ecosystem services

Modeling and mapping of ES have been listed as key elements required to move forward from the theoretical frameworks and concepts described above to practical integration of ES into institutions and decision-making (Daily and Matson, 2008). ES models and maps can indeed inform a variety of decision-making contexts, including: awareness raising and communication (e.g., Hauck et al., 2013); environmental accounting (e.g., Schröter et al., 2014); landscape, urban and conservation planning (e.g., Palomo et al., 2014); and instrument design (e.g., Locatelli et al., 2014), among others. A clear example reflecting that ES mapping is gaining prominence in the environmental policy agenda is found in the EU Biodiversity Strategy to 2020 (EC, 2011). Under Action 5, EU Member States are committed to assess and map ES in their national territories to support conservation and restoration targets through GI implementation strategies. The

MAES ('Mapping and Assessment of Ecosystem and their Services') working group has fostered the implementation of this action through the development of an indicator framework for mapping ES which has been tested in various European case studies, including urban areas (see Maes et al., 2016a; 2016b).

Given the paramount importance of ES modeling and mapping for integrating the ES approach into decision-making, there has been a rapid increase in the number of studies that model ES and map their spatial distribution, as reflected in various systematic reviews on this topic (Martínez-Harms and Balvanera, 2012; Egoh et al., 2012; Crossman et al., 2013; Malinga et al., 2015; Wolff et al., 2015). Reviews show a great variety of spatial scales, types of data sources, indicators, methods, and tools used to model and map ES. However, Crossman et al. (2013) warn that this variety also reveals a certain inconsistency in the methodological approaches to quantify and map ES that challenges comparability of different case studies and the development of a robust system of (spatial) ES indicators in national accounts and broader policy arenas. Moreover, most ES assessments have focused on studying the spatial distribution of ES supply, and only recently the ES community has started to map and model ES demand (Wolff et al., 2015).

This dissertation largely builds on the blueprint for modeling and mapping ES developed by Crossman et al. (2013) which aims to reduce this still existing uncertainty associated with the spatially explicit quantification of ES, including the demand side. The following paragraphs and **Table 1.1** outlines the main methodological approaches and tools available for modeling and mapping ES with a special focus on those used in this research work.

The first methodological approach consists on the collection of ES data through empirical methods such as direct observations and field data sources. For example, food provision is normally derived from agricultural census data (Egoh et al., 2012), as performed in Chapter V. Demand for cultural ES is commonly based on primary qualitative and quantitative data derived from questionnaires or surveys (e.g., Langemeyer et al., 2015).This empirical ES data can be linked to a given spatial unit (e.g., a land-cover class) or integrated in a more complex process-based model (see below) for mapping the distribution of ES in larger areas (Martínez-Harms and Balvanera, 2012). Empirical methods generally lead to very accurate ES maps, although data availability is a clear limitation because its collection is generally time and cost intensive (Egoh et al., 2012; Eigenbroad et al., 2010).

The approaches using ES values obtained from empirical studies at other places and other spatial scales (generally up-scaling) where data is absent or limited have been called 'look-up tables' or 'value transfer' (Troy and Wilson et al., 2006; Martínez-Harms and Balvanera, 2012). For example, Larondelle et al. (2014) used look-up tables to map the supply of three regulating ES in more than 300 European cities. In this dissertation, I used this approach to map the erosion control potential of different habitats types (Chapter V), since empirical data was not available. From an ES demand perspective, this approach was used by Costanza et al. (1997) to estimate and map the monetary value of ES at a global scale. It is a straightforward method for mapping ES where primary data is lacking, but it has been criticized for neglecting spatial differences (in terms of quality, rarity, size, proximity to population centers, etc.) of a given habitat or land cover type, hence leading to potential large errors, among other limitations (Nelson et al., 2009; Eigenbroad et al., 2010).

Participatory and expert-based approaches use the knowledge or perceptions of stakeholders and experts to value and map ES, often supported with information from literature and secondary data (Wolff et al., 2015). Expert-based approaches are commonly used in the assessment of ES involving high complexities and uncertainties and where it is difficult to obtain empirical data on natural and societal processes. Therefore, they are also often used to fill data gaps in complex process-based models (e.g., Schulp et al., 2014). One of the most commonly applied and simple expert-based approaches is the ES matrix linking ES capacity, flow and demand qualitative scores to land cover types (e.g., Burkhard et al., 2012; 2014). Schröter et al. (2014) pointed out that a critical disadvantage of applying this approach relates to the reliability of the scoring process. Participatory approaches are especially relevant for mapping ES flow and demand because they can capture the heterogeneity of human desires, values and preferences and where ES are actually used. Palomo et al. (2013), for example, used participatory mapping techniques to identify areas in which beneficiaries used or experienced certain ES. In this dissertation, an expert-based approach was used to score different components of a recreational capacity map based on the ESTIMAP tool (Chapters IV and V). As all participatory approaches, stakeholder involvement is normally resource consuming and challenged by appropriate selection of actors, while results are generally site-specific and difficult to generalize (Wolff et al., 2015).

Proxy-based methods use a single or combined secondary indicators to define and map a new proxy layer of ES (Egoh et al., 2012). Generally, these approaches incorporate the existing knowledge on the causal relationships between a certain environmental or social variable and the supply or demand of ES (Martínez-Harms and Balvanera, 2012). This approach has been commonly used to map cultural ES. For example, Casado-Arzuaga et al. (2014) used a variety of spatial indicators (naturalness, protected areas, water features, recreational paths, etc.) to map recreation provision in the Bilbao Metropolitan Greenbelt. Similarly, in this dissertation I used proxy-based methods to map outdoor recreation capacity and flow as part of ESTIMAP tool (Chapters IV and V) and most part of demand indicators developed in Chapters II-V are based on proxies. Compared to land cover based approaches, these methods can offer a major improvement on performance, but potential for substantial error is still high if the assumed causal variables are not in fact good predictors (Eigenbroad et al., 2010).

Finally, process-based models also build on the theoretical understanding of social-ecological processes, but they generally employ a sample of empirical data as response variables and a variety of environmental and social indicators as explanatory variables (Martínez-Harms and Balvanera, 2012). Process-based models can capture dynamics of ES demand and supply considering its underlying drivers and pressures that determine ES flow and induce changes in the socio-ecological system (Wolff et al., 2015). This approach has been commonly used for mapping the supply and demand of regulating ES through hydrological models (Stürck et al., 2014), water yield models (Boithias et al., 2014), or pollination models (Schulp et al., 2014), among others. Generally, these models require a comprehensive understanding of the system, important amounts of data and expert knowledge (e.g., Maes et al., 2012). Because of its complexity, process-based models are usually integrated in larger tools for ES quantification, mapping and valuation (see below). In this work, I used process-based models integrated in the i-Tree Eco tool for estimating the supply of air purification and climate regulation at the city scale (Chapters II and III) and a regression model integrated in the ESTIMAP tool to map air purification at the metropolitan scale (Chapters IV and V).

These methodological approaches are not mutually exclusive. As mentioned above, in many cases a mix of different methods is used (e.g., expert-based knowledge in process-based models). The selection of the most convenient method is generally determined by the ES type to be studied, the availability of data, the spatial and temporal scales and the overall research goals and design.

Table 1.1. Overview of methodological approaches for ES modeling and mapping and its application in this dissertation. Source: own elaboration building on Martínez-Harms and Balvanera (2012), Eigenbroad et al. (2010), and Wolff et al. (2015).

Methodological approach	Main advantages	Main disadvantages	Examples in the literature	Application in the dissertation
Look-up tables (value transfer)	Upscaling applications Enables mapping of ES in areas s where primary data are lacking	Neglects spatial differences of habitat or land cover types, hence fit to actual data may be very poor	Costanza et al. (1997) Larondelle et al. (2014)	Chapter V (Erosion control)
Empirical methods	Provides the best estimate of actual levels of ES	If data is unavailable, its collection is usually time and cost intensive Normally require a process-based model for detailed spatial representation	Eigenbrod et al. (2009; 2010); Vollmer and Grêt-Regamey, (2013)	Chapter II and III (as part of i-Tree tool) Chapter V (food production)
Participatory and expert-based approaches	Especially relevant for mapping ES flow and demand because they can capture the heterogeneity of human desires, values and preferences	Reliability of the expert scoring process Appropriate selection of stakeholders and implementation costs	Burkhard et al. (2012; 2014) Palomo et al. (2013)	Chapters IV and V (as part of ESTIMAP recreation model)
Proxy-based methods	Enables mapping of ES in areas s where primary data are lacking and performs better than look-up tables	Substantial error is still high if the assumed causal variables are not in fact good predictors	Casado-Arzuaga et al. (2013); Eigenbroad et al., (2010)	Chapters IV and V (as part of ESTIMAP recreation model) and in all chapters for demand indicators
Process-based models	Models can capture dynamics of ES demand and supply considering its underlying drivers and pressures, hence can be applied in scenario building or alternative assessment	Usually require a thorough understanding of the system and important amounts of data. Other specific limitations associated to models	Stürck et al., (2014); Boithias et al., (2014); Schulp et al., (2014)	Chapter II and III(as part of i-Tree tool) Chapters IV and V (as part of ESTIMAP air purification model)

Following the wide-spread application of methods for ES modeling and mapping, a variety of analytical tools intended to enable replicable ES analyses for decision-support purposes have been developed (for a comprehensive review and comparative assessment of ES tools see Bagstad et al., 2013). A widely applied ES mapping and valuation tool is the Integrated Tool to Value Ecosystem Services and their Trade-offs (InVEST, Kareiva et al., 2011). It is an open access GIS-based tool developed under the Natural Capital Project[7]. It includes separate models for different ES (e.g., carbon sequestration and storage, crop pollination, recreation, scenic quality, etc.) to be applied and combined to analyze how changes in an ecosystem's structure and function are likely to affect the flows and values of ES across a land- or a seascape. The models account for both ES supply (e.g., living habitats as buffers for storm waves) and demand (e.g., people and infrastructure potentially affected by coastal storms). The complexity of the models available in InVEST varies from proxy-based mapping (tier 1), biophysical production equations (tier 2), to third-party complex, site-specific process-based models (tier 3). The main inputs to InVEST are land cover data and other relevant environmental variables, and outputs are the spatially explicit estimate of ES in biophysical and in some cases monetary units. Further relevant ES mapping tools are ARIES (Bagstad et al., 2011), the ARtificial Intelligence for Ecosystem Services[8], and SolVES (Sherrouse et al., 2011), the Social Values for Ecosystem Services[9]. ARIES is a web-based ES mapping and valuation tool, which uses probabilistic Bayesian networks to analyze ES flows from point of supply to place of use and beneficiaries. SolVES is a GIS tool to assess, map, and quantify the perceived social values for ecosystems, such as aesthetics, biodiversity, and recreation.

In this dissertation, I have used two tools for modeling and mapping ES in the case study areas (Chapters II-V): i-Tree[10] (Nowak et al., 2008a) and ESTIMAP (Zulian et al., 2014). The choice of these tools was motivated by: (1) its specific design for the assessment of urban ES in the case of i-Tree; and (2) its application in the framework of the OpenNESS project in the case of ESTIMAP. i-Tree is a state-of-the-art, peer-reviewed software suite developed by the USDA Forest Service and other cooperators that quantifies urban forest structure and estimates several ES provided by urban green

[7] See http://www.naturalcapitalproject.org/invest/
[8] See http://aries.integratedmodelling.org/
[9] See http://solves.cr.usgs.gov/
[10] See http://www.itreetools.org/index.php

space (mainly urban trees and shrubs). i-Tree includes a variety of analysis tools (some GIS-based), but in this work I used i-Tree Eco (Chapter II and III) and i-Tree Canopy (Chapter III). The i-Tree Eco tool (formerly known as Urban Forests Effects - UFORE), used in more than 2800 projects across the world, especially in the United States, includes various process-based models which require the collection of different field data parameters on urban green space (e.g., identification of tree and shrub species, trees' total height and crown width, etc.). Therefore, the precision and associated error of the ES estimates depends on the sample size (i.e., fieldwork plots are randomly located within the case study area) (Nowak et al., 2008b). The most widely applied i-Tree Eco models are used to estimate air purification (amount of pollution removed by urban trees and shrubs and associated percent air quality improvement throughout a year) and global climate regulation (total carbon stored and net carbon annually sequestered by urban trees and shrubs). Outputs include estimates of ES in biophysical and monetary units, but are not directly spatially explicit (a tree cover map can be used to obtain spatial estimates). i-Tree Canopy offers a straightforward way to produce a statistically valid estimate of urban tree cover using aerial images available in Google Maps. The tool also estimates values for air purification and carbon sequestration using i-Tree Eco models, but with higher potential error because fieldwork parameters are not considered. ESTIMAP (Ecosystem Services Mapping tool) is a collection of spatially explicit ES models initially designed to support European scale policies (Zulian et al., 2014). ESTIMAP includes several proxy and process-based models to map ES such as outdoor recreation (Paracchini et al., 2014), crop pollination (Zulian et al., 2013), coastal protection (Liquete et al., 2013) and air purification (see Chapter IV). Under the framework of the OpenNESS project, ESTIMAP models were adapted to regional case studies, including the case study considered in this dissertation (Barcelona metropolitan region). ESTIMAP is based on the ES cascade model, hence it provides spatial outputs indicating ES capacity, flow and demand.

1.1.4. Case study areas: Barcelona city and metropolitan region, Spain

The spatial scope of the research carried out within the framework of this Ph.D. dissertation principally encompasses the urban area of Barcelona, located North-East of Spain on the Mediterranean coast (**Fig. 1.5**). Chapter III is the only cross-city analysis which also considers four other European urban areas. I contend that Barcelona, as a complex socio-ecological system, is an excellent testing ground for the purpose of this research for at least three reasons: (1) it is one of the most densely populated urban areas in Europe (both at the core and metropolitan urban scales), which poses great pressures and challenges in relation to the fulfillment of ES demand; (2) it still contains a rich variety of natural and agricultural habitats of high relevance in terms of ES provision at the metropolitan level (**Fig. 1.5**; **Table 1.2**); and (3) both local and regional authorities have shown interest in considering the ES approach at an operational level (e.g., Barcelona Green Infrastructure and Biodiversity Plan 2020, Barcelona City Council, 2013; Territorial Information System for the Network of Open Areas in the province of Barcelona, Barcelona Provincial Council[11]). Because multi-scale ES assessments offer a number of advantages compared to traditional single-scale evaluations (see Scholes et al., 2013), the research was conducted in the Barcelona metropolitan region (BMR) (regional or metropolitan scale) and in the Barcelona municipality (local or city scale). Chapters II and III are conducted at the local scale whereas Chapters IV and V are conducted at the regional scale. Chapter VI (General discussion and conclusions) includes a synthesis integration of the results at both scales, exploring possible cross-scale interactions.

The BMR (5.03 million inhabitants living in a total area of 3,243 km^2, Statistical Institute of Catalonia , year 2015, see also **Fig. 1.6**) embeds 164 municipalities and seven counties, but its urban core is mainly constituted by the municipality of Barcelona (1.61 million inhabitants and 101.4 km^2, year 2015) and several adjacent middle-size cities. Distribution of land covers in the BMR is highly determined by its geomorphology. Two systems of mountain ranges (Catalan Coastal Range and Catalan Pre-Coastal Range) run parallel to the Mediterranean coast, mostly covered by Mediterranean forests (mainly Pine and Holm Oak), scrubland or grassland. Prominent examples of these ecosystems include protected areas such as the Montseny massif (Pre-Coastal Range) which has the highest peaks in the BMR, or the Collserola massif (Coastal Range) which is practically enclosed by built-up areas. In contrast, coastal and inland plains are mostly covered by urban and agricultural land. For instance, the Llobregat River delta is heavily sealed by urban land and transport infrastructure (e.g., the Barcelona airport), but it still preserves valuable agricultural and wetland areas. The Penedès area (west of the BMR) is an important wine-growing region. Mediterranean landscapes such as the BMR have

[11] See http://www.sitxell.eu/en/default.asp

been subject to increasing pressures over the last decades, leading to homogenization dynamics in terms of land use (Brandt and Vejre, 2004). Since the 1950s, the BMR has experienced an accelerated urbanization process, driven by an intense industrialization of its economy and an associated migration flow from rural areas (within the BMR and beyond) to cities. This flow consolidated a compact and densely populated urban core, but also favored a dispersed or sprawled urban model in some other areas (Catalán et al., 2008). As a result, a progressive abandonment of traditional agrosilvopastoral practices took place, especially in mountainous areas, together with consequent forest densification and afforestation of open land (Otero et al., 2013). These land cover changes can be estimated thanks to the land cover map of the Barcelona province corresponding to the year 1956 (MCS56, based on aerial photographs from that year, also known as the "American flight") and the latest edition of the land cover map of Catalonia (MCSC, 2009). According to these datasets, areas covered by forests or tree plantations have risen 13.6% during this period (16,214 ha), cultivated land has decreased by 58.2% (81,741 ha) and urban covers have increased by 334.5% (57,064 ha) in the BMR.

The municipality of Barcelona is the second largest city in Spain and one of the most densely populated cities in Europe (nearly 16,000 inhabitants per km^2, see also **Fig. 1.6**). Its current compactness is largely explained by physical boundaries which hamper further urban expansion: the Mediterranean Sea in the East, the Collserola massif in the West and adjacent cities in the North and South. Probably due to this limited land availability, green space in the municipality of Barcelona is scarce. The total green space within the municipality of Barcelona (including urban parks, periurban forests and other green land covers) amounts to 27.2 km^2 representing 26.8 % of the municipal area and a ratio of 16.9 m^2 of green space per inhabitant[12] (MCSC, 2009, see **Table. 1.2**). This ratio is very low compared to other European cities -especially in northern countries -where green space can amount to up to 300 m^2 per inhabitant in some cities (Fuller and Gaston, 2009). Moreover, a substantial share of this green space corresponds to the periurban forest of Collserola (**Fig. 1.5**). The inner green space of Barcelona (excluding Collserola) amounts to 11.2 km^2 (Barcelona City Council Statistical Yearbook, 2015), which

[12] These figures slightly change depending on the data source. For example, according to the Barcelona City Council Statistical Yearbook 2015, total green space amounts to 28.2 km^2 representing a ratio of 17.6 m^2 per inhabitant. In this section, I use the MCSC (2009) dataset to allow comparability with BMR figures.

corresponds to a ratio of 7.0 m^2 of green space per inhabitant. The largest inner green space is located in the mountain of Montjuïc covering about 300 ha (**Fig. 1.5**). Nonetheless, these low levels of green space are partly counterbalanced by the high number of single street trees, accounting for 159,178 specimens in 2014, a ratio of 99.2 street trees per 1000 inhabitants. This ratio is relatively high compared to other urban areas in Europe, which mostly ranges between 50 and 80 street trees per 1000 inhabitants (Pauleit et al., 2002).

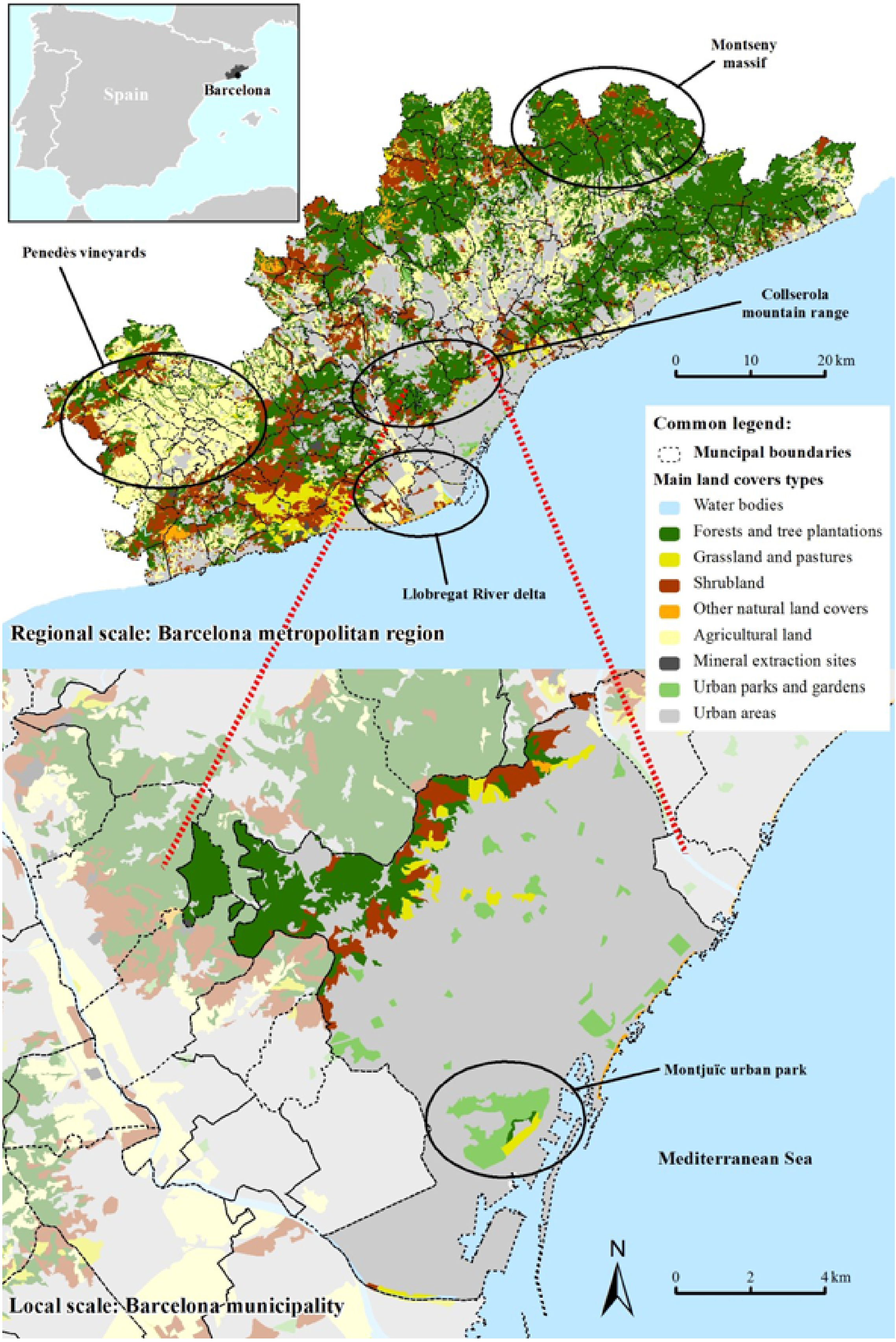

Fig. 1.5. Map of the main land covers types in the Barcelona Metropolitan Region (BMR) and the municipality of Barcelona. Source: own elaboration based on the spatial dataset "Habitats of Catalonia" (year 2013) available in SITxell.

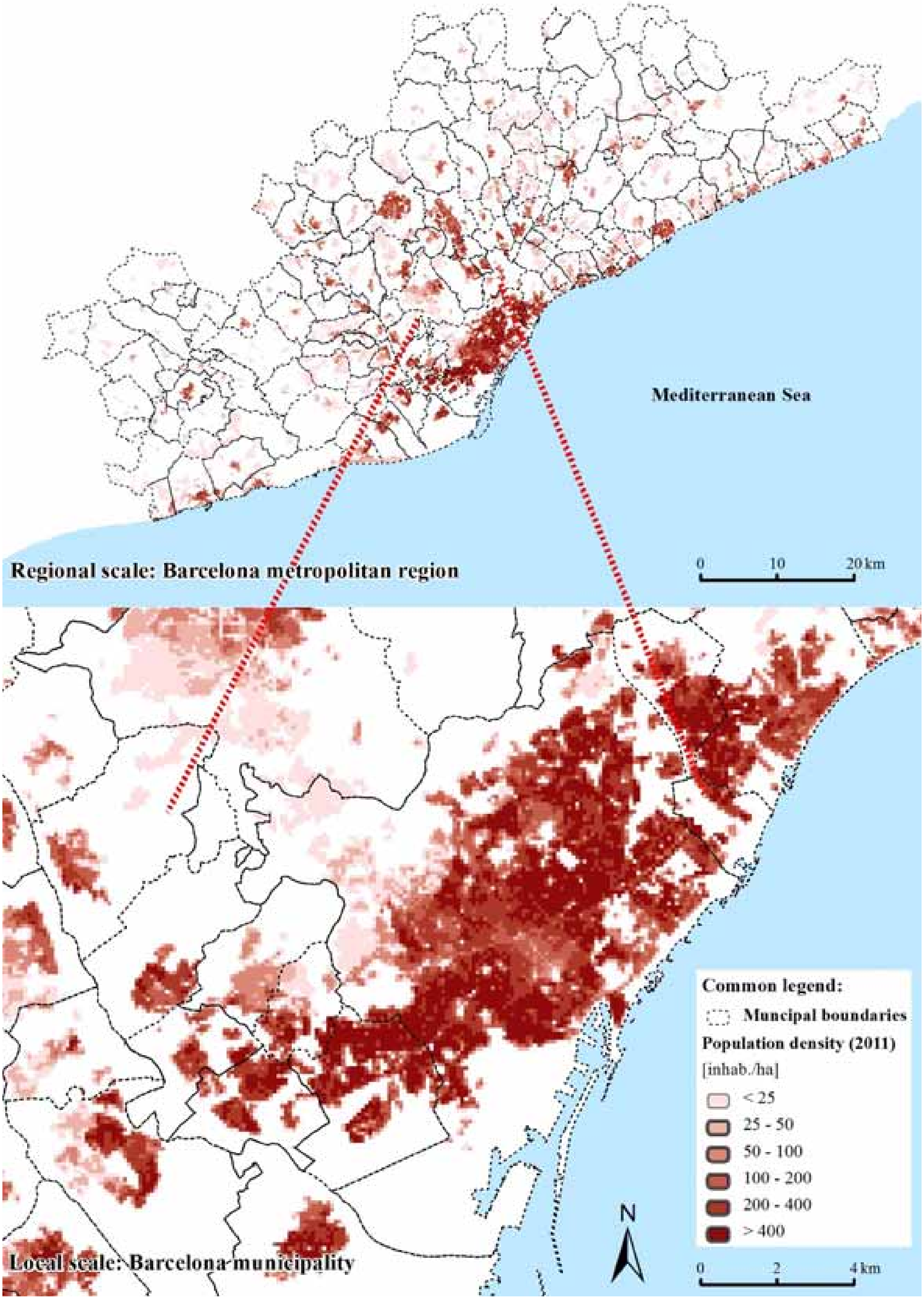

Fig. 1.6. Map of population density in the Barcelona Metropolitan Region (BMR) and the municipality of Barcelona. Source: own elaboration based on an intersect between census tract dataset (INE, 2011) and residential use classes extracted from high resolution land cover map (MCSC, 2009).

Table 1.2. Main land cover classes by area in the BMR and the Barcelona municipality. Source: own elaboration based on MCSC (year 2009).

Land cover class	BMR area (ha)	% BMR	City area (ha)	% City
Forests and tree plantations	135,717	41.84	1320	13.02
Scrubland	39,702	12.24	601	5.93
Grassland and pastures	8397	2.59	46	0.46
Water bodies (rivers, wetlands, etc.)	834	0.26	13	0.12
Other natural land covers (beaches, rocky areas, etc.)	3927	1.21	40	0.39
Crops	58,656	18.08	26	0.25
Urban green areas	2982	0.92	671	6.62
Urban land and other artificial covers	74,124	22.85	7424	73.21
Total	324,339	100.00	10,141	100.00

A multi-scale hierarchical system determines landscape and urban planning in the case study area. The BMR is one of the regional planning areas of the 'General Territorial Plan of Catalonia' (PTGC, 1995), the uppermost landscape planning instrument in the region of Catalonia. The 'Territorial Metropolitan Plan of Barcelona' (PTMB, 2010) was developed following PTGC's guidelines and approved in 2010 by the Government of Catalonia. The PTMB establishes three main planning categories, so-called "systems", for land use regulation in the BMR: (1) open areas; (2) urban land; and (3) transport infrastructure. The open areas planning system regulates the land protected from urbanization, including, fully or partially, fourteen Natura 2000 sites. The urban planning system regulates built-up land and defines strategies for urban expansion by the tentative delimitation of development areas that can be subsequently refined by municipalities through urban plans. For example, most municipalities of the urban core, including Barcelona, share a common urban plan (General Metropolitan Plan) which is currently under major revision. Finally, in a more strategic level, some municipalities such as Barcelona have approved specific plans for enhancing GI and biodiversity within their borders (e.g., Barcelona Green Infrastructure and Biodiversity Plan 2020, Barcelona City Council, 2013).

1.4. Structure of the dissertation

In terms of structure, this Ph.D. dissertation is a compilation of four original research papers preceded by the above general introduction and followed by a final chapter including a general discussion and main conclusions. At the time of submission, three research articles were already published (Chapters II, III and IV), and the fourth one was submitted for publication (Chapter V) [13]. Because the scientific articles are stand-alone publications, some degree of overlap between chapters and with this introductory chapter was unavoidable for some sections such as case study description or methodological approaches. However, I have decided to preserve each chapter in its original paper format in order to ensure their respective internal consistency. All the scientific articles have been developed under my personal lead (i.e., I am first author in all of them) with contributions by other co-authors as indicated in each Chapter's first page. I conceived, designed and performed the central research work described in each article, I analyzed the data and I wrote the first draft of the manuscripts and subsequent revised versions, ensuring the integrity of the work itself before submission to the scientific journal. Supervisors and co-authors provided expertise in the methods applied and/or contributed mostly to the introduction and discussion sections. In Chapter II, two co-authors provided the fieldwork data required for the research. In the next paragraphs, each of the following chapters is briefly presented. **Fig. 1.3** shows the main assessment domains of Chapters II-V within the conceptual framework of the dissertation. Finally, an overview of the main characteristics of the four empirical chapters is provided in **Table 1.3**.

Chapter II quantifies two regulating ES (air purification and global climate regulation) provided by urban green space in the municipality of Barcelona using the i-Tree Eco tool. The results are assessed and discussed in the context of their contribution to comply with policy targets of air quality and climate change mitigation applicable in the city. This chapter corresponds to the article 'Contribution of ecosystem services to

[13] Note: Chapter II was published in open access and hence can be reused in this dissertation. Chapters III and IV can be reused (full articles) too thanks to the Elsevier licenses nº 3920850126334 and nº 3920840969668 respectively. Chapter V will have the same reuse license if finally published.

air quality and climate change mitigation policies: the case of urban forests in Barcelona' published in the journal *Ambio* in April 2014.

Chapter III builds on the previous chapter to develop a consistent methodological approach to assess mismatches between ES supply and demand in urban areas on the basis of environmental quality standards and policy goals. The approach is applied to five European cities: Barcelona, Berlin, Stockholm, Rotterdam and Salzburg, considering three regulating ES: air purification, global climate regulation and urban temperature regulation. As advanced in Chapter II, results suggest that regulating ES supplied by urban GI are expected to play only a minor or complementary role to other urban policies intended to abate air pollution and GHG emissions at the city scale. This chapter corresponds to the article 'Mismatches between ecosystem services supply and demand in urban areas: A quantitative assessment in five European cities' published in the journal *Ecological Indicators* in April 2015.

Chapter IV presents a framework for assessing the relationships between ES capacity, flow and demand with a focus on the identification of unsatisfied demand in urban regions. The framework is tested in the BMR considering one regulating ES (air purification) and one cultural ES (outdoor recreation). For both ES, spatial indicators are developed using biophysical models from the ESTIMAP tool. The chapter contends that the mapping of ES capacity, flow and demand can contribute to the successful integration of the ES approach in landscape and urban planning because it provides a more comprehensive picture of the social and ecological factors underlying the ES delivery process. This chapter corresponds to the article 'Mapping ecosystem service capacity, flow and demand for landscape and urban planning: A case study in the Barcelona metropolitan region' published in the journal *Land Use Policy* in June 2016.

Chapter V builds on the framework presented in the previous chapter to identify, map and assess ES bundles form a supply-demand approach along the urban-rural gradient. The BMR is used again as case study area, considering a set of five ES (food provision; global climate regulation; air purification; erosion control; and outdoor recreation) and eleven spatial indicators (six at the supply side and five at the demand side). The chapter argues that, even if some ES can be provided from distant ecosystems, metropolitan regions such as the BMR have important motivations (e.g., food security, nature experience, climate adaptation and mitigation targets, etc.) to reduce their overall ES footprint. A combination of land sharing strategies and land sparing strategies is suggested as a promising approach in order to increase ES multifunctionality in urban regions and hence fulfill certain ES bundle demands of the urban population. This chapter corresponds to the article 'Ecosystem service bundles from a supply-demand approach: Implications for landscape planning and management in an urban region' submitted to the journal *Ecosystem Services* in August 2016. At the time of Ph.D. dissertation submission, the article was still under first review status.

Table 1.3. Main characteristics of the four empirical chapters included in the dissertation.

	Chapter II	**Chapter III**	**Chapter IV**	**Chapter V**
Spatial scale	City (local)	City (local)	Metropolitan (regional)	Metropolitan (regional)
Case study area	Barcelona	Barcelona; Berlin, Stockholm, Rotterdam, Salzburg	BMR	BMR
ES assessed	Air purification Global climate regulation	Air purification Global climate regulation Urban temperature regulation	Air purification Outdoor recreation	Food production Air purification Global climate regulation Erosion control Outdoor recreation
ES models and tools used	i-Tree Eco	i-Tree Eco and others	ESTIMAP	ESTIMAP and others
ES mapping	No	No	Yes	Yes
ES components assessed	Flow and demand	Flow and demand	Capacity, flow and demand	Capacity and demand (indirectly flow)
Publication status	Published (AMBIO)	Published (Ecological Indicators)	Published (Land Use Policy)	Submitted – Under review (Ecosystem Services)

Chapter VI presents the general discussion and main conclusions of this research. In this final chapter, the main conceptual and methodological contributions of the dissertation to ES science are synthesized. This chapter also outlines the key implications for planning, management and decision-making at different spatial scales of this research, highlights the main methodological limitations and caveats, and suggests potential areas for further research.

Finally, the dissertation includes the supporting information of the research chapters (Appendices A, B and C) and lists additional research achievements beyond the completion of this dissertation carried out during the Ph.D. period, including other scientific publications and reports, participation in scientific conferences, meetings and seminars, and other research activities (Appendix D).

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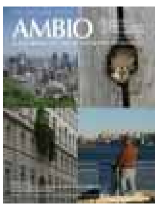
CHAPTER II

Contribution of ecosystem services to air quality and climate change mitigation policies: the case of urban forests in Barcelona

Abstract

Mounting research highlights the contribution of ecosystem services provided by urban forests to quality of life in cities, yet these services are rarely explicitly considered in environmental policy targets. We quantify regulating services provided by urban forests and evaluate their contribution to comply with policy targets of air quality and climate change mitigation in the municipality of Barcelona, Spain. We apply the i-Tree Eco model to quantify in biophysical and monetary terms the ecosystem services "air purification," "global climate regulation," and the ecosystem disservice "air pollution" associated with biogenic emissions. Our results show that the contribution of urban forests regulating services to abate pollution is substantial in absolute terms, yet modest when compared to overall city levels of air pollution and GHG emissions. We conclude that in order to be effective, green infrastructure-based efforts to offset urban pollution at the municipal level have to be coordinated with territorial policies at broader spatial scales.

Keywords: Air purification; Cities; Climate regulation; Urban ecosystem services; Urban forests; Policy targets.



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2.1. Introduction

Urban forests, encompassing all trees, shrubs, lawns and other vegetation in cities, provide a variety of ecosystem services (ES) to city-dwellers, such as air purification, global climate regulation, urban temperature regulation, noise reduction, runoff mitigation and recreational opportunities, as well as ecosystem disservices, such as air quality problems, allergies and damages on infrastructure (Escobedo et al., 2011; Gómez-Baggethun and Barton, 2013; Gómez-Baggethun et al., 2013). Specifically, a significant body of literature has stressed the contribution of urban forests in reducing air pollution levels and offsetting greenhouse gas (GHG) emissions in cities (e.g., Jo and McPherson, 1995; Beckett et al., 1998; McPherson et al., 1998; Nowak and Crane, 2002; Yang et al., 2005; Nowak et al., 2006; Paoletti, 2009; Zhao et al., 2010).

Air quality in cities is a major concern of the European Union (EU). In the last two decades, various policy instruments have been implemented at the European level to improve air quality in urban areas, mostly by regulating anthropogenic emissions of air pollutants from specific sources and sectors. These include the Directive 2010/75/EU on industrial emissions, the "Euro standards" on road vehicle emissions and the Directive 94/63/EC on volatile organic compounds emissions from petrol storage and distribution, among others. Yet, the 2013 annual report on air quality in Europe (EEA, 2013) estimated that many urban inhabitants in the EU are still exposed to air pollutant concentrations above the EU's legally binding limits (mainly set in the Directive 2008/50/EC on ambient air quality and cleaner air for Europe). For example, the report noted that 22–33% of the urban population within the EU was exposed to particulate matter (PM_{10}) concentrations above the 24-hour average limit value (50 μg m^{-3}) during the period 2009–2011. This estimation of exposure increases dramatically (85–88%) if it takes as reference the maximum levels recommended by the World Health Organization (WHO), currently set at 20 μg m^{-3} (annual mean).

As for climate change mitigation policy, the member states of the EU committed to reduce their GHG emissions by at least 20% from 1990 levels before the end of 2020 (Climate and Energy Package; EC, 2008). In an attempt to extent this commitment at the local level, the European Commission launched the 'Covenant of Mayors' in 2008. This initiative involves local authorities, voluntarily committing themselves to implement more sustainable energy policies within their territories by reducing GHG emissions at the local level by at least 20% until 2020. Such action by local authorities is deemed critical to meet global climate change mitigation targets because some 80% of worldwide energy consumption and GHG emissions are associated with urban activities (Hoornweg et al., 2011).

The focus of urban policy-making to meet the EU targets for both air quality and climate change mitigation largely remains on technical measures such as the use of the best available technology, fuel composition requirements, energy efficiency or renewable energy actions. The potential of urban green space in contributing to the compliance of these environmental targets is broadly neglected by urban policy makers (Nowak, 2006; Escobedo et al., 2011). Yet, a growing number of studies conclude that management of urban forests to enhance ES supply can be a cost-effective strategy to meet specific environmental standards or policy targets (e.g., Escobedo et al., 2008; 2010).

This research assesses ES and disservices provided by urban forests and it discusses their potential contribution in achieving air pollution regulation policy targets in cities. The objectives are twofold. First, we quantify in biophysical accounts and monetary values two ES ('air purification' and 'global climate regulation') and one ecosystem disservice ('air pollution' associated with biogenic volatile organic compounds (BVOC) emissions) generated by the urban forests in Barcelona, Spain. Second, we evaluate the potential of these ES to the achievement of environmental policy targets based on their actual contribution relative to air pollution and GHG emissions levels at the city scale. Accounting also the disservice allows having a 'net' estimate of this contribution, since BVOC emissions from urban forests can negatively impact air quality of cities (Nowak et al., 2000).

2.2. Materials and methods

2.2.1. Case study: Barcelona city

We conducted our research within the administrative boundaries of the municipality of Barcelona, Spain (**Fig. 2.1**). With 1.62 million inhabitants in an area of 101.21 km^2 (Barcelona City Council Statistical Yearbook, 2012), Barcelona is the second largest city in Spain and one of the most densely populated cities in Europe (16,016 inhabitants km^{-2}).

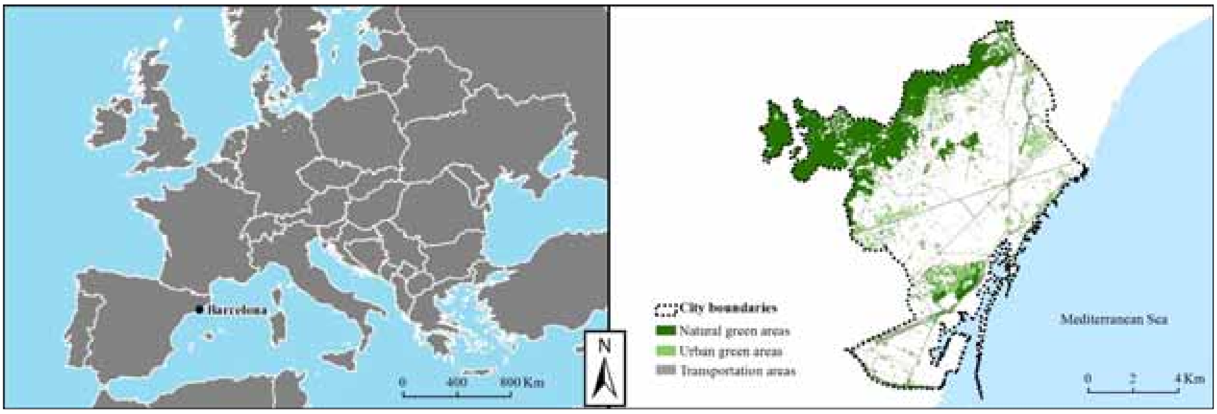

Fig. 2.1. Location of Barcelona municipality and its main green spaces. Source: own elaboration based on Natural Earth datasets (www.naturalearthdata.com) and 3rd edition of the Ecological Map of Barcelona (Burriel et al., 2006).

The total green space[14] within the municipality of Barcelona amounts to 28.93 km^2 representing 28.59% of the municipal area and a ratio of 17.91 m^2 per inhabitant (Barcelona City Council Statistical Yearbook, 2012). Most of this green space, however, corresponds to the periurban forest of Collserola (protected as a natural park). The inner-city of Barcelona (excluding Collserola) embeds only 10.98 km^2 of green space (Barcelona City Council Statistical Yearbook, 2012), which amounts to 10.85% of the municipal area and corresponds to a ratio of 6.80 m^2 of green space per inhabitant. This ratio is very low in contrast to other European cities – especially in northern countries – where green space amounts to up to 300 m^2 per inhabitant (Fuller and Gaston, 2009). Nonetheless, these low levels of green space are partly counterbalanced by the high number of single street trees, accounting for 158,896 specimens in 2011, a ratio of 98.36 street trees per 1000 inhabitants. This ratio is relatively high compared to other urban areas in Europe, which mostly range between 50–80 street trees per 1000 inhabitants (Pauleit et al., 2002). Two species, *Platanus hispanica* (46,779 trees) and *Celtis australis* (19,426 trees) account for almost one third of the street trees in Barcelona (Barcelona City Council Statistical Yearbook, 2012). Thanks to recent research (e.g., Chaparro and Terradas, 2009; Terradas et al., 2011), the role of urban forests in the provision of ES in Barcelona is starting to be acknowledged by the City Council as manifested, for example, in the *Barcelona Green Infrastructure and Biodiversity Plan 2020* (2013), a planning instrument that aims to aid the development of green infrastructure (GI) strategies in the present decade.

[14]Here 'green space' corresponds to those areas with vegetation (e.g. urban parks, gardens and other green areas) directly managed by the City Council. It includes also the natural and semi-natural areas of the Collserola Park, but it excludes green elements such as single street trees or private gardens.

As for many other large European cities (EEA, 2013), air quality improvement stands as one of the major environmental policy challenges for Barcelona. In the last decade, the city has repeatedly exceeded the EU limit values for average annual concentrations of nitrogen dioxide (NO_2) and PM_{10} pollutants (40 μg m^{-3} for both pollutants). The measures from the municipal monitoring stations during the period 2001–2011 show a steady trend for NO_2 values and a minor decrease for PM_{10} since 2006 (ASPB air quality report, 2011). During the same period, ground-level ozone (O_3) levels have frequently exceeded the EU target value for human health (120 μg m^{-3} for a daily maximum 8-hour mean period), but have never surpassed the number of allowed exceedances (25 days per year averaged over three years). Finally, carbon monoxide (CO) and sulfur dioxide (SO_2) concentrations have been historically very low in the city of Barcelona, never exceeding the EU limit values (125 μg m^{-3} in one day for SO_2 and 10 mg m^{-3} for 8-hour average for CO) (ASPB air quality report, 2011). **Fig. 2.2** synthesizes the EU limit values for air quality and the maximum levels measured in Barcelona during 2011.

In 2008, Barcelona generated approximately 4.05 million metric tons of carbon dioxide equivalent (CO_2eq) emissions, mainly due to energy consumption in the transportation, industry, housing and services sectors (PECQ, 2011). Compared to other cities world-wide, the ratio of Barcelona (2.51 t CO_2eq per inhabitant) is one of lowest proportions (Dodman, 2009; Kennedy et al., 2009). This same year, the City Council of Barcelona signed the 'Covenant of Mayors', committing to reduce by 23% GHG emissions only derived from services and activities directly managed by the City Council by 2020 (this so-called "municipal" GHG emissions include emissions from municipal buildings, street lighting, municipal vehicle fleet and waste collection, among others). In 2008 (baseline year for Barcelona), municipal CO_2eq emissions amounted to 84,403 t, a ratio of 0.052 t per inhabitant (PECQ 2011, see **Fig. 2.2**).

The Energy, Climate Change and Air Quality Plan of Barcelona (PECQ, 2011) provides the framework policy for air quality regulation and climate change mitigation during the period 2011–2020. Like other policy instruments aimed at improving indicators of environmental quality, the PECQ does not consider the enhancement of GI as a potential strategy to meet the policy targets established for air pollution concentrations and GHG emissions, as it focuses mainly on measures to improve energy efficiency and other technical fixes.

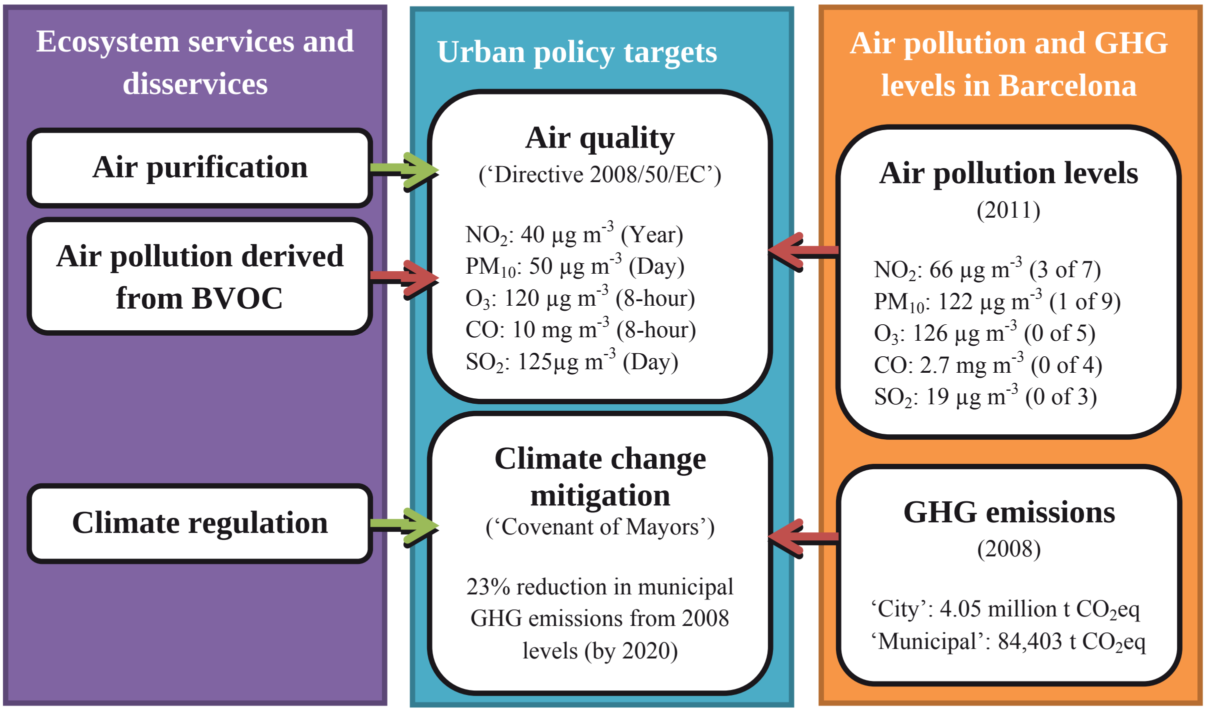

Fig. 2.2. Framework for assessing links between ES and disservices, urban policy targets and air pollution and GHG levels in Barcelona. Notes: Air quality policy limits correspond to the most stringent EU values set for the protection of human health (in brackets the averaging period applicable for each limit). Some limits are subject to a specific number of allowed exceedances (e.g., PM_{10} limit can be exceeded 35 days per year at the most). See EEA (2013) for more details. Air pollution levels in Barcelona show the highest concentration values among all the monitoring stations measuring the corresponding air pollutant during the year 2011 (in parenthesis the number of monitoring stations exceeding the air quality limit after considering the number of allowed exceedances). See ASPB air quality report (2011) for more details. Arrows represent the links between ES and disservices, air pollution and GHG levels and urban policy targets in Barcelona (red arrows represent a negative impact towards policy targets and green arrows a positive impact). Sources: own elaboration based on EEA (2013), ASPB air quality report (2011) and PECQ (2011).

2.2.2. Sample design and data collection

The i-Tree Eco model (formerly known as Urban Forests Effects - UFORE) (Nowak and Crane, 2000) was used to quantify ES and disservices in Barcelona. The i-Tree Eco model has been used in more than 50 cities across the world, especially in the United States, to assess urban forest structure and ES (Nowak et al., 2008a).

I-Tree Eco protocols (Nowak and Crane, 2000; Nowak et al., 2008a, b; i-Tree User's Manual, 2008) were followed to collect field data on urban forest structure within the municipality of Barcelona. Field data were collected within 579 randomly located circular plots (each measuring 404 m^2; 11.34 m radius) distributed across the city and pre-stratified among eight land use classes based on the third edition of the Ecological Map of Barcelona (Burriel et al., 2006, see **Fig. 2.3**). Plot centers were positioned from a random number generator of *x* and *y* coordinates for each land use class by means of a geographic information system (Miramon software, see Pons, 2006). Prior to fieldwork, plots without vegetation cover were identified using 1:5000 digital aerial ortho-photographs from the Catalan Cartographic Institute (year 2004). Only the plots with vegetation cover (trees, shrubs or herbaceous flora) were then visited for field data collection (see **Table 2.1** for sample data general figures).

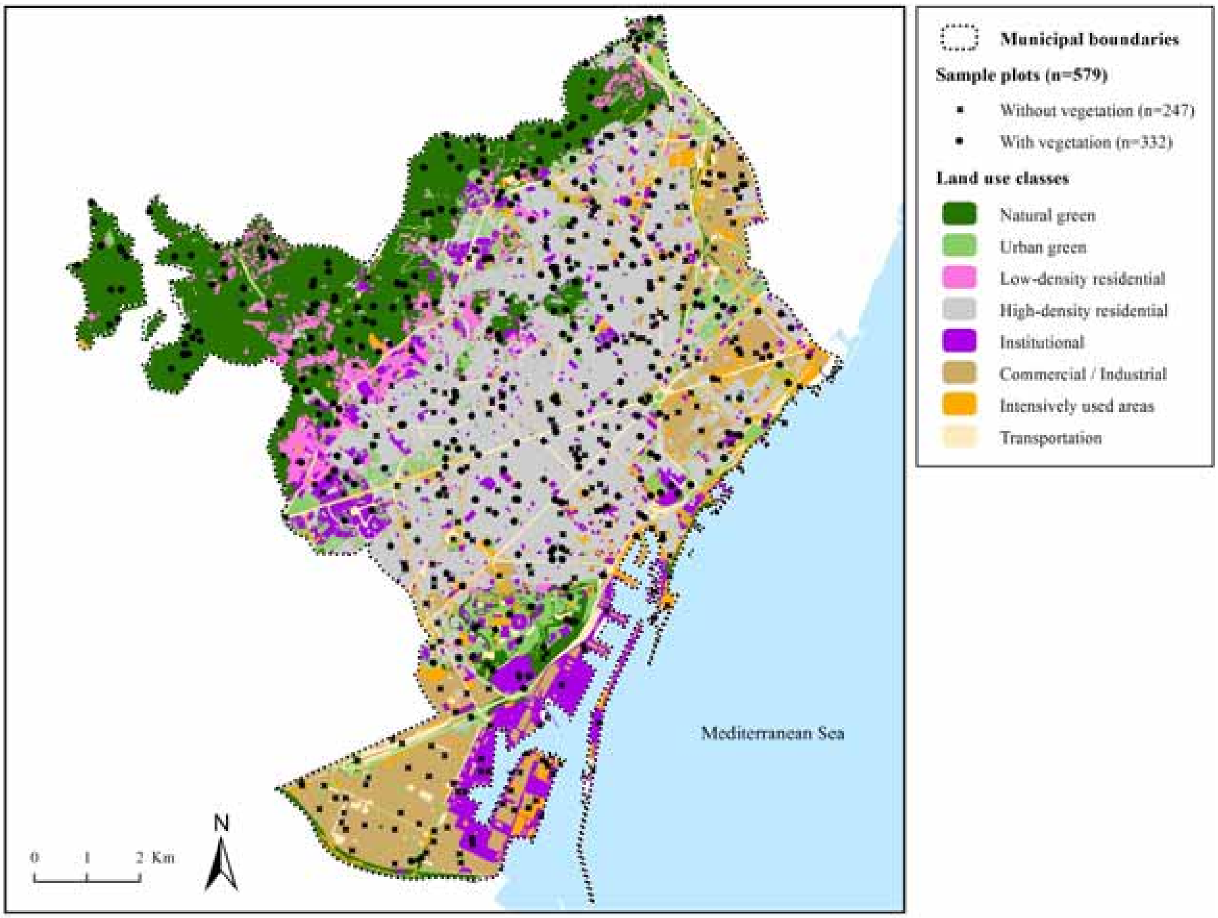

Fig. 2.3. Land use classes and location of sample plots within the municipality of Barcelona. Source: own elaboration based on the 3rd edition of the Ecological Map of Barcelona (Burriel et al., 2006).

Fieldwork was carried out from May to July 2009. Plots were located using a Global Positioning System (GPS) device supported by high resolution maps containing the precise position of the plot center and its perimeter. Inaccessible plots (due to the steep slope, lack of permission to enter private areas, impenetrable vegetation, among others) were relocated in the closest accessible area with similar land use and vegetation characteristics. The general information collected from each visited plot included, among other parameters, date of visit, GPS coordinates, actual land use (and percent of land uses if the plot fell in more than one land use class) and percents of tree cover, shrub cover, plantable space and ground cover. Main data on shrubs included the identification of species (genus at a minimum), average height, and percent area relative to total ground area. These data were collected for shrub masses (same species and height) and not at the individual level. Main data on trees included the identification of species, diameter at breast height (DBH), total height, height to crown base, crown width, percent of canopy missing (relative to crown volume), percent of impervious soil beneath canopy, percent of shrub cover beneath the canopy, and light exposure of the crown (see Nowak et al., 2008a for a complete list of data measures). Requirements of data inputs also include hourly air pollution concentrations and meteorological data (e.g., air temperature, solar radiation and precipitation averages) for a complete year. The Public Health Agency of Barcelona (ASPB) provided concentration data for CO, SO_2, O_3, NO_2, and PM_{10} air pollutants from the 13 operational monitoring stations of the city during the year 2008. Meteorological data of Barcelona was directly retrieved from the US National Climatic Data Centre (year 2008). Thus, the results from the evaluation of ES and disservices correspond to the year 2008.

Table 2.1. Sample data by land use stratification.

Land use class	Description*	Total area (ha)	Sample data				
			Sampled area (ha)	No. of plots	No. of plots with woody vegetation**	No. of trees	No. of shrub masses***
Urban green	Urban parks, lawns, allotment gardens, permanent crops, flowerbeds	806	2.02	50	50	544	89
Natural green	Woodland, scrubland, grassland, riparian vegetation, bare rock	2184	5.05	125	117	1844	329
Low-density residential	1-2 family dwellings (normally with private garden)	424	0.81	20	15	174	55
High-density residential	Multi-family dwellings with or without commercial areas	3666	8.24	204	102	531	79
Transportation	Parking lots, roads, rails and streets, stations	513	1.21	30	14	69	10
Institutional	Education, health, military, sport and other public facilities, cemeteries, port	776	1.58	39	3	21	0
Commercial/ Industrial	Factories and other industrial areas, warehouses, large shopping centers	1185	2.83	70	7	14	0
Intensively used areas	Pedestrian areas, vacant areas, areas in transformation	567	1.66	41	24	148	8
Total		10,121	23.39	579	332	3345	570

Notes: *Based on land use subclasses from the 3rd edition of the Ecological Map of Barcelona (Burriel et al., 2006). **Plots with woody vegetation account for those whether with shrubs or trees, or both. ***Data on shrubs were collected for shrub masses (same species and height) and not at the individual level.

2.2.3. Quantification and valuation of ecosystem services and disservices

Field data of urban forest structure, air pollution and meteorological data were processed using i-Tree Eco software (www.itreetools.org) to quantify the ES of air purification and climate regulation, and the disservice air pollution derived from BVOC emissions in both biophysical and economic terms. Besides, the model also provided general results on the urban forest structure of Barcelona, including information on species composition, species origin and diversity, leaf area index (LAI), and leaf biomass. The analysis of the urban forest structure of Barcelona is beyond the scope of this paper; however we refer to some relevant information in the discussion section.

The air purification service was quantified on the basis of field data, air pollution concentration and meteorological data. Fundamentally, the i-Tree Eco model estimates dry deposition of air pollutants (i.e., pollution removal during non-precipitation periods), which takes place in urban trees and shrub masses. The (removed) pollutant flux (F; in g m^{-2} s^{-1}) is calculated as the product of deposition velocity (V_d; in m s^{-1}) and the pollutant concentration (C; in g m^{-3}). Deposition velocity is a factor computed from various resistance components (for more details see Baldocchi et al., 1987; Nowak and Crane, 2000; Nowak et al., 2006; 2008a). Monetary values of the ES air purification were estimated in i-Tree Eco from the median externality values for each pollutant established for the United States (Murray et al., 1994) and adjusted by the producer's price index for the year 2007 (US Department of Labor). Externality values applied to the case study are: NO_2 = 9906 USD t^{-1}, PM_{10} = 6614 USD t^{-1}, SO_2 = 2425 USD t^{-1}, and CO = 1407 USD t^{-1}. Externality values for O_3 are set to equal the value for NO_2.

The ES of climate regulation was calculated based on the modeling results of gross carbon sequestration, net carbon sequestration (i.e., estimated net carbon effect after accounting for decomposition emission of carbon from dead trees) and carbon storage. The i-Tree Eco model calculates the biomass for each measured tree using allometric equations from the literature. Biomass estimates are combined with base growth rates, based on length of growing season, tree condition and tree competition, to derive annual biophysical accounts for carbon storage and carbon sequestration. Several assumptions and adjustments are considered in the modeling process (for more details, see Nowak and Crane, 2000; 2002; Nowak et al., 2008a). To estimate the monetary value associated with urban tree carbon storage and sequestration, biophysical accounts were multiplied by 78.5 USD t^{-1}carbon based on the estimated social costs of carbon dioxide emissions in the US for the year 2010 (discount rate 3%, EPA, 2010). Additionally, we considered GHG emissions generated by the municipal vehicle fleet dedicated to green space management (862.50 t CO_2eq according to PECQ, 2011) as a proxy of total GHG emissions directly attributable to green space maintenance. Hence, this measure was subtracted from total net carbon sequestration estimate provided by urban forests (after applying the conversion factor 1 g C = 3.67 g CO_2eq).

The emission of BVOCs from trees and other vegetation can contribute to the formation of ground-level O_3 and CO air pollutants (Kesselmeier and Staudt, 1999), hence counteracting the air purification that vegetation delivers. BVOC emissions depend on factors such as tree species, leaf biomass, daylight and air temperature (Nowak et al., 2008a). The i-Tree Eco model estimates the hourly emission of isoprene (C_5H_8), monoterpenes (C_{10} terpenoids), and other BVOCs by trees and shrubs species using protocols of the Biogenic Emissions Inventory System (BEIS; see Nowak et al., 2008a for further details). To estimate the amount of O_3 produced by BVOC emissions, the model applies incremental reactivity scales (g O_3 produced per g BVOC emitted) based on Carter (1994). CO formation from BVOC emissions is estimated for an average conversion factor of 10% based on empirical evidence (Nowak et al., 2002a). However, due to the high degree of uncertainty in the approaches of estimating O_3 and CO formation derived from BVOC emissions, no estimates of the total amount of pollution formed by urban forests are given (neither monetary costs). Only index values can be calculated to compare the relative impact of the different species on O_3 and CO formation (Nowak et al., 2002a).

2.2.4. Contribution of urban forests to air quality improvement and climate change mitigation

The relative contribution of urban forests to air quality improvement and climate change mitigation in Barcelona for the year 2008 was determined based on data of air pollution levels and GHG emissions. We considered emissions generated within the municipal area (hereafter city-based pollution) and pollution not directly attributable to city-based emissions (hereafter background pollution) to determine air pollution levels in the city. We only accounted for PM_{10} and NO_2 levels since, as described above, these are the two air pollutants whose concentrations are frequently exceeding EU value limits in the city. Data for city-based pollution and background pollution were extracted from PECQ (2011) estimations. PECQ (2011) measures include aggregated and disaggregated city-based emissions from different sectors (road transport, residential and tertiary, industry and energy generation and port activity), which in turn draws on a wide range of primary data sources (e.g., vehicle population, annual vehicle mileage, consumption of gas in households and businesses, etc.) and apply various quantitative methods (e.g., COPERT/CORINAIR model for road transport). Background pollution is measured from real pollutant concentration values recorded by the monitoring stations in the city and from one monitoring station located in the area of 'Cap de Creus' (130 km north-east from Barcelona), hence not influenced by polluting activities within the city. According to PECQ (2011), the annual average concentration of NO_2 for the year 2008 in Barcelona was mainly determined by emissions from road traffic (65.6%), while background pollution only accounted for 18.7%. In contrast, the annual average of the PM_{10} concentration was primarily determined by background pollution (88.1%).

The rate of GHG emissions was also extracted from PECQ (2011). Calculations are based on the various energy sources generating GHG emissions in the city (mainly electricity, natural gas and vehicle fuels). Electricity-related GHG emissions are calculated based on the Catalan electricity mix.

2.3. Results

2.3.1. Air purification

Total air purification is estimated at 305.6 t of removed pollutants year^{-1} with an economic value of 2.38 million USD year^{-1} (**Fig. 2.4**). PM_{10} removal is the highest among the five air pollutants analyzed (i.e., CO, NO_2, PM_{10}, O_3 and SO_2), accounting for 54% of the total biophysical value (166.0 t year^{-1}) and 46% of the total economic value (1.10 million USD year^{-1}). Pollution removal was lower for NO_2 and ground level O_3 (54.6 t, 541,000 USD for NO_2; 72.6 t, 719,000 USD for O_3), and lowest for CO and SO_2 (5.6 t, 7880 USD for CO; 6.8 t, 16,000 USD for SO_2).

Average values for monthly removal of air pollution show a similar pattern across pollutants. January, November and December were clearly the months where the uptake was lowest for all pollutants (percentages of uptake during the 3 months were 4.58 for CO, 8.45 for NO_2, 15.15 for PM_{10}, 2.69 for O_3 and 6.75 for SO_2). Spring and summer (from April to September) were the seasons with higher removal rates in average (percent of uptake during the 2 seasons was 60.96 for CO, 64.25 for NO_2, 54.43 for PM_{10}, 78.90 for O_3 and 70.46 for SO_2), although in some cases the highest monthly uptake rate corresponded to other periods (e.g., PM_{10} removal was highest in February, accounting for 10.69% of total uptake). These patterns in uptake values are normally correlated with the seasonal variation in air pollutants concentrations and the biological cycle of trees (Nowak, 1994; Yang et al., 2005). For instance, removal rates of ground level O_3 are highest in summer, when concentrations are normally higher due to a more active process of photochemical reaction forming O_3 as a consequence of warmer temperatures and due to increased leaf surface area and gas exchange at the leaf surface.

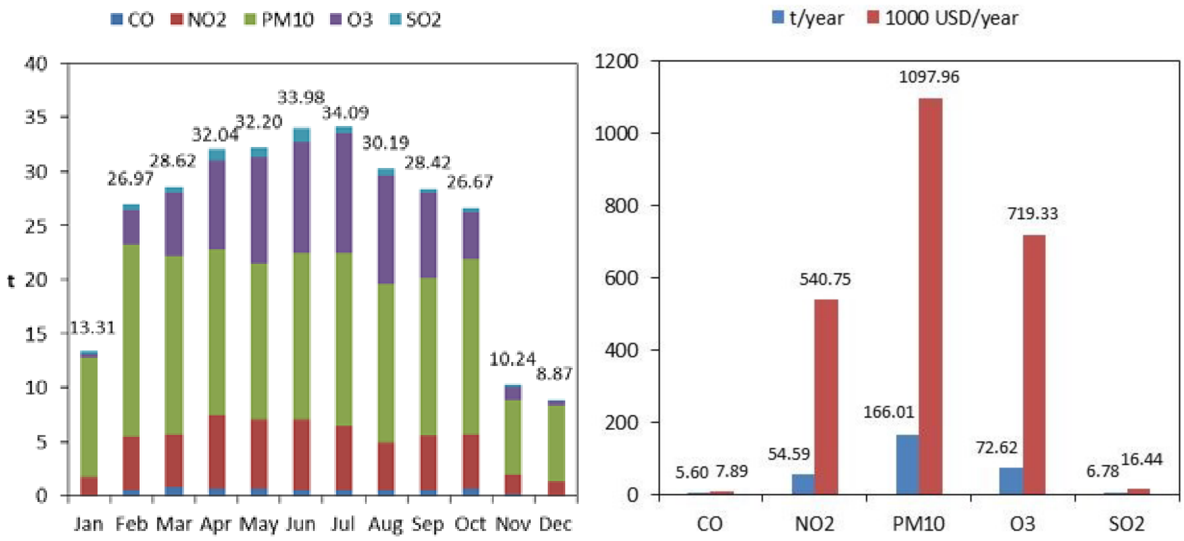

Fig. 2.4. Monthly and annual air pollution removal by air pollutant (Urban forests of the municipality of Barcelona, year 2008).

2.3.2. Climate regulation

The total biophysical value of net carbon sequestration is estimated at 5187 t C year^{-1} (19,036 t CO_2eq year^{-1}) with an economic value of 407,000 USD year^{-1} (**Table 2.2**). This total net carbon sequestration is the only value including the effect of GHG emissions of green space maintenance, since disaggregate data by land use was not available. In absolute terms urban green, natural green and high-density residential are the land use strata contributing the most to total net carbon sequestration (19%, 39% and 24% respectively). However, considering the ratio net carbon sequestration per land use area, it is the urban green class that shows the highest values among these three land uses (1.24 t ha^{-1} urban green, 0.96 t ha^{-1} natural green and 0.35 t ha^{-1} high-density residential). Surprisingly, the highest ratio among all land use classes is in the low-density residential stratum (1.33 t ha^{-1}).

Table 2.2. Carbon storage and annual carbon sequestration by land use class (Urban forests of the municipality of Barcelona, year 2008).

Land use class	Biophysical values						Monetary values	
	Carbon storage		Gross carbon sequestration		Net carbon sequestration		Net carbon sequestration	
	t	SE	t $year^{-1}$	SE	t $year^{-1}$	SE	USD $year^{-1}$	SE
Urban green	26,876	4083	1088	109	1002	100	78,688	7839
Natural green	42,108	4115	2446	207	2099	181	164,804	14,224
Low-density residential	9764	2663	613	169	565	155	44,326	12,173
High-density residential	21,014	2940	1398	157	1282	149	100,630	11,660
Transportation	3876	1213	207	56	196	54	15,366	4250
Institutional	3452	2200	76	43	-64	109	-4995	8518
Commercial/Industrial	328	153	32	15	31	14	2409	1086
Intensively used areas	6020	1693	328	65	311	62	24,396	4844
Total	113,437	19,059	6187	819	5422 5187*	823	425,625 407,177*	64,595

Notes: *Net carbon sequestration values taking into account GHG emissions of green space maintenance. SE: Standard Error.

2.3.3. Air pollution due to biogenic emissions

The total biophysical value of BVOC emissions is estimated at 183.98 t $year^{-1}$ (**Table 2.3**). Similarly to the case of carbon sequestration values, results for biogenic emissions show a major contribution of urban green, natural green and high-density residential land use strata relative to the overall biophysical value for this ecosystem disservice (17.05%; 47.46%; and 15.32% respectively). Urban green, natural green and low-density residential show to be the strata with the highest relative contribution to BVOC emissions in the city (39 kg ha^{-1}; 40 kg ha^{-1}; and 35 kg ha^{-1} respectively) considering the ratio BVOC emissions per land use area. Besides, isoprene is clearly the main BVOC emitted (51.8% of total emissions) in all land use classes (except for institutional), followed by other BVOCs (28.6%) and monoterpenes (19.6%).

Table 2.3. Annual BVOC emissions by land use class (Urban forests of the municipality of Barcelona, year 2008).

Land use class	Isoprene emissions (t year^{-1})	Mono-terpenes emissions (t year^{-1})	Other BVOCs emissions (t year^{-1})	Total BVOC emissions (t year^{-1})
Urban green	16.78	4.94	9.65	31.36
Natural green	38.79	23.65	24.87	87.31
Low-density residential	8.81	1.93	4.06	14.81
High-density residential	17.09	3.20	7.89	28.18
Transportation	4.19	0.57	1.24	6.01
Institutional	0.91	1.18	2.69	4.78
Commercial/ Industrial	1.13	0.01	0.16	1.29
Intensively used areas	7.66	0.58	2.00	10.24
Total	95.36	36.07	52.56	183.98

2.3.4. Ecosystem services contribution to air quality and climate change mitigation

From total biophysical accounts for removal of PM_{10}, NO_2 and CO_2eq, we estimated the relative contribution of urban forests ES to air quality and climate change mitigation based on air pollution and GHG emissions levels in the city (**Table 2.4**). Our results suggest that the contribution of urban forests to climate change mitigation is very low, accounting for 0.47% of the overall city-based GHG emissions. If we only account for GHG emissions derived from the sectors that are directly managed by the City Council (reference emissions to meet 'Covenant of Mayors' 23% reduction target and representing 2.10% of the total emissions) the contribution of urban forest is still modest but yet substantial, accounting for 22.55% of the emissions. Contributions of urban forests to air quality based only on city emissions differ notably depending on each air pollutant. While the overall contribution of urban forest to NO_2 removal is low relative to total emissions (0.52%), its contribution to the removal of PM_{10} amounts to a significant 22.31%. However, if we account for background pollution levels, the contribution of PM_{10} removal drops to 2.66% of total PM_{10} pollution levels.

Table 2.4. Contribution of urban forests on air quality and climate change mitigation (year 2008).

Air pollutant	**Removal biophysical value (t year^{-1})**	**Removal monetary value (USD year^{-1})**	**City-based emissions (t year^{-1})**	**Background pollution influence (%)**	**ES contribution (%)**	
					City-based emissions	**City-based emissions & background pollution**
PM_{10}	166.01	1,097,964	743.77	88.10	22.32	2.66
NO_2	54.59	540,745	10,412.94	18.70	0.52	0.43
CO_2eq	19,036	407,177	4,053,766 84,403*	N/A	0.47 22.55*	N/A

Notes: *CO_2eq emissions from services and activities directly managed by the City Council ('Covenant of Mayors' policy target baseline emissions).

2.4. Discussion

2.4.1. Urban forests potential contribution to meet air quality policy targets

Urban forests effects on air quality are still a subject of intensive research. While positive effects of air purification delivered by vegetation have been estimated at the city scale in many urban areas (e.g., Nowak et al., 2006), pollution concentration can be increased at the site scale (e.g., street canyons) depending upon vegetation configuration, pollutant emissions or meteorology, showing apparently divergent results on the effectiveness of using urban vegetation for reducing local air pollution hotspots (Pugh et al., 2012; Vos et al., 2013). Likewise, the ability of urban vegetation to remove air pollutants significantly depends on many factors, such as tree health, soil moisture availability, leaf-period, LAI, meteorology and pollution concentrations.

Our results show that the overall annual air purification rate by urban forests in Barcelona (9.3 g m^{-2} of canopy cover year^{-1}) is very similar to US cities like Columbus, Kansas City or Portland (9.2 g m^{-2} year^{-1}), although the PM_{10} removal rate (5.1 g m^{-2} year^{-1}) is significantly higher than for these cities (between 3.1 and 3.4 g m^{-2}) and closer to cities like Salt Lake City (5.2 g m^{-2}), Philadelphia (5.5 g m^{-2}) or San Diego (5.6 g m^{-2}) (Nowak et al. 2006). The higher removal rates for PM_{10}, NO_2 and O_3 compared to CO and SO_2 should be mainly attributable to the almost linear relationship between pollution removal and ambient pollution concentrations considered in the model (pollutant flux equation as $F = V_d \times C$). However, very high pollutant concentrations could severely damage vegetation or lead to stomatal closure, reducing air pollution removal ability (Robinson et al., 1998; Escobedo and Nowak, 2009). Unfortunately, these environmental thresholds are not yet factored in the i-Tree Eco model.

Our findings also show that the NO_2 removal rate by urban forests in Barcelona has a meager impact relative to actual city-based emissions (less than 1%). Therefore the potential of urban forests to contribute to the compliance of the EU limit is expected to be very low. NO_2 concentrations in the city derive largely from road transport activity (65.6% impact according to PECQ, 2011). Hence, actions focused on reduction of road traffic, technological change towards less-polluting fuels and the promotion of public transport or cycling utilities are expected to contribute more efficiently to meet policy targets. These actions can also lead to reduction in O_3 concentrations, as NO_2 is a precursor chemical to O_3 formation. PM_{10} removal rate from urban forests is notably higher than NO_2 rate, whereas city-based emissions of PM_{10} are notably lower, resulting in a substantial impact at the city scale (22.3% of total city-based emissions). However, the background pollution effect (accounting for 88.1% of the average annual PM_{10} concentration according to PECQ estimations) drastically reduces the actual impact of the urban forests service (2.7% of total PM_{10} pollution levels). Yet, we claim that there are still important reasons for which this ES should be accounted for in local policy decision-making. First, air pollution from particulate matter is a major health problem in Barcelona metropolitan area and recent research suggests that even moderate improvements in air quality are expected to report significant health benefits, together with related economic savings (Pérez et al., 2009). Second, the major role of PM_{10} background pollution in Barcelona air quality might compromise the effectiveness of municipal policies solely based on city emissions abatement. This fact also suggests that measures focused on air quality regulation should be implemented at broader spatial scales, particularly at the metropolitan level. To this end, strong coordination policies between municipal and regional authorities dealing with environmental quality and urban planning are fundamental. Third, the implementation of GI-based strategies to foster air purification (and other ES) is a realistic policy option considering the current urban context of Barcelona. I-Tree Eco results show that approximately 3.6% of the municipality area (364 ha) can be considered as available land for planting. As a complementary alternative, green roofs and walls, yet to be extensively developed in Barcelona, could be particularly appropriate in high-density neighborhoods where ground for planting is extremely scarce. Several studies have quantified the potential of green roofs for air purification in cities at the street canyon (Baik et al., 2012), neighborhood (Currie and Bass, 2008) and municipality (Yang et al., 2008) scales, besides their potential to provide many other services and benefits, such as runoff mitigation, noise reduction or urban cooling (Oberndorfer et al., 2007; Rowe, 2011). However, the technical and economic feasibility of green roofs expansion, together with possible trade-offs concerning their maintenance such as water demand, should previously be assessed in Barcelona, especially for existing buildings.

Proper management of existing green space can also contribute to air quality improvement. Yang et al. (2005) lists several factors to consider in strategies for air quality improvement based on GI, including selection of species (e.g., evergreen versus deciduous trees, dimension, growth rate, leaf characteristics or air pollution tolerance) and management practices (e.g., intensity of pruning). Previous studies in cities with high levels of air pollution, (e.g., Nowak et al., 2006; Escobedo and Nowak, 2009) suggest that meteorological conditions, mixing-layer height (the atmospheric layer which determines the volume available for the dispersion of pollutants, see Seibert et al., 2000 for a complete definition) and vegetation characteristics (e.g., proportion of evergreen leaf area, in-leaf season and leaf area index) are important factors defining urban forest effects on air quality. Further research is needed to advance our understanding of the role of morphology, function and eco-physiology of vegetation in air purification (Manning, 2008).

A further critical issue concerns the understanding of trade-offs with other ES or disservices. For example, urban parks are considered very relevant ecosystems for the provision of outdoor recreation and other cultural services in cities (Chiesura, 2004). However, highly maintained parks might remove less air pollutants and CO_2 (due to emissions from maintenance activities, Nowak et al., 2002b) than natural areas that are not intensively managed, but which can be perceived as unpleasant or even dangerous, hence providing few cultural services (Lyytimäki and Sipilä, 2009; Escobedo et al., 2011). Likewise, urban tree species with high potential for air purification can be highly invasive as well in certain cities (Escobedo et al., 2010). More generally, many specific environmental factors (e.g., soil condition, climate, water availability or longevity of the species) should be considered in urban forest management to avoid conflicts with other municipal sustainability goals (Yang et al., 2005; Escobedo et al., 2011).

The i-Tree Eco model could not provide reliable results on O_3 and CO formation rates associated to the quantified BVOC emissions. However, as mentioned above, CO levels in Barcelona (2.7 mg m^{-3} for a daily 8-hour average was the highest measure in 2011 according to ASPB air quality report, 2011) have been historically far below the EU reference value (10 mg m^{-3} daily 8-hour average). Thus it is unlikely that urban forests may compromise in any significant form the compliance of air quality relative to CO target. In contrast, ground-level O_3 levels have surpassed the EU reference value (120 μg m^{-3} daily 8-hour average) at some monitoring stations in the last decade, even if the allowed exceedances have never been reached. Although O_3 concentrations have remained steady in the last decade within the municipality of Barcelona, O_3 formation due to BVOC emissions might cause air quality problems in the long term, where BVOC emissions are expected to increase due to global warming (Peñuelas and Llusià, 2003). Nevertheless, several studies point out that the selection of low BVOC-emitting tree species can contribute positively in O_3 concentrations in urban areas because BVOC emissions are temperature dependent and trees generally lower air temperatures (Taha, 1996; Nowak et al., 2000; Paoletti, 2009). Chaparro and Terradas (2009) identified some of the tree and shrub species in Barcelona emitting less BVOC per leaf biomass. These include genera such as *Pyrus, Prunus, Ulmus* and *Celtis*.

2.4.2. Urban forests potential contribution to meet climate change mitigation policy targets

Some authors suggest that global climate regulation does not stand amongst the most relevant ES in the urban context because cities can benefit from carbon offsets performed by ecosystems located elsewhere (Bolund and Hunhammar, 1999). However, other authors argue that urban forests can play an important role in mitigating the impacts of climate change if compared to other policies at the city level (McHale et al., 2007; Escobedo et al., 2010; Zhao et al., 2010; Liu and Li, 2012).

The estimated net annual carbon sequestration per hectare of Barcelona (536 kg ha^{-1} yr^{-1}) is very similar to cities such as Baltimore (520 kg ha^{-1} yr^{-1}) or Syracuse (540 kg ha^{-1} yr^{-1}) (Nowak and Crane, 2002). It should be noted that an analysis of the overall contribution of urban GI to climate change mitigation should also account for the effects of vegetation on micro-climate regulation, which can indirectly avoid CO_2 emissions through energy saving in buildings for heating and cooling (Nowak and Crane, 2002). Hence our quantification likely underestimates the total contribution of urban forests to climate change mitigation. Analyzing the results by land-use, urban green and natural green strata are relevant for the supply of climate regulation service due to the high vegetative cover compared to the other land-use classes. High-density residential stratum also showed an important rate in net carbon sequestration, mainly attributable to its large total area (36% of the municipality) and probably, to a lesser extent, to the high presence of street trees in these neighborhoods. Finally, the high ratio of net carbon sequestration per area observed in the low-density residential stratum could be attributed to the high presence of private gardens in these areas, together with low decomposition emissions due to healthier vegetation.

In line with the results obtained in other urban studies (Pataki et al., 2009; Liu and Li, 2012), our findings show that direct net carbon sequestration in Barcelona makes a very modest contribution to climate change mitigation relative to total city-based annual GHG emissions (0.47%). Nevertheless, if we only account for the GHG emissions from services and activities directly management by the City Council (baseline emissions for the 23% reduction target from the 'Covenant of Mayors'), the contribution of urban forest is notably higher (22.55%). Similar GI-based strategies as specified for air quality improvement could also improve the contribution of urban forests to offset GHG emissions and meet the urban policy target of 23% reduction until 2020.

2.4.3. Limitations and caveats

The main advantages of the i-Tree Eco model stem from the reliance on locally measured field data and standardized peer-reviewed procedures to measure urban forest regulating ES in cities (Nowak et al., 2008a). Favored by its status as an open access model, it has been widely applied across the world (e.g., Nowak and Crane, 2002; Yang et al., 2005; Nowak et al., 2006; Currie and Bass, 2008; Escobedo and Nowak, 2009; Dobbs et al., 2011; Liu and Li, 2012).

However, i-Tree Eco has some limitations that should be taken into account when analyzing its outcomes. First, the model is especially designed for US case studies and its application in other countries is subject to some restrictions, as stated in the user's manual. For instance, although the i-Tree Eco database has over 5000 species, it did not include some tree and shrub species sampled in Barcelona, which then needed to be added to the database. Likewise, monetary valuations of air purification and climate regulation services are based on literature (see methods section above) which mainly apply to the US context and, hence, should be considered a rough estimation for Barcelona. However, these values are direct multiplier to the biophysical accounts, thus they can be easily adjusted to the case study context when data will be available. Another important limitation applying to i-Tree Eco and most dry deposition models is the level of uncertainty involved in the quantification of the air pollution removal rates due to the complexity of this process (Pataki et al., 2011). For instance, some sources of uncertainty include non-homogeneity in spatial distribution of air pollutants, particle re-suspension rates, transpiration rates or soil moisture status (Manning, 2008). Though the model outputs match well with field measured deposition velocities for urban forests, the model analyzes average effects across a city, not local variations in removal caused by local meteorological and pollution differences. However, these local fine-scale input data are often missing from urban areas and empirical data on the actual uptake of pollutants by urban vegetation are still limited (Pataki et al., 2011; Setälä et al., 2013), which makes a more accurate modeling of this ES unfeasible at the moment. For a sensitivity analysis of the i-Tree Eco deposition model see Hirabayashi et al. (2011). Estimation errors in climate regulation service values include the uncertainty from using biomass equations and conversion factors as well as measurement errors (Nowak et al., 2008a). For example, there are limited biomass equations for tropical tree species (e.g., palm trees), some of them present in Barcelona. Estimates of carbon sequestration and storage also include uncertainties from factors such as urban forests maintenance (e.g., intensity of pruning), tree decay, or restricted rooting volumes, which are not accounted for in the model's estimations (Nowak et al., 2008a; Pataki et al., 2011). BVOC emissions are estimated based on species factors and meteorological conditions (i.e., air temperature and daylight) but the uncertainty of the estimate is unknown. As mentioned in previous sections, O_3 and CO formation rates from BVOC emissions cannot be estimated with an acceptable level of reliability.

Therefore, the results presented in this paper should be considered as an approximate estimation rather than a precise quantification of the ES and disservices delivered by the urban forests of Barcelona. However, these estimates allow one to evaluate the contribution of urban forests in air quality and climate change mitigation in the city, and also to derive implications and recommendations for urban decision-making.

2.5. Conclusion

Regulating ES provided by urban forests have been widely analyzed in many cities across the world. However, the potential effectiveness of urban forests in air quality improvement and climate change mitigation is still object of debate, mainly due to the multiple factors and uncertainties involved in the actual delivery of these ES in cities, especially at the patch or site scale. Further, this potential is barely reflected in terms of its contribution to meet specific policy targets.

Our findings show that the contribution of urban forests regulating services to abate pollution is substantial in absolute terms (305.6 t of removed air pollutants year^{-1} and 19,036 t CO_2eq year^{-1}), yet modest when compared to overall city levels of air pollution and GHG emissions (2.66% for PM_{10}, 0.43% for NO_2 and 0.47% for CO_2eq). Our research further shows that the effectiveness of GI-based strategies to meet environmental policy targets can vary greatly across pollutants. For example, our results suggest that NO_2 removal potential is unlikely to contribute in any substantial way to the compliance of current EU reference values. Therefore, for combating air pollution of NO_2, synergies between GI strategies and NO_2 emission curbing strategies (e.g., targeting road traffic) need to be searched and implemented in order to effectively deal with air quality regulations. On the other hand, PM_{10} removal potential should not be neglected in urban policy-making. Its contribution to the compliance with the current EU reference value can be substantial and potentially more effective than other local policies based on emissions abatement due to the importance of background pollution in Barcelona's PM_{10} levels.

Net carbon sequestration by urban forests has a very low influence if compared to total annual GHG city emissions, but our results suggest that it can contribute considerably to meet the 23% GHG emissions reduction policy target until 2020, which only applies for emissions derived from services and activities directly managed by the City Council (2.10% of total emissions).

We determine that the implementation of GI-based strategies at the municipal level (as is aimed by the *Barcelona Green Infrastructure and Biodiversity Plan 2020*) would have a limited effect on local air quality levels and GHG emissions offsets, yet they would play a non-negligible complementary role to other policies intended to meet air quality (especially for PM_{10} levels) and climate change mitigation policy targets in Barcelona, fostering as well the provision of other important urban ES (e.g., urban temperature regulation, stormwater runoff mitigation and recreational opportunities) at no additional monetary costs. We conclude that, in order to be effective, green-infrastructure-based strategies to abate pollution in cities should be implemented at broader spatial scales (i.e., metropolitan area). However, it is critical that policy-makers consider an integrated approach in GI management, where possible trade-offs with other ecosystems services, disservices and urban sustainability goals are fully acknowledged.

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Jardins de la Torre de les Aigües, Barcelona, Spain (illustration by Amèlia Estrada Soto, published with kind permission of the author and the series of seminars on city, environment, health and drawing "Ciutat Verda" – www.ciutatverda.info)

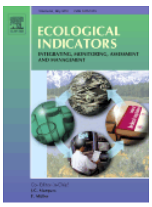
CHAPTER III

Mismatches between ecosystem services supply and demand in urban areas: A quantitative assessment in five European cities

Abstract

Assessing mismatches between ecosystem service (ES) supply and demand can provide relevant insights for enhancing human well-being in urban areas. This paper provides a novel methodological approach to assess regulating ES mismatches on the basis of environmental quality standards and policy goals. Environmental quality standards (EQS) indicate the relationship between environmental quality and human well-being. Thus, they can be used as a common minimum threshold value to determine whether the difference between ES supply and demand is problematic for human well-being. The methodological approach includes three main steps: (1) selection of EQS, (2) definition and quantification of ES supply and demand indicators, and (3) identification and assessment of ES mismatches on the basis of EQS considering certain additional criteria. While ES supply indicators estimate the flow of an ES actually used or delivered, ES demand indicators express the amount of regulation needed in relation to the standard. The approach is applied to a case study consisting of five European cities: Barcelona, Berlin, Stockholm, Rotterdam and Salzburg, considering three regulating ES which are relevant in urban areas: air purification, global climate regulation and urban temperature regulation. The results show that levels of ES supply and demand are highly heterogeneous across the five studied cities and across the EQS considered. The assessment shows that ES supply contributes very moderately in relation to the compliance with the EQS in most part of the identified mismatches. Therefore, this research suggests that regulating ES supplied by urban green infrastructure are expected to play only a minor or complementary role to other urban policies intended to abate air pollution and greenhouse gas emissions at the city scale. The approach has revealed to be appropriate for the regulating ES air purification and global climate regulation, for which well-established standards or targets are available at the city level. Yet, its applicability to the ES urban temperature regulation has proved more problematic due to scale and user dependent constraints.

Keywords: Air purification; Assessment; Global climate regulation; Green infrastructure; Human well-being; Urban temperature regulation.



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3.1. Introduction

Green infrastructure (GI) has been defined as a “network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services (ES). It incorporates green spaces (or blue if aquatic ecosystems are concerned) and other physical features in terrestrial (including coastal) and marine areas” (EC, 2013:3). In urban areas, GI elements may include parks, urban forests, allotments, street trees, green roofs, etc. (Landscape Institute, 2009). Relevant ES delivered by GI in cities include, for instance, air purification, urban temperature regulation, runoff mitigation, noise reduction and recreation (Bolund and Hunhammar, 1999, Gómez-Baggethun and Barton, 2013 and Gómez-Baggethun et al., 2013).

An increasing body of literature highlights the contribution of GI and ES in enhancing environmental quality (e.g., air quality) in cities, hence fostering a better quality of life and well-being for the urban population (e.g., Nowak, 2006, Tzoulas et al., 2007, Escobedo et al., 2011 and Pataki et al., 2011). Some studies even argue that urban policies based on the planning and management of GI can be comparable in terms of effectiveness or efficacy to other policies based on technological measures (e.g., Escobedo et al., 2008 and Escobedo et al., 2010). Yet, the assessment of the current (and potential) contribution of urban GI through ES supply as a means to meeting desired or required environmental quality conditions and goals at the city scale remains largely unexplored.

The main objective of the paper is hence the exploration of the possible contribution of ES supply to meet environmental quality standards and policy goals (hereafter referred as EQS) in urban areas. The underlying assumption derived from this objective is that EQS are to be met exclusively through ES supply. Conceptually, this hypothesis can be framed as the assessment of mismatches between ES supply and demand. This research argues that ES demand, defined here as the amount of service required or desired by society (Villamagna et al., 2013), can be expressed in relation to EQS because these provide a threshold value to determine whether the difference between ES supply and demand is problematic for human well-being. The assessment examines ES mismatches of three regulating ES which are relevant in urban areas (Gómez-Baggethun and Barton, 2013): air purification, urban temperature regulation and global climate regulation (through carbon sequestration). The methodological approach includes three main steps: (1) selection of EQS, (2) definition and quantification of ES supply and demand indicators, and (3) identification and assessment of ES mismatches on the basis of EQS considering certain additional criteria. While ES supply indicators estimate the flow or amount of an ES actually delivered (e.g., air pollutants removed by urban vegetation), ES demand indicators estimate the amount of inputs needing regulation (e.g., air pollutant concentrations) in relation to the corresponding EQS (e.g., air quality standards). The approach is applied to a case study consisting of five European cities: Barcelona, Berlin, Stockholm, Rotterdam and Salzburg. Based on the obtained results, the actual and potential contribution of urban GI to address mismatches between ES supply and demand at the city scale is discussed, as well as the advantages and limitations of using EQS to assess these mismatches.

3.2. Materials and methods

3.2.1. Conceptual framework

Recently developed conceptual frameworks in the ES literature call for a distinction between ES *capacity*, *flow* and *demand* as the main components of the ES delivery process (Villamagna et al., 2013, Burkhard et al., 2014, Schröter et al., 2012, Schröter et al., 2014 and Guerra et al., 2014). Capacity is defined as the ES potential (i.e., hypothetical maximum yield) and flow as the actual supply or use of ES experienced by people. ES demand, however, has been approached differently depending on the authors. Burkhard et al. (2014:5) define demand for ES as the "services currently consumed or used in a particular area over a given time period, not considering where ES actually are provided". Alternatively, ES demand has been described as "the amount of a service required or desired by society" (Villamagna et al., 2013:115) or "the expression of the individual agents' preferences for specific attributes of the service" (Schröter et al., 2014:541). In this paper, ES supply is conceptualized as ES flows (Hein et al., 2006) and ES demand as the required level of ES delivery by society (Villamagna et al., 2013). ES mismatches occur when the demand for ES is not totally met by the supply within a defined spatial and time scale. Thus, ES mismatches express the existence of an unsatisfied or remaining demand (Geijzendorffer et al., 2015).

According to the framework developed by Villamagna et al. (2013), the supply of regulating ES contribute to the maintenance of environmental quality within socially acceptable ranges only until a certain level of ecological pressure (e.g., air pollution). Beyond this level, ES supply cannot sustain a good environmental quality and ES demand should be considered as not totally met. Under this approach, estimating regulating ES demand requires hence information about two main elements: (1) desired conditions (i.e., good environmental quality); and (2) inputs needing regulation (i.e., ecological pressures). In line with Paetzold et al. (2010), this paper considers that EQS can be used as a threshold of desired conditions in relation to the demand for regulating ES. In general terms, EQS rely on scientific evidence and/or expert knowledge concerning the relationship between environmental quality and human well-being with the underlying aim to secure or enhance the latter (e.g., EEA, 2013). Thus, the methodological approach considered here assumes that EQS can provide a common minimum threshold value to assess regulating ES mismatches across different contexts (in this case study, different European cities). For example, World Health Organization (WHO) air quality guidelines (WHO, 2005) can be used to provide a minimum threshold to assess the mismatch between supply and demand of the ES air purification. A city where air pollution levels exceed WHO reference values reflects a mismatch in which air purification demand exceeds the current local supply. Yet, this situation does not necessarily imply that the EQS is to be achieved solely by ES supply.

3.2.2. Selection of environmental quality standards

Based on a non-exhaustive examination of European-context regulatory frameworks, relevant EQS were identified for the three ES assessed in this study (**Table 3.1**). EQS for ES air purification were derived from the European Union (EU) air quality Directive (EU, 2008) and WHO air quality guidelines (WHO, 2005). Reference values for ground-level concentrations of air pollutants are generally more stringent in the WHO standards, but only the EU standards are legally binding for the case study cities, hence the inclusion of both standards in the assessment was considered pertinent. The focus was limited to the following air pollutants: (1) particulate matter with a diameter of 10 μm or less (PM_{10}); (2) nitrogen dioxide (NO_2); and (3) tropospheric ozone (O_3), considered three of the most problematic air pollutants in terms of exposure to concentrations above the EU and WHO reference levels in Europe for its urban population (EEA, 2013).

Table 3.1. EQS selected to assess mismatches between ES supply and demand.

ES	EQS
Air purification	EU Air Quality Directive (EU, 2008) and WHO air quality guidelines (WHO, 2005) reference values: Pollutant / EU / WHO PM_{10}: 40 μg m^{-3} (year) / 20 μg m^{-3} (year) NO_2: 40 μg m^{-3} (year) / 40 μg m^{-3} (year) O_3: 120 μg m^{-3} (8-h) / 100 μg m^{-3} (8-h)
Global climate regulation	Covenant of Mayors' GHG emission reduction targets for each case study city are: • Barcelona: 23% by 2020 (baseline year 2008) • Berlin: 40% by 2020 (baseline year 1990) • Stockholm: 45% by 2020 (baseline year 1990) • Rotterdam: 50% by 2025 (baseline year 1990) • Salzburg: No explicit target found (assuming 20% by 2020, baseline year 1990)
Urban temperature regulation	Heatwave thresholds: consecutive occurrence of hot days (T-max > 35ºC) and tropical nights (T-min > 20 ºC) (Fischer and Schär, 2010).

Notes: Air quality policy targets correspond to the EU and WHO values set for the protection of human health (in brackets the averaging period applicable for each limit). EU's reference value for O_3 is subject to 25 days of allowed exceedances per year averaged over three years. See EEA (2013) for more details. GHG emission reduction targets for each case study city are based on local Sustainable Energy Action Plans (see www.covenantofmayors.eu and **Table 3.3**).

The ES global climate regulation is generally assumed to be demanded at global scale (Burkhard et al., 2012), yet city specific GHG emission reduction and offset targets can be considered as a desired condition at lower scales. Following the EU 20-20-20 targets (EC, 2008), many municipal authorities have signed up to the 'Covenant of Mayors' initiative[15], voluntarily committing themselves to reduce their GHG emissions by at least 20% until 2020 (see **Table 3.1** for specific reduction targets of the case study cities).

No explicit EQS were found in relation to urban temperature regulation at the European regulatory level, probably because human health vulnerability to temperature extremes depends on a complex interaction between different factors such as age, health status, socio-economic circumstances (e.g., housing) and regional adaptation (Kovats and Hajat, 2008 and Fischer and Schär, 2010). However, general critical temperature thresholds for health impacts in Europe have been estimated based on the spatial and

[15] See www.covenantofmayors.eu

temporal variance in excess mortality during recent heatwaves[16] episodes (Fischer and Schär, 2010). According to this research, the consecutive occurrence of days with maximum temperature above 35 °C ('hot days') and nights with minimum temperature above 20 °C ('tropical nights') has been found to explain the correlation with excess mortality. These values match well with specific temperature thresholds officially allocated to cities like Barcelona (Tobias et al., 2012), but are likely overestimated for Northern cities like Stockholm (Roklöv and Forsberg, 2008) due to regional adaptation factors. In any case, the impacts of heatwaves on human health are particularly strong in cities, both in Northern and Southern latitudes, due to the exacerbating effect of the urban heat island (UHI) (EEA, 2012).

3.2.3. Defining indicators of ecosystem service supply

ES supply was measured directly as the amount of a service delivered or experienced by people (Van Oudenhoven et al., 2012 and Villamagna et al., 2013). The indicators for ES supply were selected based on methods and data availability (see **Table 3.2**). For this analysis only terrestrial ecosystems were considered, omitting blue infrastructure elements (sea, lakes, ponds, rivers, etc.) which can also be important sources of ES supply in the urban context (Bolund and Hunhammar, 1999), especially in case study cities such as Stockholm, Rotterdam and Barcelona. The use of tools specifically designed for quantifying ES delivered by terrestrial vegetation (e.g., i-Tree Eco model) prevented a more complete assessment of urban ecosystems (i.e., including blue infrastructure).

[16] Fischer and Schär (2010) define a heatwave "to be a spell of at least six consecutive days with maximum temperatures exceeding the local 90th percentile of the control period (1961–1990)".

Table 3.2. ES supply indicators and associated quantification methods and references.

ES	Indicators	Quantification method	Sources / References
Air purification	PM_{10} removal (kg ha^{-1} year^{-1}) NO_2 removal (kg ha^{-1} year^{-1}) O_3 removal (kg ha^{-1} year^{-1})	i-Tree Eco dry deposition model based on tree canopy cover, air pollution and meteorological data	i-Tree Canopy (www.itreetools.org) AirBase v.7 (EEA, 2013b). Year 2011 Nowak et al. (2006); Baró et al. (2014); Aevermann et al. (pers. commun., 2013)
Global climate regulation	CO_2 sequestration (t ha^{-1} year^{-1}) Carbon storage (t ha^{-1})	Estimates from i-Tree assessments based on tree canopy cover and length of growing season	i-Tree Canopy (www.itreetools.org) Nowak et al. (2013); Baró et al. (2014)
Urban temperature regulation	Tree shade area (%)	Cooling effect of trees based on empirical data and tree canopy cover area estimates	i-Tree Canopy (www.itreetools.org) Bowler et al. (2010); Breuste et al. (2013)

The supply of the ES air purification was quantified using estimated air pollution removal of PM_{10}, NO_2, and O_3 by urban green space. Uptake rates were quantified using the dry deposition model of i-Tree Eco tool (Nowak et al., 2006, Nowak et al., 2008 and Hirabayashi et al., 2012). Data required for each city included hourly air pollution concentration, percentage of tree canopy cover (both deciduous and evergreen) and meteorological data. For Barcelona and Berlin air pollution removal rates were taken from Baró et al. (2014) corresponding to year 2008, and Aevermann (pers. commun., 2013) for year 2011, respectively. Air pollution concentration data from Salzburg, Stockholm and Rotterdam monitoring stations were obtained from the AirBase database v.7 (EEA, 2013b) for the year 2011. Meteorological data were retrieved from the US National Climatic Data Centre for the same year. Percentages of evergreen and deciduous tree canopy cover for these three cities were estimated using i-Tree Canopy tool[17] which allows photo-interpretation of urban land covers from Google Maps aerial imagery using a random sampling location process. A sample of 500 survey points were photo-interpreted for each city based on a categorization of three cover classes: (1) deciduous tree; (2) evergreen tree and (3) non-tree cover. This method likely

[17] See www.itreetools.org/canopy/index.php

underestimates the amount of air purification supplied since it accounts for tree canopy but not for shrubs or herbaceous vegetation which can also supply this ES (Nowak et al., 2006).

Carbon storage and annual CO_2 sequestration rates performed by urban GI were used as indicators to measure the supply of the ES global climate regulation (Nowak and Crane, 2002, Strohbach and Haase, 2012, Nowak et al., 2013 and Schröter et al., 2014). Barcelona's estimates were based on the i-Tree Eco assessment performed in 2008 using field measurements of urban forest structure, allometric equations to predict above-ground biomass and adjusted urban tree growth and decomposition rates (Baró et al., 2014). Due to limited resources for fieldwork data collection in the other case study cities, carbon storage and sequestration indicators were estimated based on the assessment carried out by Nowak et al. (2013) using urban field data from 28 cities and 6 states in United States (US), where carbon storage per square meter of tree cover averaged 7.69 kg C m^{-2} (SE = 1.36), gross carbon sequestration rate averaged 0.277 kg C m^{-2} $year^{-1}$ (SE = 0.045), and net carbon sequestration rate averaged 0.205 kg C m^{-2} $year^{-1}$ (SE = 0.041). Percentage of tree canopy cover was estimated using the i-Tree Canopy tool as described above (for Berlin, 1000 points were photo-interpreted due to its larger area). Although these rates can vary depending on variables such as tree diameter distribution or species composition in each city, the indicator estimates should be accurate as they are based on local tree cover values (Nowak et al., 2013). Further, empirical studies carried out in European cities obtained similar values (e.g., Strohbach and Haase, 2012 estimated an average carbon storage rate of 6.82 ± 1.42 kg C m^{-2} of canopy cover in Leipzig, Germany). Because tree growth (and hence CO_2 sequestration) vary depending on the local environmental conditions, sequestration rates were refined using the length of the growing season as a proxy, following the formula (Nowak, pers. commun., 2013):

$$C' = \frac{C-GS}{174} \qquad \text{(Formula 3.1)}$$

where C' = average (gross or net) carbon sequestration rate (kg C m^{-2} tree cover year); C = US average (gross or net) carbon sequestration rate (kg C m^{-2} tree cover year) (Nowak et al., 2013); GS = length of the growing season (days).

Average length of the growing season in each case study city was based on phenological data for the period 1969–1998 (Chmielewski and Rötzer, 2001). Reported trends in plant phenology in Europe and USA indicate a similar lengthening of the growing season in the last decades associated to global warming (Linderholm, 2006), thus used lengths should be considered a first-order estimate. Carbon sequestration rates were converted to CO_2 after applying the conversion factor 1 g C = 3.67 g CO_2.

The supply of the ES urban temperature regulation by green space can provide important benefits to city inhabitants by mitigating heat stress (Stone et al., 2010) and reducing UHI effects and increased temperatures resulting from climate change (Gill et al., 2007). Vegetation delivers this service mainly through the evapotranspiration process and the shading effect (basically from trees). Bowler et al. (2010) systematically reviewed the empirical evidence of this ES showing that, on average, the temperature within an urban park would be around 1 °C cooler than a non-green site in the day. Other urban GI elements such as urban forests and green roofs also show evidence of lower air temperatures compared to treeless areas and roofs without vegetation respectively (Oberndorfer et al., 2007 and Breuste et al., 2013). Tree shade area was used as a proxy indicator to quantify the supply of this service. It was estimated as tree canopy cover area using i-Tree canopy tool as described above, assuming that the cooling effect is provided mainly below tree canopy (Bowler et al., 2010).

3.2.4. Defining indicators of ecosystem service demand

Due to the different approaches to ES demand, a variety of indicators can be defined to measure it. One way is to consider population density in combination with average or desired consumption rates (Burkhard et al., 2012 and Kroll et al., 2012). ES demand can also be measured by the socio-cultural preferences directly expressed by people in interviews and questionnaire surveys (Martín-López et al., 2014) or through monetary valuation (De Groot et al., 2012). Following the conceptual framework described above, in this paper ES demand indicators express the amount or concentration of inputs (i.e., ecological pressures) needing regulation with regard to the corresponding EQS (i.e., the desired environmental conditions which secure human well-being) (Villamagna et al., 2013 and Burkhard et al., 2014). **Table 3.3** shows the selected indicators for ES demand.

Table 3.3. ES demand indicators and associated quantification methods and references.

ES	Indicators	Quantification method	Sources / References
Air purification	PM_{10} annual mean concentration (μg m^{-3}) NO_2 annual mean concentration (μg m^{-3}) 26th highest O_3 value based on daily max 8-hour averages (μg m^{-3})	Statistical data review	AirBase v.7 (EEA, 2013b) - Year 2011
Global climate regulation	Annual CO_2eq emissions per ha (t ha^{-1} $year^{-1}$) Annual CO_2eq emissions per capita (t $capita^{-1}$ $year^{-1}$)	Literature review on municipal GHG emissions and census data	Barcelona: PECQ. 2011. The energy, climate change and air quality plan of Barcelona 2011-2020. Base year 2008. Berlin: Environmental Agency of the Senate of Berlin. Base year 1990. Stockholm: Stockholm action plan for climate and energy 2010–2020. Base year 1990. Rotterdam: CDP Cities 2012 Global Report. Base year 1990. Salzburg: Energiebericht 2010 Smart City Salzburg. Base year 2010.
Urban temperature regulation	Heat wave risk (# days)	Combined tropical nights (>20ºC) and hot days (>35ºC) expected 2071-2100	Fischer and Schär (2010) and EEA (2012)

Indicators for the ES air purification were estimated on the basis of air pollution levels in each city in relation to the desired level expressed by air quality standards (Burkhard et al., 2014). These indicators express the remaining air pollution as they already include the impact of ES supply (Guerra et al., 2014 call it as "ES mitigated impact"). Annual mean concentrations for PM_{10} and NO_2 from the available traffic monitoring stations (which express the highest demand) in each case study city were extracted from the AirBase database v.7 (EEA, 2013b) using values corresponding to year 2011. O_3 levels were expressed as the 26th highest value in each city based on daily maximum 8-h averages since the current European air quality threshold includes 25 days of allowed exceedances (EEA, 2013).

Demand indicators for the ES global climate regulation were estimated on the basis of annual GHG emissions as expressed in carbon dioxide equivalent (CO_2eq) per hectare and per capita (Burkhard et al., 2014). Total emissions for each case study city were obtained from local Sustainable Energy Action Plans (SEAPs) and other municipal policy reports (see **Table 3.3** for references) corresponding to the GHG reduction target baseline year (1990 for Berlin, Stockholm and Rotterdam, 2008 for Barcelona and 2010 for Salzburg because 1990 data was not available).

Finally, demand for the ES urban temperature regulation was estimated using heatwave risk as indicator. Following Fischer and Schär (2010), heatwave risk was quantified as the number of combined tropical nights (>20 °C) and hot days (>35 °C) projected for the period 2071–2100 in Europe. This scenario was developed at a European scale and it does not take into account the UHI effect that exacerbates heatwave risk in cities (EEA, 2012). Thus, the consideration of this future scenario can roughly express a more realistic current situation of heatwave risk in the case study cities, where the UHI can reach a maximum intensity of 8 °C (e.g., Moreno-Garcia, 1994 for Barcelona).

3.2.5. Criteria for identifying and assessing ecosystem service mismatches

The assessment of matches and mismatches between ES supply and demand usually requires demand to be assessed in the same units as supply in order to obtain a budget or ratio indicating ES undersupply, neutral balance or oversupply (Paetzold et al., 2010, Burkhard et al., 2012 and Kroll et al., 2012). However, because of the EQS-based approach considered in this paper, the assessment of mismatches was determined by the following criteria: (1) in the case of non-compliance with the limit or target values stipulated by the EQS, the demand for the corresponding ES was considered to be not totally met by the current supply at the city scale, thus an ES mismatch was identified. On the contrary, in the case of standard compliance, the demand was considered to be currently met by the supply and no ES mismatch was expected at the city level; (2) due to the ES-based assumption considered here, it was also important to assess the contribution or impact of ES supply in relation to the compliance with the EQS, especially in the case of exceedance of limit or target values. In this way, informed decisions can be taken on the feasibility of increasing ES supply (e.g., increase tree canopy cover in the city) as an effective measure to address a given mismatch.

In the case of air purification, an ES mismatch between supply and demand was identified if, despite air purification delivered by urban trees, air pollution levels exceeded EU and/or WHO air quality reference values. The ES contribution to the compliance with the standards was estimated as the average air quality improvement due to air purification by urban trees from i-Tree Eco dry deposition model results (Nowak et al., 2006 and Hirabayashi et al., 2012). The estimation of this variable involved considering the mixing layer height[18] in each case city area, which was derived from radiosonde data of the closest station available in the NOAA/ESRL Radiosonde Database[19]. A "substantial mismatch" was identified if the ES contribution (air quality improvement) was lower than 10% in relation to the EQS exceedance. A "moderate mismatch" was identified if this contribution was higher than 10%. This mismatch analysis could not be done for EQS exceedances of O_3 because the standards are based on daily max 8-h averages whereas air quality improvements are based on annual averages. The criterion to assess an ES mismatch for the ES global climate regulation was defined as the deficit of urban ecological carbon sinks to contribute substantially to CO_2eq reduction targets in each city. An ES contribution lower than 10% in relation to the reduction target was considered as a "substantial mismatch". A "moderate mismatch" was identified when the contribution was higher than 10%, but lower than 100%. Finally, the uncertainty and complexity related to the impact of the ES urban temperature regulation supply at the wider city scale (Bowler et al., 2010) implies that the heatwave risk cannot be consistently compared to the cooling effect provided by GI on the basis of the heatwave thresholds at the city scale. Therefore, the mismatch assessment of this ES was excluded from the analysis.

3.2.6. Case study cities

The paper builds on five case study cities distributed along a north–south and east–west gradient across Europe: Barcelona, Berlin, Stockholm, Rotterdam, and Salzburg (**Fig. 3.1**). The cities vary in their population size, urban form, climate patterns and socio-economic characteristics (**Fig. 3.1**, **Table 3.4**), making them representative for a broad range of medium-to-large size European cities. Most of these cities have ambitious

[18] The mixing height can be defined as "the height of the layer adjacent to the ground over which pollutants or any constituents emitted within this layer or entrained into it become vertically dispersed by convection or mechanical turbulence within a time scale of about an hour" (Seibert et al., 2000).
[19] See http://esrl.noaa.gov/raobs/.

strategic plans to enhance GI and ES in the coming years (e.g., Barcelona Green Infrastructure and Biodiversity Plan 2020, Barcelona City Council, 2013). Furthermore, these are all case study cities of the URBES project (Urban Biodiversity and Ecosystem Services[20]).

The spatial scope of this analysis is the municipal or core city area (Urban Audit, 2009). An intrinsic limitation must be acknowledged when using administrative boundaries in urban ES assessments because cities are, to a large extent, influenced by ES provided beyond these boundaries, namely from the larger suburbanized and rural hinterland (Larondelle and Haase, 2013). However, the focus on the administrative areas responded to the following motivations: (1) the analysis includes indicators for which required datasets were only available at the administrative level; (2) urban policies related to green space are usually limited to city's municipal boundaries (e.g., Barcelona's Green Infrastructure and Biodiversity Plan 2020, Barcelona City Council, 2013), hence recommendations for future policies are more likely to be applicable when addressed at this spatial scale; (3) the administrative area of the case study cities corresponds well with the dense urban core of their metropolitan areas (Larondelle and Haase, 2013 and Larondelle et al., 2014).

Barcelona is the capital city of the region of Catalonia and Spain's second-largest city in terms of population. The city is characterized by a compact urban form together with a very high population density (see **Table 3.4**). Approximately a quarter of the municipal area consists of green space (parks, gardens, urban forests, etc.), most of which corresponds to the urban park of Montjuïc and the periurban forest area of Collserola. Barcelona has also a relatively high proportion of street trees compared to other European cities (Pauleit et al., 2002). Berlin is the capital city and the most populous city of Germany, located at the core center of the Berlin-Brandenburg metropolitan region. Green space amounts to one third of the city's area, including large urban parks such as Tiergarten located at the city center and larger areas of forest and water ecosystems located at the outskirts of the municipal area. The former Tempelhof airport has recently been converted into an urban park, providing new opportunities to benefit from green space to a large number of city inhabitants (Kabisch and Haase, 2014). Stockholm, awarded the first European Green Capital in 2010 by the European

[20] See www.urbesproject.org.

Commission[21], is the capital of Sweden and the country's most populated municipality. The amount of green and blue space is very relevant in Stockholm (on third of the city's areas is covered by parks, forest and other green assets and 12% by water bodies). Rotterdam is the second largest city of the Netherlands and has the largest seaport of Europe in terms of cargo volume and traffic (CRRSC, 2009). Blue space covers almost a quarter of the total city's area, mainly corresponding to the lowest course of the river Nieuwe Maas. The city is considered one of the greenest large cities of the Netherlands, having a total of 117 public parks and 747,000 trees (Frantzeskaki and Tilie, 2014). Salzburg is the fourth largest city of Austria and the capital city of the federal state of Salzburg. Almost a half of the municipal area is covered by green space, including a relevant share of forest and agricultural land which is legally protected by the City Council (Voigt et al., 2014).

[21] See http://ec.europa.eu/environment/europeangreencapital/.

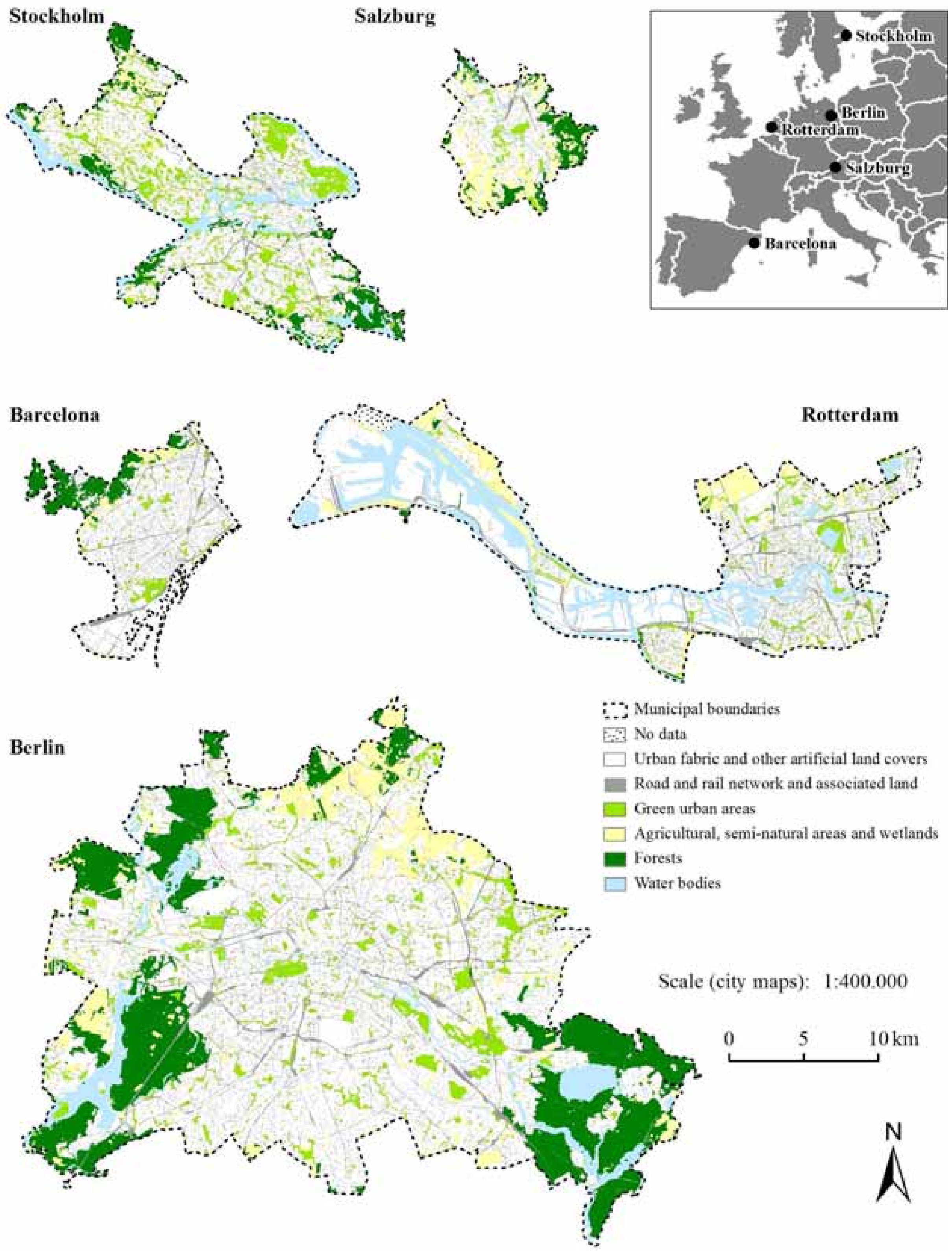

Fig. 3.1. Location of case study cities and distribution of green space covers. Source: own elaboration based on Natural Earth data (www.naturalearthdata.com) and Urban Atlas (EEA, 2010). Administrative boundaries: Catalan Cartographic Institute (www.icc.cat); Senate Department for Urban Development and the Environment (www.stadtentwicklung.berlin.de/ geoinformation/); Stockholm City Council (www.stockholm.se); Centraal Bureau voor de Statistiek – Statistics Netherlands (www.cbs.nl); Salzburg Geoinformation System (SAGIS) (www.salzburg.gv.at/sagis/).

Table 3.4. Main characteristics of the case study cities.

	Barcelona	Berlin	Stockholm	Rotterdam	Salzburg	Sources / References
Location in Europe	South-West	Central	North	North-West	Central	-
Physical geography	Coastal / River delta	Inland plains/ River	Coastal/ Lake outlet	Coastal/ River delta	Inland/ Foothill of the Alps	-
Population (#)	1,615,908	3,431,675	810,120	582,951	147,169	Urban audit 2009 (reference year 2008)
Population projection in 2050* (#)	1,672,112	3,460,046	1,648,000	621,780	161,589	Own trend calculations based on National Census, except for Barcelona (Catalan Statistical Institute – IDESCAT).
Total area (km^2)	101.6	891.1	215.8	277.4	65.7	Municipal boundaries (various sources)
Population density (inhab. km^{-2})	15,905	3,851	3,754	2,101	2,240	Urban audit 2009 (reference year 2008)
Gross Domestic Product (PPS $inhab.^{-1}$)	30,800	24,400	41,000	36,500	38,100	Urban audit 2009 (for NUTS3 region, reference years 2007-2010)
Green urban area (m^2 $inhab.^{-1}$)	3.00	16.91	43.88	23.12	25.86	Urban Atlas (EEA, 2010); Urban audit 2009
Development of green space 1990 – 2006 (ha)	-0.02	1,083	106	16	3	Kabisch and Haase (2013)
Number of private cars registered (# 100 $inhab.^{-1}$)	38.13	28.56	36.98	34.13	N/A	Urban audit 2009 (reference year 2008)
Average temperature of warmest month (ºC)	25.5	19.5	18.5	N/A	18.6	Urban audit 2009 (reference year 2008)

Note: *Except for Barcelona (highest population projection for 2021)

3.3. Results

3.3.1. Ecosystem service supply and demand across the case study cities

The quantification results of ES supply and demand indicators are partly shown in **Fig. 3.2.** The complete set of indicator results is presented in **Table A.1** (supply) and **Table A.2** (demand) of **Appendix A**.

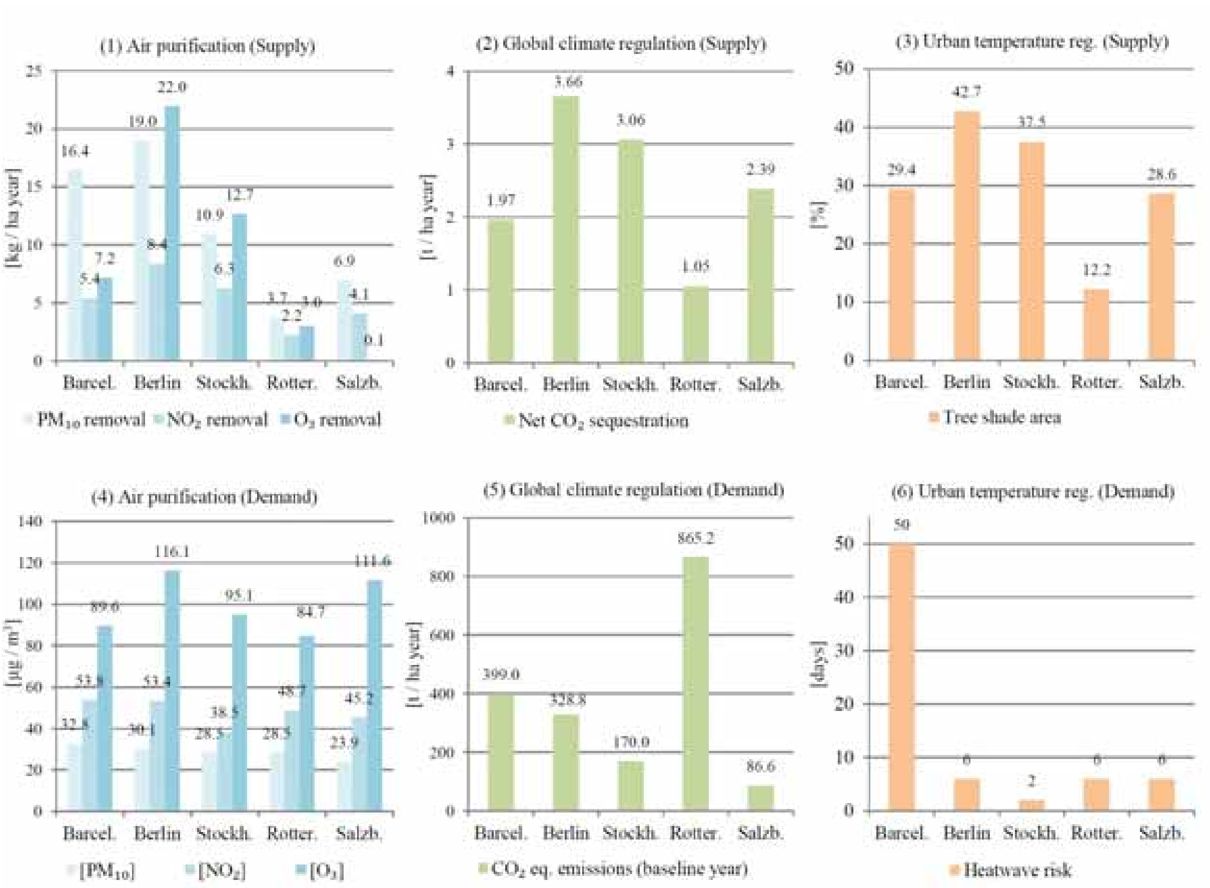

Fig. 3.2. Quantification results of ES supply and demand indicators for the five case study cities. Notes: Air purification demand values are in annual mean concentration for PM_{10} and NO_2 and in daily max 8-hour averages for O_3 (26^{th} highest value). Urban temperature regulation demand values are the maximum number of days of heatwave risk, except for the case of Barcelona which is the minimum (Fischer and Schär, 2010). Supply and demand values are not directly comparable except for global climate regulation.

Supply of the ES air purification showed the highest values in Berlin, almost doubling the average removal rate for the five case study cities when the three air pollutants are considered. The results for Barcelona and Stockholm displayed comparatively intermediate values, with a total supply of nearly 30 kg removed air pollutants per hectare annually in both cases. Rotterdam and Salzburg were characterized by the lowest values of air purification supply whatever the air pollutant considered. For example, Salzburg's O_3 removal rate was negligible compared to Berlin's (0.12 to almost 22 kg ha^{-1} $year^{-1}$) even though both cities have a relevant share of green space. PM_{10} was the air pollutant comparatively most removed in all the cities, except in Berlin where O_3 removal was slightly higher. Inversely, NO_2 was the pollutant with lowest removal rates in all case study cities, except in Salzburg where the lowest value was found for O_3. Demand indicators for the ES air purification showed different patterns compared to supply across the different case study cities. For example, NO_2 annual mean concentration levels were higher than PM_{10} values in all cities whereas supply indicators showed the opposite condition. It must be noted that PM_{10} and NO_2 have the same EU limit value (40 μg m^{-3} for annual mean concentration), thus demand indicators are comparable for this standard. The highest values for both pollutants were found in Barcelona (32.76 μg m^{-3} for PM_{10} and 53.78 μg m^{-3} for NO_2), while PM_{10} was lowest in Salzburg (23.86 μg m^{-3}) and NO_2 in Stockholm (38.50 μg m^{-3}). Results for O_3 were not comparable with NO_2 and PM_{10} values because concentrations (and standards) are based on daily max 8-h averages. Berlin (with 116.14 μg m^{-3}) and Salzburg (with 111.63 μg m^{-3}) showed the highest values for O_3. In contrast, the lowest values of O_3 were displayed by Rotterdam (84.74 μg m^{-3}) and Barcelona (89.60 μg m^{-3}).

Regarding global climate regulation supply, CO_2 sequestration indicators ranged from 1.05 t annually sequestered per hectare in Rotterdam to 3.66 t ha^{-1} $year^{-1}$ in Berlin. In the same way, carbon storage values ranged from 9.38 t ha^{-1} in Rotterdam to 32.84 t ha^{-1} in Berlin. Although Stockholm's average growing season is the shortest compared to the other cities, net CO_2 sequestration and carbon storage values were second-ranked after Berlin's. The demand side of global climate regulation showed a different picture: CO_2eq emissions per hectare were remarkably highest in Rotterdam (865.2 t ha^{-1} $year^{-1}$), most likely because of the impact of seaport activities on city's GHG emissions. On the other hand, the lowest value was found for Salzburg (86.6 t ha^{-1} $year^{-1}$). However, CO_2eq emissions per capita were lowest in Barcelona (2.51 t $capita^{-1}$ $year^{-1}$), reflecting the comparatively elevated population density of the Mediterranean city. Supply and demand indicators for this ES could be straightforwardly compared using annual net CO_2 sequestration and CO_2eq emission rates per hectare as a common unit. Results showed that demand values are approximately two orders of magnitude larger than supply.

Supply indicators for urban temperature regulation revealed also a considerable heterogeneity among case cities. The highest tree cooling area values were found in Berlin (42.70%) and Stockholm (37.50%). Rotterdam was distinctly the case study city with the lowest share of tree cooling area (12.20%). The demand for urban temperature regulation using heatwave risk as a proxy reflected clearly the different climate zones where the case cities are located. The results for Barcelona showed a very high number of expected hot days and tropical nights (>50), while heatwave risk in Stockholm is expected to be minimum (0–2 days). The values for Berlin, Rotterdam and Salzburg were higher than Stockholm's, but substantially far from Barcelona's (2–6 days).

In summary, both supply and demand indicators differed notably among the five case study cities. In most cases, Rotterdam showed the lowest supply values, followed by Barcelona or Salzburg. In contrast, the results for Berlin and, to a lesser extent, Stockholm indicated a relatively high supply of the three regulating ES analyzed. More heterogeneous results were found for demand indicators across the different cities. Barcelona and Rotterdam were clearly characterized by a high demand for urban temperature and global climate regulation respectively. Demand for air purification showed comparatively minor differences across cities. See also exemplary **Fig. 3.3** showing results for Barcelona compared to case study cities averages.

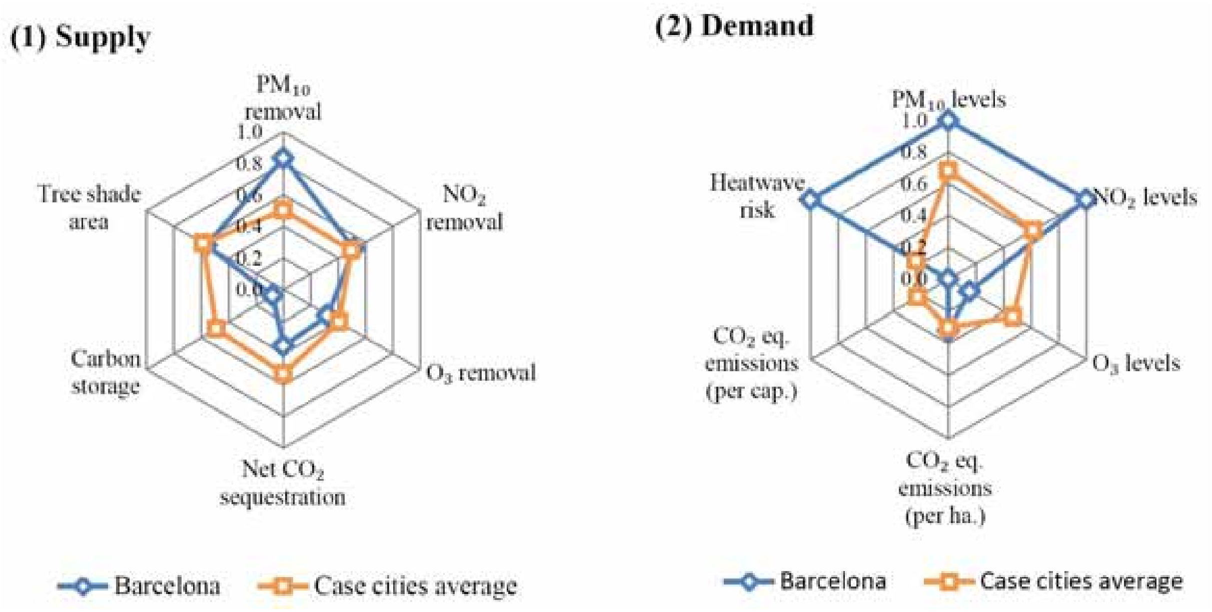

Fig. 3.3. Spidergrams comparing the standardized values of ES supply and demand indicators for Barcelona with the average values of the five case study cities. Notes: Supply and demand values are not directly comparable. Standardization is based on a linear rescaling of values in the 0-1 range on the basis of their minimum and maximum value.

3.3.2. Mismatches between ecosystem service supply and demand

Following the criteria described above, matches and mismatches between ES supply and demand were identified, showing a number of cases (12) where demand was clearly not totally met by supply considering the different case study cities (marked as red cells in **Table 3.5**). In only two cases ES demand was not totally met by supply, but the mismatch was considered minor, suggesting that the corresponding EQS could be met after the implementation of measures intended to increase ES supply (marked as yellow cells). Finally, ES supply matched with demand based on the corresponding EQS in almost half of the cases (14, marked as green cells).

Table 3.5. Identification and assessment of mismatches in ES supply and demand across the case study cities. Notes: Red cells indicate a substantial mismatch between ES supply and demand (ES contribution is lower than 10% in relation to the EQS exceedance or reduction target), suggesting that the corresponding EQS can be unlikely met by increase in ES. Yellow cells indicate a moderate mismatch between ES supply and demand (ES contribution is higher than 10% in relation to the EQS exceedance or reduction target) suggesting that the corresponding EQS could be met after the implementation of measures intended to increase ES supply. Green cells indicate that ES supply matches with demand based on the corresponding EQS. Blank cells indicate that the mismatch assessment could not be consistently done due to data limitations. See also subsection 2.5.

ES	Assessment	EQS	Barcel.	Berlin	Stockh.	Rotter.	Salzb.
Air purification	PM_{10} levels	EU					
	PM_{10} levels	WHO					
	NO_2 levels	EU/WHO					
	O_3 levels	EU					
	O_3 levels	WHO					
Global climate regulation	Contribution to city CO_2eq reduction target	City CO_2eq reduction target					
Urban temp. regulation	N/A	Heatwave thresholds					

The mismatch assessment of the ES air purification service indicated heterogeneous results across air pollutants and EQS. All cities met the EU limit value for PM_{10} annual average concentration (40 μg m^{-3}), but none of them complied with the WHO standard

(20 μg m^{-3}). Only Stockholm met the limit value for NO_2 levels (set at 40 μg m^{-3} for both standards). Tropospheric O_3 levels were below EU regulation in all case cities, but above WHO's air quality limit in Berlin and Salzburg (assuming 25 allowed exceedances per year as well), although the determination of the magnitude of the mismatch was not possible due to data limitations. The relative contribution of the ES service supply to meet air quality standards across the different case study cities is shown in **Table 3.6.** Air quality improvements due to ES supply showed the lowest values in Rotterdam and the highest values in Stockholm for all the analyzed pollutants, varying between 0.20% and 2.42% for PM_{10} levels, between 0.07% and 0.81% for NO_2 levels and between 0.10% and 1.16% for O_3 levels. According to i-Tree model results, expected air quality improvements are considerably more relevant in areas with 100% tree cover (e.g., urban forests or tree-covered urban parks). However, city-scale average annual air pollution levels in a hypothetic scenario without green space would not differ substantially from the current levels. Therefore, the ES mismatch should be minor if realistic increases in ES supply are intended to meet the standards. The results suggest that this situation only occurs for Salzburg's PM_{10} levels in relation to WHO limit value.

Table 3.6. Estimated air quality improvement due to air pollution removal by urban trees in case study cities (year 2011).

	Average percent air quality improvement at the city scale			**Average percent air quality improvement only in areas with 100% tree cover**			**Expected average annual air pollution levels without urban trees at the city scale (μg m^{-3})**		
	PM_{10}	NO_2	O_3	PM_{10}	NO_2	O_3	PM_{10}	NO_2	O_3
Barcelona	0.50	0.19	0.29	1.64	0.63	0.96	32.92	53.88	39.81
Berlin	0.73	0.21	0.30	1.67	0.49	0.70	30.33	53.49	47.41
Stockholm	2.42	0.81	1.16	6.14	2.12	2.96	29.16	38.81	55.62
Rotterdam	0.20	0.07	0.10	1.57	0.57	0.81	28.51	48.69	35.93
Salzburg	1.89	0.60	0.85	6.24	2.04	2.83	24.32	45.48	41.75

CO_2 offsets by urban GI (ES supply) compared to city-based CO_2eq emissions (corresponding to the baseline year for the reduction target) were modest in all case studies, ranging from 0.12% for Rotterdam to 2.75% for Salzburg. Similarly, the contribution of the ES supply in relation to CO_2eq reduction targets for 2020 was low in all case study cities. Salzburg was the only case where the annual sequestration rate was higher than the 10% threshold contribution (13.8%), although it must be noted that the city has the lowest reduction target among the case studies.

3.4. Discussion

3.4.1. The contribution of ecosystem service supply to human well-being in cities

The impact of urban green space on air quality in cities is a subject of scientific debate. Several empirical and modeling studies support that urban vegetation provides substantial air quality improvements followed by associated health benefits (Nowak et al., 2006, Nowak et al., 2013, Yin et al., 2011 and Islam et al., 2012). However, factors such as vegetation configuration or climate conditions can strongly limit the ability of vegetation to remove air pollutants, especially at the patch scale (Setälä et al., 2013 and Vos et al., 2013). The modeling results presented here indicate that average air quality improvements due to air purification supply is relatively low at the city scale for the three analyzed air pollutants in all case study cities (e.g., from 0.07% in Rotterdam to 0.81% in Stockholm for NO_2), although positive effects are likely to be more relevant in highly tree-covered areas such as urban forests (e.g., expected air improvements are higher than 6% for PM_{10} in Stockholm's and Salzburg's areas with an hypothetical 100% tree cover, see **Table 3.6**). Therefore, the average contribution of ES supply in regard to the compliance with air quality standards is considered modest at the local level in all case studies, suggesting a limited effectiveness to address ES mismatches by increasing ES supply (e.g., implementing tree-planting programs) unless air pollution concentration exceedance is minor (e.g., PM_{10} levels compared to WHO standard in the case of Salzburg).

A number of studies have assessed the role of urban green space as a climate change mitigation strategy by offsetting city CO_2 emissions (Pataki et al., 2009, Escobedo et al., 2010, Zhao et al., 2010 and Liu and Li, 2012). Impacts of net CO_2 sequestration rates on offsetting annual city CO_2 emissions vary from 3.4% in Gainesville, US (Escobedo et al., 2010) to 0.26% in Shenyang, China (Liu and Li, 2012). As expected, similar results have been obtained for the case study cities (ranging from 0.12% in Rotterdam to 2.75% in Salzburg). This paper has gone one step further by considering city-specific GHG reduction targets as a desired condition at the city level. Again, results show a modest contribution of ES supply (less than 15%) in all case study cities, suggesting that increases in direct carbon sequestration delivered by GI (e.g., by doubling tree density) is not likely to be an effective means for reaching local CO_2eq reduction targets (in line with Pataki et al., 2011).

Previous empirical evidence on the supply of urban temperature regulation (Bowler et al., 2010) revealed that the cooling effect of urban GI can be relatively relevant at the patch scale. For example, a maximum of 2 °C difference relative to built-up area was observed in an urban park in Stockholm (Jansson et al., 2007). However, the extension of the cooling effect of green space beyond its boundaries is uncertain, especially at the wider city scale (Bowler et al., 2010). Therefore, heatwave thresholds cannot be consistently balanced against the cooling effect provided by GI elements at the city scale. Additional empirical research is required to assess these mismatches, especially by establishing specific temperature thresholds according to each climate zone and measuring the cooling impact of GI interventions at the city scale.

The findings of this research suggest that GI can only play a minor or complementary role, at least at the core city level, to urban mitigation measures intended to abate air pollutant and GHG emissions at the source (e.g., road traffic management or energy efficiency measures) or to adaptation policies intended to cope with heat extremes (e.g., heat warning plans). Yet, there are important reasons for which the current and potential supply of these ES should not be neglected in local policy decision-making. First, GI can provide other important benefits to urban population due to its multifunctional capacity (e.g., stormwater runoff mitigation or recreational opportunities), while technological substitutes are normally designed as single-purpose. Second, although GI expansion in compact cities such as those analyzed in this paper might be challenging due to lack of available land and densification processes, measures for preserving existing green spaces and innovative ways to allocate new ones could considerably enhance ES supply at the city level (Jim, 2004). For instance, the potential of green roofs and walls to deliver a wide range of ES has been assessed in various empirical studies (Oberndorfer et al., 2007 and Rowe, 2011).

3.4.2. Strengths and weaknesses of using environmental quality standards to assess ecosystem service mismatches

The demand side is frequently omitted or underrepresented in ES assessments which usually focus on ES supply (Burkhard et al., 2014). Yet, an increasing number of studies have developed assessment methods considering both the ES supply and demand in order to provide a complete picture of the ES delivery process where mismatches between both sides can be identified (e.g., Van Jaarsveld et al., 2005, Burkhard et al., 2012, Kroll et al., 2012, García-Nieto et al., 2013, Boithias et al., 2014, Schulp et al., 2014 and Geijzendorffer et al., 2015). This paper contributes to the ES research agenda (De Groot et al., 2010) suggesting a novel methodological approach based on the use of EQS to assess mismatches between ES supply and demand with a focus on regulating ES in core city areas. Based on the assessment of ES mismatches in five European cities, strengths and weaknesses of this approach could be recognized.

This approach can be especially advantageous for regulating ES assessments because of several reasons: (1) demand for regulating ES usually cannot be indicated by direct market prices, unlike many provisioning ES for example (De Groot et al., 2012); (2) the interactions between regulating ES and human benefits are often very complex, thus ES demand is challenging to indicate (Burkhard et al., 2014). EQS are generally meaningful to society and can reasonably express a common threshold to assess regulating ES mismatches across different societal contexts as they provide a benchmark representing the minimum desirable environmental quality conditions under which some components of human well-being such as health can be secured, hence allowing comparative analyses; (3) this approach allows relatively quick assessments of ES demand if data on environmental quality is available at the city level. In contrast, other demand-side assessments like socio-cultural elicitation are usually more time consuming and resource intensive (Martín-López et al., 2014).

However, the use of EQS in ES assessments has also drawbacks. The existence of different EQS regulating the same environmental condition (or ecological pressure) can create uncertainty about which thresholds are more adequate in terms of expressing a societal demand related to human needs for well-being. In this paper, both WHO and EU standards for air quality have been used giving different ES mismatch results for some air pollutants. Although only EU standards are legally binding for case study cities, WHO standards are probably more reliable expressing a desirable or required end condition of air quality (Brunekreef and Holgate, 2002). The main shortcoming of local GHG emission reduction targets is that often they are not based on scientific evidence about possible climate change impacts, but on political reasons. Regarding urban temperature regulation, the multiple factors involved in the relationship between temperature extremes and human health vulnerability call for specific temperature thresholds to properly account for varying environmental conditions and societal demands at the local level.

More generally, the use of specific or local-based thresholds is possibly the most appropriate option when assessing ES for which demand is strongly context/user/stakeholder dependent (Paetzold et al., 2010), despite it would make cross-city comparisons less meaningful. This is clearly the case of cultural ES. For example, several standards have been suggested as thresholds for assessing the desirable amount of recreational opportunities delivered by green space in urban areas, normally based on criteria of accessibility to green space (i.e., distance) and space size (Van Herzele and Wiedemann, 2003, Söderman et al., 2012 and Kabisch and Haase, 2014). The former is commonly seen as the most important factor related to the recreational use of urban green space and a maximum 300–400 m distance from home has been observed as a threshold after which the use decreases substantially (Schipperijn et al., 2010). Some regulatory agencies have consequently recommended standards based on these criteria. For example, the European Environment Agency (EEA) recommends that people should have access to green space within 15 min walking distance (Stanners and Bourdeau, 1995) and the English standard ANGSt (Accessible Natural Greenspace Standard, Natural England, 2010) recommends that urban population should have an accessible green space no more than 300 m from home (Barbosa et al., 2007). However, these standards have been criticized because they fail to address issues such as green space quality or local context and needs (Pauleit et al., 2003). Still, some authors claim that green space recreational standards are needed but they should be locally developed according to specific social and quality criteria (Baycan-Levent and Nijkamp, 2009). Therefore, a possible extension of the approach presented in this paper beyond regulating ES should be carefully designed.

3.4.3. Spatially explicit ecosystem service mismatches

The spatial distribution of ES supply and demand at the city level has not been addressed in this paper. Yet, for some ES such as air purification or urban temperature regulation both their supply and demand can substantially vary across the urban fabric. The use of spatially explicitly indicators could show the specific location of ES mismatches at the inner-urban level (or higher scales), hence informing about ES deficit areas (demand is higher than supply) to urban planners and managers. Several attempts of mapping ES mismatches have already been developed at different spatial scales (e.g., Kroll et al., 2012, García-Nieto et al., 2013, Boithias et al., 2014 and Schulp et al., 2014). However, assessments at the core city scale are scarce, probably due to the lack of fine-resolution data for the appropriate quantification of ES supply and demand indicators.

3.5. Conclusion

This paper provides an innovative approach for assessing mismatches in regulating ES supply and demand using EQS as a common minimum threshold for determining whether the difference between supply and demand is problematic in terms of human well-being. The approach has revealed to be appropriate for the ES air purification, for which there is a large body of evidence on the health impacts of air pollution and EQS are well-established at the international level. Similarly, local GHG reduction targets can reasonably express a demand for mitigating the impacts of climate change in urban areas (global climate regulation), thus the assessment of ES mismatches was also possible. The application of the approach for the ES urban temperature regulation has proved more problematic. The demand for urban temperature regulation is strongly context and user dependent, thus common thresholds (such as heatwave thresholds) are less appropriate. Furthermore, the spatial scale to which the ES is delivered is still not totally clear in terms of scientific evidence, creating uncertainties in the ES mismatch assessment. In general, more empirical studies are needed to improve GI design and monitor its effectiveness in meeting local or international environmental standards and goals in different urban areas.

The case study of five European cities reveals mismatches between ES supply and demand in half of the 28 ES/EQS/City combinations analyzed, suggesting that further protection and restoration of urban GI will be required if ES are to play a more relevant role in meeting EQS to enhance human well-being in cities. However, the assessment indicates that ES supply contributes very moderately in relation to the compliance with the EQS in most part (12 out of 14) of the identified mismatches. Results suggest that EQS could be met after the implementation of feasible measures intended to increase ES supply only in two analyzed cases. Therefore, this research suggests that regulating ES supplied by urban GI are expected to play only a minor or complementary role (currently and potentially) to other urban policies intended to abate air pollution and GHG emissions at the city scale. Urban managers and policy-makers should take into account these considerations when designing and implementing GI programs, but recognizing at the same time the multiple benefits associated to GI in urban contexts not addressed in this assessment (e.g., runoff mitigation, noise reduction and recreational opportunities).

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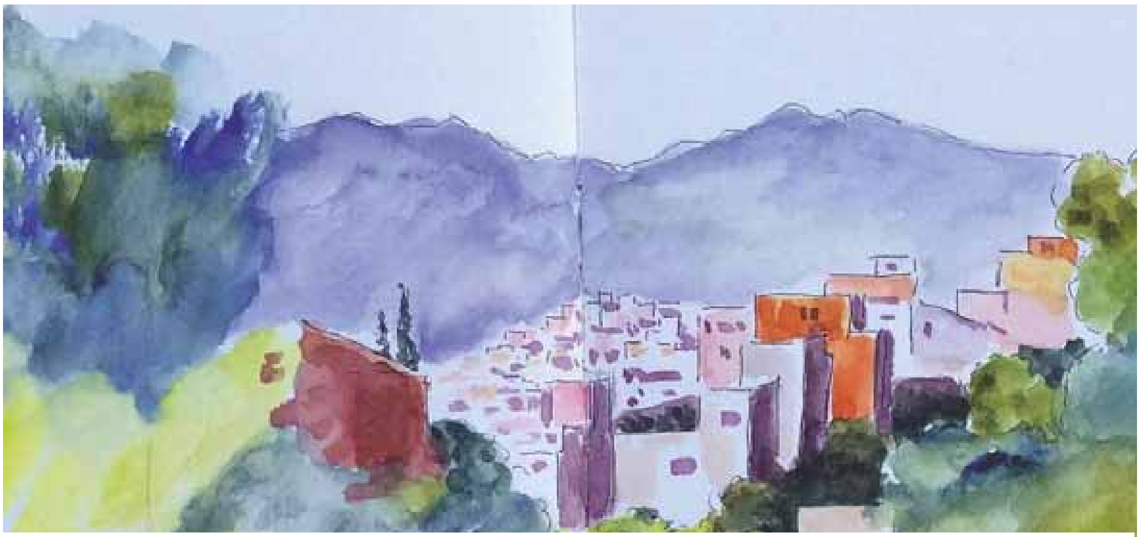

Parc dels Tres Turons, Barcelona, Spain (illustration by Núria Domingo, published with kind permission of the author and the series of seminars on city, environment, health and drawing “Ciutat Verda” – www.ciutatverda.info)

CHAPTER IV

Mapping ecosystem service capacity, flow and demand for landscape and urban planning: A case study in the Barcelona metropolitan region

Abstract

Ecosystem services (ES) mapping is attracting growing interest from landscape and urban planning, but its operationalization in actual decision-making is still limited. A clear distinction between ES capacity, flow and demand can improve the usefulness of ES mapping as a decision-support tool by informing planners and policy-makers where ES are used unsustainably and where ES flow is failing to meet societal demand. This paper advances a framework for mapping and assessing the relationships between ES capacity, flow and demand with a focus on the identification of unsatisfied demand. The framework was tested in the Barcelona metropolitan region, Spain, considering two ES of critical relevance for the urban population: air purification and outdoor recreation. For both ES, spatial indicators of capacity, flow, demand and unsatisfied demand were developed using proxy- and process-based models. The results show a consistent spatial pattern of all these components along the urban-rural gradient for the two ES assessed. The flow of both ES mainly takes place in the periurban green areas whereas the highest capacity values are mostly found in the protected areas located on the outskirts of the metropolitan region. As expected, ES demand and particularly unsatisfied demand are mostly situated in the main urban core (i.e., Barcelona and adjacent cities). Our assessment also reveals that the current landscape planning instrument for the metropolitan region mostly protects areas with high capacity to provide ES, but might lead to declining ES flows in periurban areas due to future urban developments. We contend that the mapping of ES capacity, flow and demand can contribute to the successful integration of the ES approach in landscape and urban planning because it provides a comprehensive picture of the ES delivery process, considering both ecological and social underlying factors. However, we identify three main issues that should be better addressed in future research: (1) improvement of ES demand indicators using participatory methods; (2) integration of ecological thresholds into the analysis; and (3) use of a multi-scale approach that covers both the local and regional planning levels and cross-scale interactions between them.

Keywords: Air purification; Ecosystem service mismatch; Outdoor recreation; Urban-rural gradient; Spatial modeling.

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4.1. Introduction

Ecosystem services (ES) mapping is gaining prominence in the environmental science and policy agendas (Egoh et al., 2012, Crossman et al., 2013 and Malinga et al., 2015). For example, the European Union (EU) Biodiversity Strategy to 2020 called Member States to assess and map ES in their national territory as a supporting action to maintain and enhance ecosystems (EC, 2011). ES mapping can inform a variety of decision-making contexts (Gómez-Baggethun and Barton, 2013), including: awareness raising and communication (e.g., Hauck et al., 2013); ecosystem accounting (e.g., Schröter et al., 2014); landscape and conservation planning (e.g., Palomo et al., 2014); and instrument design (e.g., Locatelli et al., 2014), among others.

In order to make ES maps operational for landscape and urban planning, recent ES literature calls for a clearer distinction between the three main components of the ES delivery process, namely ES capacity, flow and demand (Bastian et al., 2013, Villamagna et al., 2013, Burkhard et al., 2014 and Schröter et al., 2014). Most spatially explicit ES assessments have focused on studying ES capacity, i.e., the ecosystems' potential to deliver ES (see Martínez-Harms and Balvanera, 2012 for a review). In contrast, despite increased interests and efforts to assess and map ES flow and demand (e.g., García-Nieto et al., 2013, Palomo et al., 2013 and Schröter et al., 2014), the conceptualization of both components is still subject to different approaches (Villamagna et al., 2013 and Wolff et al., 2015). According to Wolff et al. (2015), ES demand can be framed either as the direct use/consumption of an ES or as the desired/required level of the ES by society. However, the conceptual framework developed by Villamagna et al. (2013) argues that only the latter approach should be considered ES demand, whereas the actual use of the ES constitutes its flow.

At the operational level, the spatially explicit distinction and assessment of ES capacity, flow and demand can enhance the integration of ES in planning, management and decision-making because it can inform planners and policy-makers about the localization of potential ES mismatches, either in terms of *unsustainable uptake* of ES or in terms of *unsatisfied demand* for ES (Geijzendorffer et al., 2015). This information can be used to design plans or policy regulations oriented to: (1) redirect ES flows from overused areas (Schröter et al., 2014), and (2) improve access to ES benefits by identifying areas where ES flows fail to meet societal demand (Kabisch and Haase, 2014).

The aim of this paper is to advance an operational framework for assessing and mapping ES capacity, flow and demand to inform landscape and urban planning. First, we build on previous conceptual frameworks to distinguish between ES capacity, flow and demand, as well as their relationships in terms of (un)sustainable uptake and (un)satisfied demand. Second, we use proxy-based and process-based models within the ESTIMAP tool (Zulian et al., 2014) to develop, test and discuss suitable spatial indicators for the three components with a focus on the identification and mapping of unsatisfied demand. Third, we assess the spatial patterns observed from the application of these indicators in a case study and discuss their implications for planning and policy.

The framework was tested in the Barcelona metropolitan region, Spain. Assessing and mapping ES capacity, flow and demand can be particularly relevant in urban landscapes, where urbanization impinges upon ecosystem's capacity to deliver sustained ES flows and where the high concentration of human population and assets usually entails high demands for ES (Kroll et al., 2012, Burkhard et al., 2012 and Haase et al., 2014). We focused on air purification and outdoor recreation, two ES of key importance for improving health and well-being in urban areas since they contribute to air pollution abatement and to the provision of opportunities for relaxation and physical activity (Bolund and Hunhammar, 1999 and Gómez-Baggethun et al., 2013).

4.2. Methods and materials

4.2.1. Conceptual distinction between ecosystem service capacity, flow, and demand

The distinction between ES capacity, flow and demand ultimately builds on the conceptual framework for ES assessment known as the "ES cascade model", which illustrates the links between ecosystems and human preferences along a chain of ecosystem properties, functions, services, benefits and values (Haines-Young and Potschin, 2010; **Fig. 4.1**). Despite the varying understanding, terminology and application of the capacity, flow and demand concepts in the ES literature (see Villamagna et al., 2013 and Wolff et al., 2015), in this paper we mostly follow the framework developed by Villamagna et al. (2013) because it provides a flexible, yet consistent approach for decision-making. Therefore, we define ES capacity as "the ecosystem's potential to deliver services based on biophysical properties, social conditions, and ecological functions", ES flow as "the actual production of the service" used or experienced by people, and ES demand as "the amount of a service required or desired by society" (Villamagna et al., 2013:116). We further developed this approach into an operational framework (**Fig. 4.1**) to inform decision-making on the basis of the relationships between capacity, flow and demand which can express two different ES mismatches (Geijzendorffer et al., 2015). On the one hand, the relationship between ES capacity and flow can indicate ES overuse or unsustainable uptake when capacity is smaller than flow, if the ES is rival or congestible (Schröter et al., 2014). On the other hand, the relationship between ES flow and demand can indicate unsatisfied demand when flow is not meeting the amount of ES demanded by society (Geijzendorffer et al., 2015). The relationship between ES capacity and demand is not explicitly considered in this framework because if demand is higher than capacity, the mismatch usually expresses an unsatisfied demand, unless flow is meeting the demand. In the latter case, the mismatch would express an unsustainable ES uptake.

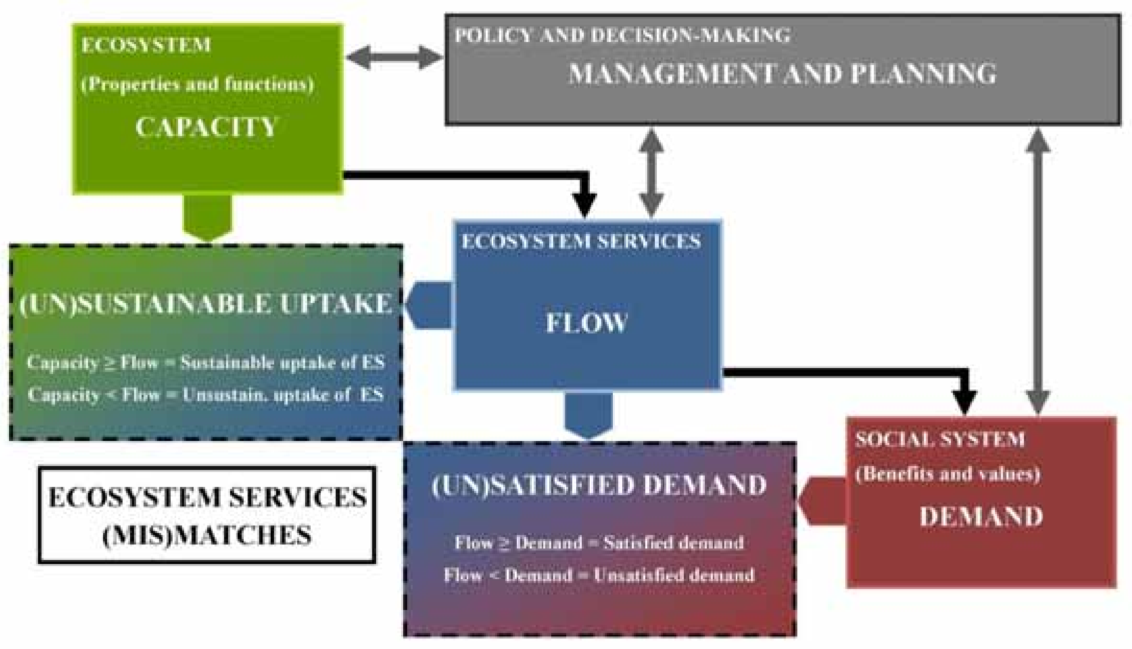

Fig. 4.1. Framework for assessing the relationships between ES capacity, flow and demand, i.e., if the uptake of ES is sustainable (capacity & flow) and if demand is being satisfied (flow & demand). Management and planning affect and are affected by ES capacity, flow and demand. Building on Haines-Young and Potschin (2010), Villamagna et al. (2013) and Geijzendorffer et al. (2015).

The conceptualization of ES demand (and unsatisfied demand) used here is inherently challenging at the operational level because it requires information about desired or required end conditions which can vary among different stakeholder groups, especially for cultural ES. For the sake of our analysis, in this paper we used environmental quality standards and recommendations as prescribed in policy as a proxy threshold to determine expected desired or required end conditions related to ES demand from a societal perspective (see Paetzold et al., 2010 and Baró et al., 2015). A risk perspective is commonly used to quantify demand for regulating ES (Wolff et al., 2015). Under this approach, demand for air purification can be indirectly indicated considering the magnitude of pressures needing regulation (i.e., air pollution levels) and population exposed to these pressures (Burkhard et al., 2014). Besides, air quality standards can be used to provide a minimum threshold to identify a possible mismatch between flow and demand for air purification (i.e., exceedance of air quality limit values in inhabited areas indicate an unsatisfied demand). Yet, this approach does not necessarily imply that air quality improvement is to be achieved solely by more ES flow because the demand driver is human-induced (Baró et al., 2015). In the case of outdoor recreation, recommended standards are related to proximity to recreational sites (e.g., Stanners and Bourdeau, 1995), as distance has been observed to be a critical variable explaining recreational use of green space in urban areas (Schipperijn et al., 2010 and Paracchini et al., 2014). Following this rationale, outdoor recreation demand can be indicated based on the availability of recreational sites close to people's home and population density assuming that all inhabitants in the case study area have similar desires in terms of (everyday life) outdoor recreational opportunities (Paracchini et al., 2014 and Ala-Hulkko et al., 2016).

4.2.2. Description of the case study area

We tested the framework in the Barcelona metropolitan region (BMR), located North-East of Spain, by the Mediterranean Sea (**Fig. 4.2**). The BMR (5.03 million inhabitants and 3244 km^2, Statistical Institute of Catalonia, year 2015) embeds 164 municipalities and seven counties, but its urban core, known as Barcelona metropolitan area, is constituted by the municipality of Barcelona (1.61 million inhabitants) and several adjacent middle-size cities (**Fig. 4.3**).

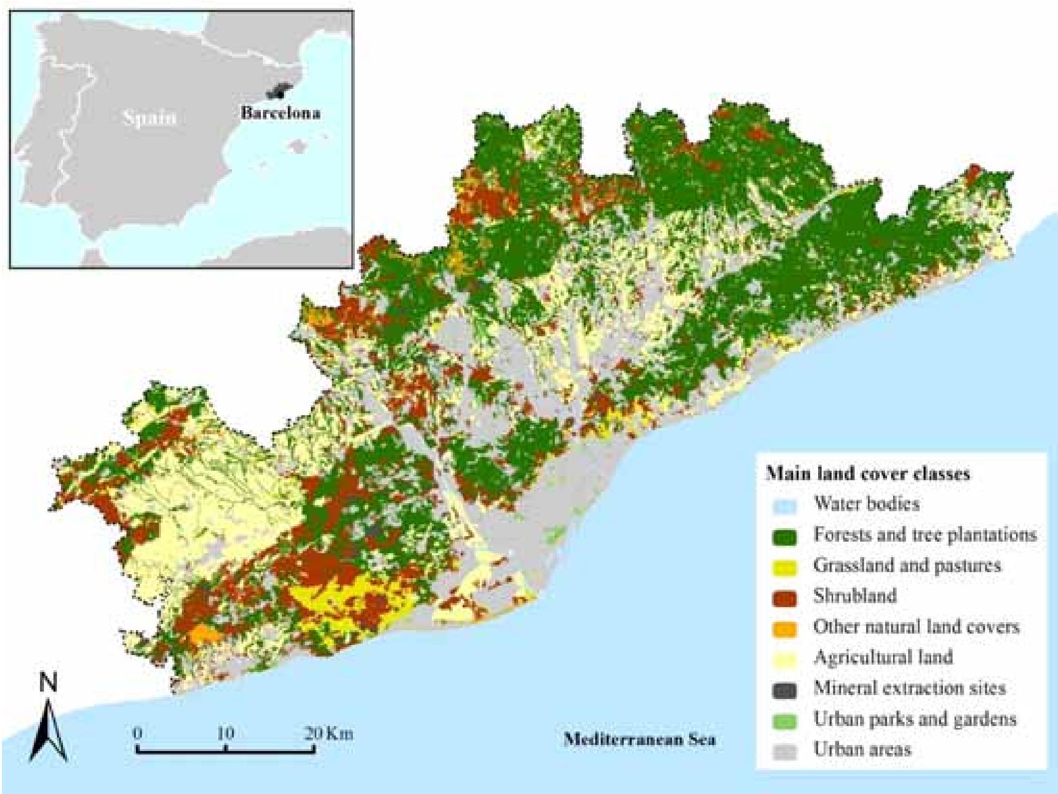

Fig. 4.2. Main land covers in the Barcelona Metropolitan Region (BMR). Own elaboration based on the spatial dataset of habitats of Catalonia (year 2013).

The BMR is one of the regional planning areas of the 'General Territorial Plan of Catalonia' (PTGC, 1995), the uppermost landscape planning instrument in the region of Catalonia. The 'Territorial Metropolitan Plan of Barcelona' was developed following PTGC's strategic guidelines and approved in 2010 by the Government of Catalonia (PTMB, 2010). The PTMB establishes three main planning categories, so-called "systems", for land use regulation: (1) open areas; (2) urban land; and (3) transport infrastructure. Because the latter system is highly dependent on transport planning and it has a limited impact in terms of land use change, the focus of this paper is on the 'open areas' and 'urban' planning systems. However, given the relationship between transportation and certain components of the selected ES (e.g., demand for air purification), some implications for transport policy are discussed. The open areas planning system regulates the land protected from urbanization, including, fully or partially, fourteen Natura 2000 sites. The urban planning system regulates built-up land and defines strategies for urban expansion by the tentative delimitation of development areas that can be subsequently refined by municipalities through urban master plans. For example, most municipalities (including Barcelona) of the urban core share a common urban master plan (General Metropolitan Plan) which is currently under major revision. See **Table 4.1** for more details and **Fig. 4.3** for the spatial representation of the two PTMB planning systems.

We contend that the BMR is an exceptional testing ground for the purpose of this research for at least three reasons: (1) the BMR is one of the most densely populated urban regions in Europe (1550 inhabitants per km^2), which poses great challenges for sustainable landscape and urban planning; (2) it contains a rich variety of natural habitats of high ecological value, including Mediterranean forests (1184.56 km^2; 36.5%) and shrub land (448.62 km^2; 13.8%), agro-systems of strategic economic importance (e.g. vineyards) (654.51 km^2; 20.2%), and inland water bodies (24.08 km^2; 0.7%) (see **Fig. 4.2**); and (3) both local and regional authorities have shown interest in implementing the ES approach in landscape and urban planning (e.g., Barcelona Green Infrastructure and Biodiversity Plan 2020, Barcelona City Council, 2013).

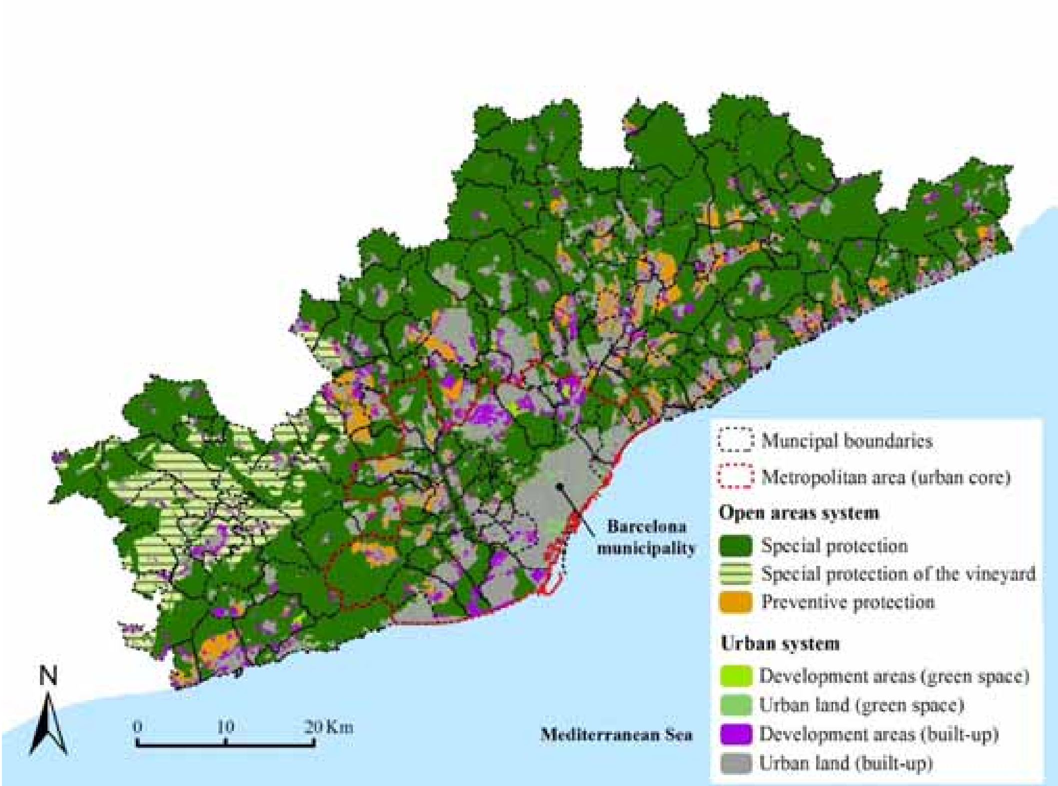

Fig. 4.3. Administrative boundaries in the BMR and planning systems of the Territorial Metropolitan Plan of Barcelona (PTMB, 2010). See also **Table 4.1**.

Table 4.1. Description of the planning systems of the Territorial Metropolitan Plan of Barcelona (PTMB, 2010).

Planning systems	Main zoning categories	Short description	Total area (km² and % of BMR)
Open areas system	Special protection	Highly protected land for its ecological and agricultural values. Includes Natura 2000 sites and other areas under different protection categories.	2031.70 (62.6%)
	Special protection of vineyard	Highly protected land for its landscape and agricultural values for the wine sector.	230.33 (7.1%)
	Preventive protection	Areas where urban development is, a priori, restricted. Normally transitional between urban and protected land. Urbanization may be possible under certain circumstances.	142.84 (4.4%)
Urban system	Urban land (consolidated)	Consolidated urban build-up land (residential, industrial, commercial, etc.), including urban green areas.	Total: 634.92 (19.6%) Green space: 84.4 (2.6%)
	Development areas	Areas designated for future urban development, including the creation of new urban green areas.	Total: 205.38 (6.3%) Green space: 36.05 (1.1%)

4.2.3. Selection of ecosystem services

The ES outdoor recreation and air purification were chosen as exemplars for the assessment because of their relevance to urban areas (Bolund and Hunhammar, 1999, Gómez-Baggethun and Barton, 2013 and Haase et al., 2014) and particularly the BMR. Moreover, unlike other ES such as global climate regulation (see Schröter et al., 2014), a meaningful distinction between capacity, flow and demand can be drawn for these two ES.

The cultural ES outdoor recreation is probably one of the most valued ES in cities, decisively contributing to enhance physical and mental health of the urban population (Chiesura, 2004, Gómez-Baggethun et al., 2013 and Triguero-Mas et al., 2015). The city of Barcelona and many of its surrounding middle-size cities are characterized by a high degree of compactness and high population density, involving a scarcity of inner green areas (Baró et al., 2014). Periurban parks and other natural suburban areas represent thus an important option for outdoor recreational opportunities in the BMR. For example, the periurban park of Collserola, located in a central position of the BMR and virtually surrounded by urban fabric, receives around two million visitors annually according to a recent study (IERMB, 2008).

The regulating ES air purification is also the subject of growing attention in the policy agenda. Abatement of air pollution is still a pressing challenge in most major urban areas worldwide, especially in regard to dioxide nitrogen (NO_2) and particulate matter (WHO, 2014). For example, the 2015 annual report on air quality in Europe (EEA, 2015) estimated that, during the period 2011–2013, 8–12% of the urban population within the EU was exposed to NO_2 concentrations above the limit value set both by the EU (EU, 2008) and the World Health Organization (WHO, 2005) in 40 $\mu g\ m^{-3}$ (annual average). The harmful impacts of air pollution on human health are consistently supported by scientific evidence (e.g., Brunekreef and Holgate, 2002 and WHO, 2013). Vegetation in urban landscapes can improve air quality by removing pollutants from the atmosphere, mainly through leaf stomata uptake and interception of airborne particles (Nowak et al., 2006). In the last decade, the city of Barcelona has repeatedly exceeded the EU limit values for average annual concentrations of NO_2 and particles with diameter of ten micrometers or less (PM_{10}). Urban trees and shrubs within the municipality of Barcelona removed 166.0 t of PM_{10} and 54.6 t of NO_2 during the year 2008 according to Baró et al. (2014) estimates.

4.2.4. Description of spatial ecosystem service models and indicators

We used the methodological framework provided by the Ecosystem Services Mapping tool (ESTIMAP) for the spatial assessment of the two selected ES (Paracchini et al., 2014, Zulian et al., 2013 and Zulian et al., 2014). ESTIMAP is a collection of spatial models for ES assessment originally developed to support environmental policies at a European scale such as the EU Biodiversity strategy (Maes et al., 2014). Because ESTIMAP is based on the conceptual ES cascade model (Haines-Young and Potschin, 2010), its spatial outputs are consistent with the ES capacity, flow, and demand framework used in this study. ESTIMAP was designed for a continental scale, therefore it was adapted to the regional scope of this paper to make it usable for urban and landscape planning.

In the following subsections we describe this adaptation and downscaling process. **Table 4.2** provides an overview of the ES indicators developed and used in the assessment and a brief description of the spatial input data. More details on the methods applied and data used to compute these indicators are provided in **Appendix B** (Supplementary information). All geoprocessing operations were carried out using ArcGIS v.10.1 (ESRI) or GRASS GIS v. 7.0 (GRASS Development Team).

Outdoor recreation

The model used here for assessing outdoor recreation focuses on nature-based recreational activities in the everyday life (Paracchini et al., 2014). Those activities include practices such as walking, jogging, bike riding, picnicking, observing flora and fauna, or simply enjoying nature, among other possibilities, but it excludes nature-related tourism activities involving long trips, which some classifications consider a distinct ES (e.g., TEEB, 2010).

Like other approaches to cultural ES mapping (e.g., Casado-Arzuaga et al., 2014), ESTIMAP-recreation assumes that all ecosystems, including natural, semi-natural and intensively managed ecosystems, are potential providers of recreational opportunities, although the capacity level depends on ecosystem features related to people's recreational preferences. The rationale for assessing recreation capacity in our model can be summarized as follows: (1) the lesser human influence on landscapes, the higher value in terms of nature-based recreational potential; (2) protected natural areas and features (e.g., remarkable trees) are considered indicators of high recreational capacity; and (3) water bodies exert a specific attraction on the surrounding areas (see Paracchini et al., 2014). Recreation capacity was hence mapped on the basis of the assessment of three components: degree of naturalness, nature protection, and presence of water. Each component was composed of one to four internal factors considered relevant in the case study of the BMR and for which spatial input data was available (see **Appendix B** in Supplementary information for a detailed description of factors and data sources). A score or weight (in the 0–1 range) was assigned to every factor standing for their relative importance or impact in terms of recreation potential. The final selection of factors and definition of scores was based on a consultation process (via focus group) with four experts working in environmental planning and territorial analysis for the Barcelona Regional Council. The experts were asked to: (1) revise a preliminary proposal of factors suggested by the research team (introducing changes if necessary); and (2) assign a score to every factor based on their thorough knowledge of the socio-ecological context in the case study area. In case of no consensus for a specific score, a compromise value was agreed (e.g., average value of suggested scores). Five factors were subject to a distance decay modeling, assuming that the recreation potential decreases as the distance from the specific feature (e.g., a beach or remarkable tree) increases (see **Appendix B** Supplementary information for details). The final dimensionless value of recreation capacity was normalized in the 0–1 range.

Mapping outdoor recreation flow is challenging because data on the actual recreational use or experience of ecosystems by people is often inexistent or limited to certain areas (but see some attempts in Palomo et al., 2013 and Schröter et al., 2014 and Wood et al., 2013). The ESTIMAP approach is based on a population analysis in which the expected ES flow is mapped by modeling the number of visitors (or trips) that reach a given recreational area considering a defined distance threshold (Paracchini et al., 2014). The adjustment applied in the case study area involved several considerations: (1) the road and track network reaches nearly every point in the BMR, hence it was not considered in the proximity analysis; (2) a distance threshold of 1 km representing close-to-home daily trips was set based on recommended standards by regulatory agencies (Stanners and Bourdeau, 1995 and Barbosa et al., 2007); (3) a population density grid was created based on an intersect between census tract dataset (INE, 2011) and residential use classes extracted from a high resolution land cover map (MCSC, 2009) assuming equal population distribution within residential land for each census tract; (4) an impedance function was applied in the modeling following Paracchini et al. (2014) (see **Appendix B** in Supplementary information for details); and (5) the expected flow was only represented in medium to very high capacity recreation areas (i.e., recreation capacity equal or higher than 0.4) assuming that inhabitants want to reach these areas and not low capacity areas (recreation capacity lower than 0.4 mostly corresponds to artificial land covers, see also **Fig. 4.4a**).

Following the rationale described above (**Subsection 4.2.1**), outdoor recreation demand was mapped based on the availability of recreational sites (i.e., recreation capacity equal or higher than 0.4) close to people's homes and population density. A spatial cross-tabulation was carried out between a reclassified raster of Euclidian distances to recreation sites and the population density grid, assuming that all inhabitants in the case study area have similar desires in terms of (everyday life) outdoor recreational opportunities, but their level of fulfillment depends on proximity to recreation sites (see cross-tabulation matrix in **Appendix B** in Supplementary information). The resulting raster indicates ES demand in residential land following a 0 (i.e., no relevant demand) to 5 (i.e., very high demand) value range. The assessment and mapping of unsatisfied demand for outdoor recreation was accomplished by selecting the number of people from the population density grid living further than 1 km (i.e., the assumed threshold distance) from any recreational site. Therefore, the spatial indicator represents the population with unfulfilled recreational expectations according to our approach.

Air purification

The ES air purification focuses on the air pollutant NO_2 for the reasons mentioned above and was modeled and mapped using the following indicators: (1) NO_2 dry deposition velocity on vegetation, considered here as a proxy to assess the ecosystems capacity to remove pollutants from the atmosphere; (2) modeled NO_2 removal flux by vegetation, considered here as measure for the ES flow; and (3) an ES demand index based on population density and exposure to NO_2 concentrations (see also Baró et al., 2015).

In many studies (e.g., Nowak et al., 2006 and Escobedo and Nowak, 2009) dry deposition velocities of the gaseous pollutants for the in-leaf season are estimated using a series of resistance formulae (Baldocchi et al., 1987) that require specific information regarding the structure and species composition of urban vegetation. Since this information was not available for the entire case study area, the capacity indicator for air purification was estimated following the approach proposed by Pistocchi et al. (2010), which estimates deposition velocity (V_d) as a linear function of wind speed at 10 m height (w) and land cover type:

$$V_d = \alpha_j + \beta_j \cdot w \qquad \text{(Formula 4.1)}$$

Where α and β are, respectively, the intercept and slope coefficients corresponding to each broad land cover type *j*, namely forest, bare soil, water or any combination thereof.

The NO_2 removal indicator (flow) was mapped based on the spatial distribution of NO_2 annual average concentrations and the capacity map. Concentrations of NO_2 were estimated using a Land Use Regression (LUR) model, a computation approach widely used for assessing air pollution at different scales (e.g., Briggs et al., 1997, Hoek et al., 2008 and Beelen et al., 2013). The LUR model was built using NO_2 concentration measurements (year 2013) from the operational monitoring stations located in the BMR (n = 40) as dependent variable, and a set of spatial predictor parameters (i.e., independent variables) related to land cover type, geomorphology, climate, population density, and road network (see **Appendix B** Supplementary information for input data details), that were considered to be the most relevant for distribution of NO_2 concentrations. Because several of the independent variables influence air pollution concentration at different spatial scales, we evaluated the correlation between each of the parameters at different scales and the measured NO_2 concentrations. We developed spatial buffers around each monitoring station from 50 to 1500 m every 50 m, and calculated for each buffer statistical values (mainly mean and sum) of the parameters. We selected the most relevant spatial buffer as the one reporting the highest R^2 between the statistical value and the measured concentration given that the correlation had the expected sign (i.e., higher concentrations with higher values of urban areas, but lower concentrations with higher values of forest areas). Within this optimal buffer, values of the original parameter were aggregated and the resulting values were used as parameters for the LUR model. Annual NO_2 removal was estimated as the total pollution removal flux in the areas covered by vegetation, calculated as the product of NO_2 concentration and deposition velocity maps (Nowak et al., 2006).

Considering the risk perspective described above (**Subsection 4.2.1**), air purification demand was mapped based on NO_2 concentration levels and population density. A spatial cross-tabulation was carried out between both variables following the same approach as for recreation, i.e., the higher NO_2 concentration and population density the higher demand values (see cross-tabulation matrix in **Appendix B** in Supplementary information). The resulting index spatially represents ES demand for air purification in the 0 (i.e., no relevant demand) to 5 (i.e., very high demand) value range. The map of unsatisfied demand for this ES was generated by selecting the population living in areas where annual mean NO_2 concentrations exceed the EU limit value (40 µg m^{-3}).

Table 4.2. Overview of ES indicators and main input data used in the assessment (building on the blueprint by Crossman et al., 2013). All indicators were mapped at a regional scale (pixel size 100x100m) using data corresponding to years 2011-2013. For further details see **Appendix B**.

Mapped ES	ES component indicator	Unit	Main input data	Mapping method	Comments and main method-ological references
Outdoor recreation (everyday life)	Recreational potential index (Capacity)	Dimension-less value (0-1)	Naturalness of habitats; Protected natural areas and features; Water features	Composite mapping	
	Expected trips to recreational sites (Flow)	Nº trips ha^{-1}	Population density grid; Recreation potential map	Distance analysis (including impedance function)	Paracchini et al. (2014); Zulian et al. (2014)
	Demand index (considering population density and distance to recreation sites)	Dimension-less value (0-5)	Population density grid; Recreation potential map	Spatial cross-tabulation	Threshold distance considered: 1 km (Stanners and Bourdeau, 1995)
	Population with low recreation opportunities (Unsatisfied demand)	Inhabitants ha^{-1}	Population density grid; Recreation potential map	Spatial extraction	
Air purifica-tion (NO_2)	NO_2 dry Deposition velocity (Capacity)	mm s^{-1}	Land cover dataset; Wind speed at 10 m height	Composite mapping	
	NO_2 removal flux (Flow)	kg ha^{-1} $year^{-1}$	Air quality monitoring stations data; Spatial predictors; Vegetation maps; Climatic and physiographical maps	Land use regression modeling (LUR)	Nowak et al. (2006); Beelen et al. (2013); Pistocchi et al. (2010); Zulian et al. (2014).
	Demand index (considering population density and NO_2 concentration)	Dimensionl ess value (0-5)	Spatial distribution of NO_2 annual average concentrations; Population density grid	Spatial cross-tabulation	NO_2 concentration limit value (annual average): 40 µg m^{-3} (EU, 2008)
	Population exposed to NO_2 concentration beyond limit (Unsatisfied demand)	Inhabitants ha^{-1}	Spatial distribution of NO_2 annual average concentrations; Population density grid	Spatial extraction	

4.2.5. Assessing urban-rural and landscape planning gradients

Urban-rural gradients have been used to analyze ecological patterns and processes in urban landscapes, including the consideration of ES indicators (Kroll et al., 2012 and Larondelle and Haase, 2013). Following these approaches, we computed urban-rural gradients of the capacity, flow, demand and unsatisfied demand of outdoor recreation and air purification using the resulting ES maps as described above. A 50-km concentric buffer with 1-km intervals was created around the city center of Barcelona (Catalunya square), covering almost all the BMR area. For each concentric ring, the average reclassified ES value (0–5 range) was calculated omitting null values. As pointed out by Kroll et al. (2012), urban-rural gradients imply a generalization of the spatial patterns existing in an urban region, but it is suitable approach to analyze major trends, relationships and variability between urban, suburban and rural areas in relation to ES provision and demand.

Assessing ES capacity, flow and demand maps in relation to current landscape planning instruments can provide relevant insights for land use policies. For example, it is possible to assess the level of protection of relevant ES providing areas in terms of capacity and flow and predict possible impacts to ES hotspots from future urbanization processes. Additionally, expected new areas of ES demand, and potentially unsatisfied demand, can be predicted from urban development areas. The intersect tool of ArcGIS v.10.1 (ESRI) was applied to extract the areas of ES capacity, flow, demand (only medium to very high values were considered, see **Fig. 4.4** and **Fig. 4.5** legends for the corresponding value ranges), and unsatisfied demand (all values were considered) allocated to the various landscape planning classes of the PTMB (see **Fig. 4.3** and **Table 4.1**).

4.3. Results

4.3.1. Spatial patterns of ecosystem service capacity, flow, and demand

Capacity, flow, demand and unsatisfied demand distribution maps for the ES outdoor recreation and air purification are shown in **Fig. 4.4** and **Fig. 4.5** respectively. Following Burkhard et al. (2014), maps show data classified into six categories, from no relevant to very high values. Classification is based on equal intervals in order to make the different classes and their values comparable with each other; except for population related indicators which required a manual classification (see break values in the corresponding map legends of **Fig. 4.4** and **Fig. 4.5**) in order to meaningfully represent the strong unevenness of urban densities (from urban sprawl to compact city).

Outdoor recreation capacity shows the highest values mainly in the forest areas located on the outskirts of the BMR (**Fig. 4.4a**). For example, the massif of Montseny, a natural park since 1977, located north-east of the study area, contains the most part of land classified as having high or very high recreational capacity (57.0%). Generally, these areas correspond to forest habitats, but a closer look also show high recreation capacity areas in aquatic habitats, such as the wetlands located in the delta of the Llobregat River, nearby the city of Barcelona. Air purification capacity values show a similar spatial pattern, yet the highest values are clearly circumscribed to the forest areas located north of the BMR (**Fig. 4.5a**). Medium capacity values in both ES are mainly distributed across the forest areas covering the coastal mountain range. For example, the periurban natural area of Collserola, a natural park since 2010, located at the core of the BMR, mostly presents medium values for both ES. Low to no relevant capacity areas generally correspond to urban and agricultural land covers. However, while lowest values in the case of outdoor recreation are clearly restricted to urban areas, the areas where air purification capacity is very low or no relevant include a broader range of land cover types, such as grassland or scrubland.

Unlike recreation capacity, the largest amount of high recreation flow values is to be found in the forest areas located in the surroundings of urban settlements (**Fig. 4.4b**). In general, riverine and coastal (e.g., beaches) ecosystems also show very high recreation flow values. Obviously, these results were expected as the flow assessment was restricted to close-to-home outdoor recreation trips for which distance to residential land is the explanatory variable. The case of air purification also shows higher flow values in periurban forest areas than other natural sites located in the hinterland, although the spatial transition is smoother as compared to recreation (**Fig. 4.5b**). Again, the natural parks of Collserola and Montseny illustrate these patterns clearly: the latter mainly contains very low to low flow values whereas the former shows mostly medium to very high values. The impact of traffic emissions over the spatial configuration of air purification flow is also noticeable on the maps, as forest areas located along the main roads have higher values in general terms. The lowest flow values for air purification are again located in urban and agricultural land, showing a similar pattern as for capacity.

As expected, the municipality of Barcelona and adjacent middle-size cities show the highest demand values in the BMR for both analyzed ES (**Fig. 4.4c** and **Fig. 4.5c**). This urban agglomeration is characterized by a compact urban form, very high population density and a relatively small share of inner green areas. The other middle-size cities, located both along the coastline and hinterland, show mostly middle to low demand values for air purification and low to high demand values for outdoor recreation. Smaller towns and sprawling urban areas mostly show very low to no relevant demand values. The impact of relevant ES providing areas, both in terms of capacity and flow, over demand distribution is also evident from the obtained maps, as residential land located close to these areas has generally lower values than more distant settlements.

Finally, results show that unsatisfied demand is circumscribed to the urban core of Barcelona and several middle-size cities (**Fig. 4.4d** and **Fig. 4.5d**). Unsatisfied demand for recreation includes a substantial portion of the city of Barcelona and other compact urban areas (163.54 km^2 in total) whereas unsatisfied demand for air purification is principally limited to the urban areas surrounding the main roads and streets of Barcelona and adjacent cities (only 46.63 km^2 in total) where NO_2 concentration is highest.

The urban-rural gradients of recreation and air purification for the BMR illustrate graphically the spatial patterns shown on the maps and described in the above paragraphs (see **Fig. 4.6**). The gradient for ES capacity is similar for both ES. The lowest values are in the first 5 km, in the Barcelona core city, and they present a gradual rise as we move away from the city center. In both cases, capacity shows a substantial increase after km 5 followed by a slight decrease after km 10–11. The periurban natural areas surrounding Barcelona (e.g., Collserola) followed by the urban and agricultural land located in the inland plains explain this pattern.

Flow gradient of air purification shows a pattern similar to the one observed for capacity, but after the first decrease (km 11–17) values follow a steady flat trend without any substantial increase. On the other hand, recreation flow shows a sharp increase in km 3 followed by a similar decline after km 5, illustrating the high flow values of periurban forests and other land covers located close to urban areas such as beaches. From km 8 until 20 a series of small picks and troughs precede a slow downward trend which corresponds to the gradual increase in the amount of recreational sites located far away from urban areas.

Demand gradients are also quite similar for both ES, showing highest values in the urban core area (1–5 km) followed by a decreasing trend as the distance to the city increases. Outdoor recreation values show a rapid decline after km 4 whereas air purification demand decreases more gradually. This result highlights that the impact of the urban core area upon ES demand is higher for recreation than for air purification in the BMR. Finally, unsatisfied demand gradients show a decreasing trend similar to the one observed for demand. The various peaks observable in mid to high distances for both ES can be attributed to the relative low amount of unsatisfied demand values, causing mean values to be more variable across the concentric rings than in the other three cases.

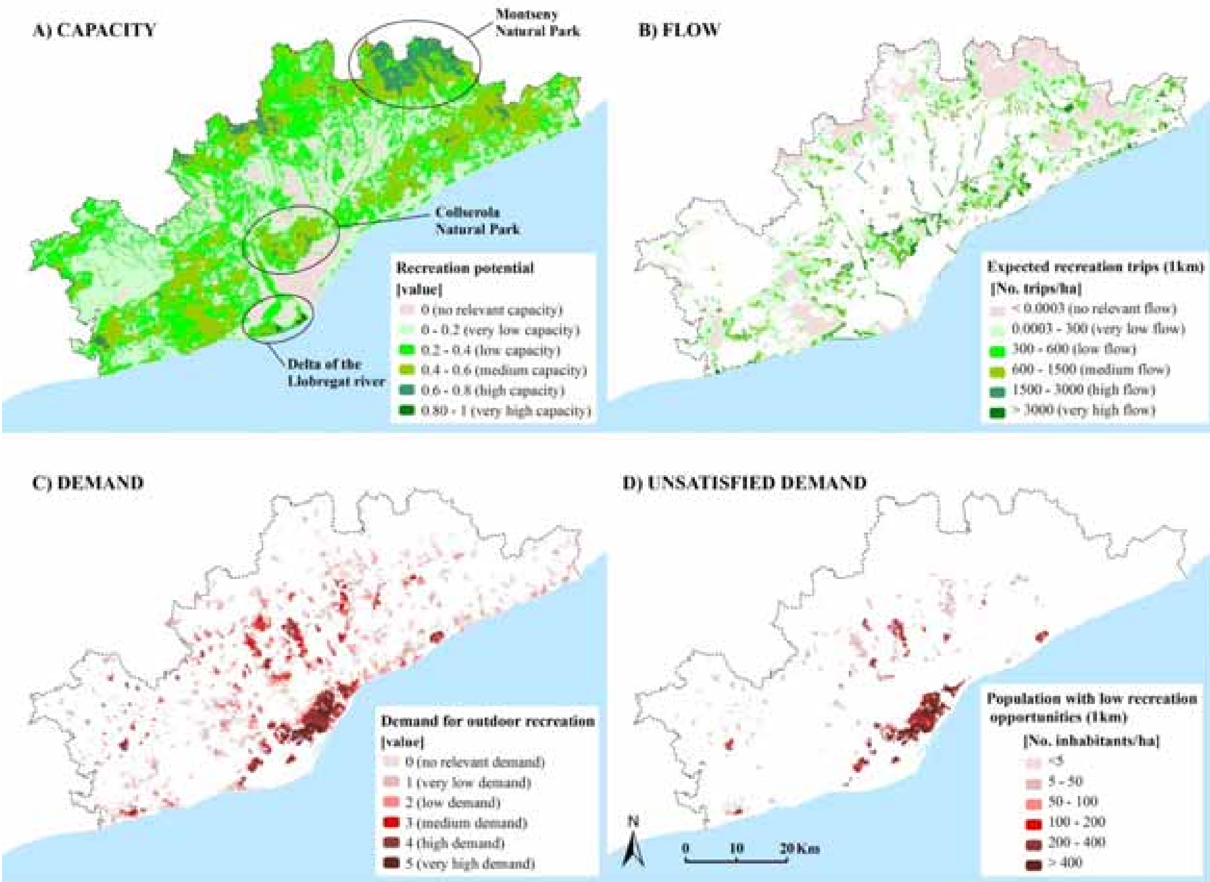

Fig. 4.4. Capacity, flow, demand and unsatisfied demand maps for the ES outdoor recreation in the BMR. See **Table 4.2** and **Appendix B** for data sources.

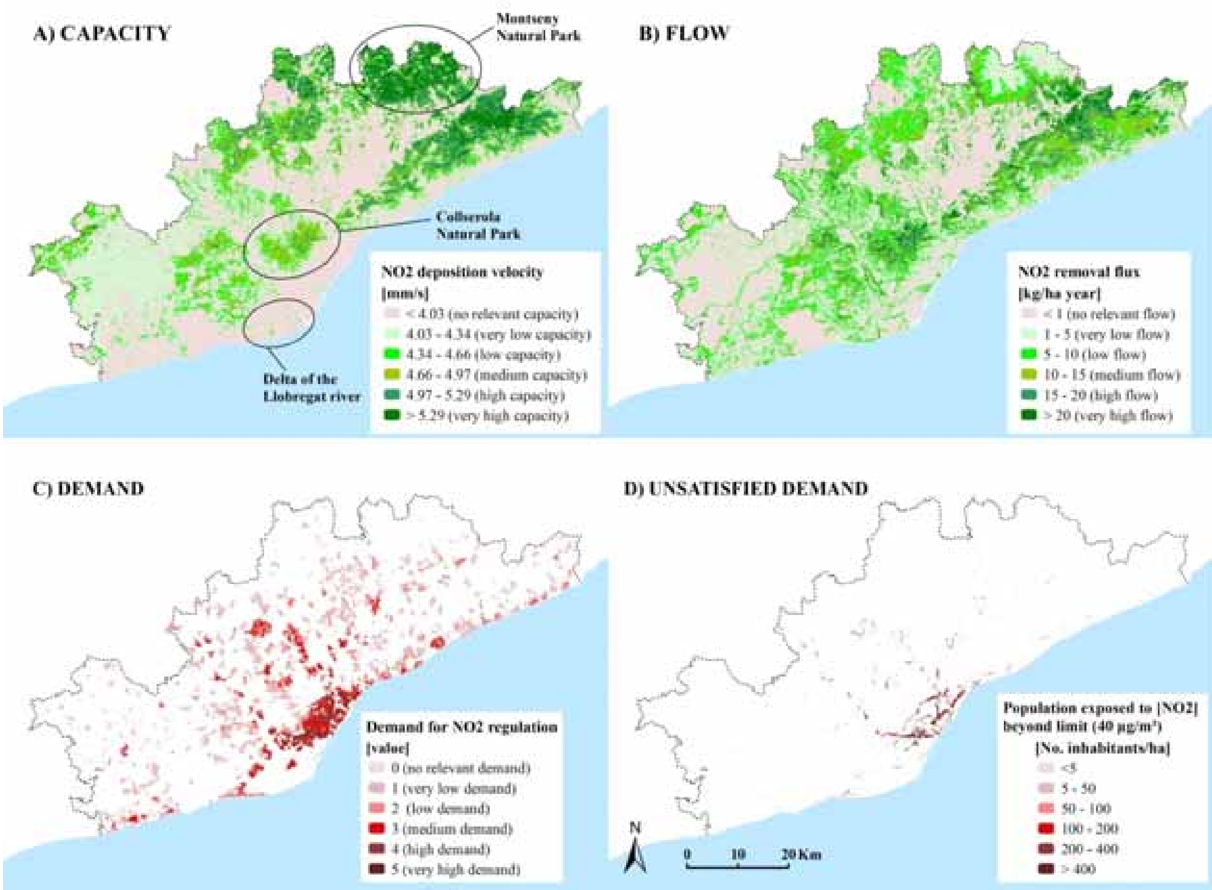

Fig. 4.5. Capacity, flow, demand and unsatisfied demand maps for the ES air purification in the BMR. See **Table 4.2** and **Appendix B** for data sources.

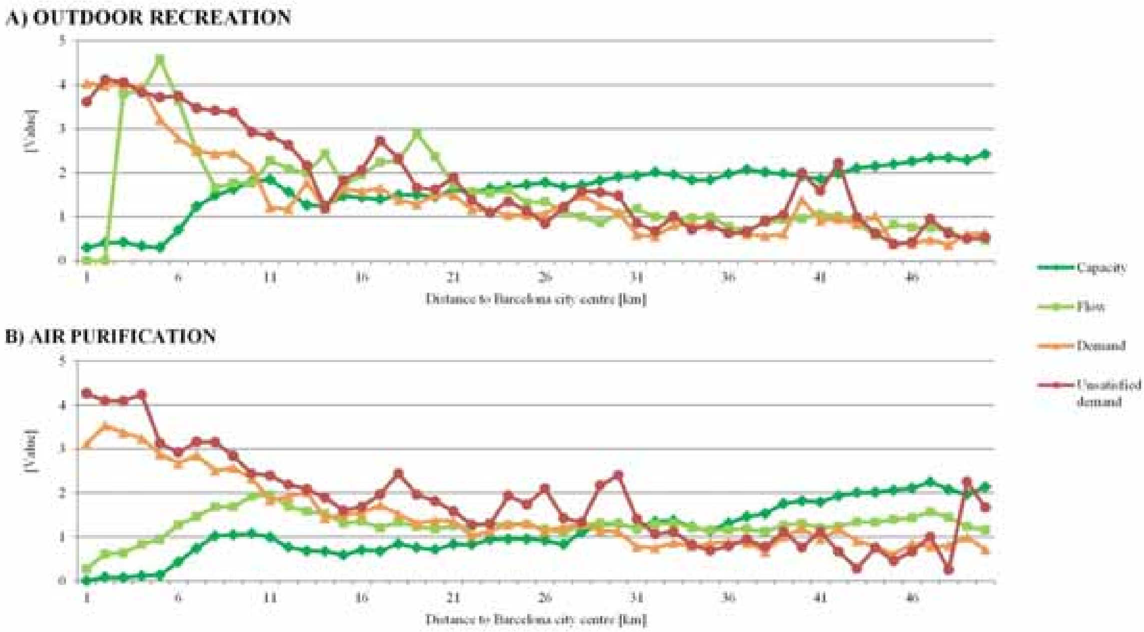

Fig. 4.6. Urban rural gradients (50 km) of the ES outdoor recreation and air purification for the BMR. Each point represents the average reclassified value (0-5 range) in the concentric ring at the respective distance from the Barcelona city center. Null values are not considered.

4.3.2. Landscape planning assessment

The total area of ES capacity, flow, demand and unsatisfied demand overlapping each of the landscape planning classes of the PTMB is shown in **Table 4.3**. Relevant areas for their capacity to provide ES are almost entirely classified as special protection (i.e., open areas planning system) since nearly 96% of the total area fall into this planning category for both ES. This result indicates that relevant ES capacity areas largely correspond to land covers that have already been protected by the PTMB due to their ecological and landscape values. Relevant ES flow areas mostly correspond to special protection land as well, yet a substantial share also corresponds to preventive protection, also open areas planning system, or the urban planning system. For example, in the case of air purification, 83.22 km^2 of flow areas (12.6%) fall into urban system categories or preventive protection whereas for ES capacity the total area is only 31.18 km^2 (4.0%). As observed on the maps, the highest flow values are generally located nearby or within suburban and urban land; hence possible impacts in terms of urbanization processes can be anticipated in these areas. As expected, demand and unsatisfied demand areas are mostly classified in the urban planning system.

Table 4.3. Total area of ES capacity, flow, demand (only medium to very high values, see **Fig. 4.4** and **Fig. 4.5** legends for the corresponding value ranges), and unsatisfied demand in relation to landscape planning classes (PTMB) (in km^2). Notes: special protection class includes special protection of the vineyard and urban classes include both urban consolidated land and development areas.

Mapped ES	ES component	Open Areas Planning System		Urban Planning System	
		Special protection	Preventive protection	Urban (green space)	Urban (built-up)
Outdoor recreation (everyday life)	Capacity	857.01 (96.0%)	7.76 (0.9%)	9.27 (1.0%)	19.00 (2.1%)
	Flow	142.04 (72.8%)	9.37 (4.8%)	10.63 (5.5%)	33.08 (17.0%)
	Demand	3.19 (2.4%)	1.23 (0.9%)	12.91 (9.6%)	117.31 (87.1%)
	Unsatisfied Demand	9.89 (6.1%)	5.67 (3.5%)	16.56 (10.1%)	131.41 (80.4%)
Air purification (NO_2)	Capacity	747.09 (96.0%)	10.18 (1.3%)	7.66 (1.0%)	13.35 (1.7%)
	Flow	579.14 (87.4%)	20.36 (3.1%)	19.53 (3.0%)	43.34 (6.5%)
	Demand	0.80 (0.7%)	0.62 (0.5%)	10.08 (8.7%)	103.98 (90.0%)
	Unsatisfied Demand	1.82 (3.9%)	1.77 (3.8%)	5.10 (10.9%)	37.94 (81.4%)

4.4. Discussion

4.4.1. Operationalization of the framework in the case study and policy implications

Our results indicate that the spatial patterns of ES capacity, flow and demand along the urban-rural gradient are similar for the two ES considered in the assessment. As expected, demand for both outdoor recreation and air purification is especially relevant in the urban core of the BMR. The actual use (i.e., flow) of both ES mainly takes place in the periurban and suburban green areas whereas the highest capacity values are mostly to be found in the protected natural areas located on the outskirts of the BMR. These findings suggest that there is a potential to increase ES capacity, and hence ES flow, in the periurban green areas of the BMR such as in the Collserola Natural Park through conservation planning and management. The current landscape planning instrument for the BMR (PTMB, 2010) classifies a substantial share of periurban areas as special protection land, thus the implementation of conservation practices intended to maintain and eventually enhance the current flow of ES could be supported. However, a considerable share of relevant ES flow areas is also located within the urban planning system or the preventive protection zoning category, indicating a potential risk of degradation due to future urbanization processes. Therefore, the revision of urban master plans such as the General Metropolitan Plan affecting the urban core of the BMR should ensure that relevant ES flows are maintained in these sensitive areas.

The assessment of ES mismatches between flow and demand shows that unsatisfied demand is mostly located in the urban core of the BMR and in several middle-size cities. We consider that planning and policy strategies intended to reconcile flow and demand at the local level should focus on different components of the framework depending on each ES.

For air purification, urban policies should focus on drivers of demand (i.e., air pollution concentrations). Previous studies (Baró et al., 2014 and Baró et al., 2015) show that average air quality improvements due to air pollution removal by vegetation is relatively low at the urban core, suggesting a limited effectiveness to address ES mismatches by increasing ES flow through strategies such as implementing tree-planting programs or selecting trees with high air pollution removal capacity. Moreover, factors such as vegetation configuration and climate conditions can limit the ability of vegetation to remove air pollutants, especially at the patch scale such as in street canyons (Vos et al., 2013). Therefore, policy interventions should focus on reducing and limiting traffic in certain areas, increasing public transport, incentivizing the use of none or low-emitting vehicles (e.g., bicycles and electric vehicles), and enhancing planning towards shorter commuting needs. The Air Quality Action Plan in the agglomeration of Barcelona (horizon 2020)[22] approved in 2014 by the Catalan Government is an important move towards the implementation of these policies in the case study area.

For outdoor recreation, different strategies could be put in place to reduce flow-demand spatial mismatches, which mainly focus on the capacity and flow aspects. For example, new protected areas and other conservation interventions such as green belts could be designed in the PTMB open areas system and urban master plans, reducing the risk of degradation due to urban sprawl processes as it occurred over recent decades (Catalán et al., 2008). An optimized fulfillment of outdoor recreation demand could also be fostered in core urban areas through strong planning and policy instruments intended to preserve existing green spaces and innovative ways to restore or create new ones. For example, the expansion of rooftop gardens in cities represents a promising solution in order to increase the delivery of a wide range of ES, including recreation opportunities (Orsini et al., 2014). The implementation of the Barcelona Green Infrastructure and Biodiversity Plan 2020 (Barcelona City Council, 2013) offers an important strategic policy framework with potential to substantially increase outdoor recreation opportunities in the municipality of Barcelona as it encourages the expansion of GI in all sorts of available land, including rooftops, inner courtyards, vacant plots, etc.

4.4.2. Methodological limitations and challenges for future research

As all data used in this study is likely available in other urban regions, it should be possible to extend this mapping approach elsewhere. Moreover, the framework can be potentially applied to other ES since capacity, flow and demand indicators have been suggested for all ES classes and groups (Burkhard et al., 2014 and Mononen et al., 2016). Based on previous applications of the ESTIMAP models (e.g., Maes et al., 2014 and Paracchini et al., 2014), we consider that the maps developed here for the two selected ES are sufficiently credible and salient for landscape and urban planning purposes in the

[22] The plan is available in English from www.airemes.net

case study area. However, several limitations and challenges for future research can be highlighted from our assessment.

One of the main limitations of this approach is that the mapping of ES demand and flow mostly relies on proxies (e.g., population density, air quality and distance) to indicate expected demand and use. Therefore, there is potential for error if the assumed causal variables are not actually good spatial predictors (Eigenbrod et al., 2010). However, there is a lack of empirical data which could be used for model validation. For example, visitor data in recreational sites is only partially available for some protected areas (e.g., see IERMB, 2008 for Collserola Natural Park). Air purification flow is based on a regression model using primary data on air pollution concentrations, but available NO_2 monitoring stations in the BMR are relatively few (n = 40), hence real heterogeneity in air pollution distribution is likely masked by the modeling process. The recreation capacity model depends strongly on expert knowledge (experts choose input data and scores), so validation or improvement could be realized through additional or complementary participatory methods as suggested below.

Improvement of results could be achieved by using other approaches and methods for mapping ES demand (see Wolff et al., 2015 for a review). For example, outdoor recreation demand indicators could be further refined by incorporating preferences, desires and expectations via household questionnaires, surveys or participatory mapping techniques (see also Vollmer and Grêt-Regamey, 2013, Burkhard et al., 2014, Brown and Fagerholm, 2015 and García-Nieto et al., 2015). These approaches can capture the diversity of demands for cultural ES and improve the spatial location of ES flows, but are usually resource intensive or site-specific (Wolff et al., 2015). Some European countries have collected data on people's recreational preferences through national visitor surveys (see Paracchini et al., 2014), but unfortunately we are not aware of any recreation survey at the regional or national level which covers the case study area. The demand approach for air purification, considering the exposure of population to air pollution levels, is consistent with most assessments of demand for regulating ES based on risk reduction (Wolff et al., 2015). However, a further refinement could be achieved by identifying and mapping specific risk groups such as children and elders, or by considering the areas where inhabitants practice outdoor activities and, therefore, where they can be exposed to air pollution (Sunyer et al., 2015).

Another issue not considered in the spatial models used here relates to ecological thresholds or tipping points (Andersen et al., 2009). An ecological threshold can be defined as a "point at which an (ecological) system experiences a qualitative change, mostly in an abrupt and discontinuous way" (Jax, 2014:1). It is often very difficult to determine when and under what conditions or pressures, ecosystems experience thresholds which can affect their ability to provide ES (Gómez-Baggethun et al., 2011). In the case of air purification, high pollutant concentrations can severely damage vegetation or lead to stomatal closure, reducing air pollution removal capacity and consequently flow (Robinson et al., 1998 and Escobedo and Nowak, 2009). In the case of outdoor recreation, the threshold is probably related to congestion. A very high number of visitors in a given recreational area, at the same time or progressively during a persistent period of time, might lead to a deterioration of the recreational experience and to the degradation of the ecosystem itself, hence jeopardizing its ability to provide this ES (Lynn and Brown 2003). The visitor carrying capacity of a given area could be defined based on expert knowledge and/or participatory approaches (Schröter et al., 2014). **Fig. 4.7** provides an illustrative outline of hypothetic patterns of ES capacity, flow and demand under increasing pressures considering ecological thresholds. In the case of air purification, capacity and flow would likely experience an abrupt decrease after the ecological threshold while for outdoor recreation the change would probably be more gradual. Moreover, air purification flow cannot exceed capacity because of biophysical constraints, but recreation flow can indeed surpass capacity and ultimately trigger its decline due to congestion (Schröter et al., 2014).

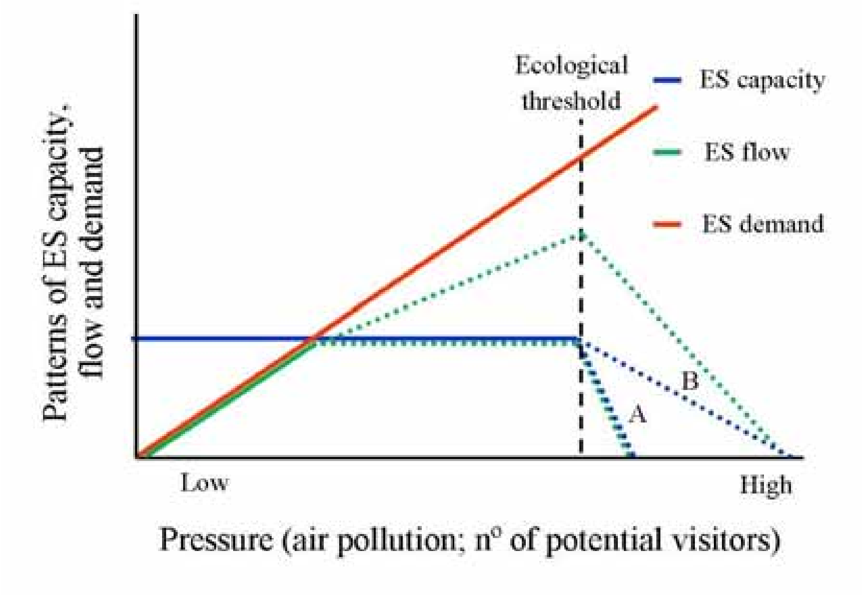

Fig. 4.7. Outline of hypothetic patterns of ES capacity, flow and demand under increasing pressures considering ecological thresholds. In the case of air purification, capacity and flow would likely experience an abrupt decrease after the ecological threshold (case A) while for outdoor recreation the change would probably be more gradual (case B). Further, air purification flow cannot exceed capacity because of biophysical constraints, but recreation flow can indeed surpass capacity and ultimately trigger its decline due to congestion.

The issue of the spatial scale of ES capacity, flow and demand maps (Geijzendorffer and Roche, 2014) also arises from this research. Our spatial results reflect that the actual use or experience of the two ES analyzed highly depends on the proximity between ES providing areas and benefiting areas (Syrbe and Walz, 2012), leading to relevant unsatisfied demands which are mainly located at the urban core of the BMR. Therefore, we argue that both the regional and local scales should be considered in these assessments in order to comprehensively support planning and policy (Scholes et al., 2013). For instance, a more detailed resolution could take into account small ES providing areas which are often overlooked in regional assessments. These areas might have a relevant impact in terms of ES flow and unsatisfied demand in the urban core. Moreover, the proposed interventions for both ES could be much more accurately designed in local scale studies. However, the lack of fine resolution spatial data for the appropriate quantification of ES capacity, flow and demand indicators is probably a major challenge for this type of analyses (Derkzen et al., 2015). This issue also calls for a strong institutional coordination between local and regional authorities dealing with urban and environmental policy and for the harmonization of planning instruments at different scales.

4.5. Conclusions

We advanced a spatial application of the ES capacity, flow and demand framework and tested its usefulness for landscape and urban planning in a case study. Our results suggest that the current landscape planning instrument for the BMR (PTMB, 2010) could foster the enhancement of relevant ES providing areas (i.e., ES capacity), but at the same time it might lead to degradation of some important ES flows due to possible future urban developments.

We argue that planning and policy strategies intended to reconcile flow and demand at the local level should focus on different components of the framework depending on each ES. For air purification, urban policies should focus on decreasing demand drivers (i.e., air pollution concentrations), whereas an optimized fulfillment of outdoor recreation demand could be fostered in core urban areas mainly through strong planning instruments intended to maintain and foster ES capacity and flow, for example by preserving and enhancing existing green spaces and restoring or creating new ones. A promising strategy could consist of a policy mix combining prescriptive policy regulations (e.g., enforcement of caps and stricter GI ratios) and economic incentives (e.g., environmental taxes, subsidies and payments), accompanied by awareness rising campaigns on the links between ecosystems and human well-being.

From our study, we contend that the mapping of ES capacity, flow and demand can contribute to the successful integration of the ES approach in landscape and urban planning because it provides a comprehensive picture of the ES delivery process, considering both ecological and social underlying factors. However, we identified three main issues that should be better addressed in this type of assessments: (1) improvement of ES demand indicators using participatory methods (i.e., incorporating different preferences or expectations); (2) integration of ecological thresholds into the analysis and models; and (3) use of a multi-scale approach that covers both the local and regional planning levels and cross-scale interactions between them.

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CHAPTER V

Ecosystem service bundles from a supply-demand approach: Implications for landscape planning and management in an urban region

Abstract

A key challenge of landscape planning and management is coping with multiple ecosystem service (ES) potentials and demands in complex socio-ecological systems such as urban regions. However, few studies have analyzed both the supply and demand sides of ES bundles, i.e., sets of associated ES that are repeatedly supplied or demanded together across time or space, from an integrated perspective. This paper advances a framework to identify, map and assess ES bundles from a supply-demand approach along the urban-rural gradient to inform landscape planning and management. The framework is applied to the Barcelona metropolitan region, Spain, covering five ES (food provision; global climate regulation; air purification; erosion control; and outdoor recreation) and eleven spatial indicators. Each indicator was quantified and mapped at the municipal level (n = 164) using available data sources and combining qualitative and quantitative methods, from expert-based matrices to process-based biophysical models. Our results show significant associations among ES, both at the supply and demand sides. Negative correlations were revealed between food provision from crops and the supply of all regulating ES and outdoor recreation, and also between demand for erosion control and demands for all other ES. Further, we identified five distinct ES supply - demand bundle types and characterized them based on the specific supply-demand relationships and the main land uses taking place in each type. Based on our findings, we call for combining land sharing strategies in urban and agricultural areas to increase landscape multifunctionality, and concurrently, assure the conservation of large periurban forest areas that are critical for delivering a wide range of ES demanded by the local urban population.

Keywords: Barcelona metropolitan region; ecosystem service mismatch; green infrastructure; spatial analysis; urban-rural gradient.

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5.1. Introduction

A key challenge of landscape planning and management is coping with multiple ecosystem service (ES) potentials and demands in complex socio-ecological systems. The last decade has seen increasing attempts to assess the relationships among different ES through the concept of ES bundles (e.g., Chan et al., 2006; Raudsepp-Hearne et al., 2010; Maes et al., 2012; Martín-López et al., 2012; García-Nieto et al., 2013; Derkzen et al., 2015). An ES bundle has been defined as "set of associated ES that repeatedly appear together across time or space" (Raudsepp-Hearne et al., 2010:5242). A key advantage of the ES bundle approach is that it allows to assess potential synergies and trade-offs by analyzing how different ES in a given area are positively or negatively associated (Bennett et al., 2009).

Assessment of ES bundles has been mostly applied to the supply side of ES (i.e., the ecosystem's potential to deliver ES based on biophysical properties and functions *sensu* Villamagna et al., 2013) using a spatially explicit approach (e.g., Chan et al., 2006; Raudsepp-Hearne et al., 2010; Haase, et al., 2012; Maes et al., 2012; Derkzen et al., 2015). In contrast, studies assessing ES bundles from a demand perspective (i.e., considering the amount of ES required or desired by society *sensu* Villamagna et al., 2013) have mostly focused on determining different socio-cultural values (e.g., Martín-López et al., 2012; Iniesta-Arandia et al., 2014), but very few have produced spatially explicit information. The reason behind this disparity probably relates to the lack of a clear methodological framework for quantifying and mapping ES demand (Wolff et al., 2015) in contrast to ES supply (Egoh et al., 2012; Crossman et al., 2013; Malinga et al., 2015).

Even fewer studies have analyzed both the supply and demand sides of ES bundles, i.e., sets of associated ES that are repeatedly supplied or demanded together across time or space, from an integrated perspective (but see García-Nieto et al., 2013; Castro et al., 2014). Yet, such approach could have important advantages for sustainable landscape planning and management in complex socio-ecological systems. These include: (1) enhanced capacity to address green infrastructure (GI[23]) planning, i.e., the identification

[23] GI is a boundary concept with various conceptual meanings (Wright, 2011), but here we follow the EU GI strategy definition: "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" (EC, 2013)

of existing key ecosystems for ES delivery (Maes et al., 2015); (2) prioritization of key areas for establishing GI projects due to existing mismatches between supply and demand of ES from a bundle perspective (García-Nieto et al., 2013); and (3) better understanding of potential trade-offs and synergies between ES considering both ecosystem's processes and societal needs (Castro et al., 2014).

Mapping and assessing ES bundles considering both the supply and the demand sides can be particularly relevant in urban regions, given their high levels of population density and pressure on available land. Assessing ES bundles in these areas can shed light on mismatches, trade-offs and synergies potentially driven by urban development processes. Even if urban areas benefit from the appropriation of vast ES providing areas beyond their boundaries (Rees, 1992; Folke et al., 1997), the local supply of ES can contribute to cope with a variety of challenges, including protection from climate extremes (e.g., moderation of heatwaves and floods), improvement of environmental quality (e.g., air pollution abatement) and healthier life styles (e.g., opportunities for recreation and relaxation) (Bolund and Hunhammar, 1999; Gómez-Baggethun et al., 2013; Haase et al., 2014).

The aim of this paper is to advance a framework to identify, map and assess ES bundles from a supply-demand perspective in order to support landscape planning, management, and decision-making in urban regions. Our framework builds on previous methodological approaches (Mouchet et al., 2014) and consists of five main steps: (1) selection, quantification and mapping of suitable ES indicators (both at the supply and demand sides); (2) assessment of spatial ES associations at both sides; (3) identification of relevant ES supply-demand bundle types; (4) analysis of ES spatial patterns along the urban-rural gradient and along a gradient of management or planning strategies; and (5) understanding of the spatial characteristics of ES bundles and their relevance for landscape planning and management. We used the Barcelona metropolitan region, Spain, as case study area, considering a set of five ES and eleven indicators (six at the supply side and five at the demand side).

5.2. Material and Methods

5.2.1. Case study area

Our research was conducted in the Barcelona metropolitan region (BMR), north-east of Spain (**Fig. 5.1A**). The BMR (5.03 million inhabitants and 3,244 km^2, Statistical Institute of Catalonia, year 2015) is a regional planning area covering 164 municipalities. Its urban core is constituted by the municipality of Barcelona (1.61 million inhabitants; **Fig. 5.1D**) and several adjacent middle-size cities. Distribution of land uses and covers in the BMR is shaped by its physical geography (**Fig. 5.1B** and **5.1C**). Two systems of mountain ranges (Catalan Coastal Range and Catalan Pre-Coastal Range) run parallel to the Mediterranean Sea coast, mostly covered by Mediterranean forests of Pine and Holm Oak trees, scrubland and grassland. Prominent examples of these ecosystems with high value for ES delivery include protected areas such as the Montseny massif (Pre-Coastal Range) which has the highest peak in the BMR (1705 m), or the Collserola massif (Coastal Range) which is virtually enclosed by urban land (**Fig. 5.1B**). In contrast, coastal and inland plains are mostly covered by urban and agricultural land. For instance, the Llobregat river delta is heavily sealed by urban land and transport infrastructure (e.g., the Barcelona airport), but it still preserves valuable agricultural and wetland areas. The Penedès area (west of the BMR) is an important wine-growing region.

The BMR is one of the regional planning areas of the 'General Territorial Plan of Catalonia' (PTGC) (PTGC, 1995), the uppermost strategic landscape planning instrument in the region of Catalonia. The 'Territorial Metropolitan Plan of Barcelona' (PTMB) was developed following PTGC's guidelines and approved in 2010 by the Government of Catalonia (PTMB, 2010). The PTMB establishes two main planning categories (called "systems") for land use regulation in the BMR: open areas and urban land (**Fig. 5.1D**). The open areas planning system (2405 km^2, 74.1% of the BMR) regulates the land protected from urbanization and includes three planning units: (1) Special protection areas (2032 km^2), which consist of land that is highly protected for its ecological and agricultural values, including Natura 2000 sites and other protected areas; (2) Special protection of vineyards (230 km^2), consisting of highly protected land for its landscape and agricultural values for the wine sector; and (3) Preventive protection areas (143 km²), for urban-rural transitional areas where urban development is restricted, except in certain circumstances. The urban planning system (840 km², 25.9% of the BMR) regulates consolidated built-up land (635 km²) and defines strategies for urban expansion by the delimitation of development areas (205 km²) that can be subsequently refined by municipalities through so-called urban master plans.

We contend that the BMR, as a complex socio-ecological system, is a suited testing area for the purpose of this research. The manifest heterogeneous spatial distributions of relevant ES providing areas (Mediterranean forests, agroecosystems, etc.) and potential beneficiaries along the urban-rural gradient can provide relevant insights for the integration of a GI perspective into future landscape planning and management instruments.

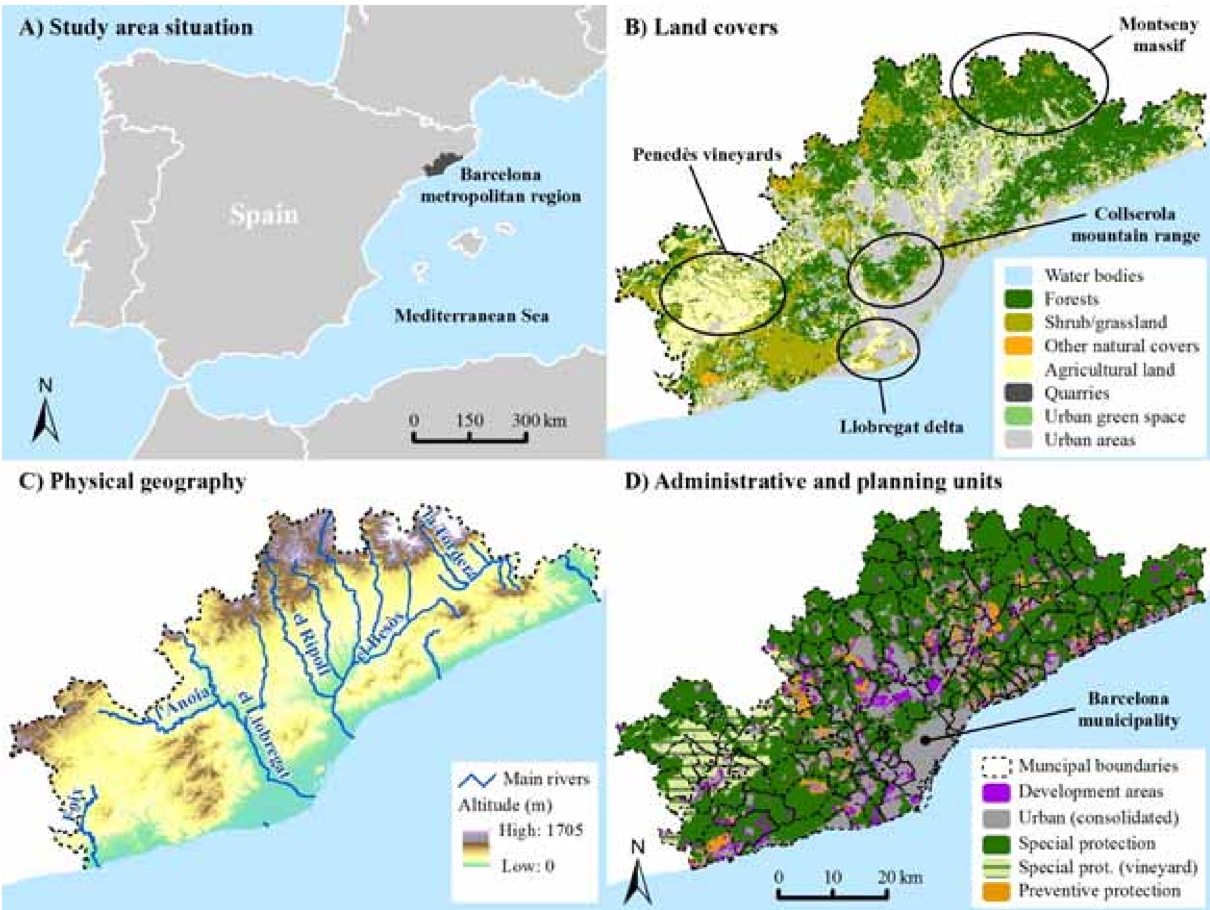

Fig. 5.1. Biophysical and administrative maps of the case study area (BMR). Own elaboration based on various spatial datasets provided by the Catalan Government and the Catalan Cartographic and Geological Institute (ICGC).

5.2.2. Selection, quantification and mapping of ecosystem service indicators

Five ES were assessed at the study area: (1) food provision; (2) global climate regulation; (3) air purification; (4) erosion control; and (5) outdoor recreation. The selection of these ES was based on three main criteria: (1) their relevance to the BMR, mainly in terms of expected demand; (2) consideration of a representative ES sample covering at least one ES from the three main ES categories of the CICES[24] classification (i.e., provisioning, regulating and maintenance, cultural services); and (3) the availability of data for both ES supply and demand sides. We consider that this selection satisfies the research goals and provides a sufficient ground for the discussion of possible relevant policy and planning implications.

For each ES, an indicator (based on direct or proxy data) was defined, measured and mapped, both at the supply and demand sides. In the case of food provision, two indicators of supply were used: crop and livestock production. Hence a total of eleven indicators were included in the analysis. **Appendix C** (Supplementary information) describes in detail the quantification and mapping methods (and provides the corresponding references) used for each ES indicator. **Table 5.1** provides an overview of the ES indicators and a brief description of main data sources. Each indicator was quantified using the most recent available datasets (typically from years 2011 to 2013). All the required geoprocessing operations were carried out using ArcGIS v.10 (ESRI) or GRASS GIS v. 7.0 (GRASS Development Team).

ES supply indicators refer here to the ecosystems' "capacity" to deliver ES (i.e. "the ecosystem's potential to deliver services based on biophysical and social properties and functions") rather to the "flow" of ES (i.e. "the actual production or use of the service") (Villamagna et al., 2013:116). The reason for using this approach is that we are interested in the long-term perspective and hence in measuring the potential of the study area in terms of ES provision regardless of whether this is actually used or experienced in the present. For example, the proxy indicator for air purification (NO_2 dry deposition velocity) indicates the capacity of ecosystems to filter air pollution, but not the actual pollutant removal. In the case of provisioning indicators (both crop and

[24] CICES (Common International Classification of Ecosystem Services) latest version is available from: http://cices.eu/

livestock production), it could be assumed that most part of the production is consumed, yet food loss and food waste represents an important problem worldwide (FAO, 2011). Similarly, all carbon sequestration ecosystems' capacity constitutes a flow because global carbon emissions are clearly exceeding actual sequestration rates (Schröter et al., 2014). In the case of erosion control, a biophysical indicator could not be calculated due to data limitations, so we applied an expert-based matrix model using land covers as spatial data following Burkhard et al. (2012; 2014). The dimensionless index for outdoor recreation is based on a composite model (Paracchini et al., 2014; Zulian et al., 2014) that estimates the capacity of ecosystems to provide recreation opportunities based on their degree of naturalness, nature protection, and presence of water (see **Appendix C** in Supplementary information for further details).

Despite there is a varying understanding of the concept of ES demand (see Wolff et al., 2015), ES demand refers here to "the amount or level of ES required or desired by society" (Villamagna et al., 2013:116). Following previous studies (e.g., Kroll et al., 2012), demand for food provision was mapped using human population density as proxy indicator. Therefore, we located demand at the site of the final beneficiary or end-consumer (Burkhard et al., 2014). We did not combine population density with average consumption rates because the focus of the research is not on self-sufficiency or balance analysis but on the assessment of the spatial patterns from a bundle approach. Demand indicators for regulating ES indicate the magnitude of pressures or inputs needing regulation (air pollution levels for air purification, carbon emissions for climate regulation and land erodibility for erosion control). This risk reduction approach is commonly applied in the ES literature (Wolff et al., 2015) and assumes that demand is oriented toward a reduction of the indicator values (Burkhard et al., 2014). A particular case is again climate regulation because the demand for this ES is global and hence could be distributed equally over the world surface (Syrbe and Walz, 2012). Yet, carbon emissions are commonly used as a proxy at lower scales (e.g., Baró et al., 2015; Zhao and Sander, 2015) as a way to indicate subglobal contributions to the need for this regulating ES. Finally, demand for experience-based cultural ES such as outdoor recreation can be estimated through the number of people wanting to experience the ES and their feasibility to do so in terms of accessibility to recreational sites (Paracchini et al., 2014; Ala-Hulkko et al., 2016). Following this rationale, here we mapped outdoor recreation demand based on the availability of recreational sites close to people's home and population density assuming that all inhabitants in the BMR have similar desires in terms of everyday life outdoor recreational opportunities (see **Appendix C** in Supplementary information for details).

Table 5.1. Overview of the ES indicators, quantification units and main data sources used in the BMR case study. Full references for data sources are provided in **Appendix C** (Supplementary information).

ES	Indicator	Quantification unit	Main data sources
Food provision (provision-ing)	Crop production (supply)	kg edible crop production ha^{-1} $year^{-1}$	Agriculture yield statistical data (year 2013) Regional land cover dataset (year 2012)
	Livestock production (supply)	Livestock units km^{-2} $year^{-1}$	Agriculture census data (year 2009)
	Population density (demand)	Inhabitants ha^{-1}	Population census tracts dataset (year 2011)
Global climate regulation (regulating)	Carbon sequestration (supply)	kg C ha^{-1} $year^{-1}$	National forest inventories data (years 1990 and 2001) Various regional spatial datasets (different sources)
	Carbon emissions (demand)	kg C ha^{-1} $year^{-1}$	Municipal Sustainable Energy Action Plans (SEAPs) (year 2012)
Air purification (regulating)	NO_2 dry deposition velocity (supply)	mm s^{-1} ha^{-1}	Regional land covers dataset (year 2012) Average wind speed data (Regional environment database)
	NO_2 concentration levels (demand)	µg NO_2 m^{-3} (annual mean)	Air quality data from BMR monitoring stations (year 2013) Various regional spatial datasets (different sources)
Erosion control (regulating)	Erosion control capacity (supply)	Dimensionless (0-5)	Expert-based data (Burkhard et al., 2012) Regional land covers dataset (year 2013)
	Land erodability (demand)	Dimensionless (0-3)	Land erodability dataset (SITxell - Geographic Information System for the Network of Open Areas in the province of Barcelona)
Outdoor recreation (cultural)	Recreational potential index (supply)	Dimensionless (0-1)	Various regional spatial datasets on habitat naturalness, protected natural areas and water features (different sources)
	Recreational demand index (demand)	Dimensionless (0-5)	Population census tracts dataset (year 2011) Various regional spatial datasets (different sources)

5.2.3. Analysis of spatial patterns and associations between ecosystem services

Individual ES indicators were mapped to visualize and compare their spatial patterns across the case study area. Although the spatial resolution of some data sources was relatively high (e.g., the regional land cover dataset was developed at a scale of 1:50,000), we used municipalities (n = 164) as the main spatial unit of analysis due to several reasons: (1) urban policies related to ES and GI in the BMR are usually implemented at the municipal level (e.g., Barcelona City Council, 2013); (2) the municipality is the smallest unit at which livestock census or carbon emissions data are available in the BMR; and (3) statistical computing limitations when dealing with data matrices derived from high resolution rasters. Therefore, ES indicators were quantified for each municipality calculating average values in case the original spatial unit was smaller and normalized by area to enable comparison across municipalities of different size. Further, ES indicators were standardized where necessary in a 0-1 range using minimum and maximum values, so that correlation or cluster analyses could be performed.

As a first step, spatial autocorrelation analysis was carried out for each ES indicator using Global Moran's I with Rook contiguity in ArcGIS v 10 (ESRI). We considered the spatial pattern to be significantly clustered if the obtained z-score (standard deviation) was higher than 1.96 (95% confidence level).

The analysis of ES associations and bundles types was carried out following Mouchet et al. (2014) and using R statistical software (R Core Team, 2015) and ArcGIS v10 (ESRI). First, associations between pairs of ES were detected using Pearson parametric correlation test both at the supply (fifteen pairs) and demand (ten pairs) sides. Overlap analysis was also applied in order to spatially visualize possible ES supply and demand "hotspots" (areas of high delivery or demand) and "coldspots" (areas of low delivery' or demand), as well as supply – demand spatial congruency. Aggregated ES supply and demand values were calculated using a simple unweighted summation of the standardized indicators' values at the municipality level. In addition, we mapped the "ES richness" at each municipal unit. To do so, we accounted for the number of ES supplied or demanded in a substantial degree (a substantial supply or demand was assumed if the indicator value was equal or higher than the average). In a second stage, we defined

ES supply - demand bundle types using cluster analysis. We classified municipalities into clusters based on similar combinations of both ES supply and demand values (i.e., ES supply - demand bundle types) using *K*-means clustering algorithm which minimizes within-group variability. The appropriate number of clusters was determined by analyzing the meaningfulness of different clustering outputs with the support of dendrograms and scree plots. The final ES supply - demand bundle types were visualized using star plots (showing average indicator values per cluster) and mapped in ArcGIS to show their spatial patterns. A principal component analysis (PCA) was also applied to analyze the relationships between the ES supply and demand indicators and the various land planning strategies (i.e., planning classes of the PTMB, 2010). Land planning strategies were included in the PCA as the area percentage of each class per municipality.

The assessment of ES spatial patterns was complemented using rural-urban gradients analysis. Following previous contribution to this research area (Kroll et al., 2012; Larondelle and Haase, 2013), we computed rural-urban gradients of the ES supply and demand indicators considered in the analysis. A 50-km concentric buffer with 1-km intervals was created around the city center of Barcelona (Catalunya square), covering almost all the BMR area. For each concentric ring, the average ES value was calculated omitting null values. In order to improve visualization of the gradients, the analysis was not performed at the municipal level but at the pixel level (using the ES data resampled at a spatial resolution of 100 m) and it was based on a reclassification of the ES values in five classes (0-4) using quintiles.

5.3. Results

5.3.1. Ecosystem service supply: spatial patterns and associations

Spatial autocorrelation results show that all ES supply indicators were spatially clustered on the case study area. The obtained z-scores (**Fig. 5.2**) indicate that there is less than 1% likelihood that the individual spatial patterns could be the result of random chance. Geographic distributions of the six ES supply indicators (**Fig. 5.2**) revealed clear similarities and dissimilarities among them. On the one hand, potential supply of regulating ES and outdoor recreation was highest in the mountainous landscapes located at the north and north-east of the BMR, mostly covered by Mediterranean forests. On the other hand, the two food production indicators followed very distinct patterns. In the case of crop production, highest values were mostly found in the flat areas of the wine-making county of Penedès (at the west side of the BMR) and in other agricultural areas located along the coast (especially in the Llobregat river delta). Livestock production was mostly clumped in low-density population municipalities located at the hinterland plains, especially at the north and west of the BMR.

The correlation results between pairs of ES supply indicators are shown in **Table 5.2**. All pairs were significantly correlated, except those including livestock production. Associations among regulating ES and outdoor recreation were highly positively correlated (Pearson coefficient > 0.5). Crop production was moderately negatively correlated with all regulating services (Pearson coefficient < -0.3 and > -0.5) and weakly negatively correlated with outdoor recreation (Pearson coefficient > -0.3).

Overlap analysis confirmed that the ES hotspots, i.e., the most multifunctional and rich areas in terms of ES provision are located at the north and north-east of the BMR (**Fig. 5.3**), including the municipalities with a high share of forest habitats and containing small settlements. In contrast, highly urbanized municipalities (e.g., in the urban core) and those mostly covered by agricultural land showed the lowest aggregated values for ES supply and none or few ES provided in a relevant amount (value ≥ mean), indicating expected ES coldspots (**Fig. 5.3**).

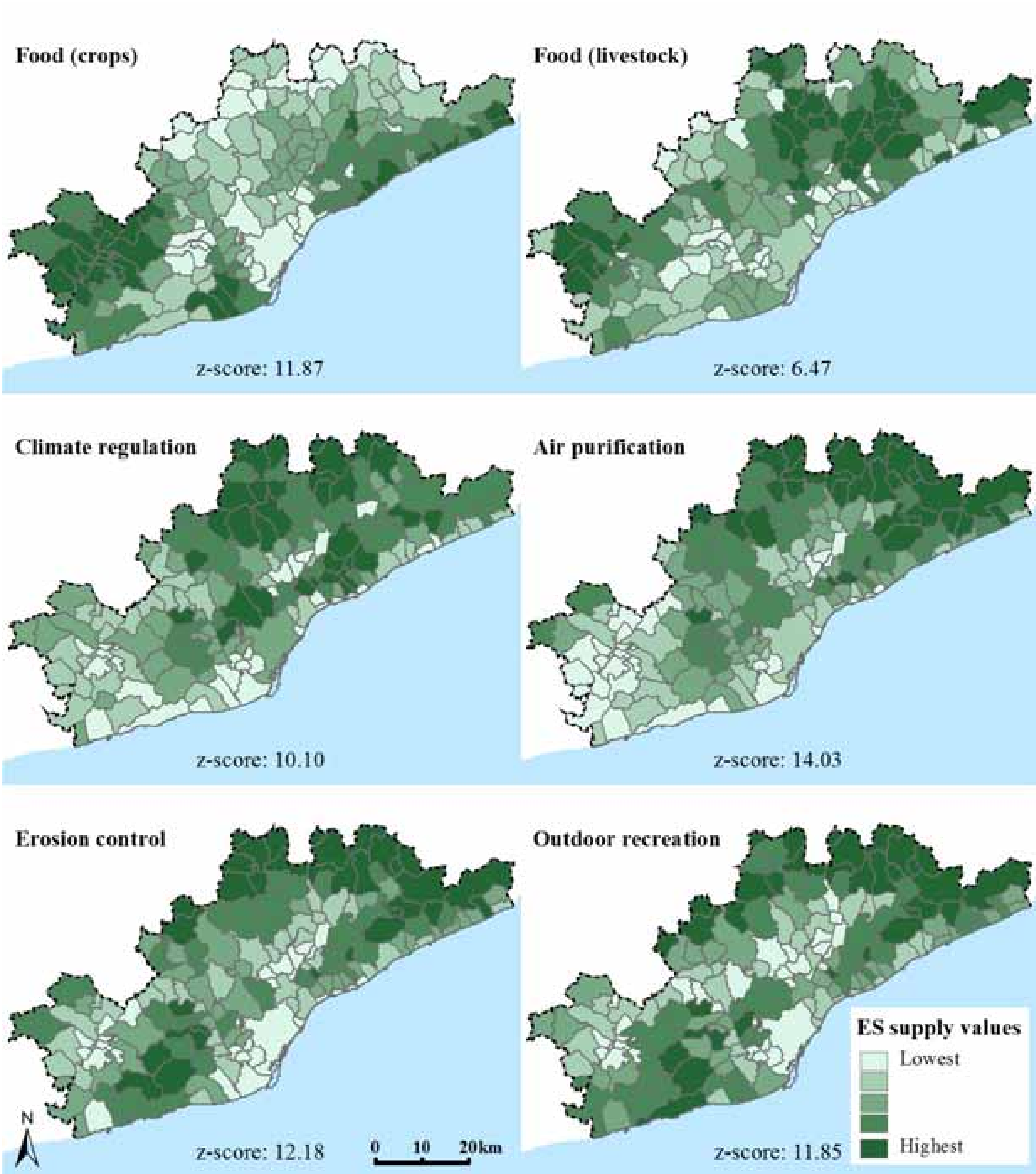

Fig. 5.2. Spatial patterns of the six ES supply indicators shown at the municipality level. Indicator values are classified in quintiles. All ES indicators are significantly clustered in space (z-score > 1.96).

Table 5.2. Significant correlations (Pearson parametric test) between pairs of ES supply indicators (*$P < 0.001$; **$P < 0.0001$).

	Food (crops)	Food (livestock)	Climate regulation	Air purification	Erosion control	Outdoor recreation
Food (crops)	1					
Food (livestock)	0.01	1				
Climate regulation	-0.36**	0.01	1			
Air purification	-0.38**	0.04	0.75**	1		
Erosion control	-0.41**	0.01	0.68**	0.86**	1	
Outdoor recreation	-0.28*	-0.09	0.65**	0.79**	0.86**	1

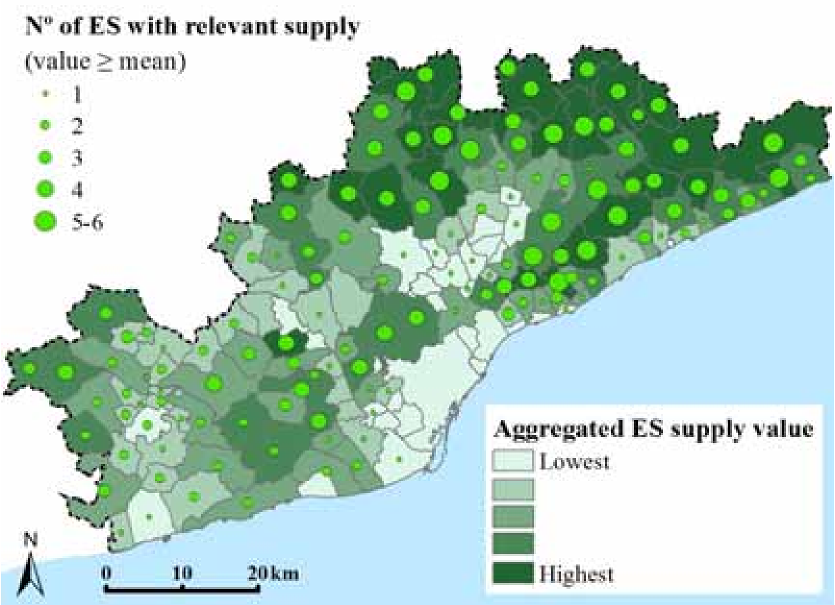

Fig. 5.3. Expected ES supply hotspots and coldspots in the BMR based on the aggregated ES supply value and the ES richness, i.e., number of ES with relevant supply (value ≥ mean) shown at the municipality level. Aggregated ES supply values are classified in quintiles.

5.3.2. Ecosystem service demand: spatial patterns and associations

All indicators of ES demand also showed a significant clustered spatial pattern on the BMR at the individual level (z-score > 1.96; **Fig. 5.4**). Furthermore, all indicators except erosion control displayed a similar spatial distribution characterized by highest values at the urban core (Barcelona and adjacent cities) and a clearly decreasing gradient towards the outskirts of the BMR (except for some municipalities, especially along the coastline). In contrast, demand for erosion control corresponded as expected mostly with the hilly areas located at the center and north-east of the BMR **(Fig. 5.4**).

All the ten possible pairwise associations between ES demand indicators were found to be significantly correlated (**Table 5.3**). Associations among food production, climate regulation, air purification and outdoor recreation were highly positively correlated (Pearson coefficient > 0.5). Erosion control was moderately negatively correlated with food production, climate regulation and outdoor recreation (Pearson coefficient < -0.3 and > -0.5) and weakly negatively correlated with air purification (Pearson coefficient > -0.3).

As expected, overlap analysis showed that the aggregated ES demand values were highest in the urban core of the BMR (**Fig. 5.5**). Additionally, this area presented the highest diversity of demands: generally four or five ES were demanded in a relevant degree (indicator value ≥ mean). Coldspots at the demand side were found mainly at the north and west of the BMR where municipalities are characterized by low population densities and a high share of agricultural or forest land covers.

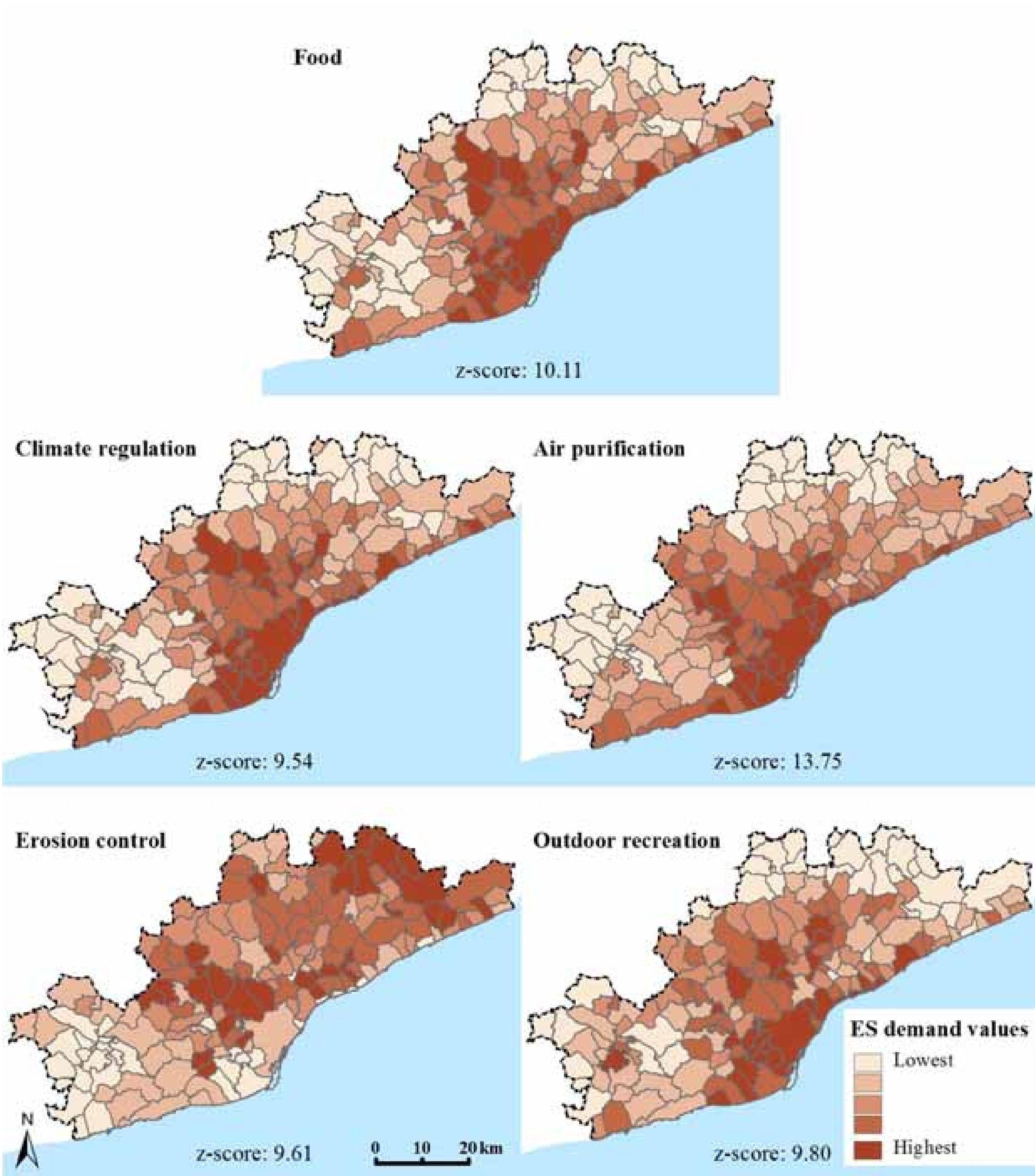

Fig. 5.4. Spatial patterns of the five ES demand indicators shown at the municipality level. Indicator values are classified in quintiles. All ES indicators are significantly clustered in space (z-score > 1.96).

Table 5.3. Significant correlations (Pearson parametric test) between pairs of ES demand indicators (*$P < 0.001$; **$P < 0.0001$).

	Food (population)	**Climate regulation**	**Air purification**	**Erosion control**	**Outdoor recreation**
Food (population)	1				
Climate regulation	0.92**	1			
Air purification	0.71**	0.67**	1		
Erosion control	-0.33**	-0.37**	-0.26*	1	
Outdoor recreation	0.90**	0.86**	0.67**	-0.33**	1

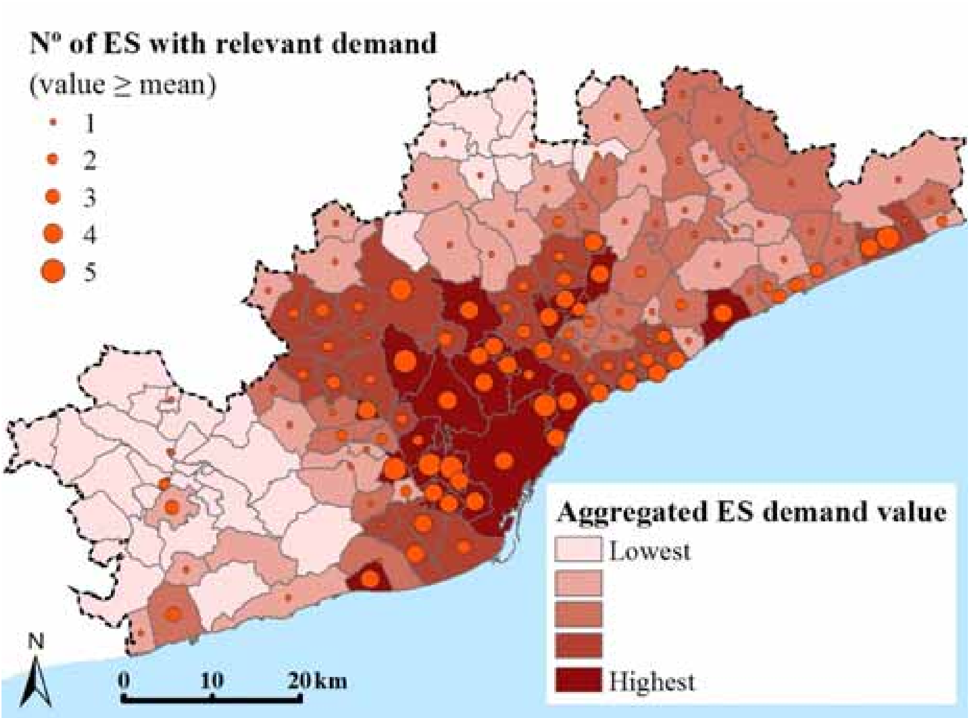

Fig. 5.5. Expected ES demand hotspots and coldspots in the BMR based on the aggregated ES demand value and the ES richness, i.e., the number of ES indicators with relevant demand (value ≥ mean) shown at the municipality level. Aggregated ES demand values are classified in quintiles.

5.3.3. Ecosystem service bundles and urban-rural gradients

Cluster analysis considering both the supply and demand indicators of ES allowed to group the 164 municipalities of the BMR into five clusters, revealing five distinct ES supply - demand bundle types (**Table 5.4; Fig. 5.6**). Spatial autocorrelation analysis determined that these five bundle types were also clustered on the BMR area (z-score = 2.28).

The five bundle types were named and characterized based on the specific supply-demand relationships and the main land uses taking place in each group. Cluster 1 was named "Urban core" because it comprises the municipality of Barcelona and several adjacent or nearby cities (*n* = 7). It is characterized by dense urbanization and very high population densities. This bundle type showed the lowest ES supply mean values and the highest ES demand values for all indicators except the demand for erosion control, revealing an overall ES mismatch from a bundle supply and demand perspective. Cluster 2 (*n* = 23), named "Suburban nodes", includes those municipalities with a very relevant amount of population and urbanized land, mostly located near the urban core or representing urban sub-centers in the BMR (Catalán et al., 2008). It displayed slightly higher ES supply mean values than the urban core and moderate ES demand values (from 0.21 to 0.27), except for air purification which was substantially higher (0.64). Cluster 3, named "Periurban green", is by far the largest bundle type by number of municipalities (*n* = 69). It comprises mostly municipalities with a relevant share of urban land, but also substantial amounts of forest and/or scrubland and, in some cases, also agricultural land. ES supply-demand relationships are characterized by low supply levels of food provision and climate regulation (yet higher than in the previous clusters), moderate to high supply values of air purification, erosion control and outdoor recreation (from 0.28 to 0.50), and a clear disparity of demands: food production, climate regulation and outdoor recreation are barely demanded while air purification and erosion control are demanded in moderate rates (0.36 and 0.44 respectively). Cluster 4 (*n* = 29), named "Cropland", groups those municipalities where land use is primarily agricultural (crops), basically located in the wine-making county of Penedès (west side of the BMR) and in other farming areas, mainly placed along the coast such as in the Llobregat River delta. All ES indicators, both at the supply and demand sides, showed low to moderate values (in the range 0.04 – 0.29), except for crop production (0.53). Finally, Cluster 5 (*n* = 36) was called "Forestland" because it comprises inland municipalities mostly covered by woodland, where urban settlements are generally small and agriculture is absent or minor. This ES bundle type showed by far the highest supply values for regulating services and outdoor recreation and the lowest ES demand values for all indicators except for erosion control which was highest (0.56).

Interestingly, this bundle mirrors the "urban core" cluster in the opposite direction regarding the relationship between supply and demand, except for food supply values.

Table 5.4. Standardized mean values for each ES indicator (both supply and demand) within each cluster or ES supply-demand bundle type. The number of municipalities per cluster is indicated with *n*.

ES		Clusters				
		Urban core (*n* = 7)	**Suburban nodes** (*n* = 23)	**Periurban green** (*n* = 69)	**Cropland** (*n* = 29)	**Forestland** (*n* = 36)
Food	Supply (crops)	0.04	0.06	0.09	0.53	0.05
	Supply (livestock)	0.00	0.04	0.09	0.09	0.09
	Demand	0.72	0.21	0.04	0.05	0.01
Climate regulation	Supply	0.03	0.05	0.18	0.04	0.43
	Demand	0.77	0.27	0.06	0.10	0.02
Air purification	Supply	0.09	0.11	0.28	0.09	0.70
	Demand	0.81	0.64	0.36	0.25	0.20
Erosion control	Supply	0.10	0.14	0.50	0.22	0.83
	Demand	0.14	0.22	0.44	0.14	0.56
Outdoor recreation	Supply	0.14	0.25	0.40	0.29	0.66
	Demand	0.76	0.25	0.10	0.11	0.02

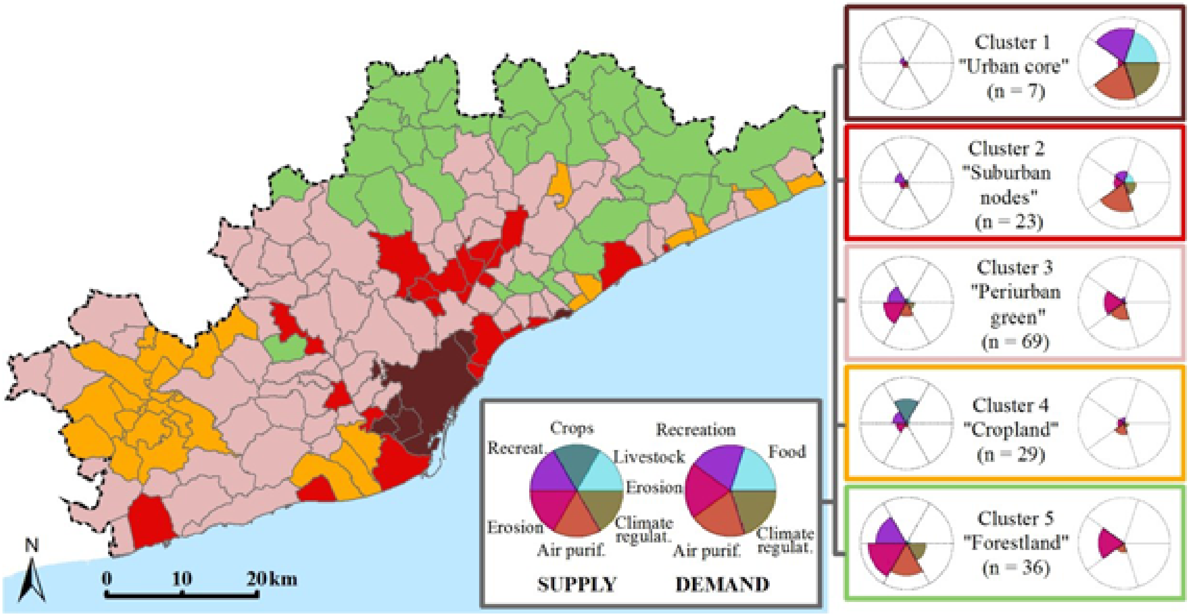

Fig. 5.6. Spatial distribution of ES supply-demand bundle types and standardized mean ES indicator values found within each cluster (represented in star plots). Outline colors of the cluster boxes link to the map classes, hence representing the map legend. The number of municipalities per cluster is indicated with *n*.

PCA results revealed two main components explaining 70.79% of the total variance in the set of eleven ES supply and demand indicators. The biplot of the PCA, representing these two first axes, is shown in **Fig. 5.7**. The first axis of the PCA (50.65% of the variance) showed a potential trade-off between the supply of regulating services and outdoor recreation (highly related to special protection planning strategy) and their demand (mostly related to urban strategies), except in the case of the demand for erosion control which contributes positively to PC1. The second axis of the PCA (20.14% of the variance) revealed a potential trade-off between the supply of provisioning services (especially crop production) and all other ES (both at the supply and demand sides). As expected, special protection of the vineyard is highly related to crop production due to the importance of the Penedès wine-making area.

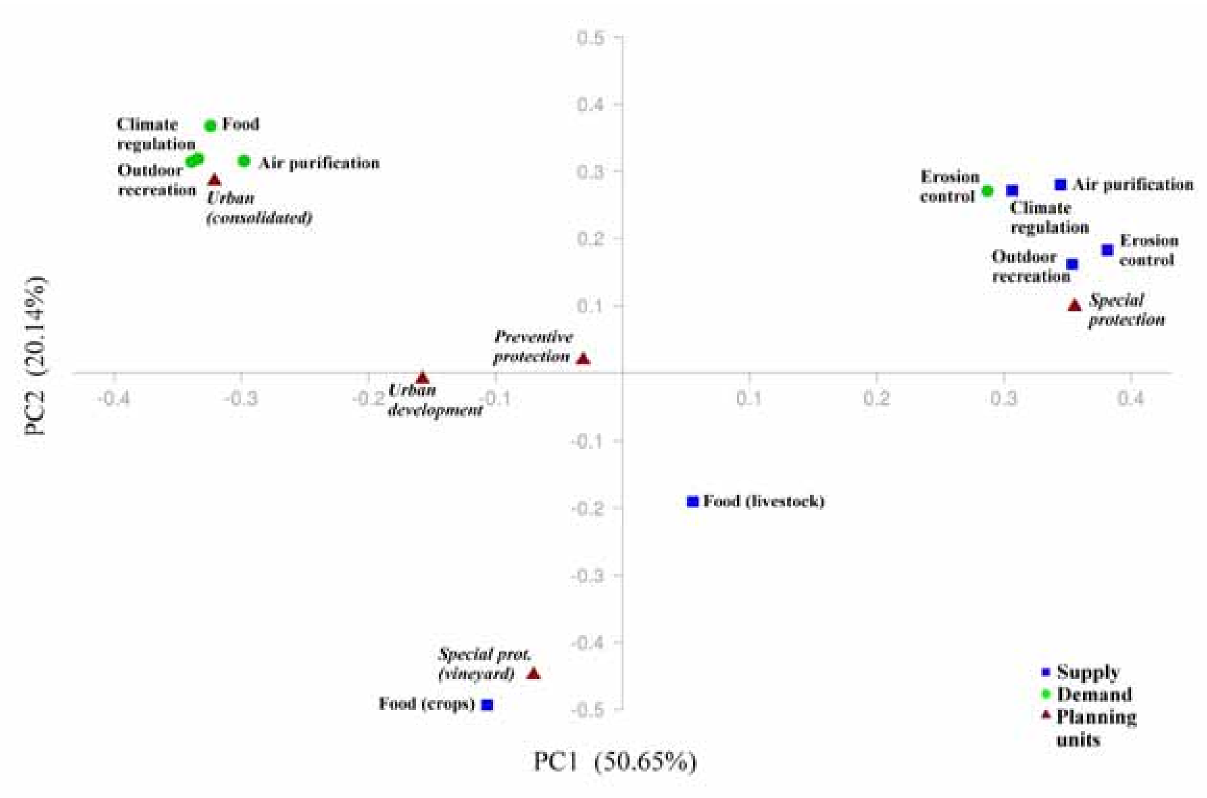

Fig. 5.7. Biplot of the principal component analysis (PCA) for the ES supply and demand indicators and their relationship with land planning strategies (PTMB).

The spatial urban-rural gradients of the ES indicators for the BMR illustrate graphically the spatial patterns shown in the maps and described above. The gradients for ES supply showed a similar mounting common trend in all indicators as distance to the urban core increases (**Fig. 5.8A**). In all cases (except crop production), gradients revealed a remarkable increase after km 5-6 followed by a slight decrease after km 10-11 only lasting 3-4 km before regaining the growing trend. This pattern can be explained by the periurban areas surrounding the urban core, mainly covered by forests (e.g., Collserola mountain range), scrubland or grassland, which precede the urban and agricultural land located in the inland plains. Demand gradients also showed a common similar pattern for all indicators, except erosion control (**Fig. 5.8B**). Values were highest in the urban core followed by a decreasing trend as distance increases. Outdoor recreation and food production gradients performed a sharp decline in the first 10 km whereas air purification and climate regulation decreased more gradually because are less dependent to population density. Erosion control demand gradient revealed a similar pattern as for supply, but following a steady trend after km 11 rather than a growing one.

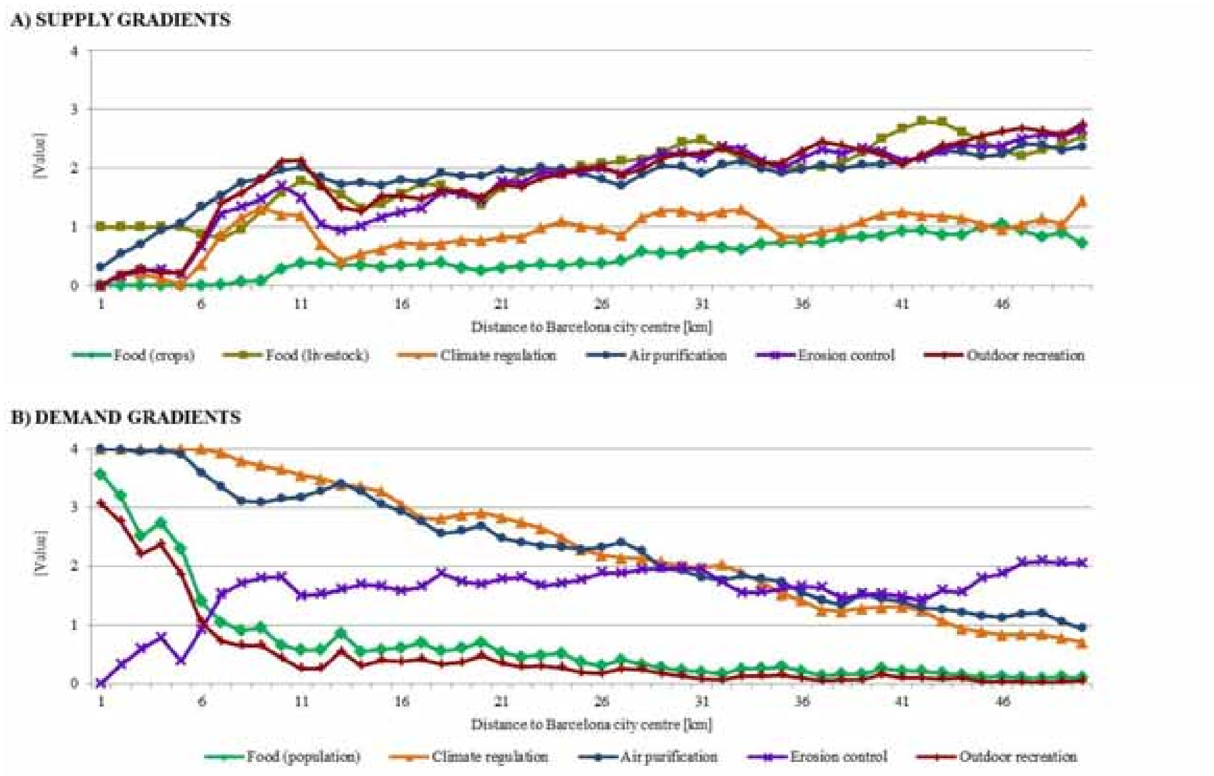

Fig. 5.8. Urban-rural gradients (50 km) of the ES supply and demand indicators for the BMR. Each point represents the average reclassified value (0-4 range) in the concentric ring at the respective distance from the Barcelona city center. Null values are not considered.

5.4. Discussion

5.4.1. Understanding ecosystem service bundles along the urban–rural gradient

Our results show that land cover and the underlying socio-ecological conditions decisively shape supply and demand patterns of ES in the BMR. Interestingly, the resulting ES supply-demand bundle types can be interpreted from a "land sharing" versus "land sparing" approach (Lin and Fuller, 2013). Municipalities under the "Urban core", "Cropland" and "Forestland" clusters follow largely a sparing landscape model based on one predominant land cover whereas the municipalities grouped into the "Suburban nodes" and "Periurban green" clusters could be classified as land sharing-based spatial configurations consisting of a mix of land covers.

These patterns are the result of complex historical processes. Mediterranean landscapes such as the BMR have been subject to increasing pressures over the last decades, leading to homogenization dynamics in terms of land use (Brandt and Vejre, 2003; Gómez-Baggethun et al. 2011). Since the 1950s, the BMR has experienced an accelerated urban development, driven by industrialization and associated migration from rural areas (within the BMR and beyond) to cities, especially to the urban core (Catalán et al., 2008). As a result, a gradual abandonment of traditional agrosilvopastoral practices took place, especially in mountainous areas, together with consequent forest densification and afforestation of open land (Otero et al., 2013). Only the most productive, easily-irrigable and accessible land parcels (mostly located in the lowlands) preserved their agricultural use (Marull et al., 2010).

Currently, "Cropland" municipalities are characterized by a high share of agricultural land which basically provides crop products and are relatively poor in terms of capacity to deliver other ES. On the other hand, "Forestland" municipalities are mostly covered by woods and have a high potential to sequester carbon, remove air pollution, control erosion and provide recreation opportunities. Other assessments of ES supply bundles have showed similar results (e.g., Raudsepp-Hearne et al., 2010; Maes et al., 2012) indicating a clear positive association (i.e., synergy) between all the analyzed regulating ES and outdoor recreation and a significant negative association (i.e., trade-off) between crop production and these ES. At the same time, both "Cropland" and "Forestland" municipalities are sparsely urbanized and populated, which explains the low values they present for ES demand. An exception is erosion control demand, which (unlike the other ES) is not related to urban intensity factors but to geomorphologic aspects (e.g., topographic slope). Consequently, "Forestland" municipalities, mostly located in hilly landscapes, have substantially higher demand values than "Cropland" municipalities which are basically situated in flat areas. Our results also show that, as expected, the widespread and dense urbanization characterizing "Urban core" municipalities reflect the highest ES mismatches between supply and demand when both are analyzed from a bundle perspective (again with the exception of erosion control), a result that is consistent with previous studies focused at the city level (Baró et al., 2014; 2015).

"Suburban nodes" and especially "Periurban green" municipalities are characterized by higher landscape heterogeneity and mix of land uses and covers. As a result, ES bundles show a rather "balanced" budget between supply and demand mean values, with some relevant exceptions such as air purification (especially in the "Suburban nodes" bundle), stressing the fact that air pollution problems are not only confined to highly urbanized land. However, it should be noted that a quantitative ES (mis)match or budget analysis as performed in other studies (e.g., Burkhard et al., 2012; 2014; Kroll et al., 2012) is not possible here because supply and demand indicators are not directly comparable. The only exception is climate regulation where both indicators have the same unit (kg C ha^{-1} $year^{-1}$). Ratios showed that the carbon emissions considered are higher than carbon offsets provided by the local vegetation in all municipalities but five (all of them included in the "Forestland" cluster"). Therefore, our research should not be interpreted in terms of self-sufficiency, but strictly as the assessment of the spatial patterns and associations between ES supply and demand indicators from a bundle approach. From the analysis of ES bundles, it is also worth pointing out that livestock production is not particularly prominent in any cluster. Unlike crop production, livestock farming does not necessarily require extensive land parcels (especially for pork or poultry); hence it probably holds a higher spatial compatibility with other land uses. However, results also indicate a likely trade-off with dense urbanization, probably because: (1) urban communities usually are unwilling to live close to industrial animal production sites (Raudsepp-Hearne et al., 2010); and (2) regional land use regulation directly establishes minimum distances between these sites and urban areas (which depend on the type of animal and other factors).

5.4.2. Implications for landscape planning and management

The spatial relationship between ES supply and demand is also a key issue for landscape planning and management (Syrbe and Walz, 2012). Previous studies (Costanza, 2008; Fisher et al., 2009; Burkhard et al., 2014) have classified ES according to their spatial characteristics suggesting several differentiated categories. Below, we analyze the spatial characteristics of the selected ES and discuss its implications for landscape planning and management in the BMR in the light of the obtained results. Crop and livestock productions are classified as "*decoupled*" ES because, as most provisioning ES, they can be transported from the place of production to the place of consumption over long distances, involving in many cases complex supply chains (Burkhard et al., 2014). This characteristic allows metropolitan regions such as the BMR to let their food supply rely largely on food imports, at the same time that it allows that a substantial part of its food production is exported elsewhere (e.g., wine products from Penedès are exported worldwide). However, preserving farming areas in urban regions can also play an important role in terms of food security and resilience which should be considered in strategic planning (Barthel and Isendahl, 2013; Camps-Calvet et al., 2016). Additionally, Mediterranean agricultural landscapes hold important cultural values such as aesthetic appreciation, sense of place which and local ecological knowledge (Gómez-Baggethun et al., 2010), that are not included in this assessment. These aspects are often recognized in landscape planning and also reflected in consumer preferences for local food (Feldmann and Hamm, 2015). In the BMR, the Penedès vineyards and other agricultural areas are explicitly protected in regional planning instruments such as the PTMB (2010). Climate regulation was classified by Costanza (2008) as a "*global non-proximal*" ES because the benefits derived from carbon sequestration and storage by ecosystems are realized globally. Cities and urban regions, including the BMR, are generally far from having a net zero carbon footprint (see Escobedo et al., 2010; Liu and Li, 2012; Baró et al., 2015) and many of them have set substantial CO_2 emissions reduction targets over the coming years (see for example the Covenant of Mayors initiative in Europe[25]). With regard to land use planning and decision-making, BMR's budget for climate regulation does not necessarily require achieving carbon neutrality, but regional and local policies could foster carbon reduction and offsetting actions both inside and beyond metropolitan boundaries so global climate regulation goals can be met in the long-term (currently municipal Sustainable Energy Action Plans define measures at the local level). Air purification can be considered a "*local proximal*" or "*omnidirectional*" ES because benefits are realized in the ES providing area or its surrounding landscape without directional bias (Fisher et al., 2009). In terms of spatial planning, that means that urban green space and periurban green areas are key providing areas because it is where the ES is actually delivered due to higher air pollution levels (Baró et al., 2016). Even if the reduction of air pollution emissions should be the first priority in urban policy, GI planning in the BMR can contribute to improve air quality if a land sharing approach is considered in urban development (Stott et al., 2015) and, concurrently, large periurban green areas such as the Collserola massif remain protected from urbanization (Depietri et al., 2016). The Barcelona Green Infrastructure and Biodiversity Plan 2020 (Barcelona City Council, 2013) is an interesting initiative towards a land sharing model in the urban core because it fosters the expansion of GI in all sorts of available land, including rooftops, inner courtyards,

[25] http://www.covenantofmayors.eu/index_en.html

vacant plots, etc. Erosion control corresponds to an "*in situ"* ES because the benefit (soil retention) is realized in the same location of provision (Burkhard et al., 2014), but can also be considered "*directional"* (Costanza, 2008) because it can prevent erosion-related events such as landslides which benefit downhill areas. In this paper, we have basically analyzed the former condition due to indicator characteristics, showing an apparent synergetic relationship between supply and demand spatial patterns. This can be explained because the areas with higher risk of erosion due to geomorphologic factors (e.g., steepness in mountain ranges) are mostly covered by ecosystems with a high potential to control this process (e.g., forests) whereas land covers with low capacity (e.g., agro-ecosystems) are usually located in topographically less vulnerable areas. Regional urban planning regulation in the BMR currently favors this situation forbidding urban developments in areas where slope is higher than 20%. Finally, outdoor recreation is classified as "*in situ*" or "*user movement related*" ES (Costanza, 2008) because, as most part of cultural ES, users need to actively reach providing areas in order to experience the related benefits. Therefore, accessibility is a key aspect for the assessment of outdoor recreation supply-demand relationships (Paracchini et al., 2014). Some studies have observed that beyond a threshold of 300-400 meter distance from home, the (everyday) recreational use of urban green space decreases substantially (Schipperijn et al., 2010). Furthermore, size of the providing area is also relevant because some outdoor activities (e.g., walking the dog, playing some sports, relaxation) can be realized in relatively small recreational patches (e.g., pocket parks), but others such as hiking or cycling require much larger areas. Therefore, in terms of spatial planning, this ES would require a combination of land sparing and land sharing models, as already considered by the English standard ANGSt (Accessible Natural Greenspace Standard, Natural England, 2010) or by other regional decision-support instruments (Van Herzele and Wiedemann, 2003). In the BMR, an effective harmonization of regional planning instruments such as the PTMB (2010) with municipal GI plans (e.g., Barcelona City Council, 2013) is required in order to achieve this arrangement of urban and periurban spaces.

5.5. Conclusions

To our knowledge, this study presents the first assessment of ES bundles that integrates both the supply and demand sides in an urban-rural gradient. Our results show that urban and agricultural intensity is associated to lower potential and richness in terms of ES supply. Conversely, forest landscapes are characterized by a high multifunctionality, especially in regard to regulating ES, but most of these ES are barely demanded in some of these areas. Urbanization is also a clear driver at the demand side, as higher population densities and pressures (e.g., air pollution) inevitably entail increased needs for provisioning, regulating and cultural ES, generally leading to larger local mismatches between supply and demand. From an aggregated urban-rural gradient perspective, our case study shows inverse spatial patterns of ES supply and demand for all the analyzed ES, except for erosion control. This was already observed in other urban regions considering specific ES groups (e.g., Kroll et al., 2012).

With regard to landscape planning and management, a key aspect is considering the spatial scale relationships between ES supply and demand. The urban population needs nearby ecosystems in order to benefit from air purification or outdoor recreation services and, even if food or climate regulation can be provided from distant ecosystems, metropolitan regions such as the BMR have important motivations (e.g., food security, nature experience, climate adaptation and mitigation targets, etc.) to reduce their overall ES footprint. Based on these considerations, we argue that a promising approach could consist of combining land sharing strategies in urban and agricultural land in order to increase their multifunctionality and resilience (e.g., stricter GI ratios in urban development plans and fostering the provision of cultural ES in agricultural landscapes), and concurrently, assure the conservation of large patches of multifunctional periurban areas (such as the Collserola massif in the BMR) which are vital for the fulfillment of certain ES bundle demands of the urban population, but generally more vulnerable to urbanization processes.

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Parc de les Infantes, Barcelona, Spain (illustration by Beatriz Acevedo Cantero, published with kind permission of the author and the series of seminars on city, environment, health and drawing "Ciutat Verda" – www.ciutatverda.info)

CHAPTER VI

General discussion and conclusions

6.1. Conceptual and methodological contributions

This dissertation contributes to bridge the so-called supply-demand knowledge gap in research of urban ES, as identified by Elmqvist et al. (2015). In their global assessment on urbanization and ES, these authors noted that a mounting body of knowledge exists on the provision of urban ES at different scales, but that there is little information on needs and demands of ES in cities and metropolitan areas. In this context, the review of urban ES research carried out by Luederitz et al. (2015) also called for an integrated approach which "captures all the elements of the ES production chain and draw holistic conclusions for planning and management of urban ES".

Building on previous conceptual frameworks and particularly on the ES Cascade model (Haines-Young and Potschin, 2010), the dissertation contributes to improve the identification of mismatches in urban ES assessments by means of the operational distinction between ES capacity, flow and demand (see **Figure 4.1** in Chapter IV). Despite this distinction was previously suggested by other authors (e.g., Villamagna et al., 2013; Schröter et al., 2014) and it has already been used for analyzing ES mismatches in other case studies (see Geijzendorffer et al., 2015), this study represents one of the first attempts to operationalize it in the urban context, both at the city (Chapters II and III) and metropolitan (Chapters IV and V) scales.

As described in the Introduction Chapter (see **Fig. 1.3**), the dissertation particularly focuses on the relationship between ES flow and ES demand, i.e., the analysis of (un)satisfied demand. The conceptualization of ES demand (and unsatisfied demand) used here follows the approach developed by Villamagna et al. (2013) and therefore requires information about desired or required end conditions in regard to ES delivery rather than direct use or consumption. To this end, this work provides a novel methodological approach based on the use of EQS and policy goals as a proxy threshold to determine these desired conditions from a societal perspective. Therefore, my approach to the notion of demand for ES departs from the market-oriented approach that frames it as the aggregated result of individual preferences (often through estimations of individual willingness to pay for such services) to frame demand in relation to EQS and policy targets defined in policy processes, in line with concepts such as safe-minimum standards (*sensu* Ciriacy-Wantrup, 1952) or safe operating boundaries (*sensu* Rockström et al., 2009). Even if the thresholds included in EQS and policy targets are often based on expert knowledge and/or scientific evidence, these standards are expected to reflect, at least to some extent and in the context of representative democracies, the *demands* of citizenship for securing human well-being, including health aspects (e.g., air quality standards represent a societal demand for breathing clean air considering the scientific evidence in regard to air pollution and health). This approach has been applied at the city scale considering three regulating ES (Chapter III). Although a risk reduction perspective has been used in several studies to quantify demand for regulating ES (see Wolff et al., 2015), the explicit identification and assessment of mismatches between ES flow and demand for urban planning and management was clearly under-researched in the literature. The approach is further applied at the metropolitan scale (Chapter IV) using spatially explicit data and considering a cultural ES (outdoor recreation).

Chapter V also contributes to the operationalization of the ES bundle concept from a supply-demand approach. Assessment of ES bundles has been mostly applied to the supply side of ES (e.g., Raudsepp-Hearne et al., 2010), but very few studies have analyzed also the demand side. Particularly, this chapter provides (to my knowledge) the first attempt to identify relevant ES supply-demand bundle types using spatial ES indicators at both sides.

In regard to urban ES modeling and mapping, this dissertation brings forth two main methodological contributions. First, it includes one of the first applications of the i-Tree Eco tool (Nowak et al., 2008) in Europe (Chapters II and III). The models included in i-Tree Eco were especially designed for U.S. case studies, hence its application elsewhere contributed to improve its adaptability and upgrading (e.g., by including non-U.S. species in the model database). Currently the tool is used in many case studies worldwide to assess urban forests structure and ES (see www.itreetools.org). Second, this study also comprises one of the first attempts to downscale and adapt the ESTIMAP tool (Zulian et al., 2014) at a regional scale (Chapters IV and V). Previously, the tool had been used to support environmental policies only at a European scale (Maes et al., 2014).

6.2. Limitations and caveats

The tools and models used in this dissertation for ES quantification and mapping (i.e., i-Tree Eco and ESTIMAP) have been applied in previous peer-reviewed assessments (e.g., Nowak and Crane, 2002; Nowak et al., 2006; Paracchini et al., 2014; Zulian et al., 2014), hence I consider that their outputs are sufficiently credible and salient for planning and management purposes. However, these methodological approaches have also limitations and caveats that are synthesized in this section.

One of the main limitations of this study is that quantification and mapping of ES indicators (covering capacity, flow and demand) is commonly based on model-based proxies. Eigenbrod et al. (2010) and Schulp et al. (2014) warn that there is potential for error if the assumed proxy variables are not actually good ES predictors, especially in spatially explicit approaches. However, validation of results was challenged by a lack of suitable empirical or observed data on ES at the corresponding spatial scales.

Probably the modeling and mapping of cultural ES in urban areas involves the highest level of uncertainty since capacity, flow and demand of these ES are strongly context/user dependent. The outdoor recreation capacity model used in Chapters IV and V was primarily based on an expert knowledge approach. However, the use of additional or complementary participatory methods (e.g., visitors' surveys on green space preferences) would have improved the quality of the final map. Outdoor recreation flow and demand were mapped based on the assumption that all inhabitants in the BMR have similar behaviors and desires in terms of (everyday life) recreational experience and that distance (as a proxy of accessibility) to green space is a critical variable explaining its recreational use in urban areas. Promising approaches to overcome this methodological simplification include participatory mapping (e.g., Palomo et al., 2013), the use of social media data such as geo-tagged photographs showing recreational activities (e.g., Wood et al., 2009) or questionnaires on outdoor recreational expectations in urban areas (Burkhard et al., 2014).

Air purification has been assessed both at the municipal level (Chapters II and III) and at the metropolitan level (Chapters IV and V) using dry deposition models. Both i-Tree Eco and ESTIMAP models use the same approach based on the quantification of a removed pollutant flux (flow) as the product of dry deposition velocity (proxy of capacity) and pollutant concentration (proxy of demand). As for other models attempting to simulate complex biophysical processes, there are many uncertainties and limitations in dry deposition models which prevent a more accurate determination of air pollution uptake by urban vegetation. For instance, some sources of uncertainty include non-homogeneity in spatial distribution of air pollutants, particle re-suspension rates, soil moisture status, transpiration rates or leaf boundary resistance (Manning, 2008). However, local and regional fine-scale input data for these variables are not usually available and empirical data on the actual uptake of pollutants by urban vegetation is still limited (Pataki et al., 2011; Setälä et al., 2012).

Estimates of direct carbon sequestration by urban GI (Chapters II, III and V) also face multiple uncertainties and limitations. Urban vegetation is usually exposed to unique environmental conditions (e.g., restricted rooting volumes and higher temperature and CO_2 concentration than in rural areas) and maintenance characteristics (e.g., intensity of pruning and irrigation) which can positively or negatively impact their total carbon offsetting capacity (Pataki et al., 2011; Tang et al., 2016). Allometric and growth equations used to quantify carbon storage and sequestration are mostly based on non-urban conditions, yet adjustment factors are often considered in the modeling to minimize error (e.g., i-Tree Eco, Nowak et al., 2008). In addition, fossil fuel emissions associated to urban green space maintenance (e.g., pruning) and decomposition rates of removed or decaying trees can eventually compensate sequestration gains or even generate negative carbon balances (Nowak et al., 2002), but are also difficult to estimate. Further, the assessments of global climate regulation included in this dissertation only consider the carbon flux associated to urban trees and other vegetation, omitting the contribution related to soils. Even if urban soils can act as relevant carbon sinks (Pouyat et al., 2006), soil respiration can constitute an important emission source too (Velasco et al., 2016), thereby adding a new layer of complexity in urban carbon budget estimates.

The use of EQS and policy targets in ES mismatch assessment (Chapters II, III and IV) has also shortcomings. Despite the human health effects of ambient air pollution have been subject to intense research in the last decades (e.g., Brunekreef and Holgate, 2002), there are still different EQS or limit values for the same pollutant depending on the institutional context (e.g., WHO Air Quality Guidelines are generally more stringent than EU reference values). Similarly, GHG emission reduction targets can vary widely among cities and they are still based on political voluntary commitments. The multiple scale/user dependent factors involved in defining a societal demand for certain ES such as urban cooling (Chapter III) or outdoor recreation (Chapters IV and V) also call for local-based standards rather than general reference values or recommendations.

Finally, another issue not factored in the models used in this dissertation relates to ecological thresholds or tipping points (Andersen et al., 2009). Chapter IV discusses how increasing pressures on ecosystems might trigger ecological thresholds and therefore alter the patterns of ES capacity, flow and demand for air purification and outdoor recreation (see **Fig. 4.7**). However, it is often very difficult to determine when and under what conditions or pressures, ecosystem experience thresholds which can eventually affect their ability to provide ES (Gómez-Baggethun et al., 2011).

6.3. Multi-scale implications for planning, management and decision-making

Assessing current and potential mismatches between ES supply and demand has direct implications for GI planning, management and decision-making in urban areas, especially within the context of the emerging concept of NbS (Kabisch et al., 2016). This dissertation has assessed ES mismatches at two main urban scales: city / local (Chapters II and III) and metropolitan / regional (Chapters IV and V). The site or patch scale is also indirectly considered in this work based on some of the modeling results and evidence from reviewed literature.

Chapters II and III indicate that the contribution of regulating ES provided by urban GI to mitigate climate change-related pressures (i.e., carbon emissions, air pollution and heat stress) is limited (less than 3% considering total carbon emissions and air pollution in all case study cities) and uncertain (see above) at the city scale. In addition, the positive impact of urban GI on environmental quality can be challenged by disservices and trade-offs such as BVOC emissions or GHG emissions associated to urban vegetation management practices (see Chapter II).

Chapters IV and V show mismatches between ES supply and demand at the metropolitan scale too, even if the proportion of urban GI versus built-up land is substantially higher in the BMR than at the Barcelona municipal level (see **Fig. 1.5** and **Table 1.2**). For example, the assessment of direct carbon sequestration by forests in the BMR reveals a marginal impact in the overall metropolitan carbon budget (less than 1%). The estimated high air purification capacities of large metropolitan GI blocks (e.g., protected natural areas) are generally 'underused' due to their distance from demand sites (i.e., residential areas most affected by air pollution). Similarly, areas with highest potential to deliver outdoor recreation opportunities are often inaccessible to most BMR inhabitants (considering an everyday life approach, i.e., threshold distance of 1km).

Modeling results from this dissertation (e.g., **Table 3.6**) and evidence from empirical studies is largely supportive that urban GI, especially urban trees, can contribute to improve air quality and reduce heat stress at the site level, especially within and around green spaces (e.g., Bowler et al., 2010; Yin et al., 2011). Yet, factors such as vegetation selection, design and management practices can have a critical impact on the performance of ES provision (see Janhäll, 2015 for air purification). Besides regulating ES, small green spaces such as pocket parks, allotments and rooftop gardens can provide manifold recreational opportunities and other cultural ES at the site scale with potential impacts at the city level (e.g., Orsini et al., 2014; Camps-Calvet et al., 2016). These findings align well with synthesis assessments on urban GI (e.g., Pataki et al., 2011; Demuzere et al., 2014) and they indicate that improving our understanding on the scale at which urban GI can indeed offer effective NbS is essential to link greening strategies to appropriate levels of planning and decision-making (Scholes et al., 2013).

On the basis of the results of this dissertation and their associated uncertainties regarding the potential of urban GI to cope with different challenges in urban areas at various spatial scales, the following implications for planning, management and decision-making can be drawn (see also an overview in **Fig. 6.1**). First, urban climate change and air pollution mitigation policies should primarily focus on the drivers of demand (built infrastructure and transport systems leading to increased pollution and GHG emissions), not on the sinks (urban GI absorbing carbon and pollutants). This dissertation argues that air pollution problems and local GHG emissions are to be principally dealt with emission reduction policies (e.g., road traffic management, energy efficiency measures, emission caps, and incentives to public transport and clean vehicles) at the city scale. The role of urban GI strategies can be complementary to these policies, but not alternative. Additionally, carbon offsets associated to GI can be fostered by local and metropolitan authorities beyond urban boundaries (Seitzinger et al., 2012). Second, even if ES such as food or global climate regulation can be provided from distant ecosystems due to their decoupled character, metropolitan regions such as the BMR may still have important motivations to reduce their overall ES footprint and increase their resilience, for example in terms of food security, self-capacity to provide opportunities for nature experience, progress towards climate adaptation and mitigation targets. Based on the findings of this dissertation, I argue that a promising approach could consist of combining land sharing strategies in urban and agricultural land in order to increase their multifunctionality and resilience (e.g., stricter GI ratios in urban development plans and fostering the provision of cultural ES in agricultural landscapes), and concurrently, assure the conservation of large patches of multifunctional periurban forest areas (such as the Collserola massif in the BMR). This work shows that these areas are vital for the fulfillment of certain ES demand bundles of the urban population, but they are also generally more vulnerable to urbanization processes. Third, urban GI can contribute to site-scale strategies related to air quality, heat stress and outdoor recreation. For example, wise planning of pocket parks, street trees and rooftop gardens could contribute to the creation of 'healthy' areas within dense urban cores. In very compact cities such as Barcelona, the potential of rooftop gardens and green walls can be particularly relevant due to lack of available land (Berardi et al., 2014; Orsini et al., 2014). Another strategy could consist of freeing up streets and spaces currently used by cars and turn them into green areas. Barcelona is already implementing this strategy through the 'superblocks' concept, i.e., areas in which traffic will be restricted and public spaces will be redesigned for leisure or other social purposes[26]. This plan can partially regain the original idea of Ildefons Cerdà, the town planner who designed the gridded neighborhood of 'la Eixample' (currently one of the most compact districts and with higher deficit of green space in Barcelona) in the late 19th century. His design of open, healthy residential blocks with publicly accessible green spaces was finally undermined by real-estate market interests (Neuman, 2011). Furthermore, fostering the connectivity

[26] See the following press article for more information: https://www.theguardian.com/cities/2016/may/17/superblocks-rescue-barcelona-spain-plan-give-streets-back-residents

between these GI elements (i.e., creation of greenways or corridors) can eventually generate relevant impacts at the city level as well. And fourth, trade-offs and disservices related to ES should be considered in GI planning and management in order to estimate 'net' impacts to urban quality of life. Even if most urban GI elements, such as urban trees, are multi-functional in relation to most of the ES considered in this dissertation, some trade-offs and disservices have also been identified (Chapters II and V). The literature on urban ES already provides a comprehensive analysis of synergies, trade-offs and disservices associated to different types of urban GI (e.g., Demuzere et al., 2014; von Döhren and Haase, 2015). For example, dense tree canopies provide a high shading effect, but they are also associated to lower dispersion rates of air pollution in street canyons (Vos et al., 2013; Jin et al., 2014).

The scope of this dissertation is limited to several relevant ES in urban areas, but urban GI can provide additional ES and benefits to the urban population. Other studies on urban GI (e.g., Pataki et al., 2011; Gómez-Baggethun et al., 2013; Demuzere et al., 2014) suggest that ES such as runoff mitigation, water purification, and the health and social benefits associated to other cultural ES such as sense of place, social cohesion or cognitive development can play a relevant role as NbS in urban areas. Unlike standard 'grey' or technological infrastructures which are normally designed as single-purpose, the main added value of urban GI resides on its multi-functionality.

Finally, I contend that urban GI planning and management requires a holistic approach, considering the whole range of ES potentially provided by different types of urban GI and the interactions between them, together with the different spatial scales at which these ES can be relevant for the resilience, sustainability and livability of urban areas. This issue calls for a strong multi-scale and multi-disciplinary institutional coordination between all the authorities dealing with urban and environmental policy and for the harmonization of planning and management instruments at different levels.

6.4. Concluding remarks and future research

The limitations described above show that more empirical research is needed in order to decrease the current levels of uncertainty associated to the modeling and mapping of ES capacity, flow and demand in urban areas, especially at the city and metropolitan scales. Currently these knowledge gaps hamper a sound assessment on the potential or effectiveness of GI strategies to cope with different urban challenges, yet future methodological improvements probably will not substantially vary the policy implications described above and synthesized in the following **Fig. 6.1**.

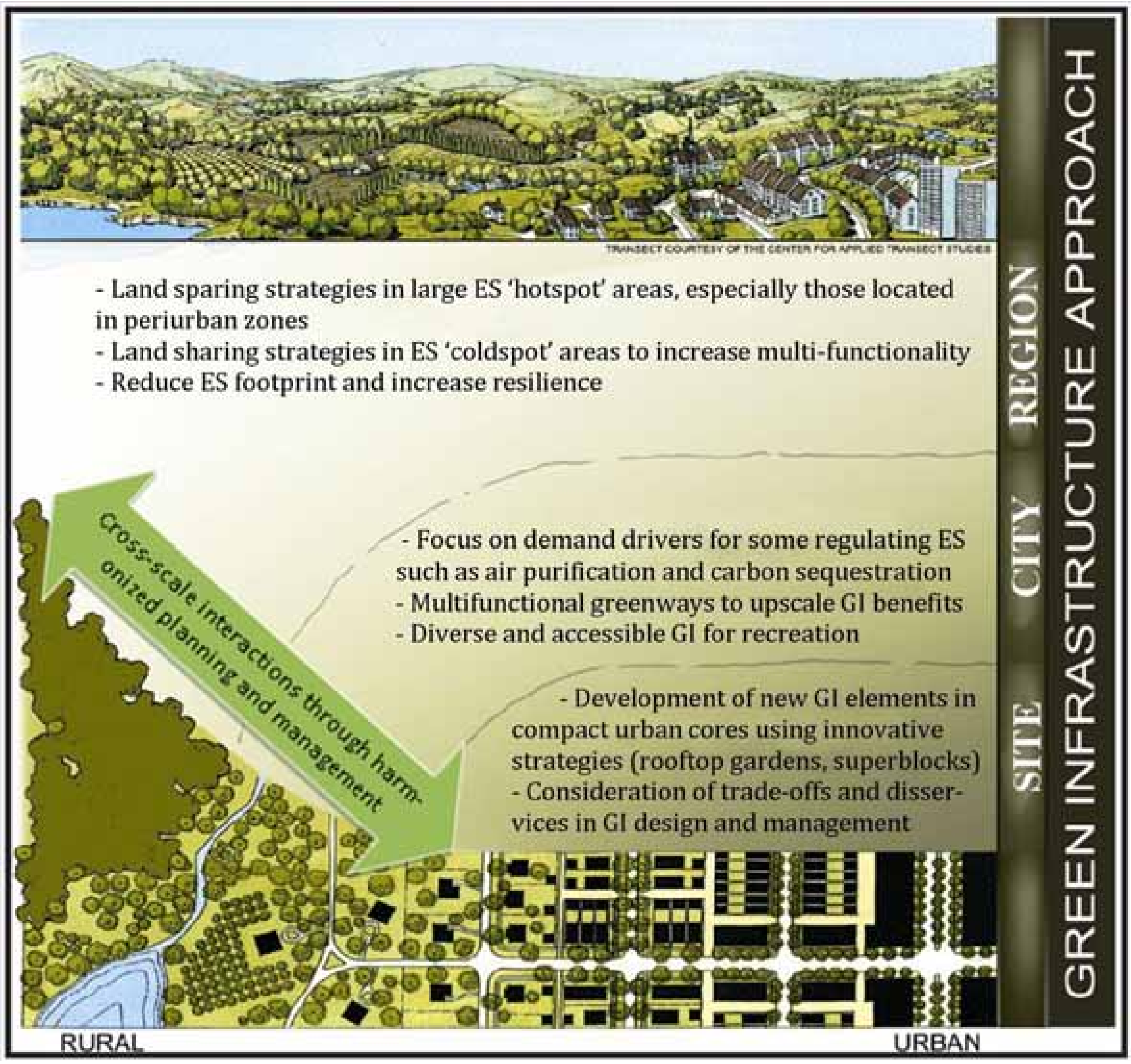

Fig. 6.1. Main conclusions in terms of GI planning, management and decision-making on the basis of the results of this dissertation. Modified from Allen (2014).

From this work, I contend that future research should focus on improving the understanding of the temporal and spatial dynamics of urban ES, considering feedbacks between supply and demand (Haase et al., 2014; Wolff et al., 2015). For example, the impact of seasonal or even shorter temporal dynamics associated to ES capacity, flow and demand (e.g., air pollution and heatwave episodes, 'recreational seasons' associated to fishing, mushrooming, etc.) is highly relevant for mismatch analysis and hence for urban planning and management (Burkhard et al., 2014). Deep understanding of these dynamics should also account for the ecological thresholds described in previous chapters (e.g., Chapter IV).

This dissertation has focused on the 'urban scale', including the metropolitan and city levels. However, as mentioned in the Introduction Chapter, cities depend and impact on ecosystems (and human populations) located far beyond urban boundaries. These urban teleconnections (Seto et al., 2012) need to be better understood and accounted for, for example through the assessment of life cycle impacts (LCIA) on ES (Othoniel et al., 2016). The distinction between ES capacity, flow and demand, is potentially useful in this context. The quantification of flows and demands of ES which can be 'imported' or 'appropriated" by cities (e.g., provisioning ES, global climate regulation) can contribute to existing approaches for measuring these urban teleconnections, including the concepts of ecological footprint (Rees, 1992; Folke et al., 1997) and ecological debt (Goeminne and Paredis, 2010), and potentially overcome some of its limitations (Galli et al., 2016). Besides, the assessment of ES demands from urban areas as a driver of land use change at the global scale is needed in order to understand land use transformations beyond the traditional approach focused on agricultural production (Wolff et al., 2015).

Finally, another key research challenge that should be addressed in future research relates to the improvement of ES demand assessments in urban areas considering an environmental justice perspective. The indicators presented in this dissertation in regard to ES demand (and unsatisfied demand) provide only a general insight on the magnitude and distribution of ES needs and deficits in urban areas. However, these results could be analyzed further considering the diverse socio-economic groups existing in most cities. Particularly, future research should focus on the identification of factors and institutional barriers (e.g., property rights, exclusion mechanisms, planning instruments, etc.) that lead to trade-offs and synergies in the demand for particular bundles of ES (especially cultural ES) by different social groups in urban settings (see Kabisch and Haase, 2014). This will play a major role in supporting equity policies, particularly for the urban poor as well as other vulnerable groups. In this regard, the UN Sustainable Development Goal 11 ("Make cities inclusive, safe, resilient and sustainable"[27]) specifically claims that "By 2030, provide universal access to safe, inclusive and accessible green and public spaces, in particular for women and children,

[27] See http://www.un.org/sustainabledevelopment/cities/

older persons and persons with disabilities". To this end, interdisciplinary approaches involving different stakeholder perspectives and preferences throughout the research process are increasingly recognized as the way forward (Anguelovski, 2013; Haase et al., 2014; Luederitz et al., 2015).

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APPENDICES

Appendix A. Supplementary information for Chapter III

Table A.1. ES supply indicators for the five case study cities.

ES	Indicator	Barcel.	Berlin	Stockh.	Rotter.	Salzb.	Mean
Air purification	PM_{10} removal kg ha^{-1} year^{-1} (Mg year^{-1})	16.42 (166.01)	18.97 (1690)	10.93 (235.77)	3.71 (101.74)	6.92 (45.46)	11.39 (447.80)
	NO_2 removal kg ha^{-1} year^{-1} (Mg year^{-1})	5.40 (54.59)	8.36 (745)	6.29 (135.78)	2.24 (61.37)	4.12 (27.05)	5.28 (204.76)
	O_3 removal kg ha^{-1} year^{-1} (Mg year^{-1})	7.18 (72.62)	21.96 (1,957)	12.67 (273.44)	2.99 (81.94)	0.12 (0.78)	8.98 (477.16)
Global climate regulation	Net CO_2 sequestration t ha^{-1} year^{-1} (t year^{-1})	1.97 (19,986)	3.66 (325,726)	3.06 (66,131)	1.05 (29,218)	2.39 (15,673)	2.43 (91,347)
	Carbon storage t ha^{-1} (Mg)	11.22 (113,437)	32.84 (2,925,924)	28.84 (622,326)	9.38 (257,071)	21.99 (144,421)	20.85 (812,636)
Urban temperature regulation	Tree shade area % (ha)	29.40 (2,973)	42.70 (38,048)	37.50 (8,093)	12.20 (3,343)	28.60 (1,878)	30.08 (10,867)

Note: see references and corresponding time-ranges in **Table 3.2.**

Table A.2. ES demand indicators for the five case study cities.

ES	Indicator	Barcel.	Berlin	Stockh.	Rotter.	Salzb.	Mean
Air purification	PM_{10} annual mean concentration $\mu g\ m^{-3}$	32.76	30.11	28.45	28.45	23.86	28.72
	NO_2 annual mean concentration $\mu g\ m^{-3}$	53.78	53.38	38.50	48.66	45.21	47.90
	26th highest O_3 value based on daily max 8-hour averages $\mu g\ m^{-3}$	89.60	116.14	95.14	84.74	111.63	99.45
Global climate regulation	CO_2eq. emissions per ha. $t\ ha^{-1}\ year^{-1}$	398.99	214.70	128.59	1,067.35	86.59	379.25
	CO_2eq. emissions per capita $t\ capita^{-1}\ year^{-1}$	2.51	5.40	3.40	48.51	3.82	12.73
Urban temperature regulation	Heat wave risk days	>50	2-6	0-2	2-6	2-6	N/A

Note: see references and corresponding time-ranges in **Table 3.3.**

Appendix B. Supplementary information for Chapter IV

Mapping of outdoor recreation

As shown in **Figure B.1**, the model follows a composite mapping procedure based on the aggregation of the three components (i.e., degree of naturalness, nature protection and water). Each component was developed through one or several factors considered relevant in the case study of the BMR and for which spatial input data was available (**Table B.1**). All the scores assigned to the water component factors and to the 'remarkable trees' factor of the nature protection component were subject to a distance decay modeling, assuming that the recreation potential decreases as the distance from the specific feature (e.g., a beach) increases. The following inverse logistic function was applied to these factors:

$$f\ (d) = \frac{1+K}{K+e^{\alpha \cdot d}} \cdot w \qquad \text{(Formula B.1)}$$

Where: d is the distance from the specific feature, α and K are the size and shape parameters of the function adjusted according to a distance threshold assessment, and w is the assigned score. The parameters α and K were respectively set at 0.0035 and 30 for the factor 'beaches', corresponding to a distance thresholds of 1000 m at which the score is decreased by 50% and 2000 m at which the score is zero, and 0.008 and 30 for the rest of factors, corresponding to distance thresholds of 500 m and 1000 m (see also **Table B.1**). The distance thresholds were defined based on the expert consultation process.

Factors within each component were aggregated by a simple linear summation method and normalized between 0 and 1 following equation (2). The three components were aggregated in the same way in order to obtain the final recreation potential index (RPI). Unlike the factors, all the components were given equal weights under the assumption that they cover complementary aspects of the recreational potential (Paracchini et al., 2014).

$$v' = \frac{v - min}{\max - min} \qquad \text{(Formula B.2)}$$

Table B.1. Recreation potential components and data sources. Factors, scores and distance function thresholds defined in the expert consultation process.

Component	Factors (spatial dataset)	Data source	Assigned score	Distance function thresholds (m)		Comments
				50%	0%	
Degree of naturalness	Closeness to potential native vegetation	SITxell database – Habitats dataset (Barcelona Regional Council)	0-1	N/A	N/A	Based on an ecosystem assessment of its ecological succession stage. Originally scores ranged from 0 (artificial land covers) to 4 (climax ecosystem).
Nature protection	Areas designated as natural parks	Environment geodatabase (Catalan Government)	1	N/A	N/A	
	Areas designated as regional protected areas or Natura 2000 sites	Environment geodatabase (Catalan Government)	0.8	N/A	N/A	It excludes areas which are also designated as natural parks.
	Remarkable protected trees	Environment geodatabase (Catalan Government)	0.8	500	1000	
	Areas designated as geological heritage	Environment geodatabase (Catalan Government)	0.5	N/A	N/A	
Water	Lakes, ponds, reservoirs and wetlands	SITxell database – Habitats dataset (Barcelona Regional Council)	1	500	1000	
	Beaches	SITxell database – Habitats dataset (Barcelona Regional Council)	1	1000	2000	
	Main river network	Environment geodatabase (Catalan Government)	0.5	500	1000	Rivers usually having permanent water flow all the year.
	Angling river areas	Environment geodatabase (Catalan Government)	0.3	500	1000	

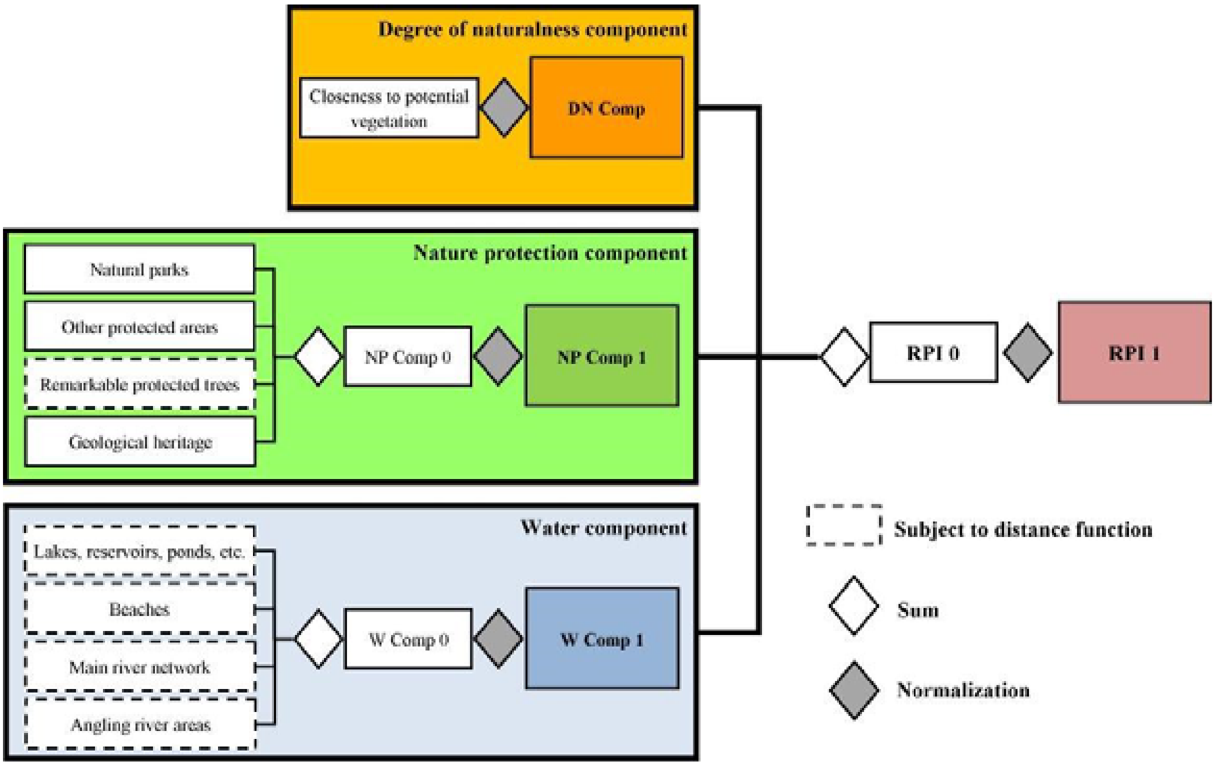

Fig. B.1. Flowchart of the procedure to obtain the recreational potential map (adapted from Paracchini et al., 2014).

Potential trips for mapping the expected outdoor recreation flow were estimated using a neighbor operator with a custom matrix. The custom matrix was based on the distance decay function (B.1) considering $\alpha = 0.008$ and $K = 30$ (see also **Fig. B.2**).

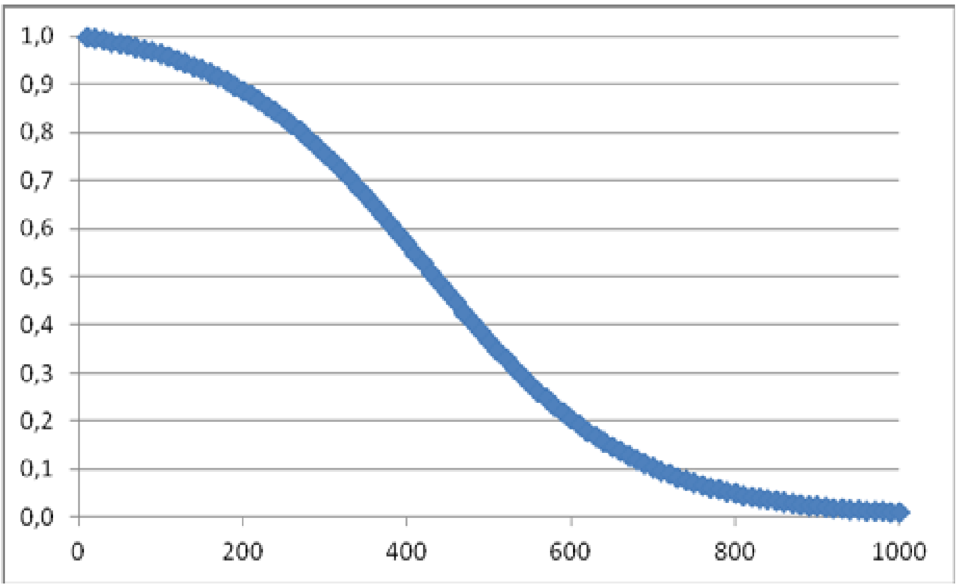

Fig. B.2. Distance function (in m) applied for mapping the expected outdoor recreation flow. The function shape shows that the probability of travelling beyond 500 m decreases below 0.5 (Paracchini et al., 2014).

Table B.2. Cross-tabulation matrix between a reclassified raster of Euclidian distances to recreation sites and the population density grid used to obtain the outdoor recreation demand values (legend on the right). It assumes that all inhabitants in the case study area have similar desires in terms of (everyday life) outdoor recreational opportunities, but their level of fulfillment depends on proximity to recreation sites. Distance breaks consider the recommended standards by regulatory agencies (Stanners and Bourdeau, 1995; Barbosa et al., 2007).

	Distance to recreation sites (m)					
Population density (inhab. ha^{-1})	< 300	300 - 600	600 - 900	900 - 1200	1200 – 1500	> 1500
< 5	**0**	**0**	**0**	**0**	**0**	**0**
5 - 50	**0**	**1**	**1**	**2**	**2**	**3**
50 – 100	**0**	**1**	**2**	**2**	**3**	**4**
100 - 200	**0**	**2**	**2**	**3**	**3**	**4**
200 – 400	**0**	**2**	**3**	**3**	**4**	**4**
> 400	**0**	**3**	**4**	**4**	**4**	**5**

0	Lowest demand
1	
2	
3	
4	
5	Highest demand

Mapping of air purification

Table B.3 includes the complete list of the parameters considered for the modeling. Some of the predictor variables reflect sources or sinks of air pollution such as the road network, different types of land use and population density. The latter was considered also a proxy for traffic flow levels since no complete information is currently available. Furthermore, factors such as elevation, topographical exposure, distance to the sea, annual mean temperature, and annual mean wind speed also influence the spatial concentration of pollutants and were included in the modeling.

Annual air pollution removal was estimated as the total pollution removal flux in the areas covered by vegetation, where the removal flux (F; in t ha^{-1} $year^{-1}$) is estimated as:

$$F = V_d \cdot C \cdot 0.365 \qquad \text{(Formula B.3)}$$

where V_d is the deposition velocity of the pollutant to the leaf surface (in m s^{-1}) and C is the pollutant concentration (in µg m^{-3}), and 0.365 a coefficient used for units adjustment. Areas covered by vegetation were calculated by a combination of detailed land cover maps of urban green areas and forest, aggregated to 100 m resolution. For urban vegetation, the green layers of the Global Human Settlement Layer (GHSL) (JRC, IPSC, Ferri et al., 2014) were used. For forests, the High Resolution Global Forest map developed by Hansen (2013) was used. Both GHSL and Hansen map are by now the most detailed information available on vegetation cover in the case study area. In overlapping areas, the maximum value of both maps was applied. Final map of vegetation had values between zero (i.e., no vegetation) and one (i.e., totally covered by vegetation).

Table B.3. Input data for the processing of the air purification model. All the input variables were computed at 100 m of resolution (pixel size).

Component	Data description	Data source	Comments
Air pollution measure-ments	Average annual pollutant concentrations in BMR monitoring stations (NO_2) - year 2013	Air quality database (Catalan Government - http://qualitatdelaire.cat)	Exported to vector data (points)
Spatial predictors	Land cover dataset	SITxell database (www.sitxell.eu) (Barcelona Regional Council)	Converted from vector data (polygons)
	Digital Elevation Model (DEM)	SITxell database (www.sitxell.eu) (Barcelona Regional Council)	Resampled from 15x15 m raster (bilinear resampling)
	Average mean temperature (annual)	Climatic Digital Atlas of Catalonia (www.uab.es/atles-climatic)	Resample from 180x180 m raster (bilinear resampling)
	Average mean precipitation (annual)	Climatic Digital Atlas of Catalonia (www.uab.es/atles-climatic)	Resample from 180x180 m raster (bilinear resampling)
	Average wind speed at 60 m altitude from land surface (annual)	Environment geodatabase (Catalan Government)	Resample from 200x200 m raster (bilinear resampling)
	Population density grid	Census tract dataset (INE, 2011) Residential use classes extracted from land cover map (MCSC, 2009)	Intersect assuming equal population distribution within residential land for each census tract
	Road network	TeleAtlas® MultiNet™ dataset (update 2014)	
Vegetation map	Urban vegetation	Global Human Settlement Layer (GHSL JRC, IPSC, Ferri et al., 2014)	
	Forest vegetation	High Resolution Global Forest map (Hansen, 2013)	
	Permanent crops	SITxell database (www.sitxell.eu) (Barcelona Regional Council)	Extracted from land cover dataset

Table B.4. Cross-tabulation matrix between NO_2 concentration levels and population density used to obtain the air purification demand values (legend on the right). It assumes that the higher NO_2 concentration and population density the higher demand values. NO_2 concentration break consider the current NO_2 concentration limit in Europe (EU, 2008).

Population density (inhab. ha^{-1})	NO$_2$ concentration (µg m^{-3}) < 10	10 - 20	20 - 30	30 - 40	40 - 50	> 50
< 5	0	0	0	0	0	0
5 - 50	0	1	1	2	2	3
50 – 100	0	1	2	2	3	4
100 - 200	0	2	2	3	3	4
200 – 400	0	2	3	3	4	4
> 400	0	3	4	4	4	5

Legend	
0	Lowest demand
1	
2	
3	
4	
5	Highest demand

References

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Appendix C. Supplementary information for Chapter V

Description of ecosystem service mapping methods and data sources

Food (crops)

Crop production (supply indicator) in the BMR was estimated and mapped using two publicly available data sources: (1) 2013 agricultural yield statistical data (Catalan Ministry of Agriculture, 2013); and (2) a regional land cover dataset (Catalan Ministry of Territory and Sustainability, 2012). In order to clearly distinguish between crops for human food consumption and crops for other uses (fodder, materials, etc.), we received expert support from a regional farmers' union (*Unió de Pagesos*). Since the crop classes considered in the statistical data are more detailed than in the land cover map, we applied a table of correspondence between both categorizations following a previous study carried out by the farmer's union (Unió de Pagesos, 2013). For example, the statistical crop classes 'irrigated cereals', 'irrigated leguminous' and 'irrigated potatoes' were grouped into the agricultural land cover class 'irrigated herbaceous crops'. An average agricultural yield per agricultural land cover class (in kg ha^{-1} $year^{-1}$) was estimated and mapped considering the different corresponding statistical crop yields weighted by their relative areas (Unió de Pagesos, 2013).

Food (livestock)

Livestock production (supply indicator) data were taken from the 2009 Spanish Agricultural Census (INE, 2009). Unlike crop production, the share of total livestock production directly allocated to human food consumption is very difficult to estimate; hence we used total livestock units (LSU) as a proxy indicator. Eurostat[1] defines the livestock unit as "a reference unit which facilitates the aggregation of livestock from various species and age as per convention, via the use of specific coefficients established initially on the basis of the nutritional or feed requirement of each type of animal". The species considered in the case study area were bovine animals, sheep and goats, equidae, pigs, poultry and rabbits (breeding females). We mapped livestock units directly at the municipality level (normalized by area) because it is the smallest unit at which livestock census data were available. The number of livestock units produced per

[1] See http://ec.europa.eu/eurostat/statistics-explained/index.php/Glossary:Livestock_unit_(LSU)

farm and its localization, as used in other studies (e.g., van Oudenhoven et al., 2012), were not available.

Food (population density)

Food provision demand was mapped using population density as a proxy indicator. A population density grid was generated based on an spatial intersect between a census tract dataset (INE, 2011) and residential land classes extracted from a high resolution land cover map (MCSC, 2009) assuming equal population distribution within residential land for each census tract.

Global climate regulation

The annual rate of carbon sequestration as supply indicator of global climate regulation was estimated based on above-ground tree biomass maps for the province of Barcelona from Pino (2007). The author used empirical data from two Spanish forest inventories (IFN2 and IFN3) and applied a land use regression (LUR) model considering various spatial predictors such as land cover, elevation and various climate variables. Carbon sequestration was estimated and mapped from tree biomass net growth between the two inventories considering a biomass-carbon ratio of 0.5 which approximates the proportional mass of carbon in the tree species of the case study (Gracia et al., 2000-2004).

Demand for climate regulation was based on annual carbon emissions estimated for each BMR municipality. Estimates were collected from municipal Sustainable Energy Action Plans (SEAPs) corresponding to the year 2012 by the Barcelona Regional Council[2] accounting for emissions from sectors such as housing, transportation, services or waste management. Unfortunately, SEAPs' data did not include emissions from some relevant sectors such as industry or agriculture; therefore total values provide a first order estimate of the magnitude of carbon emissions at the municipal level.

Air purification

Methods and data sources for mapping the supply (NO_2 dry deposition velocity) and demand (NO_2 concentration levels) indicators of air purification in the BMR are fully described in Baró et al. (2016), hence here only a brief overview is provided. The supply indicator was estimated following the approach proposed by Pistocchi et al. (2010),

[2] See http://www.diba.cat/en/web/mediambient/pactealcaldes

which estimates deposition velocity (Vd) as a linear function of wind speed at 10 m height (w) and land cover type.

$$V_d = \alpha_j + \beta_j \cdot w \qquad \text{(Formula C.1)}$$

Where α and β are, respectively, the intercept and slope coefficients corresponding to each broad land cover type *j*, namely forest, bare soil, water or any combination thereof.

Concentration of NO_2 (demand) was estimated using a LUR model, a computation approach widely used for assessing air pollution at different scales (e.g., Beelen et al. 2013). The LUR model was built using NO_2 concentration measurements (year 2013) from the operational monitoring stations located in the BMR (n = 40) as dependent variable, and a set of spatial predictor parameters (i.e., independent variables) related to land cover type, geomorphology, climate, and population, that were considered to be the most relevant for distribution of NO_2 concentrations.

Erosion control

A biophysical indicator could not be calculated for the supply of erosion control due to data availability limitations, so we applied an expert-based matrix model (Burkhard et al., 2012) using the regional land cover dataset as spatial data (Catalan Ministry of Territory and Sustainability, 2012). We applied a table of correspondences between the CORINE land cover types used in Burkhard et al. (2012) and the regional land cover types.

Demand for erosion control was mapped using a land erodibility index map developed for the Geographic Information System for the Network of Open Areas in the province of Barcelona (SITxell[3]). The index is based on geomorphological and lithological factors, but it does not include climate variables such as rainfall. It defines four levels of erodibility, from 0 (negligible erodibility) to 3 (very high erodibility).

Outdoor recreation

Methods and data sources for mapping the supply (recreational potential index) and demand (recreational demand index) indicators of outdoor recreation in the BMR are fully described in Baró et al. (2016), hence here only a brief overview is provided. The model used here for assessing outdoor recreation focuses on nature-based

[3]See http://www.sitxell.eu/en/mapa_geologia.asp

recreational activities in the everyday life (Paracchini et al., 2014; Zulian et al., 2014). The rationale for assessing recreation capacity in this model can be summarized as follows: (1) the lesser human influence on landscapes, the higher value in terms of nature-based recreational potential; (2) protected natural areas and features (e.g., remarkable trees) are considered indicators of high recreational capacity; and (3) water bodies exert a specific attraction on the surrounding areas (Paracchini et al., 2014). Recreation capacity is hence mapped on the basis of the assessment of three components: degree of naturalness, nature protection, and presence of water. Each component was composed of one to four internal factors considered relevant in the case study of the BMR and for which spatial input data was available (see Baró et al., 2016). A score or weight (in the 0–1 range) was assigned to every factor standing for their relative importance or impact in terms of recreation potential. The final selection of factors and definition of scores was based on a consultation process (via focus group) with four experts working in environmental planning and territorial analysis for the Barcelona Regional Council. The final dimensionless value of recreation capacity was normalized in the 0-1 range.

Demand for outdoor recreation was mapped based on the availability of recreational sites (i.e., recreation capacity equal or higher than 0.4) close to people's homes and population density. A spatial cross-tabulation was carried out between a reclassified raster of Euclidian distances to recreation sites and the population density grid, assuming that all inhabitants in the case study area have similar desires in terms of (everyday life) outdoor recreational opportunities, but their level of fulfillment depends on proximity to recreation sites (see cross-tabulation matrix in Baró et al., 2016). The resulting raster indicates ES demand in residential land following a 0 (i.e., no relevant demand) to 5 (i.e., very high demand) value range.

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Appendix D. Additional research achievements during the Ph.D. period

My research achievements during the Ph.D. period (2013 - 2016) go beyond the completion of this Ph.D. dissertation. During this period, I had the opportunity to participate in other scientific publications and reports, and also to present my research in various conferences, meetings and seminars. Moreover, I have reviewed for several scientific journals, participated in project proposals and carried out other research activities. All these achievements are listed below.

D.1. Articles in international peer reviewed journals (published / submitted)

*Baró, F., Palomo, I., Zulian, G., Vizcaino, P., Haase, D., Gómez-Baggethun, E. 2016. Mapping ecosystem service capacity, flow and demand for urban planning and policy: a case study in the Barcelona metropolitan region. *Land Use Policy 57:405-417*. doi: 10.1016/j.landusepol.2016.06.006

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*Baró, F., Chaparro, L., Gómez-Baggethun, E., Langemeyer, E., Nowak, D. J., Terradas, J. 2014. Contribution of Ecosystem Services to Air Quality and Climate Change Mitigation Policies: The Case of Urban Forests in Barcelona, Spain. *Ambio* 43:466–479. doi: 10.1007/s13280-014-0507-x

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Depietri, Y., Kallis, G., Baró, F., Cattaneo, C. 2016. The urban political ecology of ecosystem services: The case of Barcelona. *Ecological Economics* 125:83–100. doi: 10.1016/j.ecolecon.2016.03.003

Langemeyer, J., Baró, F., Roebeling, P., Gómez-Baggethun, E. 2015. Contrasting values of cultural ecosystem services in urban areas: The case of park Montjuïc in Barcelona. *Ecosystem Services* 12:178–186. doi: 10.1016/j.ecoser.2014.11.016

Rodríguez-Rodríguez, D., Kain, J.H., Haase, D., Baró, F., Kaczorowska, A. 2015. Urban self-sufficiency through optimised ecosystem service demand. A utopian perspective from European cities. *Futures* 70:13–23. doi: 10.1016/j.futures.2015.03.007

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*Baró, F., Gómez-Baggethun, E., Haase, D. Ecosystem service bundles from a supply-demand approach: Implications for landscape planning and management in an urban region. *Ecosystem Services*. Submitted August 2016.

Haase, D., Kabisch, S., Haase, A., Baró, F., et al. Greening cities – to be socially inclusive? About the paradox of society and ecology in cities. *Current Opinion in Environmental Sustainability*. Submitted.

D.2. Book chapters, scientific reports and other publications

Baró, F., Gómez-Baggetun, E., Assessing the potential of regulating ecosystem services as nature-based solutions in urban areas: A multi-scale perspective. In: Kabisch, N., Bonn, A., Korn, H., Stadler, J. (eds.). Nature-based Solutions to Climate Change in Urban Areas - Linkages of science, society and policy. Under review.

Baró, F. 2015. A multi-scale assessment of regulating ecosystem services in Barcelona. In: Sergi Nuss-Girona, Mita Castañer (eds.): *Ecosystem Services: Concepts, methodologies and instruments for research and applied use*. Girona: Documenta Universitaria, 2015. (Quaderns de medi ambient; 6). ISBN 978-84-9984-308-7 Available via: https://www.documentauniversitaria.cat/botiga.php?a=llibre&id=825

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Kopperoinen, L., Stange, E., Rusch, G., Baró, F., Garcia-Blanco, G., Mederly, P. 2015. Integrating Nature-based Solutions in urban planning. OpenNESS project policy brief nº3. Available via: http://www.openness-project.eu/sites/default/files/OpenNESS_brief_03.pdf

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Potschin, M., Kretsch, C., Haines-Young, R., Furman, Berry, P., Baró, F. 2015. Nature-based solutions. In: Potschin, M. and K. Jax (eds): *OpenNESS Ecosystem Service Reference Book*. EC FP7 Grant Agreement no. 308428. Available via: www.openness-project.eu/library/reference-book

D.3. Oral presentations in conferences, meetings, courses and seminars

Baró, F. [author and speaker], Gómez-Baggethun, E., Haase, D. Ecosystem service bundles along the urban-rural gradient: Implications for landscape planning and management. European Ecosystem Services Conference, *Helping nature to help us*, University of Antwerp, Belgium, 19-23 September 2016.

Baró, F. [author and invited lecturer] Advance International Course on "Mapping ecosystem services for landscape planning". Lecture: Indicator based mapping (2h). International Centre for Advanced Mediterranean Agronomic Studies (CIHEAM), Zaragoza, Spain, 15-19 February 2016.

Baró, F. [author and invited speaker] A multi-scale assessment of regulating ecosystem services in Barcelona. XV International Summer School on the Environment (ISSE 2015), *Ecosystem Services: Concepts, methodologies and instruments for research and applied use*, University of Girona, Spain, 2-3 October 2015.

Baró, F. [first author and speaker], Langemeyer, J., Gómez-Baggethun, E. Integrating ecosystem services and green infrastructure in urban planning: Case study in the Barcelona Metropolitan Region. OpenNESS Annual meeting, 20-24 April 2015, Barcelona, Spain.

Baró, F. [author and speaker] Monetary & non monetary valuation of urban ecosystem services. URBES 3rd Training Session - *An in-depth insight in the results of the URBES Project*. Brussels, Belgium, 14 January 2015.

Baró, F. [author and speaker], Martín-López, B., Palomo, I. Ecosystem service supply and demand spatial (mis)matches. OpenNESS WP 3/4/5 Integrated Assessment & Valuation Workshop. Edinburgh, UK, 17-21 November 2014.

Baró, F. [first author and invited speaker] Chaparro, L., Gómez-Baggethun, E., Langemeyer, J., Nowak, D. J., Terradas, J. Contribución del verde de Barcelona a la calidad del aire y la mitigación del cambio climático. XVI *Congreso Nacional De Arboricultura*, Valencia, Spain, 22-24 October 2014.

Baró, F. [author and invited speaker] URBES. Estudi sobre urbanització i ecosistemes. "Ecotendències" conference. *Els ecosistemes naturals i la biodiversitat: un recurs economic?* Barcelona, Spain, 3 June 2014.

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Baró, F. [first author], Langemeyer, J., Gómez-Baggethun, E., Chaparro, L., Terradas, J. 2013. Quantifying regulating ecosystem services provided by urban forests in Barcelona, Spain. The 10th biennal conference of the ESEE 2013 Ecological Economics and Institutional Dynamics. Lille, France, 18-21 June 2013.

Turkelboom, F., Baró, F. [coauthor], et al. When you cannot have it all: Ecosystem services trade-offs in local planning contexts. European Ecosystem Services Conference, *Helping nature to help us*, University of Antwerp, Belgium, 19-23 September 2016.

Sanyé-Mengual, E., Baró, F. [co-author], et al. Enhancing ecosystem services in cities through multifunctional rooftop gardens. Insights from a co-designed pilot project in Barcelona, Spain. Growing in cities: Interdisciplinary Perspectives on Urban Gardening International Conference. Basel, Switzerland, 9-10 September 2016.

Turkelboom, F., Baró, F. [coauthor], et al. Ecosystem services real-world applications: The proof of the pudding is in the eating! The 8th Ecosystem Services Partnership (ESP) World Conference, *Ecosystem Services for Nature, People and Prospertiy.* Stellenbosch, South Africa, 9-13 November 2015.

Gómez-Baggethun, E., Baró, F. [coauthor] Ecosystem services provided across urban-rural gradients. Workshop Ecogradientes, UAM, Madrid, Spain, 7 July 2015.

Martín-López, B., Baró, F. [coauthor], Palomo, I. Integrated mapping of ecosystem services supply and demand. AGFORWARD (Agroforestry for Europe) WP 7 Meeting, Copenhagen, Denmark, 12 November 2014.

Langemeyer, J., Gómez-Baggethun, E., Baró, F. [coauthor] Integrating multiple ecosystem service values by social multi-criteria evaluation – an application to inform urban policy making. The 11th Biennial Conference of the International Society of Ecological Economics (ISEE), *Well-being and equity within planetary boundaries*. University of Iceland, Iceland, 13-15 August 2014.

Langemeyer, J., Haase, D., McPhearson, T., Kremer, P., Baró, F. [coauthor], Rall, E., Gómez-Baggethun, E., Frantzeskaki, N. Multi-criteria framework for enhancing relevance of urban ecosystem service assessments for policy-making. Society for Urban Ecology (SURE) World Conference, Berlin, Germany, 25-27 July 2013.

Frantzeskaki, N., Haase, D., Baró, F. [coauthor], Gomez-Baggethun, E., Artmann, M., Schewenius, M., Elmqvist, T., Kaczorowska, A., Kain, J-H., van Ham, C. 2013. Socio-ecological transitions of cities. Exploring spatial, ecological and governance dynamics in European cities. *The 4th International Conference on Sustainability Transitions*. Zurich, Switzerland, 19-21 June 2013.

Langemeyer, J., Baró, F. [coauthor], Gómez-Baggethun, E. 2013. Valuing urban ecosystem services: Three examples from Barcelona–Urban Gardens, Parks & Forests. Resilience Cities Conference, Bonn, Germany, 31 May – 2 June 2013.

D.4. Poster presentations in conferences and meetings

Baró, F. [first author and presenter], Langemeyer, J., Gómez-Baggethun, E. Integrating ecosystem services and green infrastructure in urban planning: Case study in the Barcelona Metropolitan Region. 2nd OpenNESS Annual meeting, Barcelona, Spain, 20-24 April 2015.

Baró, F. [first author and presenter], Langemeyer, J., Gómez-Baggethun, E. Sustainable planning of urban green infrastructure in the metropolitan region of Barcelona. 1st OpenNESS Annual meeting, Budapest, Hungary, 24-28 March 2014.

Baró, F. [first author], Haase, D., Gómez-Baggethun, E., Frantzeskaki, N. Assessing the mismatch between supply and demand of urban ecosystem services at city scale. A comparative analysis of European cities. Society for Urban Ecology (SURE) World Conference, Berlin, Germany, 25-27 July 2013.

Kopperoinen, L., Baró, F. [coauthor and co-presenter], et al. Integrating nature-based solutions in urban planning: examples from five case studies. *Nature-based solutions to climate change in urban areas and their rural surroundings - linkages between science, policy and practice*. European Conference on Biodiversity and Climate Change (ECBCC), Bonn, Germany, 17-19 November 2015.

Langemeyer, J., Baró, F. [coauthor], Gómez-Baggethun, E. Multi-Criteria Evaluation of Ecosystem Services – Informing Urban Green Infrastructrue Policies. 11th Biennal Meeting of the European Society for Ecological Economics (ESEE 2015): *Transformations*. Leeds, UK, 30 June – 3 July 2015.

D.5. Other research activities

Reviewer for the scientific journals: Plos One; Ecological Indicators; Environmental Impact Assessment Review; Moravian Geographical Reports.

Research stay in the Department of Landscape Ecology, Institute of Geography of the Humboldt-Universität zu Berlin, under the supervision of Dr. Dagmar Haase. September-October 2014.

Co-supervisor of the master student Mingyuan Zhao (MSc in Urban Environmental Management, Wageningen University) in her internship report "Mapping recreational opportunities in the province of Barcelona". April – July 2014.

Participation in the development of the project proposal “ENABLE” (Enabling green blue infrastructure in complex social ecological regions system solutions to wicked problems). BiodivERsA ERA-Net Joint Call 2015 (www.biodiversa.org).

Participation in the development of the project proposal “SCENERY” (Synergies and tradeoffs between food supply, ecosystem services and biodiversity in a European green, blue and grey infrastructure context). BiodivERsA ERA-Net Joint Call 2015 (www.biodiversa.org).

Participation in various scientific seminars of the research group LASEG, including: (1) The bibliometric approach to a researcher’s CV (10th December 2014); (2) How to write a scientific paper? (19th January 2015); (3) Ethics in research (in the field and in the office) (15th June 2015).

URBAN GREEN INFRASTRUCTURE

Modeling and mapping ecosystem services for sustainable planning and management in and around cities

In an increasingly urban planet, many cities and their inhabitants are facing multiple pressing threats within their borders, including heat stress, pollution and growing disconnection with the biosphere. Improving sustainability, resilience and livability in urban areas should be thus a major goal on the policy agenda, from local to global authorities. The operationalization of the ecosystem services framework, building on the concepts of 'green infrastructure' and 'nature-based solutions', is claimed by a mounting number of policy-makers, practitioners and scientists as the way forward to address many of these urban challenges. However, the extent to which urban green infrastructure can offer relevant solutions to these challenges is rarely considered in ecosystem service assessments, and therefore unknown to decision-makers. This dissertation critically examines the role and contribution of green infrastructure to cope with diverse urban challenges (with a focus on air pollution, greenhouse emissions, heat stress and opportunities for outdoor recreation), both at the city and metropolitan scales. The spatial scope of the research carried out within the assessment framework of this dissertation principally encompasses the urban area of Barcelona, Spain.

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