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Integrating Ecosystem Services in urban planning

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Del carattere degli abitanti d'Andria meritano di essere ricordate due virtù: la sicurezza in se stessi e la prudenza. Convinti che ogni innovazione nella città influisca sul disegno del cielo, prima d'ogni decisione calcolano i rischi e i vantaggi per loro e per l'insieme delle città e dei mondi.

Italo Calvino, "Le città invisibili", 1993 (1° ed. 1972), Oscar Opere di Italo Calvino, Mondadori Editore, p.147

As for the character of Andria's inhabiants, two virtues are worth mentioning: self-confidence and prudence. Convinced that every innovation in the city influences the sky's pattern, before taking any decision they calculate the risks and advantages for thenselves and for the city and for all worlds.

Italo Calvino, "Invisible cities", English translation by W. Weaver (1974), Harcourt Brace & Company

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Summary

In the last decade, urban ecosystem services (ES) have been a growing research field, and many authors acknowledge the potential of scientific findings for guiding decision-making towards a more sustainable urban development. In real-world cases, ES knowledge has demonstrated to improve decision-making at different levels: raising stakeholders' awareness and reframing dialogues, providing support for plans and policies, and guiding specific decisions. Nevertheless, the integration of urban ES in policy-making processes and governance practices is still at the beginning, and shortcomings emerge regarding the actual effectiveness of this science-policy interface. Among the decision-making processes that affect urban ES, spatial planning is arguably the most relevant. Land-use decisions made during planning processes determine both the availability of urban green and blue infrastructure, hence the supply of ES within the city, and the distribution of demand and beneficiaries. Integrating knowledge and concerns for urban ES in planning practices is therefore essential to secure their provision and to sustainably promote wellbeing and quality of life in cities. Exploring the integration of ES knowledge and approaches in urban planning processes and tools is the overall objective of this thesis.

The thesis is structured around four specific objectives. The first objective is to investigate the current level of integration of ES knowledge and approach in urban planning practices, thus understanding what ES information is already used and how a further integration of the ES approach could improve planning decisions. To this aim, a methodology for the content analysis of planning documents was developed and applied to a sample of 22 recently-approved urban plans of Italian cities. The review considered the inclusion of nine urban ES across three plan components (i.e., information base, vision and objectives, and actions). The high number of actions to address urban ecosystem services and the variety of tools for implementation that were found demonstrate that a certain level of integration already exists. However, only some ES (i.e. recreation and some regulating services linked to typical urban environmental problems) are widely addressed, while others are

hardly considered. Although the potential of ecosystem-based actions to tackle a wide range of urban issues is acknowledged, the low quality of information base, the lack of analysis of needs and demand, and the reference to good practices without any adjustment to the local conditions may lead to sub-optimal or even unwanted results, e.g. when potential trade-offs are overlooked. Future urban plans would benefit from a further appropriation of the ES approach by practitioners and decision-makers, and from a better integration of the growing ES knowledge in planning practices and tools.

The second objective is to enhance the usability of scientific findings on urban ES for urban planners. The aim is twofold: on the one hand, to identify and systematize scientific findings relevant for planning in a coherent conceptual framework; on the other hand, to provide guidance on how such findings can be used to support planning decisions. Moving from the results of the review, the analysis focused on urban regulating ES and addressed two specific barriers that prevent their integration in planning processes: the complexity of their biophysical foundations and the lack of indicators to explicitly account for the demand. An own framework was built by combining the 'Cascade conceptual model' with the supply-demand approach for mapping ES. The framework illustrates the different roles of urban green infrastructure, environmental conditions, and urban population and activities in determining the supply and demand of urban regulating ES, and identifies appropriate indicators to describe their features and interactions in a spatially-explicit way. Moreover, it identifies the entry points and the pathways through which urban planning affects the provision of regulating ES and related benefits in cities. The framework is applied to distil and systematize a fragmented scientific evidence on urban regulating ES, collected through a review of a wide literature. Planning-relevant knowledge, methods, and indicators are organized according to the main components of the framework, thus providing planners with a useful tool to support the design of planning actions and the assessment of their expected outcomes.

The third objective aims to test the integration of the ES approach to support real-life urban planning decisions. Different stages of the planning process are identified as possible entry points for ES knowledge to inform planning decisions, with different associated requirements. The research focuses on the use of ES assessments as criteria to evaluate planning scenarios. The case study deals with the prioritization of greening interventions to regenerate brownfield sites in the city of Trento (Italy). The benefits generated by alternative planning scenarios in terms of improved cooling effect by vegetation and enhanced opportunities for nature-based recreation are quantified based on the number of beneficiaries broken down into different vulnerability classes, and then compared through a multi-criteria analysis. The application demonstrates the potential of beneficiary-based indicators, coherent with social-oriented planning objectives, to integrate ES knowledge in the stage of urban planning processes where decisions among alternative options are to be made. Moreover, it shows the benefits of multi-criteria analysis techniques, which allow integrating ES information with other diverse inputs, exploring different stakeholder perspectives, and balancing the trade-offs between the enhancement of urban ES and other competing objectives.

The fourth objective is to frame the integration of ES in urban planning in the wider context of European spatial strategies for sustainable urban development. Beyond the 'green city' strategy supported by the ES approach, five other main spatial strategies agreed-upon at the EU level are identified through a content analysis of 30 policy documents. These are: 'compact city', 'urban regeneration', 'functional mix', 'no land take', and 'high density'. A set of indicators based on land use-land cover data allows measuring the level of coherence between the strategies and the recent spatial development trends of 175 European cities. The results reveal relationships between the observed trends, population dynamics, and geographical location of the cities, and suggest the presence of multiple factors, including path dependencies and land-use legacies, that may catalyse or hinder the implementation of the strategies. Furthermore, the findings highlight potential conflicts, as well as synergies and trade-offs among the strategies, which should be carefully considered. Hence the need for a simultaneous monitoring of multiple spatial features when assessing urban development trajectories and the importance for urban planning of accounting for the mutual relations among multiple strategies toward sustainability.

Despite the increasing call for ecosystem-based actions and the synergy with other approaches already accepted in the planning practice (e.g., green infrastructure), integrating ES in current planning processes and tools still requires some efforts from both sides of the science-policy interface. This thesis contributed by identifying common gaps and shortcomings in the way urban ES are currently addressed, providing guidance to include considerations for urban regulating ES in planning practices, testing the usability ES assessments to support real-life planning decisions, and identifying the relation of the ES approach with other spatial strategies for sustainable urban development. However, case studies, samples, and methods considered are necessarily limited, and allowed only for a narrow view on such a broad topic. Further work is needed to test the validity of the results in different contexts and, most importantly, their usability in real-world decision-making processes.

Chapter 1

Scope of the thesis

1.1 Background

1.1.1 Ecosystem services: conceptual framing

Human life on Earth depends on ecosystems. This is the main message conveyed by the concept of ecosystem services (ES), which has gained an ever-increasing attention in the scientific (McDonough et al., 2017) and policy debate (e.g., CBD, 2011; European Commission, 2006, 2010) of the last two decades. The success of the term ‘ecosystem services’ is arguably due to its encompassing all “the direct and indirect contributions of ecosystems to human wellbeing” (TEEB, 2010a), thus providing a comprehensive framework within which the multiple relations between humans and nature can be described and analysed.

Although the concept was not entirely new - the *IUCN Conservation Strategy* in 1980 had already mentioned goods and services provided by nature - the term ‘ecosystem services’ appeared for the first time in 1981 in a book by Ehrlich and Ehrlich as an evolution of the term ‘environmental services’ (Ehrlich and Ehrlich, 1981). Its fortune was not immediate and, in the following years, the ES concept remained confined within the disciplinary boundaries of conservation ecology, somehow in the background of the debate around sustainable development. As an evidence of this, Marion Potschin and colleagues, in the brief history at the beginning of their Handbook of ecosystem services (Potschin et al., 2016), note its surprising absence in the *Brundtland report* (UN, 1987), and the explicit reference to ES in the *Agenda 21* five years later (UN, 1992). But it is only in the late Nineties that ES are brought to the forefront of the scientific debate thanks to two pioneering works. In 1997, Gretchen Daily provided the first comprehensive overview of the ES through which nature underpins human wellbeing (Daily, 1997), while a group of ecologists and economists made the first attempt to estimate, based on ES, the total economic value of the biosphere (Costanza et al., 1997), generating a rapidly-growing interest in the topic.

In 2005, the publication of the *Millennium Ecosystem Assessment* report (MA, 2005) under the umbrella of the United Nations Environmental Programme (UNEP) put ES high on the world policy agenda. The ES concept was proposed as an innovative way to communicate the growing concerns for the unprecedent rates of ecosystem degradation and biodiversity loss, thus providing an additional justification for nature conservation based on what nature does *for* people (Mace, 2016, 2014). The framework developed by the MA details the pathways through which biodiversity affects the different constituents of human wellbeing (i.e., security, basic material for good life, health, and good social relations), ultimately determining people's freedom of choice and actions. The pathways represent the different ES, classified and characterized in terms of current trends, main drivers and pressures, and expected changes under plausible scenarios for the future.

A step forward in framing the ES concept was the so-called 'Cascade conceptual model' proposed by Haines-Young & Potschin (2010). The flow of ES from nature to society sketched out by the MA is here detailed following a stepwise approach. At the origin of the cascade, within the natural domain, biophysical structures and processes generate ecosystem functions that support the provision of ES. Downstream, within the socio-cultural domain, are the benefits produced by ES, which can be associated to different values. By breaking down the concept into its component parts, the 'Cascade' helped to gain clarity on the mechanisms that determine the provision of ES, and on the terminology adopted to describe ES-related processes. Despite some shortcomings can be highlighted in the simplistic view suggested by the metaphor, where ES seem to flow effortlessly to passive beneficiaries (Braat and de Groot, 2012), the ‘Cascade’ set a common basis for the development of the ES science in the following years, and proved to be a useful tool in the communication and operationalization of the concept (Jax et al., 2017; Potschin-Young et al., 2017).

Further refinement of the ES concept in the following years indeed built upon the ‘Cascade’, mostly focusing on human inputs required ES production and feedback loops from the socio-cultural sphere to the ecosystem and biodiversity domain. This is the case of the ‘TEEB diagram’ (de Groot et al., 2010b), which highlighted the role of institutions in determining ES use and management interventions on the supporting ecosystems, based on the perception of ES values and the related preferences and desires. The integration of the ‘Cascade’ with the DPSIR (Driver-Pressure-State-Impact-Response) model (Müller and Burkhard, 2012) and its reverse reading as a stairway of different management activities (Spangenberg et al., 2014) followed a similar rationale. These expansions of the ES framework allowed for a more holistic view of the relationships between nature and society, and opened to the use of the ES concept for informing and supporting a wider range of decisions than those strictly related to nature conservation (see Section 1.1.3).

An example of initiatives aimed at mainstreaming the ES concept in policy- and decision-making is *The Economics of Ecosystems and Biodiversity - TEEB*, initiated in 2007 under the auspices of the G8+5. Focusing on the feedback loops that connect ES values to ES use and ecosystem management, the TEEB moved from the assumption that economic values could be valid arguments to oppose the loss of biodiversity and ES resulting from short-sighted policies. The aim was to demonstrate that, if biodiversity and ES values are explicitly considered, in a medium-to-long term perspective, biodiversity conservation and sustainable use of natural resources are cost-effective options that allow increasing the overall benefits. The TEEB approach consists of three steps: recognizing, demonstrating, and capturing values (TEEB, 2010a), which encompass the whole decision-making process. Its application in different settings have produced a long series of studies, covering a wide range of topics in many countries and regions across the globe (http://www.teebweb.org/).

The predominantly-linear shape of the ES framework was overcome in most recent year through its full integration within the concept of socio-ecological systems. An example is the circular scheme by Reyers et al. (2013), where 'nature' and 'society' are no more presented as separate entities, but complex socio-ecological processes determine ES provision, use, and management. The formalization stresses the presence of ES bundles and of socio-ecological production functions, coherently with the paradigm of 'nature *and* people' that is progressively emerging as a substitute of 'nature *for* people' also in the biodiversity conservation discourse (Mace, 2014). This framing of the ES concept in the socio-ecological system approach, indicated as a needed advancement beyond that offered by the MA (Carpenter et al., 2009), is today seen as the most mature and promising ground from which the discipline can continue to grow (Bennett and Chaplin-Kramer, 2016; Mace, 2016).

Since its appearance, the concept of ES has been subject to many criticisms, the most frequent of which have been well summarized, with related counter-arguments, by Schröter et al. (2014). Most of the critiques arise from the anthropogenic perspective of the concept, which could imply a latent justification for the commodification of nature and the potential conflict with biodiversity conservation objectives. In the most recent framing of the ES concept within the socio-ecological system perspective, many of these critiques appear weakened, when not fully overcome. Moreover, although some of them may be valid as hypothesis, 20 years of applications have demonstrated that good intentions have largely prevailed over misleading uses. Despite not bringing significant advancements to traditional scientific disciplines, the ES concept has proved to be a fertile ground for trans-disciplinary studies (Fürst et al., 2017; Jacobs et al., 2016; Schröter et al., 2014) that today represent an established branch within the field of sustainability science (Kates et al., 2001), addressing the relation between human life and ecosystems with the ultimate aim of promoting a sustainable use of natural resources (Abson et al., 2014; Bennett and Chaplin-Kramer, 2016; Crouzat et al., 2014).

1.1.2 Classifying and assessing ecosystem services

One of the main concerns that followed the raise of the ES concept was that of classification. Since any ES-based application starts with selecting the ES to be assessed from a (defined or implicit) list (La Notte et al., 2017), finding agreed definitions and classification systems is a prerequisite of any attempt to measure and map ES (Burkhard and Maes, 2017). Moreover, the integration of ES into policies at various levels requires standardized definitions for monitoring, communicating, and comparing results (La Notte et al., 2017). The first classification proposed by the Millennium Ecosystem Assessment (MA, 2005) distinguished among four main ES categories: provisioning ES (e.g., provision of food, water, and timber), regulating ES (e.g., climate regulation, water purification, flood regulation), cultural ES (e.g., recreation, sense of place, aesthetic quality), and supporting ES (e.g., nutrient cycling, soil formation, habitat provision), largely taken-up by the successive literature. Upon the MA classification are based both the classification systems proposed by TEEB, where supporting ES were merged with regulating (TEEB, 2010a), and the classification of Nature's Contributions to People into material, non-material, and regulating proposed by the *Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (IPBES, 2017).

A more structured approach to the classification problem was proposed by the widely-adopted *Common International Classification of Ecosystem Services* (www.cices.eu, last accessed February 2018), based on five hierarchical levels. Version 4.3 of the CICES, released in 2013, identified 48 ES classes, which have been further expanded to consistently include abiotic ecosystem outputs in the most recent version 5.1 (January 2018). The hierarchical structure of the CICES supports ES accounting and allows accommodating local concerns and specificities at the lower levels (i.e., class and class type) while maintaining the required coherency for comparing aggregate ES values among larger geographical units at the higher levels (i.e., division and group). To avoid overlaps and double-counting, CICES defines and classifies only 'final services', i.e. ES according to the rationale of the 'Cascade model', while supporting services and functions (also called 'intermediate services') are not considered. A similar approach was also followed by the US-EPA *Final Ecosystem Goods and Services Classification System* (FEGS-CS, https://www.epa.gov/ecoresearch/final-ecosystem-goods-and-services-classification-system, last accessed February 2018), whose main specificity is the detailed classification of beneficiary typologies (Landers and Nahlik, 2013).

Much of the scientific efforts around ES have focused on the development, selection, and application of appropriate methods and indicators to assess them. While at the beginning the focus was mostly on the analysis of ecosystem functions and processes leading to ES, already the MA classification highlighted the multitude of values associated to ES (Jacobs et al., 2016) and the problem of capturing and measuring all of them, which emerged even more clearly under the economic lens of the TEEB initiative (TEEB, 2010a). Considering the type of values that they aim to capture, ES assessment methods are commonly classified in biophysical, socio-cultural, and economic methods (Harrison et al., 2017). Biophysical methods quantify ES in biophysical units based on the analysis of structural and functional traits of ecosystems, or on biophysical modelling (e.g., hydrological and ecological models, production functions). Socio-cultural methods capture individual or social preferences expressed by stakeholders in non-monetary terms (e.g., time use assessments, photo series analysis). Economic methods quantify ES values in monetary units (e.g., market prices, replacement cost, hedonic pricing). Although the distinction is sometimes blurred (e.g., methods to investigate social preference can be used to assign monetary values), it helps to understand the variety of methods from different disciplinary backgrounds that can be adopted in ES assessments.

Considering the scope, ES assessment methods can be categorised according to the focus on ES supply (Martínez-Harms and Balvanera, 2012) or demand (Wolff et al., 2015). The supply side includes the investigation of both the potential or capacity of ecosystems to provide ES and their actual use or flow (Bastian et al., 2012; Villamagna et al., 2013). The demand side may consider the actual beneficiaries of ES or the total (satisfied and un-satisfied) demand in a certain area, arising from certain activities or population groups. While most ES assessments have targeted the supply side, the analysis of demand reveals who benefits from ES and allows detecting winners and losers of changes in ES provision. Moreover, a comparison between supply and demand can highlight mismatches in terms of unsatisfied needs or unsustainable management practices, but requires a clear definition of the problem and the selection of comparable indicators (Baró et al., 2015; Burkhard et al., 2012).

Many ES assessments explicitly consider the spatial dimension, producing ES maps that show the variation of ES values across space. Since both ecosystems and the pressures and impacts on them are spatial entities, mapping ES has become one of the most prominent field of ES research (Burkhard and Maes, 2017). Mapping approaches and techniques reflect the variety of methods earlier described and combines a wide range of spatial and non-spatial information, from remotely-sensed data to GPS tracks. Mapping ES is an effective way to visualize and communicate results of ES assessments, hence a powerful tool to inform decision-making (Burkhard et al., 2013; Hauck et al., 2013b), especially analysing the expected consequences of planning actions at different scales (Albert et al., 2015; Baró et al., 2016).

While today many mapping and assessment methods are well-established in the research field and have demonstrated their potential to provide useful information to decision-making, the challenge now is on how multiple ES assessments can be integrated to contribute to answer real-world policy questions. On the one hand, decisions usually affect not a single but a bundle of ES (Jopke et al., 2015; Spake et al., 2017), hence assessments able to account for multiple ES and their multiple values are needed to investigate synergies and trade-offs potentially arising from decisions. On the other hand, ES assessments should be able to reflect views and opinions of the different stakeholders involved in the decision-making process, including those that are normally under-represented (Jacobs et al., 2016). Some approaches that can be used to this purpose have been identified (e.g., multi-criteria decision analysis, Bayesian Belief Networks, participatory scenario development) (Dunford et al., 2017; Harrison et al., 2017; Langemeyer et al., 2018; Saarikoski et al., 2016), but advancements in this direction are still needed and represent one of the main challenges for the next development of ES science (Burkhard et al., 2018; Jacobs et al., 2017).

1.1.3 Ecosystem services informing decisions*

What characterized the ES concept since its origin was the explicit link with decision-making. Gretchen Daily and colleagues identified in this link the main innovation of the ES approach, where ES values are acknowledged and assessed with the specific purpose of informing decisions (Daily et al., 2009). Highlighting the dependency of human wellbeing on nature, the ES concept definitely makes clear that no trade-off can exist between sustainable human development and nature conservation (de Groot et al., 2010a). Consequently, identifying, mapping, quantifying, and valuing ES is expected to improve decision making, ultimately promoting more sustainable development trajectories (Díaz et al., 2015; Guerry et al., 2015; TEEB, 2010b). In the last years, efforts have been made to include ES in different decision-making processes to support the identification and comparison among costs and benefits of different policies (TEEB, 2010b) and to contribute to the assessment of their impacts (Geneletti, 2013).

At the international level, the acknowledgement of the need to secure a sustainable and fair provision of ES was explicitly at the basis of the adoption of the *Aichi-targets* by the *Convention on Biological Diversity* (2010) and of the creation of the *Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (2012). The European Union is at the forefront in pursuing these obligations and is leading the way toward mainstreaming the ES approach by progressively embedding the ES concept in its policies (Bouwma et al., 2017). Through the *EU Biodiversity strategy to 2020*, EU Member States committed to map and assess ES in their territory, thus setting the base for a continuous monitoring and the inclusion of ES in the system of national accounting and reporting across the EU (Maes et al., 2016, 2012). The MAES working group was established as a scientific support to develop a coherent framework to be consistently applied across the EU (Maes et al., 2013). The MAES working group also provided practical guidance to implement the framework, and a list of possible indicators to map and assess the ES provided by different types of ecosystems (Maes et al., 2016).

Comprehensive ES assessments have also been realized at national level, both in the EU, to comply with target 2 of the *EU Biodiversity strategy to 2020* (Schröter et al., 2016), and in other parts of the world,

* The title was inspired by the paper: Guerry et al. (2015). Natural capital and ecosystem services informing decisions: From promise to practice. *Proceedings of the National Academy of Sciences, 112*(24), 7348–7355. http://doi.org/10.1073/pnas.1503751112

especially in relation to the TEEB initiative (http://www.teebweb.org/, last accessed February 2018). Furthermore, several local experiences have proven the effectiveness of the ES approach in driving policy changes toward more sustainable outcomes in different contexts and scales (Ruckelshaus et al., 2015). Addressed topics include river basin management, climate change adaptation and mitigation, green infrastructure planning, and corporate risk management, to name just a few (Dick et al., 2017; Ruckelshaus et al., 2015), with a wide range of stakeholders involved in different decision-making processes, from landscape and urban planning (Albert et al., 2014a; Hansen et al., 2015) to impact assessment (Geneletti, 2016; Rozas-Vásquez et al., 2018). Overall, thanks to its capacity of tailoring ecological knowledge to decision makers who have as main concerns social or economic objectives (Schleyer et al., 2015), the ES concept has progressively emerged as a tool to support and inform a wide range of decisions.

Investigating the mechanisms through which ES knowledge is integrated in decision-making processes, Mckenzie and colleagues identified different levels of integration, corresponding to different purposes (Mckenzie et al., 2014). At the conceptual level, ES knowledge helps raising stakeholders' awareness and reframing dialogues, thus enabling people to develop new ideas and values. At the strategic level, it promotes and provides justification for plans and policies. At the instrumental level, it guides specific decisions taken by rational and informed decision-makers. Hence different levels of integration correspond to different expected impacts on the two sides of this science-policy interface, from the co-production of knowledge for research use, to the definition of specific actions and policies that improve biodiversity and ecosystem health along with human wellbeing (Posner et al., 2016b). Although distinctions are rarely as clear as classifications assume, and different modes of knowledge use are generally combined within different stages of the decision-making process (Mckenzie et al., 2014), the scheme in Figure 1.1 helps to frame the integration as a process itself and to understand what benefits it can provide.

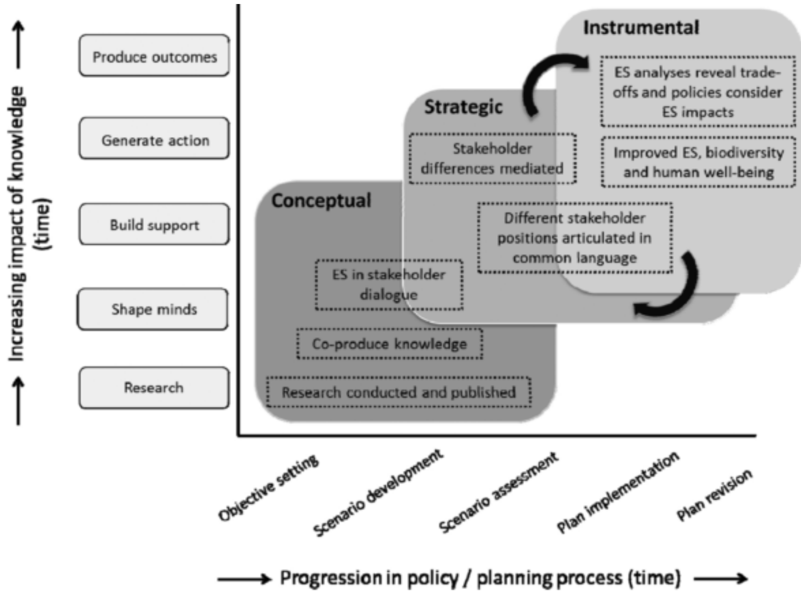


FIGURE 1.1: *The three levels of integration of ES knowledge in decision-making processes: main tasks, corresponding stages of the planning process, and expected impacts of the conceptual, strategic, and instrumental modes of knowledge use. (Source: Mckenzie et al., 2014).*

1.2 Ecosystem services and urban planning

1.2.1 Understanding the relationship between ecosystem services and urban planning

Even though cities may seem to have little to do with the concept of ES, except for largely benefitting from them while threatening through urbanization processes their provision (MA, 2005), this view has progressively shifted during the last years. While the ES science was developing, cities started to be seen not just as consumers of ES supplied from outside urban areas, but also as producers themselves, as already noted in the seminal work by Bolund and Hunhammar (1999). The study of urban ES, i.e. of the "ES provided by urban ecosystems and their components" (Gómez-Baggethun and Barton, 2013), became a focus of ES research (Haase et al., 2014b; Luederitz et al., 2015). Regulating and cultural ES emerged as the most relevant in urban areas (Elmqvist et al., 2016; Gómez-Baggethun and Barton, 2013). By regulating stormwater runoff and flows, purifying the air, regulating microclimate, reducing noise, and moderating environmental extremes, urban ecosystems affect the quality of the urban environment and control the associated hazards. Moreover, by providing suitable space for recreation, increasing the aesthetic quality of urban spaces, offering opportunities for cultural enrichment, and preserving local identity and sense of place, they provide a range of non-material benefits that are essential for human and societal wellbeing in cities (Elmqvist et al., 2016; Gómez-Baggethun and Barton, 2013).

Preserving, restoring, and enhancing urban ES is therefore necessary to ensure liveable, sustainable, and resilient cities (Botzat et al., 2016; Frantzeskaki et al., 2016; McPhearson et al., 2015). Urban ES and associated benefits are linked to many of the most pressing challenges for cities. Mitigating and adapting to climate change, promoting citizens' heath, enhancing social inclusion, and reducing the environmental footprint of cities, to name just a few, all have a direct relation with the provision of urban ES (Bowler et al., 2010; Demuzere et al., 2014; McPhearson et al., 2014). Furthermore, many urban ES produce effects only at the local level (Andersson et al., 2015) and man-made substitutes, when existing, have often high costs and impacts (Elmqvist et al., 2015). Considering the growing demand determined by a booming urban population, maintaining healthy and functioning urban ecosystems appears of utmost importance to guarantee that sustainability goals are met.

Urban planning affects urban ES in multiple ways. First, the provision of urban ES depends on the availability and spatial distribution of urban ecosystems and their components, hence on the strategic decisions on land-use allocations that are made during urban planning processes (Langemeyer et al., 2016). Second, by defining the spatial arrangement of land uses, urban planning also determines the distribution of population and urban functions, which affects the demand for urban ES (Baró et al., 2016). Third, spatial planning also determines other properties of the city physical structure (e.g. accessibility) that play a key role in defining who can benefits from urban ES (Barbosa et al., 2007). Hence, making urban planning aware of ES and their values, and assessing the impacts of planning actions on their provision, is fundamental to ensure that benefits from ES are preserved and enhanced.

Acknowledging the presence of nature within cities as beneficial is not an innovation in the urban planning discipline, and references to the importance of green spaces in cities and to their positive influence on the wellbeing of urban population can be traced back to the very initial stage of modern planning. It should be enough to cite here Howard's 'garden city' model that, "being a healthy, natural, and economic combination of town and country life", aimed "to raise the standard of health and comfort of all true workers of whatever grade" (Howard, 1902). However, in the last century, a view of nature in cities as only related to aesthetic and recreational values prevailed, and a strong focus on urban form as a determinant of the environmental performance of cities made other strategies, such as compactness, density, and functional diversity, prevail even when the then new paradigm of sustainability emerged (Jabareen, 2006). Only recently, also thanks to a growing scientific evidence, 'greening the city' has become an imperative for urban planning. The concepts of 'ecosystem-based actions' (Brink et al., 2016; Geneletti and Zardo, 2016) and 'nature-based solutions' (Raymond et al., 2017) applied to cities suggest the active promotion of urban ES and related benefits to sustainably tackle a wide range of urban challenges. The inclusion of such approaches in urban planning, design, and management is receiving wide support, to the point that the European Union is financing their implementation through a funding line of the Horizon 2020 Work Programme.

Within this framework, the integration of ES knowledge and approach in urban planning is indicated from many sides as a valuable strategy to address some of the 'wicked' problems of todays' urban development, from the necessary transition to resilience (Collier et al., 2013) to the need for sustainable approaches to address urban peripheries (Geneletti et al., 2017). That's why the inclusion of ES in urban plans started to be considered an indicator of their quality (Woodruff and BenDor, 2016), ultimately measuring their capacity to put in place strategic actions towards more sustainable and resilient cities (Frantzeskaki et al., 2016).

1.2.2 Integrating ecosystem services in urban planning: expectations and challenges

Integrating the ES concept in urban planning processes is expected to provide multiple benefits. First, to clarify the ecological - structural and functional - foundations of ES provision, thus highlighting the links between human wellbeing and the state of ecosystems (Haines-Young and Potschin, 2010), hence the role of ecological knowledge in supporting effective planning actions (Schleyer et al., 2015). Second, to raise awareness on the whole range of ES and associated benefits that are produced by urban ecosystems, thus providing a comprehensive understanding of the values at stake and of the trade-offs that may arise from land-use decisions (de Groot et al., 2010a). Third, to support the explicit identification of beneficiaries, including those normally under-represented in decision-making processes, thus promoting concerns for environmental justice (Ernstson, 2013) and strengthening planners' arguments in balancing public and private interests (Hauck et al., 2013c).

From an operational perspective, ES science has produced a large variety of methods and tools that are available for decision-makers to implement the approach (Grêt-Regamey et al., 2017; Harrison et al., 2017). Urban planning benefits particularly from the possibility of mapping ES, i.e. of localizing supply and demand across the city and making the flow of the services and the variations in the associated values spatially explicit (Burkhard and Maes, 2017; Hauck et al., 2013b; Martínez-Harms and Balvanera, 2012; Wolff et al., 2015). Mapping ES should help to identify weaknesses and opportunities for planning actions, and lead to a more informed assessment of their expected outcomes.

Various stages of the planning process are expected to benefit from a successful integration of ES knowledge and approach. These include: the analysis of current conditions and the identification of existing needs, the definition of goals and expected performances, the design and assessment of alternatives, the prioritization of the most effective solutions, as well as the monitoring and follow-up on decisions (Adem Esmail and Geneletti, 2017; Geneletti, 2015; Langemeyer et al., 2016). Improvements are not only related to the contents, but also to the process itself. In fact, the ES concept can provide a common language and act as a 'boundary object' that facilitates communication and the integration of multiple views and values, leading to inclusive and collaborative decision-making and, ultimately, to more robust and better decisions (Adem Esmail et al., 2017; Mckenzie et al., 2014; Schleyer et al., 2015).

Despite all these expectations, the integration of ES in decision-making is still scarce (Albert et al., 2014a; Ruckelshaus et al., 2015; Saarikoski et al., 2017; Slootweg, 2015). Part of the reason is in a research discipline not yet fully established, as demonstrated by the still uncertain, sometimes ambiguous and even contested terminology (Schröter et al., 2014). Focusing on urban ES research, Haase et al. (2014), Kremer et al. (2016), and Luederitz et al. (2015) summarized the main challenges to face. Among others, they identified the need for more appropriate methods and indicators able to capture the heterogeneity and fragmentation of urban ecosystems, a scarce investigation of the relation between urban ES and biodiversity, the uncertainty about the degree of transferability of data and results, and the lack of analyses that account for ES demand by integrating people's preferences and values, particularly in the assessment of cultural ES. Hence, scientific advancements are still needed to support a full integration of the ES concept and approach in urban planning.

On the other side of the science-policy interface, Saarikoski et al. (2017) discussed the main institutional challenges to the integration of ES knowledge in decision-making, most of which require time and the joint effort of practitioners and institutions to be overcome. However, when analysed from the specific perspective of urban planning, many of the described barriers appear less strong. An example are the difficulties that arise in involving multiple perspectives in the decision-making process. Urban planning is commonly a multi-actor process with a certain degree of participation ensured by legally-binding procedures (e.g., Strategic Environmental Assessment). Furthermore, despite competing interests in the use of land, actions aimed at conserving, restoring, and enhancing urban ecosystems are generally supported by citizens and communities owing to the multiple benefits they provide, while trade-offs between provisioning and regulating or cultural ES are not so relevant in urban areas. Finally, the same use of the term 'ecosystem service', which may be contested in certain contexts, is more easily accepted in urban planning processes where it can be associated to the concept of public services and facilities that administrations must guarantee.

Overall, urban planning today appears both a promising context where to integrate the ES approach, and a discipline in need of integrating ES knowledge to implement innovative strategies and actions toward sustainable urban development. While scientific advancements would make ES knowledge more credible, and overcoming institutional barriers to its use in decision-making processes would make it more legitimate, applied research working on the interface between science and policy is the only way to increase the relevance of scientific findings as a prerequisite for their actual use to inform real-life decisions. As Cowling and colleagues put it: “As a mission-oriented discipline, ES research should be user-inspired and user-useful, which will require that researchers respond to stakeholder needs from the outset and collaborate with them in strategy development and implementation” (Cowling et al., 2008). The thesis commits to this mission, and deals with the problem of making scientific findings on ES relevant for urban planning.

1.3 Objectives and structure of the thesis

1.3.1 Objectives and research questions

The overall objective of the thesis is to explore the integration of ES knowledge and approaches in urban planning, understanding how knowledge, concepts, methods, and tools developed within the ES science can offer a valid support to improve urban decision-making toward sustainable and resilient cities. As briefly explained in the previous section, several challenges characterize the integration of ES knowledge and approach in urban planning from both sides of the science-policy interface. This thesis aims to contribute to overcome some of them, by providing a critical view on what such integration can be useful for, what it actually means with respect to urban planning practices, how it can be operationally pursued in real-life planning processes, and what results it can be expected to produce. Accordingly, the thesis is structured around four specific objectives and associated research questions.

The first objective is to investigate the level of integration of ES knowledge and approach in current urban planning practices, thus understanding what ES information is already used, how, and for which purpose. This should reveal what needs are still to be addressed, what opportunities and potentials of the ES approach are still to be exploited, and what benefits can be expected from a further integration of ES in urban planning. This first objective sets the basis of the whole research, and is functional to refine the focus of the other objectives. The research questions associated to the first objective are:

What is the current level of integration of ES knowledge in urban planning practices? In which aspects a further integration of the ES approach could improve urban planning contents and decisions?

The second objective is to enhance the usability of scientific findings on urban ES for urban planners. Usability is a pre-requisite for knowledge to be integrated in decision-making processes, hence to become influential and produce impacts. The aim here is twofold: on the one hand, to select scientific findings relevant for planning and systematize them in a coherent conceptual framework; on the other hand, to provide guidance to planners on how such findings can be used to inform and support planning decisions. The research questions associated to the second objective are:

What scientific findings on urban ES are relevant for urban planning? How can relevant ES knowledge be operationalized in the design of planning actions?

The third objective is to test the application of the ES approach to real-life planning decisions. Exploring the integration of ES knowledge in urban planning through a case study helps to answer operational questions related to the role that ES knowledge can play within the planning process. In fact, different stages of the decision-making process define different requirements for ES knowledge to be considered relevant and usable. Hence, also the most appropriate methods and indicators vary across the stages. The research question associated to the third objective is:

Are ES assessment methods and indicators suitable to support a real-life urban planning decision?

The fourth objective is to frame the integration of ES in urban planning in the wider context of spatial strategies for sustainable urban development. Even when looking just at the aspects related to the urban form and the impact of land use arrangements on city performance, the strategy of preserving and enhancing urban ecosystems supported by the integration of ES in urban planning is not sufficient, alone, to make the spatial development of cities sustainable. Other issues besides the quantity and spatial distribution of urban green infrastructure must be addressed (e.g., the expansion of urban areas and the reuse of urban voids) and other strategies are needed and promoted, at multiple institutional levels, to guide sustainable urban planning. This objective aims to go beyond the ES approach explored so far to identify the main spatial strategies for sustainable urban development and to investigate the relations among them, thus highlighting synergies and trade-offs and understanding to which extent they are mutually reinforcing, or rather conflicting. Accordingly, the research questions associated to the fourth objective are:

In addition to the enhancement of urban ecosystems, what other spatial strategies exist to pursue sustainable urban development in urban planning? What synergies and trade-offs can be expected among the different strategies?

The thesis is grounded on the hypothesis that the integration of ES in urban planning has the potential to enhance the quality of decision-making. This conviction will be questioned, demonstrated, and detailed in the following chapters. However, a wider perspective is needed to avoid focusing on what is just a tool, and missing the true goal of "making cities and human settlements inclusive, safe, resilient and sustainable" (UN General Assembly, 2015, Sustainable Development Goal n.11).

1.3.2 Outline

The thesis is organized into five chapters. Chapter 2, Chapter 3, Chapter 4, and Chapter 5 address the four specific objectives mentioned in the previous section and contribute to explore different levels of integration.

Chapter 2 investigates the current level of integration of ES knowledge and approach in urban planning by reviewing the content of a sample of recent urban plans. Although some research works have already investigated this issue from different perspectives, most of them approached the problem as the uptake of a new concept in existing processes. The analysis here moves from the hypothesis that the ES concept is not completely new in urban planning and proposes a methodology that looks at implicit references to urban ES in the different components of the plans. Most of all, it focuses on planning actions that address urban ES, identifying and classifying their purposes, target areas, and implementation tools. The chapter reveals what ES knowledge is already included in current plans and what advancements can be expected from a further integration.

Chapter 3 moves from the results of the review and addresses the specific barriers that limit the integration of scientific findings on regulating ES in urban planning. The chapter presents a conceptual framework that describes the process determining the provision and use of urban regulating ES, and identifies the entry points and pathways through which it can be affected by planning decisions. The framework is applied to the analysis of a wide scientific literature and helps to distil planning-relevant findings and to systematize them in a coherent overview of urban regulating ES. Thanks to the analysis, knowledge, methods, and indicators to assess the different values associated to urban regulating ES are made accessible, and guidance is offered to planners on how to use them to inform planning decisions.

Chapter 4 presents a case study that explores the use of ES knowledge to inform a real-life planning decision. The case study concerns the prioritization of re-greening interventions on existing brownfields based on the expected benefits in terms of improved cooling effect by vegetation and enhanced opportunities for nature-based recreation. The 'integration' is here investigated in its multiple meaning of integration of ES knowledge in the planning process, integration across multiple ES affected by planning decisions, and integration of multiple values associated to ES. The analysis shows how different methods for ES assessment, including modelling approaches and participatory methods based on the involvement of experts and stakeholders, can be combined to inform the prioritization of alternative scenarios in an urban planning process.

Chapter 5 frames the integration of ES in urban planning in the wider context of existing spatial strategies for sustainable urban development. The main spatial strategies agreed-upon within the European Union are identified through the analysis of 30 policy documents addressing urban development published since the foundation of the EU. The strategies, including the ‘green city’ strategy supported by the ES approach, are compared to the recent development trends of a large sample of 175 EU cities. The comparison allows assessing the level of coherence between international commitments and their on-the-ground implementation, and highlights synergies and trade-offs among the strategies. The analysis reveals additional aspects that should be monitored when implementing a ‘green city’ strategy, to ensure that planning decisions are really promoting a more sustainable urban development.

Finally, Chapter 6 concludes the thesis by providing a summary of the main findings and discussing some implications for both ES science and urban planning.

Chapter 2

Exploring ecosystem services in urban plans: what is there, and what is still needed for better decisions*

2.1 Introduction

This chapter aims to set the basis for the thesis by providing an overall picture of the opportunities and challenges of integrating ecosystem services (ES) in urban planning practices. More specifically, by reviewing recent planning documents, the chapter aims to shed light on what ES information, if any, is already used to support planning actions, and on what is still needed to improve plan contents and decisions. From the ES science perspective, the analysis is expected to highlight research areas that still need to be strengthened before findings can be effectively taken-up and operationalized in planning practices. From the planning perspective, it is expected to highlight planning-relevant issues that would benefits from a further integration of the ES approach.

* This chapter is based on: Cortinovis, C., Geneletti, D. (2018). Ecosystem services in urban plans: What is there, and what is still needed for better decisions. *Land Use Policy, 70*, 298-312. http://doi.org/10.1016/j.landusepol.2017.10.017

Scientists have monitored the uptake of ES in planning practices mainly following two approaches. The first approach investigates how practitioners, decision-makers, and stakeholders understand the concept of ES. Perceived opportunities and limitations in the use of ES in planning are usually elicited from key informants through interviews. For example, Niemelä et al. (2010) identified advantages and disadvantages according to the opinion of 24 professionals working on land-use planning and environmental management in Finland. Other successive studies report on interviews to policy-makers from the European Commission and Member States (Hauck et al., 2013a), German landscape and regional planners, (Albert et al., 2014b), Portuguese regional planners (Mascarenhas et al., 2014), and Swedish stakeholders and planners at the municipal level (Beery et al., 2016; Palo et al., 2016). Addressing self-reported perceptions and opinions, these studies do not measure the actual implementation of the ES concept into planning practices. However, their results can be useful to understand the mechanisms according to which this integration may take place.

The second approach reviews the content of documents, including strategic plans (Piwowarczyk et al., 2013), environmental policies (Bauler and Pipart, 2013; Maczka et al., 2016), Environmental Impact Assessment and Strategic Environmental Assessment reports and guidelines (Honrado et al., 2013; Mascarenhas et al., 2015), environmental laws (Matzdorf and Meyer, 2014), and, more recently, urban plans (Hansen et al., 2015; Kabisch, 2015; Rall et al., 2015). These studies usually apply a content or keyword analysis. Some searched for the explicit use of the term 'ecosystem service' inside the documents as an indicator of the influence of the ES paradigm on the policy discourse. However, this method does not reveal if and how well the concept is actually applied (Hansen et al., 2015). Moreover, a lack of explicit reference to ES does not necessarily mean that the underlying concept is missing. Previous results suggest that planners may perceive a high level of ES integration even when the term is absent from planning documents (Mascarenhas et al., 2014), and that linguistic preferences related to local habits or established practices may limit the explicit mention of ES even when the concept is accepted and acknowledged (Niemelä et al., 2010). Hence, one may gain a better understanding of the integration of the ES concept in planning by accounting for its implicit use, either through larger sets of keywords (Maczka et al., 2016; Mascarenhas et al., 2015) or through deeper content analyses (Hansen et al., 2015).

However, most of existing analyses strengthen the newness of the ES concept and frame the problem as the adoption of a new planning paradigm. Quite the opposite, as it clearly emerges from both planners opinions (Beery et al., 2016) and historical analyses of planning documents (Wilkinson et al., 2013), ES-inclusive approaches have routinely been used in planning, even though under different names, and planning has a tradition of accounting for – at least some - ES. Hence, focusing on the uptake of ES as a new planning paradigm may lead to overlook what is already there, and to direct research to objectives that do not support its operationalization. To understand how the ES approach can contribute to improve the current planning practices, it is necessary to identify which urban ES are addressed and how, and to what extent the conceptual framework of ES is already integrated in urban plans. To this aim, this chapter investigates the contents of plans searching for implicit references to ES and classifying the information based on their use within the plan. Section 2.2 describes the methods adopted and the selection of the sample. The main findings of the analysis are presented in Section 2.3 and discussed in Section 2.4, focusing particularly on what is already there in terms of actions and tools for their implementation, and what is still needed for an effective integration of ES in urban plans. Finally, conclusions are drawn in Section 2.5.

2.2 Materials and methods

The analysis of planning documents is based on a directed qualitative content analysis, which aims to interpret the contents of a communication starting from an existing theoretical framework (Hsieh and Shannon, 2005). The framework provides the key categories that are used to classify the contents based on similar meanings, thus following a deductive approach (Elo and Kyngäs, 2008; Hsieh and Shannon, 2005). Since urban plans are "communicative policy acts", this analysis is a suitable way to systematically investigate and assess their contents (Norton, 2008), as shown by previous applications in plan quality evaluation (Lyles and Stevens, 2014). The analysis is composed of three steps, described in the following sub-sections.

2.2.1 Assessing the breadth of ecosystem service inclusion in urban plans

The key categories of interest in this research are urban ES and plan components. That is, it aims to analyse how different urban ES are addressed in different plan components. Urban ES are identified from the lists provided by Elmqvist, Gómez-Baggethun, & Langemeyer (2016) and Gómez-Baggethun & Barton (2013). Accordingly, the analysis considers nine urban ES: food supply, water flow regulation and runoff mitigation, urban temperature regulation, noise reduction, air purification, moderation of environmental extremes, waste treatment, climate regulation, and recreation. All of them are ES "provided by urban ecosystems and their components" (Gómez-Baggethun and Barton, 2013) that are directly affected by planning decisions and actions at the urban scale. The identification of plan components follows previous content analyses of urban plans (Baynham and Stevens, 2013; Berke and Conroy, 2000; Geneletti and Zardo, 2016; Heidrich et al., 2013; Woodruff and BenDor, 2016), which refer to three main plan components: information base, vision and objectives, and actions. The *information base* component illustrates the background knowledge that supports planning decisions. The *vision and objectives* component states the long-term vision of the plan and the targets (either qualitative or quantitative) that the plan pursues. The *actions* component illustrates decisions taken by the plan, including strategies and policies (projects, regulations, etc.) that are envisioned to achieve the objectives.

Urban ES and plan components are cross-tabulated in a table (Table A.1), which is filled for each plan under investigation by analysing both its textual and cartographic documents, and reporting the relevant content. The number of filled cells in the table allows to measure the overall breadth of inclusion of the analysed ES, according to the formulation of the breadth score indicator proposed by (Tang et al., 2010) and later applied by (Kumar and Geneletti, 2015). The breadth score was calculated both for the whole plans and for each component individually. Then, the inter-component coherence, i.e. the presence of the same ES across the different components of the plans, was assessed applying a Chi-squared test for independence to all the possible combinations of two components (*information base*/*vision and objectives*, *vision and objectives*/*actions*, *information base*/*actions*).

2.2.2 Assessing the quality of ecosystem service inclusion in urban plans

Following a common approach in the existing literature on plans evaluation (Baker et al., 2012; Berke and Conroy, 2000; Geneletti and Zardo, 2016; Kumar and Geneletti, 2015), a scoring protocol was developed to assess the quality of ES inclusion in the plans. Quality is conceptualized as the presence of desired characteristics, described through criteria that high-quality plans are expected to meet (Berke and Godschalk, 2009). Building on the scoring protocol developed by Baker et al. (2012), the analysis adopted a 5-point scale, with scores ranging from 0 (no inclusion) to 4 (high-quality inclusion). Table 2.1 presents the scoring protocol used to assess the quality of inclusion in the *information base* component. A plan is awarded the highest score in this component when it acknowledges the links between ecosystems and human wellbeing, identifies functions and processes that determine the provision of ES, and applies this knowledge to a quantitative assessment of the local provision that also includes an analysis of demand and beneficiaries. Meeting these requirements, a high-quality *information base* component provides an appropriate and locally-relevant knowledge base for defining targeted policies and designing effective actions (Bassett and Shandas, 2010).

TABLE 2.1: *Scoring protocol for the* information base *component. Modified after* (Baker et al., 2012; Geneletti and Zardo, 2016). *The examples are taken from the analysed plans (translation by the authors): plan ID codes are reported in Table A.3.*

score	description	example
0	The plan contains no evidence of the ES concept.	-
1	The plan acknowledges the link between ecosystems and ES supply, either explicitly as part of the information base, or implicitly in the description of objectives and actions.	"Urban green areas […] guarantee protection of biodiversity inside the city as well as recreation and compensation of anthropogenic impacts." [explicit] Source: P12 "Acoustic green belts with a minimum length of 50m […] must be composed of evergreen broadleaves hedges or trees, with preference for fast growing, indigenous species with large crowns". [implicit in the description of actions] Source: P21
2	The plan mentions functions and processes on which ES provision depends, and identifies the elements that define ES potential. However, it lacks local application and analysis.	"Urban microclimate […] can be enhanced by the presence of vegetation […]. A continuous green network that crosses the city, linked to the countryside, constitutes a ventilation corridor that enhance urban microclimate. The most relevant biophysical process that determines the effects of vegetation on urban climate is the transpiration (…)". Source: P06
3	The plan shows a limited level of locally specific application of the ES concept. A basic qualitative assessment of the current state of ES is performed, but detailed analysis, quantitative measurements, and clear identification of demand and beneficiaries are lacking.	"Land-use changes determine an increase in soil sealing with higher storm water run-off. […] The increase in soil sealing and, consequently, in the flow rates produced by the reference rain event were quantified based on the distribution of sealed surfaces (e.g. streets, roofs) and permeable surfaces (e.g. parks) in each transformation area, as proposed by the draft masterplan". Source: P20
4	The plan shows an in-depth application of the ES concept in the analysis of the local provision of urban ES, including quantitative measurements, detailed assessment, and identification of demand and beneficiaries.	Spatially-explicit mapping of the accessibility to recreational areas (5 classes of accessibility), and quantification of beneficiaries broken down by age group (< 3; between 4 and 7; between 8 and 14; > 64 years). Source: P04

Table 2.2 presents the scoring protocol used for the *vision and objectives* component. Here, a plan is awarded the highest score when it defines locally-specific principles and quantitative targets for the enhancement of ES provision. A high-quality *vision and objectives* component is expected to coordinate public and private land-use decisions to achieve the defined goals (Berke and Godschalk, 2009), and, more specifically, to guide the choice of the best planning alternatives in terms of both "what" and "where" (Kremer and Hamstead, 2016). For the *actions* component, a binary score was assigned to record the presence, for each urban ES, of at least one action (as in Wilkinson et al. (2013)). Then, the overall quality of the component was defined as the share of ES addressed by at least one action in the plan. This is because a good inclusion of a specific ES in the *actions* component depends on local factors, e.g. related to the need for enhancement of the specific ES, as well as to the limits and opportunities for actions in the specific context. Therefore, defining a list of expected actions and using them as criteria to measure plan quality would be pointless.

For each plan, the results of the application of the scoring protocol are reported in a table (Table A.2). The overall quality of inclusion in the sample is expressed according to the measure proposed by Tang et al. (2010), which calculates the average score considering only the plans with a non-zero score in the component. The depth score indicator was calculated for each urban ES for the *information base* and for the *vision and objectives* components. Finally, the inter-component consistency, i.e. the coherence in the quality of inclusion across the different plan components, was assessed by computing the Pearson's correlation coefficient on the normalized scores obtained by the plans in the three components (Kumar and Geneletti, 2015; Tang et al., 2010).

TABLE 2.2*: Scoring protocol for the* vision and objectives *component. Modified after* (Baker et al., 2012; Geneletti and Zardo, 2016). *The examples are taken from the analysed plans (translation by the authors): plan ID codes are reported in Table A.3.*

score	description	example
0	The plan contains no evidence of objectives related to the ES.	-
1	The plan defines objectives of ecosystem conservation/enhancement, which are expected to positively affect ES provision, but does not directly refer to ES.	"Allow the restoration of river sides, particularly of potential flooding risk areas and retention areas that control overflows". Source: P11
2	The plan defines objectives directly related to ES provision. However, they are entirely descriptive, and lack local application and analysis.	"Tree planting, enlargement of existing green areas, and hedge planting must be encouraged to enhance the local microclimate (including air purification, noise abatement, mitigation of the heat island caused by impermeable surfaces)". Source: P07
3	The plan defines qualitative objectives directly related to ES provision through a locally specific analysis and application of the ES concept.	[In the peri-urban areas] "the municipal administration envisions the drafting of a specific plan […] for the safeguard and enhancement of green recreational areas and green belts, aimed at increasing the absorption of particulate matter and the reduction of the urban heat island effect." Source: P10
4	The plan defines objectives and quantitative targets related to ES provision through a locally specific analysis and application of the ES concept.	"The objective of increasing the amount of public green areas up to three times the existing can also be reached by making the 22% of the actual inaccessible green areas accessible and usable. This way, the green area per inhabitant doubles and exceeds the 30 sqm/inh". Source: P09

2.2.3 Analysing planning actions to address ecosystem services

To gain more insights on ES-related planning actions, three action properties were further investigated, namely typology, target area, and implementation tool. The *typology* describes the type of intervention on urban ecosystems, i.e. conservation, restoration, enhancement, or new ecosystem. The *target area* describes the scale of the planning action and the spatial distribution of the interventions within the city, i.e. widespread over the whole territory, targeting specific areas, or limited to specific sites. The *implementation tool* describes the type of legal instruments provided to implement the action, i.e. regulatory tools, design-based tools, incentive-based tools, land acquisition programs, or other tools. Categories of implementation tools are derived from Brody (2003) and Brody, Highfield, & Carrasco (2004), with the addition of the category of design-based tools that are typical of urban plans. Table 2.3 provides more details on the categories and sub-categories adopted. A list of planning actions addressing each of the nine urban ES considered in the analysis was compiled for each plan. Then, actions were classified with respect to the three properties, and recurrent combinations were identified both in the whole sample and for each urban ES.

TABLE 2.3: *Categories and sub-categories adopted for classifying planning action based on typology, target area, and implementation tool.*

Category	Description
Typology	
conservation	Action aimed at preserving the current state of urban ecosystems in order to secure the provision of ES. - *e.g. preserving existing wetlands*
restoration	Action aimed at recovering the health and functionality of urban ecosystems in order to get back to a level of ES provision offered in the past. - *e.g. de-paving sealed surfaces*
enhancement	Action aimed at improving the state of existing urban ecosystems in order to enhance the provision of ES. - *e.g. enlarging existing urban parks*
new ecosystem	Action aimed at creating new urban ecosystems in order to provide new ES in an area. - *e.g. planting street trees*

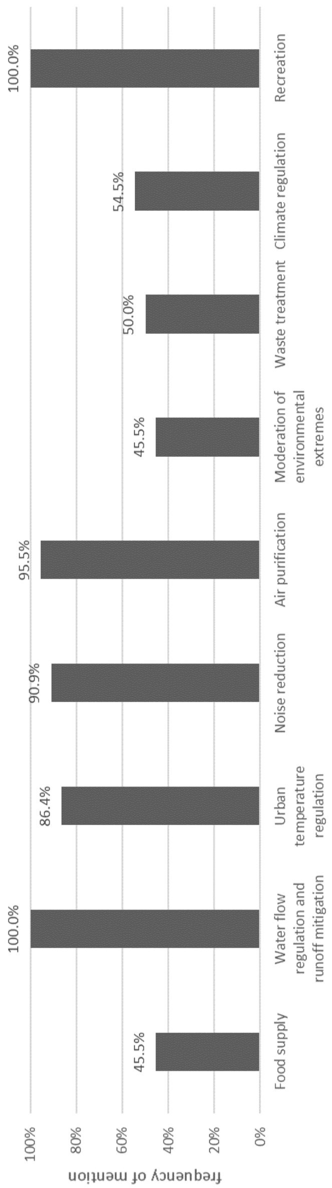
TABLE 2.3 *(continued)*

Category	Description
Target area	
widespread	The action targets all the future interventions of a certain typology. - *e.g. new building interventions, demolitions and reconstructions, large urban transformations*
specific areas	The action targets one or more zones in which the plan divides the city, or areas in the city identified by the presence of a specific issue. - *e.g. industrial sites, agricultural fragments*
specific sites	The action targets a specific project site or transformation area envisioned by the plan. - *e.g. a specific urban park, a specific brownfield to re-develop*
Implementation tool	
regulatory tools	
building code standard or requirement	Definition of a standard or a requirement in the building code that must be met when developing or re-developing an area.
compensation measure	Definition of a compensation measure (e.g. payments for realizations, mandatory land property transfers), including its rationale and quantification.
conservation zone or protected area	Definition of a boundary for a conservation zone or a protected area, and of the rules (restrictions and limitations) that must be respected within this area.
other regulatory tools	All the other types of actions undertaken through regulatory tools (e.g. density regulations, permitted and forbidden uses related to zoning).
design-based tools	Definition of specific design solutions to implement either in public projects or in privately-lead urban developments.
incentive-based tools	
preferential tax treatment	Definition of a financial incentive in the form of a preferential tax treatment (usually a reduction in planning fees).
density bonus	Definition of a non-financial incentive in the form of an increase in the surface (or volume) that is allowed in the area.
transfer of development rights	Definition of a ‘transfer of development rights’ mechanism: the development right is assigned to an area as a compensation for the placement of a conservation easement that prevents further development, and can be applied in other areas or sold. Participation is on a voluntary basis.

Category	Description
other incentive-based tools	This category includes all the other types of incentive-based tools, such as the possibility of realizing specific interventions under certain conditions.
land acquisition programs	Definition of a program for land acquisition by the public administration, with the aim of realizing a public project.
other tools	
principles for public space design	Definition of design principles and guidelines (non-compulsory) that should be applied in the realization of public spaces.
principles for territorial management	Declaration of principles that the municipal administration will follow in the management of the territory (e.g. commitment in administrative processes or in the implementation of future planning documents). It also includes assessment criteria for proposed interventions, when no incentive is envisioned.
promotion of good practices	Suggestion of principles, good practices, best available techniques, etc. (non-compulsory) to apply in private areas.

2.2.4 The sample

I applied the described methods to a sample of 22 recent urban plans of Italian cities. Urban plans in Italy are comprehensive spatial planning documents drafted at the municipal level, fairly similar in content to analogous documents around the world. Their main tasks are: to define the land-use zoning of the territory, including areas for new developments, and the related rules; to design and coordinate the system of public spaces and public services; to detail and integrate dispositions and regulations agreed at the upper administrative levels (provincial and regional coordination plans) on specific issues (e.g. infrastructures, protected areas, ecological networks) (Bragagnolo et al., 2012). To capture the influence of the growing global debate on ES following the publication of the Millennium Ecosystem Assessment (MA, 2005) and of the first TEEB report (TEEB, 2010a), the study was limited to the most recent planning processes. All the urban plans of the 118 Italian provincial capitals were checked and only the plans approved (at least in draft version) since 2012 were selected fot the analysis, which resulted in the final sample of 22 plans. The list of the plans and some data about the respective cities are reported in Appendix A (Table A.3).



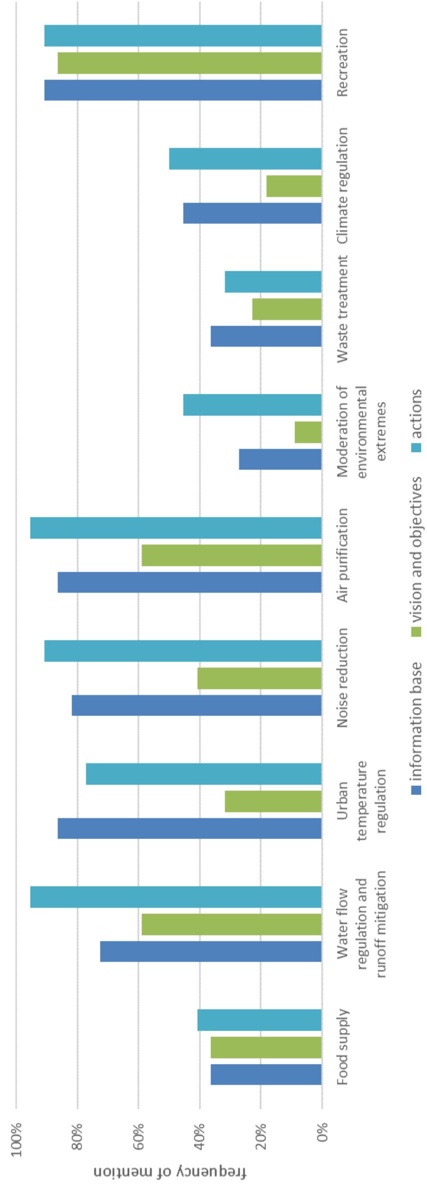


Figure 2.1: *Breadth score indicator measuring the inclusion of urban ES in at least one component of the analysed plans.*

Figure 2.2: *Breadth score indicator measuring the inclusion of urban ES in the three components of the analysed plans.*

2.3 Results

2.3.1 Breadth of ecosystem service inclusion in urban plans

The most frequent number of ES found in a plan is seven (nine plans), with only two plans addressing eight and nine ES respectively. Figure 2.1 shows the breadth score indicator measuring the overall inclusion in plans (i.e. inclusion in at least one component). Urban ES are clearly divided into two groups: five urban ES are included in almost all plans in the sample (breadth score > 85%); the other four urban ES are considered by around half of the plans (breadth score between 45% and 55%). Figure 2.2 breaks down the breadth score by plan component (for a detail on each plan refer to Figure A.1). The frequency of mention in the *information base* and in the *actions* components is similar across ES, although values for the latter are slightly higher. The frequency of mention in the *vision and objectives* component is generally lower, with the only two exceptions of food supply and recreation, which are mentioned evenly in the three components.

The inter-component coherence measured through the Chi-squared test (Table 2.4) describes a significant relation between the presence (and absence) of ES inclusion across the three components. However, the value of the test statistic is much higher for the pair *information base* and *actions*, while the *vision and objectives* component is less coherent with both the other components in terms of ES inclusion.

TABLE 2.4: *Chi-squared test for independence to measure the inter-component coherence in the presence of ES [**significance at the 0.01 level].*

	information base	*vision and obj.*	*actions*
information base	-		
vision and objectives	39.140**	-	
actions	89.262**	31.774**	-

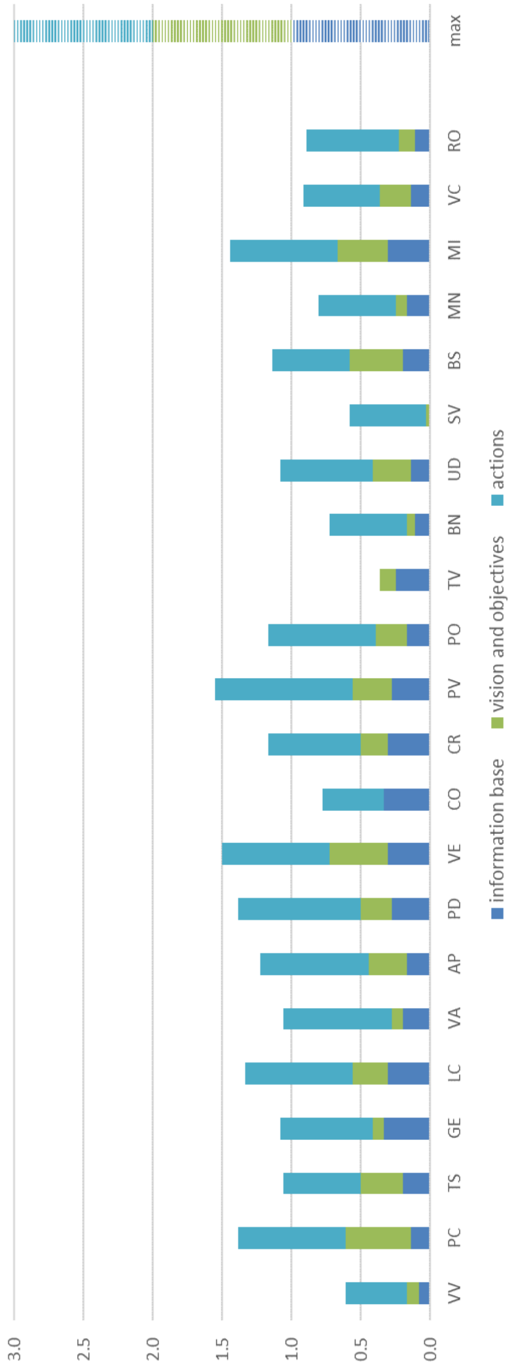


FIGURE 2.3: *Overall quality of ES inclusion (sum of the normalized scores of the three components).*

2.3.2 Quality of ecosystem service inclusion in urban plans

The overall quality of ES inclusion (Figure 2.3) is generally low, with only two plans in the sample reaching the score of 1.5 in the 0-3 range obtained by summing the normalized scores in the three components. The *actions* component receives the highest average normalized score (0.65), while normalized scores for the *information base* and the *goals and objectives* components are lower than 0.5 in all plans. The inter-component consistency estimated through the level of correlation among the quality scores in the three components is also low (Table 2.5): only the quality scores for the *visions and objectives* and the *actions* components show a significant correlation, which is however quite low (0.44).

TABLE 2.5: *Pearson's correlation calculated on the normalized quality scores to measure the inter-component consistency in the quality of ES inclusion [*significance at the 0.05 level].*

	information base	*vision and obj.*	*actions*
information base	1		
vision and objectives	0.196	1	
actions	0.185	0.444*	1

Figure 2.4 shows the distribution of quality scores for the different urban ES in the *information base* (top) and in the *goals and objectives* (bottom) components. The most common quality score in the information base component is 1, but the same pattern discussed for the breadth indicator emerge with respect to the different ES. Although the overall performance is quite poor, five ES (namely water flow regulation and runoff mitigation, recreation, air purification, noise reduction, and urban temperature regulation) are addressed in this component more often and with a higher quality compared to the others. Water flow regulation and run-off mitigation and recreation are the only ones for which some of the plans were given the highest scores. However, only analyses of recreation show, in some cases (around 30%), consideration for demand and beneficiaries. In the *vision and objectives* component, the pattern is less clear. Here, the most common quality score is 0, which indicates the absence of any reference to ES.

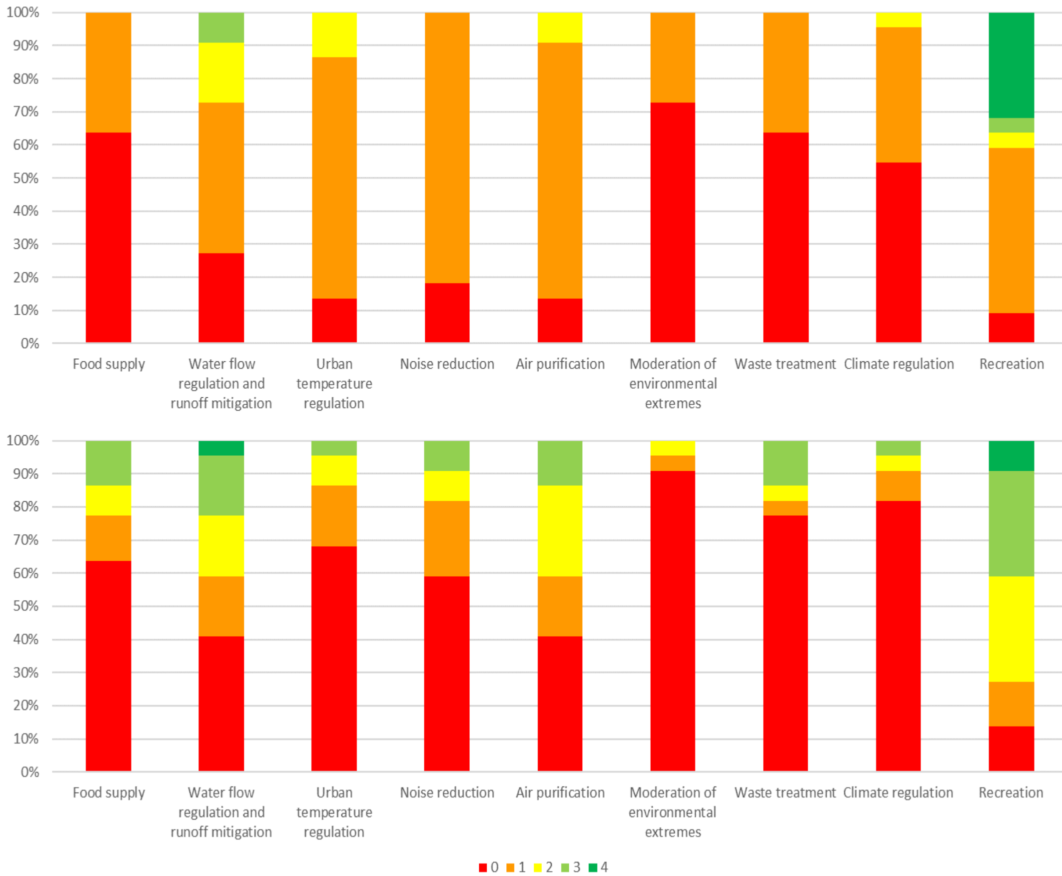


FIGURE 2.4: *Distribution of quality scores in the* information base *(top) and in the* vision and objectives *(bottom) component for the different urban ES.*

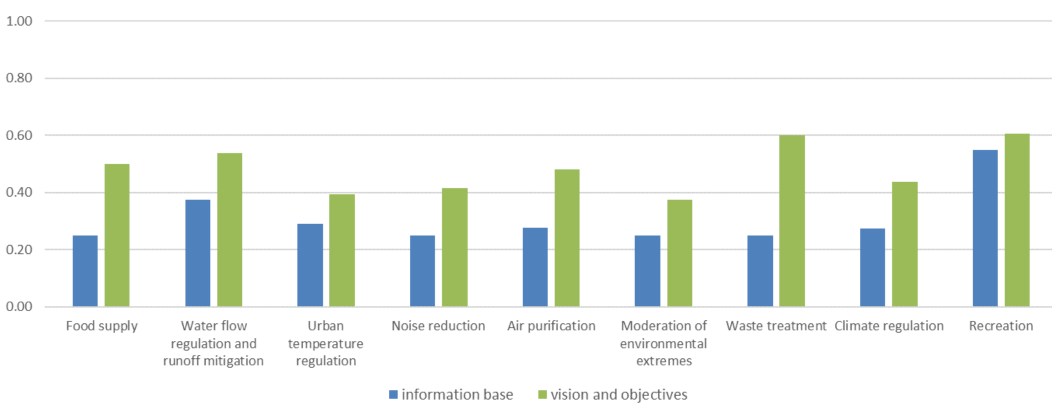


FIGURE 2.5: *Depth score indicator measuring the quality of inclusion of urban ES in the* information base *and in the* vision and objectives *components.*

However, the highest scores (3 and 4) are more frequent than in the *information base* component, and are found at least in one plan for almost all ES, even though a quality score of 4 is again obtained only by water flow regulation and runoff mitigation and recreation. The depth score indicator (Figure 2.5) confirms that, when ES are considered, the average quality of the *vision and objectives* component is higher compared to the *information base* component.

2.3.3 Actions to address ecosystem services in urban plans

Considering the whole sample of 22 plans, a total of 526 actions addressing urban ES were recorded, distributed as shown in Figure 2.6. Recreation is by far the most commonly address ES, with an average of more than eight actions per plan. An average of three to four actions per plan address water flow regulation and runoff mitigation, noise reduction, and air purification, with implicit acknowledgement of the demand for mitigation of these common urban environmental problems. The other services are addressed on average by less than two actions per plan. Table 2.6 lists groups of actions for each urban ES, based on the type of intervention proposed.

TABLE 2.6: *Groups of actions based on the type of intervention proposed.*

Urban ES and related actions	Number of plans
Food supply	
realization of new allotment gardens	6
protection of existing allotment gardens and residual agricultural patches	4
definition of peri-urban agricultural parks	3
restoration of urban and peri-urban patches for zero-mile agricultural produce	2
protection of traditional eco-compatible fishing and fish farming areas	1
Water flow regulation and runoff mitigation	
prescription of a minimum share of unsealed surfaces to maintain in new developments	14
prescription of permeable pavements for parking areas, cycling paths, etc.	9
realization of green roofs	6

TABLE 2.6 *(continued)*

Urban ES and related actions	**Number of plans**
Water flow regulation and runoff mitigation (continued)	
realization of bio-retention basins or other ecosystem-based approaches to storm-water management	6
de-paving	5
use of vegetation to control storm-water runoff	3
reduction of existing plans for new urban developments	2
protection of existing unsealed areas within the city	2
design of green areas to serve as natural basins (storage capacity and hydraulic connection to sealed areas)	2
use of nature-based solutions to guarantee hydraulic invariance of new developments	2
Urban temperature regulation	
provision of trees to shade parking areas	10
creation of new green areas/enlargement of existing green areas	7
conservation of unsealed surfaces	3
use of tree to shade buildings, thus reducing energy demand for internal cooling	3
realization of green roofs and green walls	3
de-paving	3
increase of vegetation density in existing green areas	2
use of trees and hedges as wind barriers	2
realization/restoration of lines of street trees	2
use of trees to shade public spaces	2
use of green for microclimate regulation (generic)	2
use of water areas for cooling	1
Noise reduction	
realization of green barriers/areas for noise shielding from infrastructures	15
realization of green barriers/areas for noise shielding from factories and plants	15
soil modelling for noise protection	4
use of green for noise shielding (generic)	4
conservation of existing green areas	3
Air purification	
realization of green barriers/areas for air purification from traffic missions	15
realization of green barriers/areas for air purification from industrial emissions	13
creation of woodlands and urban forests	5
use of green for air purification (generic)	4
conservation of existing green areas	4
realization of green roofs and green walls	4
use of trees and vegetation in parking areas	3

Urban ES and related actions	Number of plans
Moderation of environmental extremes	
enlargement of river areas and conservation/reclamation of floodplains	8
renaturing of river banks and reinforcement of bank vegetation as a protection system	3
use of vegetation to mitigate tides and protect from coastal floods	1
design of green areas to provide additional water storage capacity during extreme events	1
restoration of peri-urban agricultural areas as safeguards against wild fires	1
Waste treatment	
renaturing of riverbeds and restoration of banks and floodplains	3
realization of phyto-depuration areas or constructed wetlands for production plants	3
conservation/restoration/enhancement of buffer zones (strips) along water courses	3
use of ecosystems for wastewater treatment (generic)	3
realization of roadside vegetation strips for rainwater treatment	1
Climate regulation	
realization of Kyoto-forests and new woodlands	8
increase of public green areas	5
enhancement of private green areas	3
conservation and enhancement of public green areas	2
realization of green roofs	1
Recreation	
realization of new public green spaces and urban parks	16
strengthening walking and cycling accessibility among green areas and with the rest of the city	16
increasing fruition of green spaces through new walking and cycling paths	14
restoration of existing green areas aimed at increasing their use	14
promotion of new functions and uses in the existing green spaces	12
enlargement of existing green spaces	8
identification of opportunities for recreation in agricultural areas	8
realization of peri-urban parks	7
opening of existing private/unused gardens and green spaces to public use	6
realization of new urban gardens	3
identification of protection areas based on their public use	3

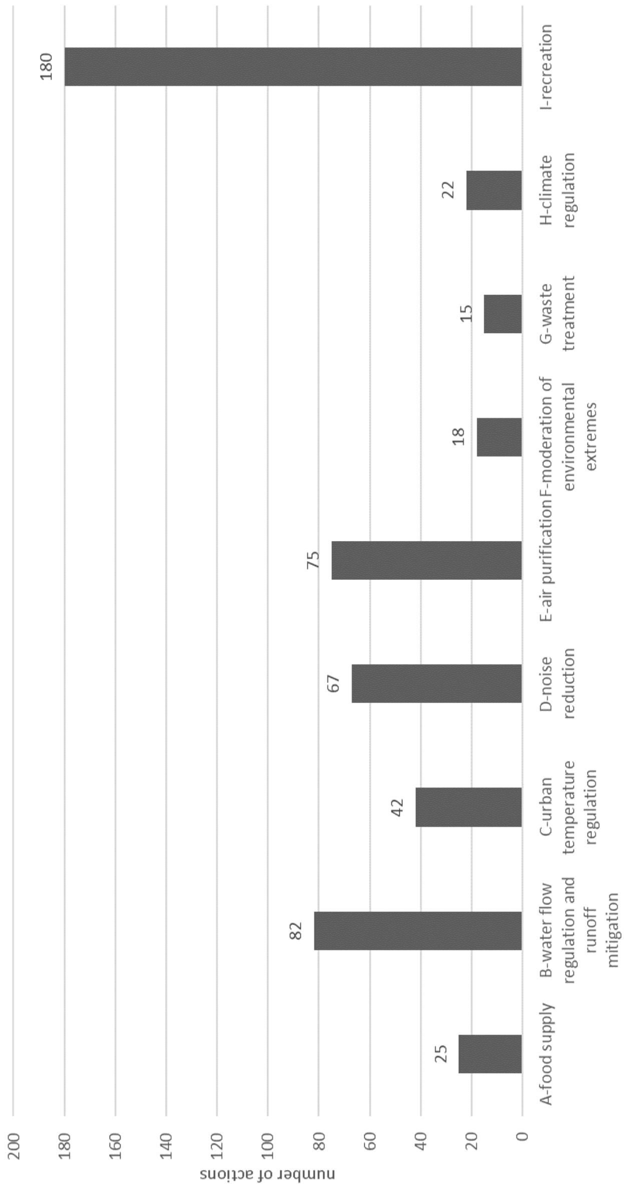


FIGURE 2.6*: Number of actions addressing each ES in the whole sample of plans.*

Figure 2.7 describes the distribution of action according to the three properties (typology, target area, and implementation tool). New interventions, such as the realization of new green areas, represent the most common typology of action (53%). 44% of the actions rely on design-based implementation tools (e.g. projects included in the plan), through which the public administration can control action implementation with a quite high level of detail. Regulatory tools, particularly the definition of standards and other specific requirements in building codes, and other tools, such as the suggestion of good practices, are also among the most common, both with 25% of the sample. Incentive-based tools (e.g. density bonuses) and land acquisition programs are the least adopted tools, and accounts for only 4% and 3% respectively. In terms of target areas, specific sites are the most common and represent the target of 50% of the actions. These include, for example, the restoration of specific ecosystems, the identification of conservation areas, and the realization of new urban parks. 29% of the actions target specific areas in the municipal territory, such as regulations to be applied in industrial areas or safeguards to protect agricultural patches. Finally, 21% of the actions are widespread. These include requirements for all new building interventions and rules to respect in case of demolitions and reconstruction.

Actions on specific sites are usually implemented through design-based tools, while actions on specific areas are generally implemented through regulatory tools or other ‘soft’ tools such as the suggestion of good practices. Soft tools also clearly prevail in the case of widespread measures. Concerning typologies, conservation actions are more often implemented through regulatory tools, while for both enhancement and restoration activities the preferred tools are design-based. For example, new conservation areas are often defined through a boundary in the maps and a set of rules, while restoration measures are often proposed through a more detailed design.

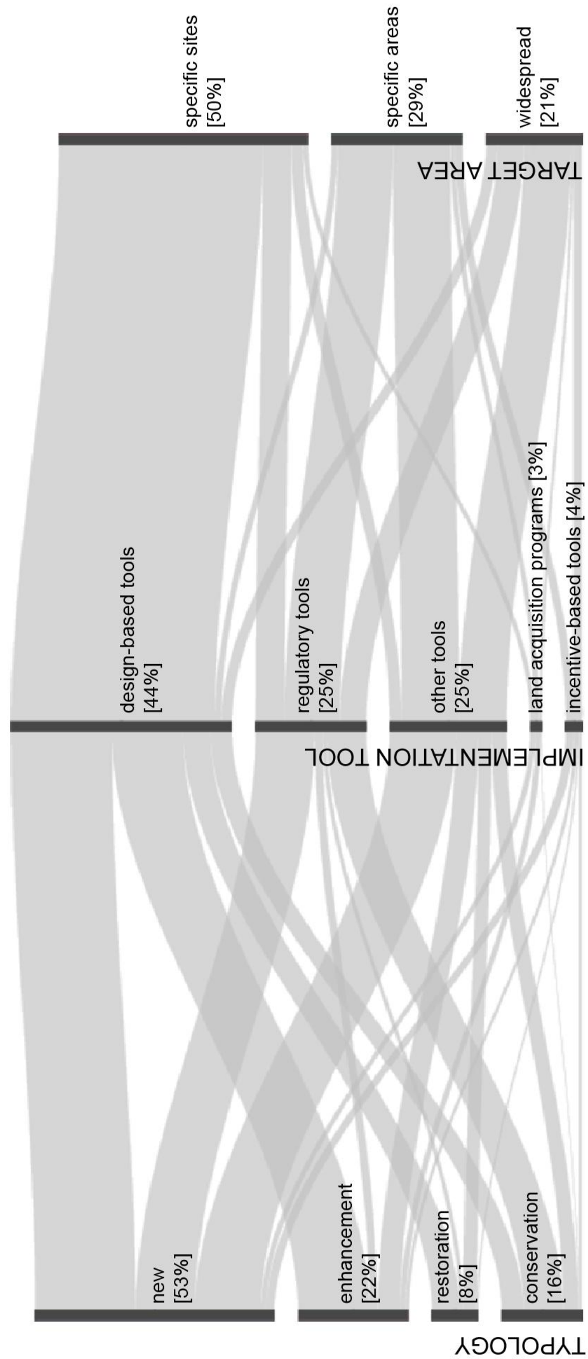


FIGURE 2.7: *Distribution of planning actions per typology, target area, and implementation tool, and recurring combinations in the whole sample. A further detail on the sub-categories of implementation tools is provided in Figure A.3.*

This general picture is partly different when looking at single ES (Table 2.7). Conservation actions are the preferred typology for improving food supply (conservation of agricultural patches) and water flow regulation and runoff mitigation (conservation of existing unsealed surfaces), while recreation is mostly promoted through enhancement interventions on existing green and blue areas. Water flow regulation and runoff mitigation also differs in term of target areas and, consequently, implementation tools, mostly prescriptions related to the share of unsealed surfaces to maintain in new developments. Two other ES do not have design-based as the preferred tools: food supply, for which 40% of the actions consist in principles for territorial management, and waste treatment, which is commonly addressed through the promotion of good practices.

Table 2.7: *Action properties broken down by ES. Values are expressed as percentage of the actions addressing each ES. Bold indicates the most frequent properties for actions addressing each ES.*

	Food supply	Water flow regulation and runoff mitigation	Urban temperature regulation	Noise reduction	Air purification	Moderation of environmental extremes	Waste treatment	Climate regulation	Recreation
typology									
conservation	**41.3**	**55.2**	2.4	7.8	9.0	**33.3**	3.3	13.6	3.9
restoration	13.3	12.5	9.5	1.9	3.0	25.0	13.3	6.8	8.1
enhancement	5.3	13.1	9.5	4.1	7.7	8.3	20.0	13.6	**46.7**
new ecosystem	40.0	19.2	**78.6**	**86.2**	**80.3**	**33.3**	**63.3**	**65.9**	41.4
target area									
widespread	24.0	**56.1**	35.7	13.4	16.0	11.1	13.3	**52.3**	3.3
specific areas	**52.0**	28.0	35.7	**43.3**	**44.0**	38.9	**60.0**	18.2	12.2
specific sites	24.0	15.9	28.6	**43.3**	40.0	**50.0**	26.7	29.5	**84.4**
implementation tool									
building code standard	0.0	**43.3**	26.2	29.9	24.0	8.2	20.0	0.0	1.7
compensation measure	4.0	2.4	0.0	6.0	5.3	0.0	6.7	27.3	1.7
conservation zone	12.0	1.2	0.0	1.5	1.3	16.7	0.0	0.0	3.6
other regulatory tools	0.0	3.7	0.0	1.5	1.3	0.0	0.0	0.0	0.0
design-based tools	20.0	13.4	**31.0**	**31.3**	**30.7**	**44.4**	13.3	**36.4**	**77.5**
preferential tax treatment	0.0	2.4	4.8	1.5	1.3	0.0	0.0	13.6	0.0
density bonus	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.0
transfer of dev. rights	0.0	0.0	0.0	1.5	1.3	0.0	0.0	4.5	0.6
other incentive-based tools	8.0	1.2	0.0	0.0	0.0	5.6	0.0	0.0	0.3
land acquisition programs	0.0	3.7	0.0	1.5	1.3	5.6	0.0	0.0	4.7
pr. for public space design	12.0	3.0	9.5	0.0	0.0	2.8	0.0	0.0	0.0
pr. for territorial mgmt	**40.0**	6.1	14.3	6.0	12.0	0.0	13.3	9.1	10.0
promotion of practices	4.0	19.5	14.3	19.4	20.0	16.7	**46.7**	9.1	0.0

2.4 Discussion

The research analysed a sample of 22 Italian urban plans. Following a common approach in studies investigating the uptake of the ES concept in planning practices (Beery et al., 2016; Mascarenhas et al., 2014; Piwowarczyk et al., 2013; Young, 2013), the sample comprises cases from a single country. Hence, results are not representative of the whole spectrum of urban planning practices. Nevertheless, comparing the results with findings from similar investigations carried out in other contexts, and more widely with the existing literature on urban ES, allows highlighting similarities and differences, and drawing some general remarks on the inclusion of urban ES in planning practices. Cross-city comparison is considered fundamental to advance ES research and promote an effective operationalization of scientific findings (Kremer and Hamstead, 2016). So far, similar studies have mostly focused on large cities renowned for their commitment in environmental policies (Hansen et al., 2015; Kabisch, 2015; Rall et al., 2015; Wilkinson et al., 2013). These cities may act as light-houses in spreading innovative concepts; however, significant differences in terms of critical mass, resources, and institutional capacity may limit transferability of good practices to small and medium-sized cities (Giffinger and Fertner, 2007). This study offers an original perspective that mainly encompasses small and medium-sized cities, thus contributing to a wider understanding of potentials, gaps, and limitations related to the inclusion of ES in urban planning practices.

The main findings of the analysis are summarized in Table 2.8 and discussed in the following sections.

TABLE 2.8: *Summary of the main findings of the review of urban plans.*

what is already there	what is still needed
• urban planning addresses urban ES through a high number and a great variety of actions • a wide range of local problems can be addressed through ES-based actions • urban planners are already equipped with a large set of tools to implement ES-related actions • recreation provided by urban ecosystems, although not linked to the ES concept, is widely acknowledged and promoted by planning actions • a set of key regulating ES to address pressing urban environmental problems (i.e. water flow regulation and runoff mitigation, air purification, urban temperature regulation, and noise reduction) are widely acknowledged and addressed	• scientific knowledge is only partly transferred to planning practices • there is little guidance on how to incorporate information on ES into planning processes • usable methods to assess urban ES at a relevant scale while accounting for multi-functionality of ecosystems are still lacking • plans contain no analyses of ES demand and of the existing and expected beneficiaries (with the only exception of recreation) • ES are not considered a strategic issue in urban planning

2.4.1 What is already there: the current state

Actions to address ES

This full list of actions addressing urban ES (Table 2.6) is much more comprehensive than the list of possible nature-based interventions in urban environments proposed by Sutherland et al. (2014) and later expanded by European Commission (2015), demonstrating the capacity of planning practices to creatively address urban ES. Interestingly, by looking at current plans, the findings expand not only the number of solutions that are proposed, but also the range of issues that are usually considered when proposing ecosystem-based solutions. Issues and respective solutions such as safeguarding traditional food supply in cities through sustainable fishery, providing wind shielding by vegetation, protecting against wildfires by maintaining agricultural practices, just to name few, indicate local problems to which ecosystem-based actions may offer a sustainable solution.

Tools for implementation of ES-related actions

Strategies and tools to implement principles of ecosystem management within existing planning frameworks have often been ignored by research (Brody, 2003). ES research too, which is more and more focused on translating ES principles and approaches into concrete actions, has nevertheless largely overlooked the question of how such actions are to be practically implemented. Actions and tools represent the core of a plan (Brody et al., 2004), and the quality and probability of success of an action depend on the type of tool through which it is implemented. The review identified five broad categories of tools that reflect different levels of compulsoriness and different roles of the involved stakeholders. Planners are, in fact, equipped with a large toolbox, assembled during the last two centuries of urban planning history and already put to use in the operationalization of other concepts and approaches, such as sustainable development (Berke and Conroy, 2000). Most of these tools are already applied to address ES, even though there is a large prevalence of the most traditional, public-driven tools (i.e. regulatory and design-based tools). Despite limitations, for example in terms of budget availability, may hamper the adoption of certain tools, all of them are in principle available to local administrations to implement planning actions. Looking creatively at the whole toolbox can help exploring new possibilities to also address the least considered ES, and to imagine innovative actions that exploit opportunities to further engage local stakeholders and communities.

Consideration of recreation as an ES

Possibly the most predictable finding concerns recreation, which has been among the main concerns of urban planners since the very beginning of the discipline. All plans in the sample address recreation. This is in line with results from analyses of urban plans in other cities around the world, including diachronic analyses following the development of planning documents through time. In the review of urban plans for Stockholm and Melbourne carried out by Wilkinson et al. (2013), recreation and fresh water provision are the only ES mentioned in all the analysed documents, the oldest of which dates back to 1929. A similar result also emerged from an international sample of cities including champions of green planning such as Stockholm, Berlin and New York (Hansen et al., 2015). Kabisch (2015) obtained the same result from the analysis of planning and strategic documents related to green spaces in Berlin: recreation has been addressed by all plans in the last 20 years, without any apparent relation between the inclusion of the service and the explicit acknowledgement of the ES concept.

This frequent consideration for recreation is a consequence of the widely recognized importance of green spaces for the wellbeing of urban population (Kabisch et al., 2015). The familiarity of planners and decision-makers with this service since well before it was labelled as an ES determined the availability of models, techniques, and indicators to measure the performance of cities, to investigate current needs, and to define specific objectives for its enhancement (Barbosa et al., 2007; Kabisch et al., 2016; La Rosa, 2014). In the analysed sample, the information base about recreation often shows a quantitative and spatially-explicit analysis of supply, demand, and beneficiaries, although the term 'ecosystem service' is hardly ever used. As also emerged from a study involving Portuguese regional planners, it is hard for practitioners to link the already familiar issue of recreation to the newly introduced concept of cultural ES (Mascarenhas et al., 2014).

The result partly contrasts with the fact that, in scientific publications on urban ES, cultural ES are the least explored (Haase et al., 2014b), which has determined a lack of scientifically-sound methods for their analysis (Martínez-Harms and Balvanera, 2012). Due to their intangible dimension, their relation with non-material values, and their inherent subjectivity (Chan et al., 2012), cultural ES are considered as difficult to capture through scientific models and indicators (La Rosa et al., 2015). The sphere of citizens' perceptions and preferences that determine cultural ES values can only be investigated through stakeholder involvement (Luederitz et al., 2015; Wolff et al., 2015). However, this is most probably also the reason why cultural ES are always present in planning documents at the local scale, which are expected to capture values and beliefs of the local community, and reflect them in an agreed vision for the city (Norton, 2008). Although an implicit inclusion of cultural ES in the planning process may limit their visibility, hence consideration, for example in balancing trade-offs (Chan et al., 2012), cultural ES, at least for what regards recreation, do not seem to be under-represented in current planning practices at the urban scale.

Consideration of key urban regulating ES

Four regulating ES are acknowledged by almost all the reviewed plans, namely, water flow regulation and runoff mitigation, urban temperature regulation, noise reduction, and air purification. These ES are related to environmental issues specific of urban contexts, such as soil sealing, urban heat island, and noise and air pollution, which have been key topics in the discourse around urban sustainability and resilience during the last years. Publications both at the EU and at the global level (e.g. EEA, 2012; Science for Environment Policy, 2015; UN-HABITAT, 2009), and initiatives like URB-ACT (http://urbact.eu/), Mayors Adapt (http://mayors-adapt.eu/), and 100 Resilient Cities (http://www.100resilientcities.org/) have in fact contributed to raising awareness and spreading knowledge about these issues among practitioners and policy-makers.

The breadth of inclusion of the four regulating ES can also be linked to the growing popularity of ecosystem-based actions to address the related environmental issues (Brink et al., 2016; Geneletti and Zardo, 2016; Morani et al., 2011; Van Renterghem et al., 2015). In fact, the number of actions addressing these ES in the analysed plans is significantly higher compared to all the other ES, except for recreation. In general, actions addressing the four regulating ES are more common and easier to implement compared to other regulating ES, due to several reasons. First, most of them (e.g. cooling by vegetation, noise barriers, permeable surfaces) require only local interventions (Andersson et al., 2015), with no need for interjurisdictional cooperation that may represent a barrier (Kremer et al., 2016). Moreover, they are supported by strong scientific and empirical evidence (Demuzere et al., 2014), and easy-to-use methods are available to quantify the expected results, which is still not the case for other regulating ES less frequently included in urban ES literature (e.g. moderation of extreme events and waste-water treatment) (Haase et al., 2014b). Finally, such actions usually produce positive and measurable local benefits (Faehnle et al., 2014), thus easily gaining support from local stakeholders.

2.4.2 What is still needed: potential improvements

Strengthening the information base

The depth of ES inclusion in the *information base* component can be considered an indicator of the level of knowledge transfer from science to practice. The average low score indicates that, despite the exponentially-growing number of studies on urban ES (Haase et al., 2014b; Luederitz et al., 2015), a successful transfer is still lagging behind, and the operationalization of the ES concept is far from being in place (Geneletti, 2011; Kremer et al., 2016; Ruckelshaus et al., 2015). A locally-specific application of the ES concept is a clear gap in the *information base* of current plans, where existing methods, models, and tools for ES mapping and assessment are almost completely overlooked even in cases where the enhancement of ES is an explicit goal. This is a warning for ES science, which has already been criticized for claiming the applicability of the developed methods to real-world practices as a justification for research, without then caring about their actual use (Laurans et al., 2013; Slootweg, 2015). The result confirms findings from a recent study in Sweden, where urban planners expressed the need for an active support from research, more than the simple exchange of information and data, to integrate ES in the current planning practices (Palo et al., 2016). To this aim, little guidance is provided by planning guidelines, e.g. existing guidelines on sustainable planning: at present, they provide no indication on which type of information on ES should be collected, and how it should be incorporated into the planning process (Woodruff and BenDor, 2016). Nevertheless, enhancement of existing guidelines would be an effective mean to summarize the existing scientific knowledge and promote a better consideration for ES in urban plans.

Analysing the multi-functionality of urban ecosystems at a relevant scale

Looking at the results of the review, and particularly at the actions proposed, two main issues emerge for current methods and approaches to urban ES mapping and assessment. The first one is the relation between scales of analysis and action. Most actions found in the reviewed plans are implemented through design-based tools that act at the very local scale. Such actions produce changes that are not captured by land use-land cover data, the most common source of information for ES analyses (Martínez-Harms and Balvanera, 2012). To effectively support urban planning in the operationalization of the ES approach, usable methods are needed to map current conditions of urban ES and to measure expected and actual outcomes of planning actions at a relevant scale (Haase et al., 2014b). The second issue is multi-functionality. The common assumption in land-use planning that to each area corresponds one single function, as well as the current approach to ecosystem-based actions as solutions to specific issues, conflict with the multi-functionality of urban green infrastructure (Hansen and Pauleit, 2014). Methods to map urban ES should better integrate consideration for multi-functionality, providing ways to simultaneously assess the provision of multiple ES under different planning scenarios (Kremer and Hamstead, 2016).

Accounting for ES demand

The explicit consideration for the demand side of ES and the identification of beneficiaries should be among the main improvements brought to the urban planning practice by the ES concept. Referring directly to benefits experienced by citizens would strengthen planners' arguments against other sectoral interests, especially in balancing public and private benefits (Hauck et al., 2013c). Urban planning is one of the social arrangements that establish who in the city benefits from ES, hence urban plans are a strong determinant of environmental justice within cities (Ernstson, 2013). To this respect, an effective *information base* should necessarily consider not only ES supply within the city, but also the distribution of beneficiaries, and their different levels of demand for each specific ES (Kabisch and Haase, 2014). With the only exception of recreation, analyses of the existing and expected beneficiaries, and of the differentiated needs of urban population, are lacking. Demand for regulating ES is implicitly acknowledged in the definition of some actions, particularly mitigation measures related to the presence of specific sources of environmental risk (e.g. noise, air pollution, flooding). Although such ecosystem-based actions somehow implicitly recognize the principle of risk reduction or prevention as the type of demand characterizing regulating ES (Wolff et al., 2015), operationalization does not go beyond the empirical level and the scale of single sites, without any baseline analysis conducted to support decisions. Methods and indicators exist in the literature to assess demand for regulating ES, however applications at the urban scale, especially in spatially-explicit, multi-ES assessments able to reveal distributional inequalities, are only few (Baró et al., 2016, 2015), and too demanding to be applied in real-world practices.

Including ES in the strategic vision

The breadth score indicator shows that there is a lack of consideration for ES in the strategic component (i.e. *vision and objectives* component). Only three ES, which represent the 'hottest' topics for urban planning in the analysed cities, are addressed in the *vision and objectives* components of more than half of the plans, namely recreation, water flow regulation and runoff mitigation, and air purification. Moreover, the level of coherence between *vision and objectives* and the other two components is particularly low, indicating that the *information base* is more often directly linked to the formulation of actions, rather than used to support the definition of goals and objectives. Other studies (e.g. Beery et al. (2016)) confirm that the strategic component is the weakest point in the uptake of ES by urban planning. Considering the results, this may be linked to the different level of ES inclusion in the *actions* and the *vision and objectives* components. The presence of many actions that implicitly address urban ES in current planning practices makes probably easier for stakeholders and decision-makers to recognize the instrumental use of ES knowledge compared to its strategic use.

A weak strategic vision, lacking specific objectives and targets for ES enhancement, undermines the perspective of a long-term commitment that could guarantee action implementation and persistence of ES consideration beyond the time horizon of the single plan (Wilkinson et al., 2013). In the reviewed plans, the distance between the high number of actions addressing some ES and the low quality of their baseline assessments and strategic objectives indicate that current approaches are largely based on the reference to general good practices, rather than on the analysis of current needs and the consequent formulation of local strategies. Similarly, Geneletti & Zardo (2016) pointed at the lack of analyses to support the design and the location of ecosystem-based interventions in urban climate adaptation plans. Even though the reference to good practices may be an important source of knowledge and co-learning among cities, replicating the same solutions without tailoring them to the specific local context may lead to suboptimal results.

2.5 Conclusions

The review of 22 urban plans focused on the use of the ES concept as a tool to support decision-making (Mckenzie et al., 2014), as opposed to the explicit uptake of the term ‘ecosystem services’. Similarly to what has been observed for the concept of sustainable development (Persson, 2013), the hypothesis was that an effective integration should build on what is already there, and follow a mechanism of ‘internalization’ that does not necessarily require rethinking or reshaping current practices. The findings, summarized in Table 2.8, reveal that current urban plans already include a high number of ES-related actions and a variety of tools for their implementation. This indicates that planners have the capacity and the instruments to enhance the future provision of urban ES. Actions in the analysed plans often go beyond those ordinarily mentioned as good practices, and the range of issues that they address is wider. This demonstrates a certain level of creativity that, combined with traditional ecological knowledge and the understanding of local social-ecological systems, enables the design of locally-relevant interventions.

However, the study unveils a two-speed integration of urban ES, with a set of services that are widely addressed by urban plans (recreation, above all, but also regulating ES linked to environmental problems typical of urban areas), and others that are hardly considered. The least considered (e.g. waste treatment and moderation of environmental extremes) are also the least popular in the scientific literature (Haase et al., 2014b), and when they are included in urban plans, their treatment is very shallow (e.g. suggestion of one-fits-all good practices). This can be ascribed, at least partly, to gaps in the scientific literature, which has not produced methods and guidance that fit urban planning practices. To this purpose, future developments in methods for ES assessment are particularly needed, in terms of i) scale and resolution, to better match those of planning actions and of their outcomes; ii) analysis of demand and beneficiaries, especially of regulating ES; iii) consideration of multi-functionality of urban green infrastructure, to allow the assessment of multiple ES under different planning scenarios.

Concerning planning practice, a further understanding and appropriation of the ES approach by urban planning would benefit future practices in many respects. First, it could promote consideration of a larger set of urban ES, at least in the initial phases of planning processes, thus increasing awareness of all values at stake, highlighting co-benefits and trade-offs that may arise from planning actions, and making prioritization more transparent. Second, it could strengthen the consideration of ES as a strategic issue for urban planning, thus promoting the definition of objectives and targets for ES enhancement, and ensuring long-term commitment in the implementation and monitoring of planning actions. Finally, it could support the explicit identification of ES demand and beneficiaries, thus improving baseline information to address urban environmental equity, and providing planners and decision-makers with stronger arguments against conflicting interests on land use decisions.

Chapter 3

Operationalizing scientific knowledge on urban regulating ecosystem services: a framework for planners*

3.1 Introduction

The multiple benefits of urban green infrastructure, including flood control, air pollution reduction, microclimate regulation, and noise mitigation, are more and more frequently acknowledged by planners (see Chapter 2), and actions aimed at enhancing their provision are becoming common in urban plans (Geneletti and Zardo, 2016; Hansen et al., 2015). However, real-life applications rarely take advantage of the growing scientific literature on urban ecosystem services (ES) and of the methods and tools available for their assessment (Albert et al., 2014b; Davies et al., 2017; McPhearson et al., 2014). Most planning actions are based on the reference to good practices and show a predominant heuristic approach that makes little use of scientific knowledge. In line with findings from Chapter 2, authors report

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inadequate analyses to support the design and location of interventions (Geneletti and Zardo, 2016) and the lack of preliminary assessments of their potential effectiveness (Jones et al., 2012). Failing to tailor planning actions to the specific context of application limits their effectiveness and may lead to unplanned or even unwanted results. Indeed, from a planner's point of view, "we have plenty of information about our environments but need to learn how to convert it to knowledge and then how to use it to take wise actions" (Steiner, 2016).

The aim of this chapter is to bridge the gap between scientific findings on urban regulating ES and their operationalization, thus supporting urban planners in the design of effective actions to enhance the provision of urban regulating ES and associated benefits in cities. To achieve this objective, it is first necessary to identify what barriers hinder the integration of scientific findings in urban planning. The overall challenges of this science-policy interface have been already summarized in Section 1.2. However, two barriers specifically characterize urban regulating ES. The first barrier is the complexity of the biophysical and functional foundations of their provision, whose insufficient understanding limits planners' capacity to exploit the full potential of urban green infrastructure (Trinomics B.V., 2016). Compared to other ES, ecosystem functions and processes that support urban regulating ES are more complex, also due to the interaction with the highly-variable environmental conditions of urban areas (Andersson et al., 2015). The second barrier relates to the demand. Studies addressing urban regulating ES have mostly focused on the supply side, while the analysis of demand and benefits is often overlooked (Schmidt et al., 2016). The poor identification of demand and beneficiaries and the lack of explicit consideration for mismatches between supply and demand limit the operationalization of the ES concept and undermine the usability of regulating ES assessments in decision-making (Bagstad et al., 2014; McPhearson et al., 2014).

Over the last years, the wide literature on urban regulating ES (Haase et al., 2014b; Luederitz et al., 2015; Pulighe et al., 2016) has produced relevant scientific findings for urban planning. Indicators and proxies exist to describe the supply of regulating ES in urban areas (Albert et al., 2015; Dobbs et al., 2011; Gómez-Baggethun and Barton, 2013), and spatially-explicit methods and tools for mapping and assessment of urban regulating ES are included in a growing number of research applications (e.g., (Baró et al., 2016; Derkzen et al., 2015; Holt et al., 2015; Larondelle and Lauf, 2016)). Some limitations have been pointed out, particularly the inadequate resolution of methods transferred from regional-scale applications (Martínez-Harms and Balvanera, 2012) and the prevalent focus on ecosystem structure and functions rather than on the actual services produced (Luederitz et al., 2015). However, urban ES science is rapidly advancing in the refinement, providing planners with valuable information to integrate considerations of urban ES in plans and policies (Pulighe et al., 2016).

To enhance the usability of these scientific findings, their implications for the design of planning actions must be clarified. Here, a conceptual framework is proposed that describes the process of urban regulating service provision and use, identifying the key elements involved and their interactions. The framework highlights the entry points for planning and the paths through which it affects the intensity and spatial distribution of regulating ES and associated benefits across the city. The framework is then applied to the analysis of seven urban regulating ES, and serves as a guidance to navigate a wide scientific literature. Planning-relevant information is collected and systematized, providing a description of the specific process that characterize the provision and use of each urban regulating service.

The chapter is structured as follows. Section 3.2 describes the methodology adopted to build (Section 3.2.1) and to apply (Section 3.2.2) the framework. Section 3.3 describes the framework and its components. Section 3.4 presents the results of the application of the framework to seven urban regulating ES, including possible indicators to describe each component and hints on how to operationalize the approach in planning practices. Finally, Section 3.5 discusses the findings and Section 3.6 draws some conclusions.

3.2 Methods

3.2.1 Building the framework

The proposed framework was built by combining two existing models and approaches for ES assessment: the Cascade conceptual model (Haines-Young and Potschin, 2010) and the supply-demand approach for ES mapping and assessment (Baró et al., 2016; Burkhard et al., 2012). The Cascade conceptual model provides the stepwise description of the supply side of urban regulating ES, which 'flow' from the functional characteristics of urban ecosystems, and supports a clear distinction between services and benefits. The supply-demand approach for ES mapping and assessment provides the concept of service benefitting area, which is used to spatially describe ES as the overlap between supply and demand.

From a planning perspective, assuming the stepwise approach of the Cascade model allows navigating the framework in both directions, thus understanding not only the expected consequences of planning actions, but also what actions are needed to achieve a defined objective (Potschin-Young et al., 2017; Spangenberg et al., 2014). At the same time, the spatially-explicit description of urban regulating ES supported by the supply-demand approach, specifically formulated in the context of spatial analysis of ES (Syrbe and Walz, 2012) and already applied in a number of mapping studies (Burkhard, Crossman, Nedkov, Petz, & Alkemade, 2013; García-Nieto, Garcìa-Llorente, Iniesta-Arandia, & Martín-López, 2013; Palomo, Martín-López, Potschin, Haines-Young, & Montes, 2013 among others), is critical toward their operationalization in urban planning (Haase et al., 2014b).

Two elements complete the framework, namely environmental conditions, and urban population and activities. Environmental conditions play a key role in the provision of urban regulating ES, affecting both the supply and the demand side. For example, a high concentration of air pollutants produces negative health effects on both urban green infrastructure (thus affecting the capacity to provide the service of air purification) and urban population (thus determining the demand for the service). Urban population and activities are included in the framework to describe the demand for urban regulating ES: starting from them, the demand side is structured in a 'parallel cascade' that mirrors that of supply. This would help overcoming the still limited availability of methods and indicators for assessing the demand for urban regulating ES (Olander et al., 2018; Schmidt et al., 2016), and provide planners with valuable information to understand actual and potential beneficiaries. The explicit consideration for environmental conditions and the description of the demand side are the main innovative contributions brought by the framework to the conceptualization of urban regulating ES.

3.2.2 Analysing urban regulating services from a planning perspective

The framework was applied to the analysis of seven regulating ES, identified among those supplied by urban ecosystems (Gómez-Baggethun and Barton, 2013; Haase et al., 2014b; Luederitz et al., 2015). The analysed ES are listed in Table 3.1, together with the respective ecosystem functions and the supporting biophysical structures and processes (Haines-Young and Potschin, 2010). The application is based on the review of a wide scientific literature on urban regulating ES, which was selected through a snowball search starting from the references cited in Table 3.1, supplemented by other recent studies. Following the components of the framework, information relevant to planning was distilled in a coherent and comprehensive overview.

TABLE 3.1: *List of the analyzed urban regulating ES and identification of the respective ecosystem functions and supporting biophysical structures and processes. Modified after* (Elmqvist et al., 2016; Gómez-Baggethun and Barton, 2013). *Functions in italics are not further considered in the study. * For waste treatment, among the high number of existing typologies, the analysis is restricted to the illustrative cases of wetlands and vegetation strips.*

urban regulating ES	**ecosystem function**	**biophysical structure (process)**	**key references**
air purification	uptake of gaseous air pollutants	leaves	(Nowak et al., 2006)
	deposition of particles	vegetation	(Nowak et al., 2006)
global climate regulation	carbon sequestration	vegetation (photosynthesis) and soil	(Jo and McPherson, 1995; Nowak et al., 2013)
	carbon storage	vegetation and soil	(Pouyat et al., 2006; Strohbach and Haase, 2012)
moderation of extreme events	physical barrier (absorption of kinetic energy)	trees	(Danielsen et al., 2005; Dobbs et al., 2011)
noise reduction	reflection and diffraction of noise	vegetation and soil	(Van Renterghem et al., 2012)
	noise absorption	vegetation (mechanical vibration) and soft soil	(Van Renterghem et al., 2012)
runoff mitigation and flood control	water infiltration	permeable surfaces	(Yang et al., 2015)
	rainfall interception	tree canopies	(Xiao and McPherson, 2002)
	reduction of flood velocities	vegetation	(Nisbet and Thomas, 2006)
	water storage	floodplains	(Blackwell and Maltby, 2006)
urban temperature regulation	evapotranspiration	vegetation	(Coutts et al., 2012)
	shading	tree canopies	(Shashua-Bar and Hoffman, 2000)
	evaporation	water	(Saaroni and Ziv, 2003)
	heat transfer (storage and release)	*water bodies*	(Saaroni and Ziv, 2003)
	wind blocking	*trees*	(Huang et al., 1990)
waste treatment*	removal of storm water pollutants (sedimentation, filtration, sorption, assimilation and degradation)	ponds, wetlands, vegetated surfaces	(Clar et al., 2004a; Hemond and Benoit, 1988)
	decomposition of solid organic litter	*soil*	(Vauramo and Setälä, 2011)

3.3 The framework

The framework (Figure 3.1) conceptualizes the city as a socio-ecological system, where urban ES emerge as the result of complex interactions between the ecological, the socio-economic, and the governance spheres. Environmental conditions are at the centre of the framework. The presence of urban green infrastructure within given environmental conditions determines the supply of the services, which in turn performs a feedback regulatory effect on the environmental conditions of the city. On the opposite side of the framework, urban population and activities exposed to undesirable environmental conditions determine the demand for urban regulating ES. Both supply and demand can be spatially described, and the actual use of the services depends on the matching of the two respective areas. Benefits are therefore limited to such service benefitting areas, and their intensity depends on specific characteristics of the beneficiaries.

Urban planning responds, among others, to an unsatisfied demand for urban regulating ES, and acts on them through two main entry points: i) on the supply side, by determining the properties and spatial distribution of urban green infrastructure, and ii) on the demand side, by defining the spatial arrangement of urban population and activities. Concerning the supply side, conservation, restoration, enhancement, and creation of urban green infrastructure components are actions that planners can put in place to secure and enhance the provision of urban regulating ES (see Chapter Chapter 2). Concerning the demand side, planners can arrange land uses in a way that the demand from urban population and activities matches the existing supply (Rodríguez-Rodríguez et al., 2015). Through these decisions, urban planning determines the intensity and spatial distribution of urban regulating ES and of the related benefits across the city (Langemeyer et al., 2016).

The following sections details the components of the framework and their interactions.

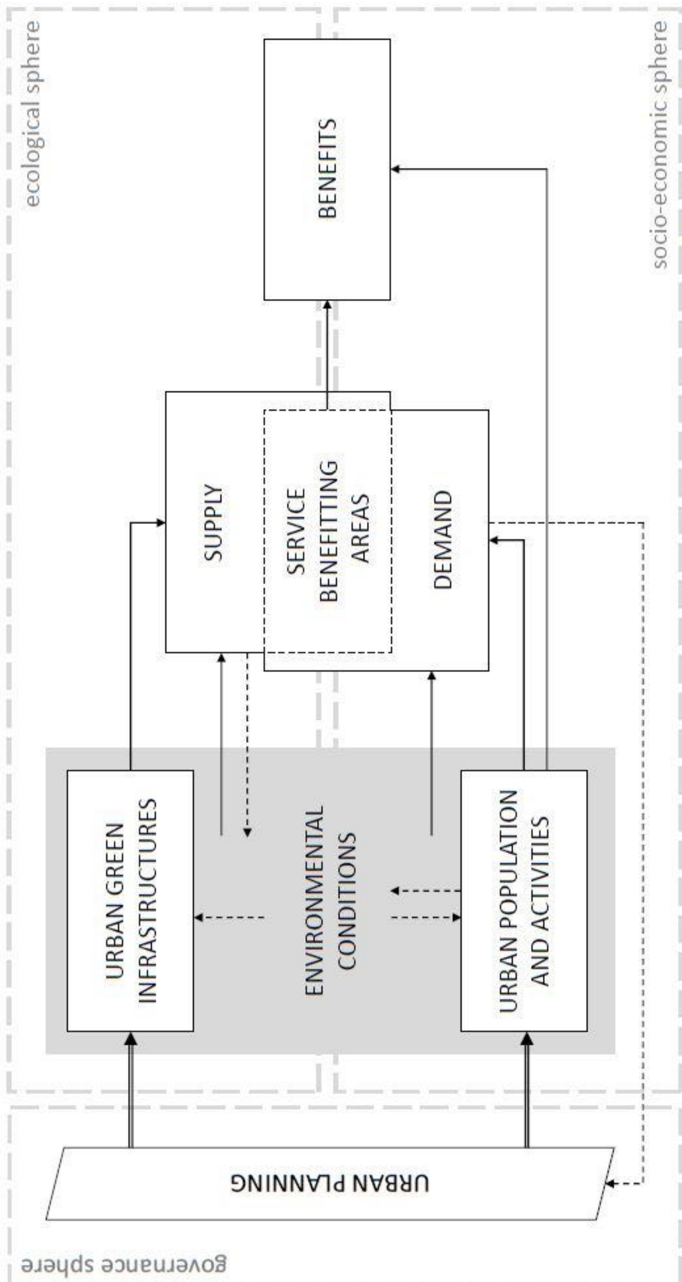

FIGURE 3.1: *Framework describing the key elements and interactions that determine the intensity and spatial distribution of urban regulating ES within the city. Arrows indicate primary (solid line) and secondary (dashed line) interactions in the process of ES provision and use. Thick arrows indicate the two main entry points for urban planning to enhance urban regulating ES.*

3.3.1 Environmental conditions

Environmental conditions define the context in which urban green infrastructure and urban population and activities are located, and exert multiple influences on both the supply and the demand side of urban regulating ES. On the supply side, environmental conditions act on green infrastructure components mainly through physical or chemical processes that produce modifications of their structure (Tzoulas et al., 2007), thus affecting their capacity to provide the services. From this perspective, they can be interpreted as pressures on urban ES, as commonly understood by other ES frameworks (Albert et al., 2015; Spanò et al., 2017). Additionally, the efficiency of some ecosystem functions depends on the intensity of environmental conditions, which therefore have a direct effect on the supply of the services. When this is the case, the effect depends on the function underpinning each service and may be positive (ES supply increases when the intensity of the respective environmental condition increases), or negative (ES supply decreases when the intensity of the respective environmental condition increases).

On the demand side, being factors that contribute to the suitability of land for the location of both residential and non-residential uses, environmental conditions affect the distribution of population and activities within the city. Furthermore, undesirable environmental conditions represent a risk for human health and security, ultimately affecting social and economic wellbeing. Hence, undesirable environmental conditions generate the need for regulation, which is at the basis of the demand for urban regulating ES.

3.3.2 Urban green infrastructure and the supply side

Urban green infrastructure components involved in the provision of ES can be defined as service providing units (SPU), i.e. “the components of biodiversity necessary to deliver a given ES at the level required by service beneficiaries” (Luck et al., 2003; Vandewalle et al., 2013). Different green infrastructure components are characterised by different ES potential, i.e. capacity to perform the ecological functions involved in the service (Bastian et al., 2012; Burkhard et al., 2014). The ES potential is determined by the biophysical, ecological, and dimensional properties of urban green infrastructure components, as shown in Figure 3.2.

Biophysical properties of urban green infrastructure components mainly refer to the typology of the components acting as SPU (Bartesaghi Koc et al., 2016; Braquinho et al., 2017). Although differences in the ES potential can be found among different species and individuals, detailed information is often unavailable for planning purposes. Hence, average performances based on typology are frequently used (e.g., Derkzen, van Teeffelen, & Verburg, 2015; Escobedo & Nowak, 2009). Classifications based on the identification of SPU (as opposed to land use-based assessments) commonly mention the following typologies: woodland/forest/coarse vegetation, trees, (tall and short) shrubs, grass/herbaceous vegetation/fine vegetation, bare soil/permeable surfaces, wetlands, and water, sometimes including mixed typologies based on management like private gardens or urban agriculture, to overcome data limitations (Baumgardner et al., 2012; Davies et al., 2011; Derkzen et al., 2015; Kremer and Hamstead, 2016; McPhearson et al., 2013).

Ecological properties of urban green infrastructure components refer to the baseline level of ecological organization required for ES supply (Andersson et al., 2015). For some urban regulating ES, individual components are not able to provide the service. This may happen in two cases. The first case is when urban regulating ES are emergent properties that require a minimum dimension and the interaction of different (biotic and a-biotic) factors to perform the underpinning functions (Escobedo et al., 2011). In this case, the smallest SPU may be an entire ecosystem. The second case is when, although individuals are able to perform the underpinning functions, the single contribution does not reach the minimum level of supply that is required, from the beneficiary perspective, to consider the performed function as an actual service (Luck et al., 2003; Vandewalle et al., 2013). In this case, the smallest SPU is generally a population.

Finally, dimensional properties of urban green infrastructure components refer to the sizes of the SPU that predominantly affect their biophysical potential. Depending on the urban regulating service, one or more properties (e.g., area, width, or length) of the SPU are usually adopted to calculate ES supply, either as proxies or as inputs for production functions and models (Maes et al., 2014; Nahuelhual et al., 2015). Hence, they are the most relevant from a planning perspective, and can be used to compare SPU with the same biophysical and ecological properties but different spatial extent. The relation between properties and ES potential may be linear or non-linear, with non-linear relations showing a decreasing efficiency, usually described by logarithmic functions.

3.3.3 Urban population and activities and the demand side

The concept of ES is strictly related to the presence of someone that gains benefits from the functions performed by urban green infrastructure. From this perspective, urban population and activities are at the basis of the demand for urban regulating ES, and their characteristics and spatial distribution affect the variability of demand and benefits across the city.

In general, demand for regulating ES is defined as “need for protection, achievement of predetermined conditions, mitigation” (Wolff et al., 2015), hence it indicates vulnerability to existing conditions (Bagstad et al., 2013; Nedkov and Burkhard, 2012) and distance from a desired state, with explicit reference to the concept of risk and hazard (Baró et al., 2016). The presence of urban population and activities exposed to undesirable environmental conditions indicates an unsatisfied demand. In principle, all the three components of vulnerability (i.e., exposure, sensitivity, and resilience (Turner et al., 2003)) affects the level of demand, which can be therefore differentiated on an individual basis. However, it is common practice in the analysis of urban regulating ES to assess demand by focusing only on the exposure component, and with reference to a desired state equally valid for the whole study area.

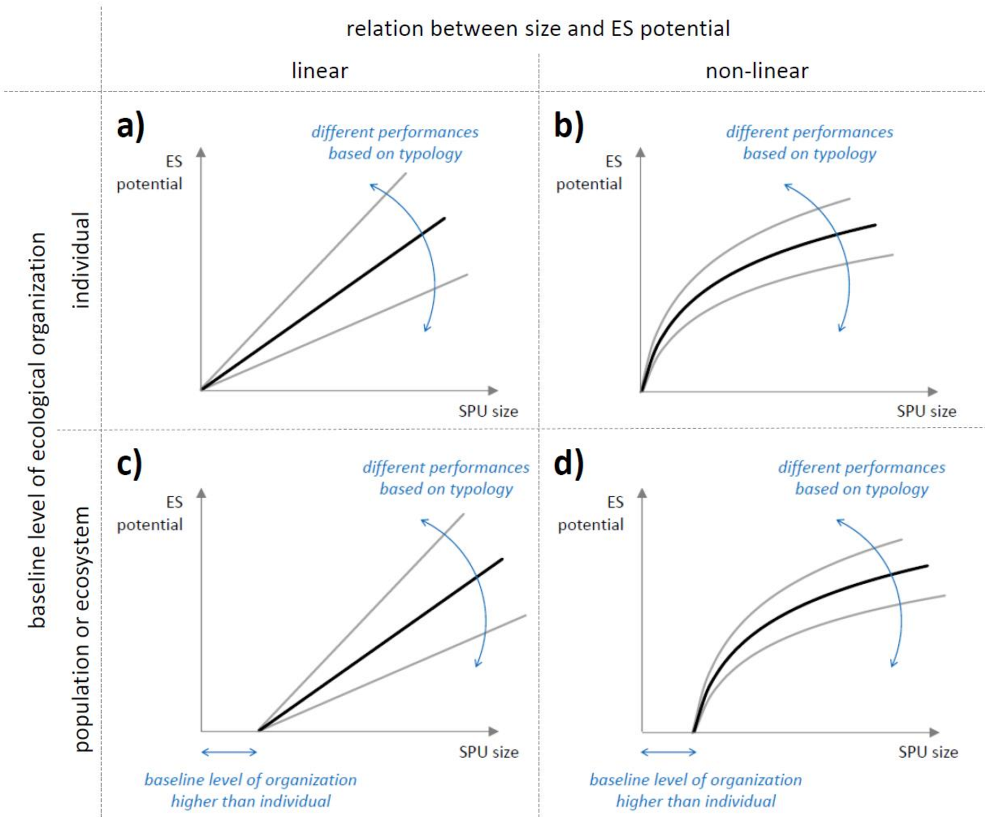

FIGURE 3.2: *Schematic representation of the possible relations between properties of urban green infrastructure components and their ES potential: a) linear relationship between size and ES potential; b) non-linear relationship between size and ES potential; c) linear relationship between size and ES potential with a baseline level of ecological organization higher than individual required for service supply; d) non-linear relationship between size and ES potential with a baseline level of ecological organization higher than individual required for service supply. Different slopes indicate the different ES potential of different typologies of urban green infrastructure components.*

Since benefits from most urban regulating ES are enjoyed unconsciously, the desired state is generally defined at the institutional level through environmental quality standards or targets to be achieved (Geijzendorffer and Roche, 2014). Thus, the distance between the standards or targets and the actual environmental conditions is adopted as an indicator of the intensity of demand (Baró et al., 2016, 2015; Burkhard et al., 2014). The total demand in a certain area can be quantified by multiplying the intensity of demand by the amount of urban population and activities exposed (Baró et al., 2016).

3.3.4 Service benefitting areas and benefits

Service benefitting areas (SBA) are those areas where ES are actually enjoyed by beneficiaries (Burkhard et al., 2014; Syrbe and Walz, 2012). From a spatial perspective, this requires areas covered by ES supply and areas hosting the demand for ES to overlap (Burkhard et al., 2012). Areas covered by ES supply, i.e. areas where environmental conditions are affected by the presence of urban green infrastructure, are potential SBA. Depending on the specific ES under consideration, potential SBA are characterized by different spatial relations with SPU and different spatial scales. Spatial relations between SPU and SBA have been classified into four classes: in situ, omnidirectional, directional upstream-downstream, directional buffer (Fisher et al., 2009; Syrbe and Walz, 2012). The type of spatial relation depends on the type of ecosystem functions that support service provision (e.g., mechanical, chemical, bio-physical functions) and on the environmental component that is regulated (e.g., air, water). The scale of SBA ranges from the very local scale of a single household to the global scale.

Contrarily to what happens to tradable ES, such as provisioning services, SBA of regulating ES are never decoupled from SPU (Burkhard et al., 2014). However, the presence of man-made infrastructure may mediate between the two areas, as in the case of the urban water sector. Once the potential SBA overlaps with demand areas, it is possible to calculate to what extent the supply covers the demand. Supply-demand ratios (Zhao et al., 2015) and budgets (Burkhard et al., 2012), and the level of unsatisfied demand (Baró et al., 2016) are possible ways to measure the efficiency in the provision of the services within each SBA, although the results of such approaches, especially when applied to regulating ES, require careful interpretation based on a clear understanding of the underlying hypotheses (Schröter et al., 2012).

The benefits generated by the provision of urban regulating ES depend on the vulnerability of beneficiaries to the regulated environmental condition. The vulnerability profile of beneficiaries can be described in terms of exposure-response ratio, i.e. in terms of expected outcome from the exposure to the undesirable environmental condition, which depends on the specific levels of sensitivity and resilience that characterize each beneficiary (Turner et al., 2003). Higher levels of sensitivity and lower levels of resilience determine a higher vulnerability, hence a greater benefit from urban regulating ES. Since urban regulating ES are purely non-rival (i.e., the use of the service by an individual does not affect the quality and quantity available to others) (Kemkes et al., 2010), benefits are not limited by crowding or congestion in SBA. Hence, in principle, a total benefit can be calculated as the sum of the benefits experienced by each beneficiary without accounting for variations in the level of supply due to use.

3.4 Application of the framework to seven urban regulating services

The framework was applied to the analysis of the seven urban regulating ES listed in Table 3.1. The main results are summarized in Tables 3.3-3.7 and Figure 3.3. The information can be navigated “service-wise”, by tracking a single service across the tables, or following the order of the main parts of the framework, thus gaining a transversal overview across the analysed services. The two ways of reading the findings complement one another and help to answer different planning questions, as exemplified in Table 3.2. The accompanying text comments on relevant issues and implications for planning.

TABLE 3.2: *Examples of planning questions associated to the different components of the framework that can be answered through a service-wise or a transversal reading of the findings. The first group includes planning questions linked to the enhancement of a specific urban regulating service and associated benefits, while the second group includes questions that explore the relation between multiple urban regulating ES.*

components of the framework	**examples of planning questions that can be answered through a SERVICE-WISE reading of the findings**	**examples of planning questions that can be answered through a TRANSVERSAL reading of the findings**
environmental conditions **Table 3.3**	• What environmental conditions are controlled by urban regulating ES and what are their effects on urban green infrastructure and urban population and activities?	• Are environmental conditions likely to produce negative effects on multiple urban regulating ES?
urban green infrastructure **Table 3.4**	• What green infrastructure components should be prioritized as service providing units? • What are the main features to consider in the design of green infrastructure as service providing units?	• What co-benefits in terms of multiple urban regulating ES can be expected from interventions on urban green infrastructure?
supply **Table 3.5**	• How does the dimension of urban green infrastructure components affect the supply of urban regulating ES? • Do environmental conditions have a direct effect on the supply of urban regulating ES?	• Is the supply of multiple urban regulating ES expected to show the same increase from the enhancement of urban green infrastructure?
urban population and activities **Table 3.6**	• What indicators can be used to describe the distribution of urban population and activities when assessing the demand for urban regulating ES?	• What is the influence of the spatial distribution of urban population and activities on the demand for multiple urban regulating ES?
demand **Table 3.6**	• What target values can be used to measure the demand for urban regulating ES?	• Is there any correlation among the levels of demand for multiple urban regulating ES?
service benefitting areas **Figure 3.3**	• Where should urban green infrastructure components and urban population and activities be located to maximize the number of beneficiaries of urban regulating ES? • What is the right scale to map and assess urban regulating ES?	• What bundles of urban regulating ES can be expected to show a similar distribution across the city? • What urban regulating ES can be analyzed at the same scale?
benefits **Table 3.7**	• What vulnerable groups/areas should be targeted to gain the highest benefits from the enhancement of urban regulating ES? • How can benefits associated to the enhancement of urban regulating ES be quantified?	• What vulnerable groups/areas should be targeted to maximize the benefits from multiple urban regulating ES?

3.4.1 Environmental conditions

Table 3.3 describes the environmental conditions linked to the urban regulating ES analysed, their spatial distribution within the city, and their influence on both urban green infrastructure and urban population and activities. Environmental conditions related to urban regulating ES are generally human-induced factors, often directly related with urbanization processes, although their scale varies from local to global phenomena. Most of the environmental conditions listed in Table 3.3 are commonly monitored in the context of spatial and sectoral plans, with the aim of assessing the quality of the urban environment (e.g., air, water, noise pollution) or the presence of risks, specifically those related to climate change (e.g., heat waves, floods, extreme events) (Galler et al., 2016). Thus, indicators of environmental conditions are usually available for mapping and assessment of urban regulating ES, and able to inform decision-making processes (Albert et al., 2015; Kandziora et al., 2013). The information on the spatial variability of environmental conditions, combined with their effects on supply and demand, is fundamental to define the best location of planning interventions, as it will be discussed in the following paragraphs.

3.4.2 Urban green infrastructure and the supply side

Table 3.4 describes the typologies of urban green infrastructure components involved in the provision of the analysed urban regulating ES, and the relevant biophysical, ecological, and dimensional properties that affect the supply. The approach, based on SPU, helps planners to address the high heterogeneity and fragmentation of the so-called urban ecosystems, which often are not ecosystems in proper ecological terms but patches and scattered elements (Cadenasso et al., 2007; Gómez-Baggethun and Barton, 2013; Müller et al., 2013). The identification of SPU allows distinguishing the smallest distinct homogeneous elements that can be addressed by planning and management (Andersson et al., 2015). At the same time, although resulting from a strong simplification of the biophysical functions and processes behind, the three relevant properties can be adopted as a first guideline to assess the ES potential of urban green infrastructure when more detailed information is not available.

As shown in Table 3.4, different typologies of urban green infrastructure, sometimes performing different ecosystem functions, are involved in the supply of the same service. This complexity, which may represent a challenge for ES assessments, is also an opportunity for planning. In most cases, different options exist to enhance the supply of urban regulating ES. Moreover, Table 3.4 highlights the multifunctionality of urban green infrastructure. Most of the typologies considered are multifunctional, i.e. they support the provision of a bundle of ES (Luederitz et al., 2015). Due to multifunctionality, similar spatial distributions of different urban ES emerge in cities (Holt et al., 2015) and synergies rather than trade-offs can be expected among urban regulating ES, as well as between them and some cultural and supporting services (Demuzere et al., 2014; Derkzen et al., 2015). Synergies among ES and the resulting multiple benefits are one of the main strengths of ecosystem-based approaches (European Commission, 2015; Geneletti and Zardo, 2016; Iacob et al., 2014), which planners can exploit when designing planning actions. Furthermore, accounting for synergies can improve the valuation of urban green infrastructure, and the assessment of alternative planning actions against multiple objectives (Kremer and Hamstead, 2016).

Table 3.5 describes how the properties of SPU and the intensity of environmental conditions affect the supply of urban regulating ES, based on methods and indicators commonly applied to measure the supply. The relation with the properties of SPU supports planners in the choice and design of urban green infrastructure components. In the case of ecosystem functions performed at the individual level and with a linear relation between SPU key size and ES potential, SPU quantity or dimension can be balanced by performance, i.e. bigger SPU of lower performance can be replaced by smaller SPU of higher performance. This happens, for example, for global climate regulation (Davies et al., 2011) and air purification (Weber et al., 2014). In other cases, the presence of non-linear relations or the need for a minimum dimension of the SPU entail the need for a careful choice of green infrastructure typologies. The relation with the intensity of environmental conditions, combined with information on their spatial variability (Table 3.3), provides essential knowledge for the location of urban green infrastructure. For example, in the case of air purification, since the amount of air pollution removed is directly proportional to pollution concentration, knowing the distribution of air pollutants in different areas of the city allows creating or enhancing urban green infrastructure where they can be more effective (Tallis et al., 2011).

3.4.3 Urban population and activities and the demand side

Table 3.6 lists indicators that can be used to describe the relevant properties of urban population and activities and the demand for urban regulating ES. The spatial assessment of demand in terms of exposure combines information on the spatial variability of environmental conditions (Table 3.3) with information on the spatial distribution of urban population and activities. Environmental conditions are measured with respect to environmental quality standards or targets that may be expressed at the local, national, or international levels. The spatial distribution of urban population and activities across the city is mostly described through indicators that are of common use in traditional planning practices. This should simplify their adoption, and could promote the emergence of new indicators and approaches through a cross-fertilization between planning and the ES science.

The assessment of demand, e.g. for assessing alternative scenarios or prioritizing planning interventions, is the stage where multiple objectives can be included. Objectives may encompass a wide range of social and economic goals, including equity (Kabisch and Haase, 2014) and poverty alleviation (Adem Esmail and Geneletti, 2017). For example, different weights can be assigned to demand areas with disadvantaged conditions in terms of green infrastructure availability or socio-economic status, independently from the enhancement of specific urban environmental conditions. Assigning multiple objectives to planning actions primarily aimed at increasing the provision of urban regulating ES denotes acknowledgement of the synergies among urban ES and of the multiple co-benefits of ecosystem-based actions. The exploitation of synergies and co-benefits generated by the multifunctionality of urban green infrastructure is favoured by the fact that, as demonstrated by the indicators in Table 3.6, high levels of demand for multiple ES are often concentrated in the same areas of the city.

TABLE 3.3: *Environmental conditions linked to the analyzed urban regulating ES, their spatial distribution within the city, and their influence on urban green infrastructure and urban population and activities. Key: uniform = affecting the whole city with the same intensity; variable = affecting the whole city with different intensities depending on the location; local = affecting only certain areas of the city.*

urban regulating ES	environmental condition	spatial distribution	main effects on urban green infrastructure	main effects on urban population and activities
air purification	concentration of air pollutants (PM_{10}, $PM_{2.5}$, NO_2, O_3, CO, SO_2)	variable	Elevated ozone concentrations reduce tree biomass and leaf area. (Wittig et al., 2009) Concentrations of air pollutants delays spring phenology. (Jochner et al., 2015)	Ambient air pollution is responsible for 14% of the disease burden of lung cancer, 23% of ischemic heart disease, 25% of stroke and 9% of chronic obstructive pulmonary disease worldwide. (Prüss-Üstün et al., 2016)
global climate regulation	concentration of greenhouse gases (CO_2)	uniform	-	Climate change acts as a driver of more frequent and intense extreme events, including extreme precipitation and heat waves, with negative effects on both human health and the economy. (Patz et al., 2014; Revi et al., 2014)
moderation of extreme events	storms, floods and waves	local	Floods, waves, and storms damage trees and remove vegetation. (Escobedo et al., 2009; Yanosky, 1982)	Extreme events put at risk people, infrastructures, and economic activities. (Jahn, 2015)
noise reduction	noise	local	-	Traffic noise induces annoyance, stress, and sleep disturbances, and increase the risk for ischaemic heart disease, stroke, and hypertensive diseases. Noise disturbance also produces a significant decrease in housing and renting prices. (Vienneau et al., 2015)

TABLE 3.3 *(continued)*

urban regulating ES	environmental condition	spatial distribution	main effects on urban green infrastructure	main effects on urban population and activities
runoff mitigation and flood control	stormwater runoff	variable	-	Urban flooding damages buildings and infrastructures and has negative effects on transport systems. (ten Veldhuis and Clemens, 2010) Urban flooding causes health effects in terms of mortality (drowning), injuries, and infections, plus a wide range of psychological and mental health effects. (Fewtrell et al., 2008)
urban temperature regulation	urban heat island and heat waves	variable	Droughts and limited water availability may lead to leaf senescence, reduced transpiration, loss of canopy cover, and vegetation death. (Coutts et al., 2012)	Mortality rates and hospital admissions for heat-related, cardiovascular, and respiratory diseases increase during heat waves. (D'Ippoliti et al., 2010; Mastrangelo et al., 2007) Urban heat island exacerbates the negative effects of heat waves in urban areas. (Tan et al., 2010)
waste treatment	concentration of stormwater contaminants	variable	-	Stormwater discharge is a major cause of pollution in receiving waters, damaging ecosystems, contaminating drinking water supplies, and making recreational areas unsafe and unpleasant. (Barbosa et al., 2012)

Table 3.4: *Urban green infrastructure components as service providing units for urban regulating ES: typologies, baseline levels of ecological organization, and sizes. Modified and expanded after Andersson et al. (2015). 'Soil' is to be interpreted as bare (permeable) soil.*

urban regulating ES	urban green infrastructure components		
	typology	baseline level of ecological organization	key size for planning
air purification	trees, shrubs	individual	area
global climate regulation	trees, shrubs, soil	individual	area
moderation of extreme events	trees, wetlands	population	width of the buffer zone
noise reduction	trees, shrubs, herbaceous vegetation	population	width of the buffer zone, length parallel to the source (for linear sources, e.g. traffic)
runoff mitigation and flood control	trees, shrubs, soil, wetlands	population	area (for interception and infiltration), volume (for storage)
urban temperature regulation	trees, shrubs, herbaceous vegetation, wetlands, water courses, water bodies	individual	area, shape index
waste treatment	herbaceous vegetation, soil, wetlands	ecosystem	wetland-to-watershed area (for wetlands) / length in the direction of water flow (for vegetation strips)

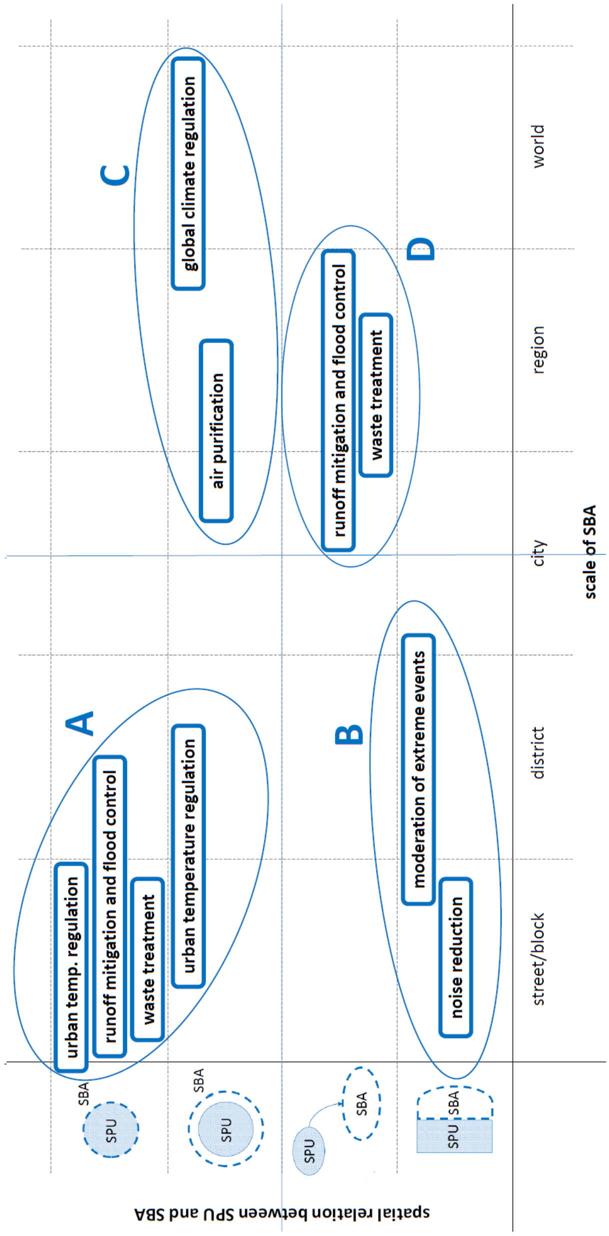

FIGURE 3.3: *Spatial scale of potential SBA (x-axis) and spatial relation between SPU and SBA (y-axis) for urban regulating ES. The categories of spatial relations between SPU and SBA follow the classification proposed by Fisher et al. (2009) and Syrbe and Walz (2012), namely (from top to bottom) in situ, omnidirectional, directional upstream-downstream, directional buffer. The scale of SBA is identified by mean of five illustrative definitions. The four clusters correspond to local services with homogeneous effects (A), local services with directional effect (B), supra-local services with homogeneous effect (C), and supra-local services with directional effect (D).*

TABLE 3.5: *The supply of urban regulating ES: possible indicators, and relations with the properties of SPU and with the intensity environmental conditions. Types of relation with the properties of SPU are described in Figure 3.2. Types of relation with the intensity of environmental conditions are indicated according to the following key: ↑ = increase, ↓ = decrease, ⇔ no effect.*

urban regulating ES	indicator of ES supply [unit]	relation ES supply/SPU properties	relation ES supply / environmental condition
air purification	pollution removal [t/yr.]	a) Pollution removal increases linearly with the area of canopy cover. The total amount of air pollution removed in a certain period of time through dry deposition can be calculated multiplying the total area of canopy cover by the average value of the pollutant flux. (Nowak et al., 2006)	↑ The amount of air pollution removed through deposition is directly proportional to pollution concentration. (Nowak et al., 2006)
global climate regulation	carbon storage [t], carbon sequestration [t/yr.]	a) Carbon storage and carbon sequestration increase linearly with the area covered by trees. The total amount of carbon stored/sequestered in a certain period of time can be calculated multiplying the total area of tree cover by average values of carbon storage/sequestration per square meter. (Nowak et al., 2013)	↑ Growth rate, hence carbon sequestration, increases with CO_2 concentration. (Curtis and Wang, 1998)
moderation of extreme events	wave height reduction [%]	d) Wave height reduction increases less than linearly (quadratic function) with the width of the buffer element. (Barbier et al., 2008)	⇔
noise reduction	excess attenuation [dBA]	c) / d) Relative attenuation of noise increases with the width of the tree belt. Studies report both linear and less-than-linear relations. (Aylor, 1972; Fang and Ling, 2003; Van Renterghem, 2014)	⇔

TABLE 3.5 *(continued)*

urban regulating ES	**indicator of ES supply [unit]**	**relation ES supply/SPU properties**	**relation ES supply/environmental condition**
runoff mitigation and flood control	avoided runoff	a) / c) (Farrugia et al., 2013)	↓ Infiltration decreases with increasing moisture conditions due to previous events. (Liu et al., 2014)
urban temperature regulation	Δt [°C]	b) The intensity of the cooling island produced by parks and wetlands increases less than linearly (logarithmic function) with the area and with the inverse of the shape index. (Cao et al., 2010; Sun et al., 2012)	↓ ↑ Evapotranspiration decreases with temperature and dryness. (Coutts et al., 2012) The cooling effect due to shading increases with background temperature. (Shashua-Bar and Hoffman, 2000)
waste treatment*	pollution removal efficiency [%]	d) Pollutants removal efficiency of wetlands is correlated with the logarithm of the wetland-to-watershed area ratio. Pollutants removal efficiency of vegetation strips follows a similar trend with respect to the length of the strips. (Carleton et al., 2001; Clar et al., 2004b)	↓ Treatment efficiency decreases above a certain water load. (Clar et al., 2004b)

TABLE 3.6: *Exemplary indicators for the spatially-explicit assessment of demand for urban regulating ES.*

urban regulating ES	spatial distribution of population and activities	environmental quality standard or target
air purification	population density (Baró et al., 2016; Morani et al., 2011)	air quality targets (e.g., European Union, 2008) (Baró et al., 2016)
global climate regulation	census population; transportation, agricultural and industrial intensity per census tract (Zhao et al., 2015)	emission reduction targets (e.g., Covenant of Mayors) (Baró et al., 2014) target equal to carbon emissions (Zhao et al., 2015)
moderation of extreme events	population density, road density, percentage of artificial surfaces, number of historical and cultural sites (Liquete et al., 2013)	acceptable risk based on the return time of the event (Liquete et al., 2013)
noise reduction	residential and recreational areas (Syrbe and Walz, 2012)	target noise levels, e.g. (WHO, 2009)
runoff mitigation and flood control	flood-vulnerable properties (Bagstad et al., 2014); built areas (Syrbe and Walz, 2012); households (Syrbe and Walz, 2012)	acceptable risk based on the return time of the event (Olsen et al., 2015)
urban temperature regulation	census population (Geneletti et al., 2016)	based on a common definition of heatwave (e.g., Fischer and Schär (2010)) (Baró et al., 2015) critical heat index (Bodnaruk et al., 2017)
waste treatment	-	quality standards for the receiving waters (e.g., European Union, 2000) post-construction stormwater standards (e.g., US EPA, 2011)

3.4.4 Service benefitting areas and benefits

Figure 3.3 shows the spatial scale of potential SBA and their spatial relation with SPU. Potential SBA are areas where environmental conditions are positively affected by the supply of urban regulating ES: only the overlap with demand areas, i.e. the presence of beneficiaries, makes them actual SBA. Identifying the location of SBA and populating them with the actual beneficiaries (being them people, properties, or other ecosystems) represents a necessary step for a meaningful valuation of urban regulating ES, as well as for a complete understanding of the effects of spatial transformations on their provision (Bagstad et al., 2014; Olander et al., 2018). Knowing where beneficiaries are is useful to define policies on the use and management of urban regulating ES (Fisher et al., 2009), and to identify winners and losers of land use changes (Bagstad et al., 2014).

Table 3.7 identifies vulnerable groups and areas, and indicators that can be adopted to measure the benefits provided by urban regulating ES. Direct benefits from urban regulating ES are mostly in terms of increased human health and security, while indirect benefits also involve the spheres of materials for good life and good social relations (MA, 2005). Accounting for the different levels of demand that correspond to different users and uses of urban areas, and to the related vulnerability profiles, allows maximizing the benefits produced by urban regulating ES. As it can be observed, most vulnerable groups are the same across the different services, hence targeting them as beneficiaries could significantly increase the benefits produced by planning actions.

The identification of SBA, beneficiaries, and benefits based on the spatial relation between supply and demand is useful for planning, since it allows defining the location of green infrastructure components based on the areas where they are more needed or desired. Moreover, it supports the use of the second entry point for planning: the distribution of population and activities across the city. As exemplified by (Geneletti et al., 2016) for microclimate regulation, the supply of urban regulating ES may be used as a positive location factor, for example in the prioritization of urban infill interventions. Given the non-rival character of urban regulating ES (Kemkes et al., 2010), also “placing more beneficiaries across the landscape” (Bagstad et al., 2014) may be an effective strategy to increase their provision and the related benefits, although the increased reliance on the same SPU has to be taken into account (Bagstad et al., 2014).

A final remark comes from Figure 3.3, which highlights that beneficiaries and benefits may be not limited to the urban scale. SBA of some urban regulating ES go well beyond the boundaries of the city, up to the global scale in the case of global climate regulation. Enhancing the provision of supra-local urban regulating ES is therefore a positive contribution that cities can offer to the quality of a wider environment. This strengthen once more the need for planning processes that overcome administrative boundaries, and opens to the implementation of ecosystem-based actions in cities as parts of mitigation and compensatory schemes at the landscape scale (Knight and Landres, 2013).

Table 3.7: *Specific vulnerable groups and areas to be considered and exemplary indicators for the assessment of benefits from urban regulating ES.*

urban regulating ES	specific vulnerable groups/areas	benefit indicator
air purification	foetuses and children, elderlies, and persons with pre-existing cardiorespiratory diseases, diabetes, or asthma (Makri and Stilianakis, 2008)	reduction of premature deaths and hospital admissions (Mindell and Joffe, 2004; Tiwary et al., 2009) avoided externalities (Nowak et al., 2006)
global climate regulation	-	monetary value based on carbon market prices (Zheng et al., 2013) or on estimated marginal social costs of carbon dioxide emissions (Nowak et al., 2008)
moderation of extreme events	vulnerable areas based on number of people and total cost of damage (Wei et al., 2004)	reduction of human deaths (Das and Vincent, 2009) replacement cost of engineering structures (Narayan et al., 2016)
noise reduction	children, elderly, chronically ill (WHO, 2009)	number of person with change from annoyed to not annoyed or dB(A) change per person/household per year, and related economic value based on hedonic pricing (Veisten et al., 2012)
runoff mitigation and flood control	vulnerable areas based on damage cost (Olsen et al., 2015)	avoided damage, based on the total value of properties protected (Nedkov and Burkhard, 2012) or on specific depth-damage functions for different land use-land cover types (Olsen et al., 2015) replacement cost of manmade substitutes (Silvennoinen et al., 2017)
urban temperature regulation	infants; elderlies; people with obesity, hypertension, pulmonary, or cardiovascular disease; people with restricted mobility; people living alone and lacking social contacts; low-income groups (Basu and Samet, 2002; Kenny et al., 2010) urban areas with more intense heat island effect based on density and lack of green spaces (EEA, 2012)	reduction in cumulative population-risk weighted exceedance heat index (Bodnaruk et al., 2017) total number of people and number of vulnerable people exposed to the cooling effect of urban green infrastructure (Geneletti et al., 2016)
waste treatment	-	savings based on replacement cost (Breaux et al., 1995)

3.5 Discussion

This study responded to the growing demand for frameworks to support planners in designing effective actions that enhance the provision of ES (Koschke et al., 2012; Langemeyer et al., 2016). Although good practices of planning for urban regulating ES are spreading, also in response to a growing call for the implementation of nature-based solutions (Raymond et al., 2017), a scientifically-sound design of planning actions tailored to the specific socio-ecological context of application is often overlooked, ultimately limiting their effectiveness (Chapter 2; Geneletti and Zardo, 2016). The proposed framework builds on two among the most popular models and approaches that have demonstrated applicability to planning contexts: the Cascade conceptual model and the supply-demand approach for mapping ES (Burkhard et al., 2012; Potschin-Young et al., 2017; Spangenberg et al., 2014). Elements from the two are combined, thus taking a step forward to their unification, and detailed to meet the specific characteristics of urban regulating ES.

By focusing on urban regulating ES, the study contributes to overcome the main barriers to their operationalization in planning. A first barrier is related to complexity. Several authors highlighted the importance of providing simple and easy-to-use information, models, and tools, to support decision-making (Ruckelshaus et al., 2015; Slootweg, 2015). In the case of urban regulating ES, most of the complexity is due to the number of variables involved. The proposed framework breaks down this complexity by identifying the key elements involved in the process of ES production and use, and describing their roles and interactions. Identifying the causal relations among the components of the framework supports the assessment of planning and management decisions based on how their effects are expected to propagate, ultimately enhancing or reducing benefits from urban regulating ES (Olander et al., 2018).

Unlike many of the most common ES conceptual frameworks (e.g., the MAES (Maes et al., 2013), the ‘cascade-integrated’ DPSIR (Müller and Burkhard, 2012), the EPPS (Bastian et al., 2012), and the ES-in-Planning (Albert et al., 2015)) that include environmental conditions in a general definition of drivers and pressures affecting the provision of ES, the proposed framework describes the specific effects of environmental conditions on both supply and demand. This should clarify the use of indicators of environmental conditions in the context of urban regulating service assessments. Indicators of environmental conditions are frequently adopted as proxies of regulating ES, but it is often unclear whether they measure demand or supply, and to what stage of the cascade they refer (see for example the list in Kandziora, Burkhard, & Müller (2013)).

Moreover, it is commonly assumed that biophysical indicators provide adequate measures of ES provision, while social aspects related to needs and values are overlooked (Olander et al., 2018). A poor definition of the demand side has been recognized as a key barrier to the operationalization of scientific knowledge on urban regulating ES in planning (Bagstad et al., 2014; McPhearson et al., 2014). The proposed framework advances the understanding of the demand for urban regulating ES by drawing a 'parallel cascade' from urban population and activities to ES and benefits, and detailing the causal links between vulnerability to undesirable environmental conditions and demand for urban regulating ES (Bagstad et al., 2013).

To help planners decide "where to put things" (Polasky et al., 2008), the indicators selected to characterize each component of the framework are mostly retrieved from spatially-explicit analyses of urban regulating ES. Spatially explicit concepts and indicators are considered essential to integrating the ES approach in urban planning (Haase et al., 2014b) and this research demonstrates that, on both the supply and the demand side, spatial distribution is as much important as quantity in determining the benefits from urban regulating ES. But more than a collection of useful indicators and illustrative applications for urban regulating service assessments, the framework offers planners guidance to enhance the provision of urban regulating ES and associated benefits in cities. Two entry points are identified: acting on the supply side, by improving urban green infrastructure availability and efficiency, and acting on the demand side, by enabling people to more effectively benefit from the services. The application of the framework allows understanding, based on the specific service of interest and on the existing conditions, what are the most relevant variables on which the results of planning actions depend, and which path can be expected to produce the highest benefits.

Among the potential aims of conceptual frameworks listed by (Potschin-Young et al., 2017), this chapter mainly refers to its use as an ‘organizing structure’ that provides “a shared language and a common set of relationships and definitions to make complex systems as simple as they need to be for their intended purpose” (Díaz et al., 2015). Here, the intended purpose is to support effective planning actions, and organizing the available scientific knowledge appears a first step toward its operationalization. However, potential users of the framework should be aware of the degree of simplification that this implies. The simplification is evident in the description of the complex biophysical functions and processes at the basis of ES supply, boiled down to three key properties (namely typology, level of ecological organization, and size). Although this may seem a strong limitation, the three properties were identified based on a review of models, methods, and indicators typically available for urban planners, who often cannot perform in-depth analyses. More detailed information, whenever existing and usable, can feed the application of the framework to real-world planning contexts.

Furthermore, it should be noted that the chapter focuses only on regulating ES provided by green infrastructure within the city, and on planning processes at the urban scale. However, the availability, spatial distribution, and functionality of urban green infrastructure are also affected by planning decisions at wider scales. At the same time, a ‘good’ urban planner should consider the effects of planning actions beyond the territorial boundaries of the city. Not only, as highlighted by the application of the framework, the service benefitting areas of some urban regulating ES can be bigger than the city, or located outside its boundaries, but also the localization of urban population and activities may produce consequences on a wider scale. Finally, despite the effort to describe the main interactions and feedbacks, the framework schematizes only the main and the most direct relations in the production of ES (Ernstson, 2013). The ‘urban planning’ component of the framework, in particular, should be intended as a complex decision-making process (Mckenzie et al., 2014) rather than simply as its outcomes. Applications to real-world case studies are needed to test on the ground the usability of the framework in the different stages of the planning process, and to assess the benefits of its adoption compared to more traditional planning approaches (Geneletti et al., 2017).

3.6 Conclusions

Overall, the study recomposed a fragmented scientific evidence on urban regulating ES and provided guidance for urban planners to integrate relevant knowledge in planning practices. A successful transfer of scientific knowledge on ES is expected to improve various stages of the planning process: analysing conditions and identifying existing needs, defining goals and expected performances, designing and assessing alternatives, and prioritizing the most effective solutions (Adem Esmail and Geneletti, 2017; Geneletti, 2015; Langemeyer et al., 2016). By describing the elements involved in the process of urban regulating ES provision and their interactions, the proposed framework identified the entry points and the pathways through which planning decisions affect the provision of urban regulating ES and associated benefits in cities. This should make planners aware of the socio-ecological processes behind (Ernstson, 2013; Kremer et al., 2016), and of the levers on which they can act. The indicators proposed, albeit only illustrative of the approach, proved to be informative, and can be adopted to assess current and expected conditions, ultimately supporting the design of planning actions and the assessment of their impacts on the provision of urban regulating ES. Within the context of a progressive spreading of ecosystem-based actions (Chapter 2; Geneletti and Zardo, 2016) and a growing call for the implementation of nature-based solutions (Kabisch et al., 2017; Raymond et al., 2017; van den Bosch and Sang, 2017), this could support the operationalization of existing ES knowledge towards more effective planning actions.

Chapter 4

Applying an ecosystem service approach to support real-life planning decisions: the case of brownfield regeneration in Trento

4.1 Introduction

After conceptually exploring the relationship between ecosystem services (ES) and planning actions in Chapter 3, the aim here is to investigate if and how the ES approach can provide a valid support for guiding real-life planning decisions. Among the pathways through which ES knowledge can impact policy- and decision-making, the chapter focuses on the use of ES knowledge to 'generate actions' and 'produce outcomes' (Posner et al., 2016b). At this level, the expected results of knowledge integration are new or updated plans and policies that consider impacts on ES and promote their balanced provision, ultimately improving human health and wellbeing along with biodiversity and nature conservation (Posner et al., 2016b). Drawing from a set of case studies in which ES knowledge was used to support decision-making, Barton et al. (2017) provide some examples of the

* This chapter is based on: Cortinovis, C., Geneletti, D. (in review). Mapping and assessing ecosystem services to support urban planning: A case study on brownfield regeneration. *OneEcosystem.*

tasks that ES assessments can perform along these pathways. When used with a 'decisive' role, ES knowledge can support the formulation and structuring of the decision problem; help to identify criteria for screening, ranking, and spatial-targeting of the alternatives; and provide arguments for negotiations, shared norms, and conflict resolution. When used with a 'design' role, ES knowledge can set the basis for a wide range of implementation tools, from the definition of standards and policy targets, to the design of regulations, certifications, pricing, and incentives, to the establishment of damage compensations (Barton et al., 2017). The identified tasks correspond to different stages of the decision-making process, with different requirements for knowledge to be considered useful and usable.

Indeed, ES assessment is a process itself (Rosenthal et al., 2015), and the interface with the different stages of the decision-making process that it aims to support determines the relevance, usability, and potential impacts of its results (Adem Esmail and Geneletti, 2017; Mckenzie et al., 2014). Cowling et al. (2008) propose an operational model for mainstreaming ES in land-use planning based on three stages, namely, *assessment*, *planning*, and *management*. The three stages correspond to a progressively smaller scale of analysis and to different levels of stakeholders' involvement, from informed to empowered. Rosenthal et al. (2015) further detail the process and identify six steps for decision-relevant ES assessments: i) scope, ii) collect and compile data, iii) develop scenario, iv) analyse ES, v) synthesize results, and vi) communicate knowledge. Although these descriptions help the 'ES side' to structure a rational process, they provide little information on how to manage the interface with the decision-making process that it aims to support.

Geneletti (2015) provides an example of how the interface between ES assessments and decision-making processes can be framed. Focusing on strategic decisions and associated Strategic Environmental Assessment (SEA), Geneletti first describes the SEA process, characterized by a consolidated sequence of stages and respective objectives, and then identifies the role of ES assessments with respect to each stage. A similar approach can be applied to spatial planning, for example considering the 'Ecological Planning model' proposed by Frederick Steiner (Steiner, 2008). The author himself summarizes the model as a process that "involves setting goals, assessing the environment, analyzing suitabilities, exploring options, selecting a course of action, testing those actions through design, and implementing a plan. In a democracy, the public is involved throughout the process." (Steiner, 2016). By further generalizing, the main stages of *objectives*, *analyses*, *decisions*, *implementation*, and *administration* can be identified. As shown in Figure 4.1, a certain correspondence between the main stages of the planning process and the operational model defined by Cowling et al. (2008) emerges. Moreover, different tasks of ES assessments (Barton et al., 2017) can be linked to the different stages of the planning process (Figure 4.1).

Within this framework, urban ES literature appears unbalanced in favour of the stage of *analyses*. In fact, most ES studies lack the identification of specific planning questions and stakeholders to which they may be relevant, and their recommendations are often limited to the generic assertion that findings should be somehow taken into account by planning and management (Haase et al., 2014b). Very few studies are explicitly aimed at supporting the phase of *decisions* by assessing future scenarios formulated during the planning process, as done for example by Kain et al. (2016) and Sanon et al. (2012). This use requires defining appropriate indicators for measuring the expected outcomes of planning scenarios in terms of changes in human wellbeing, which is still a challenge for ES science (Ruckelshaus et al., 2015). Moreover, while most ES assessments focus on the supply of a single ES (Haase et al., 2014b), evaluating planning scenarios requires assessing the consequences of planning interventions on both the supply and the demand of multiple ES (see Chapter 3), explicitly addressing potential trade-offs among different ES and competing land uses (Kain et al., 2016; Sanon et al., 2012; Woodruff and BenDor, 2016).

The chapter presents an application of ES assessments to support a real-life planning decision related to the prioritization of re-greening interventions on brownfield sites. Two illustrative ES are assessed in the current condition and under future planning scenarios, and the results are combined through a multi-criteria analysis. Section 4.2 introduces the case study, and Section 4.3 describes the methods applied for the assessment. The results are presented in Section 4.4 and discussed in Section 4.5. Finally, Section 4.6 draws some concluding remarks.

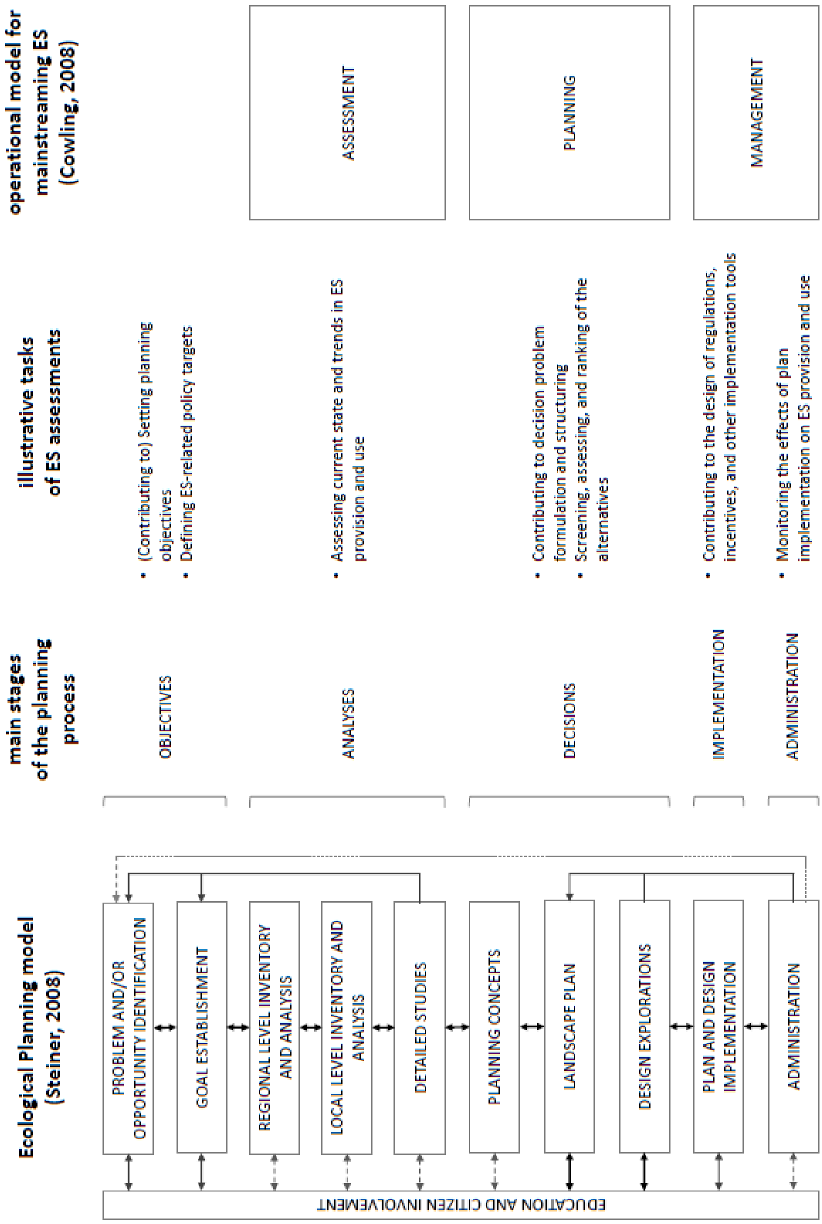

FIGURE 4.1: *The 'Ecological Planning model' by F. Steiner* (Steiner, 2008) *(left) and its generalization into five main stages, with corresponding tasks for ES assessments (expanded after* (Barton et al., 2017)*). The relation between the generalized planning process and the operational model for mainstreaming ES proposed by* Cowling et al. (2008) *is highlighted in the right column.*

4.2 Case study

The case study takes inspiration from a real-life planning issue in the city of Trento, an alpine city in the North-Eastern part of Italy. The following sections briefly describe the context, the specific planning issue, and the objectives of the analysis, and justifies the approach and the selection of the ES to consider for the assessment.

4.2.1 Context and objectives of the analysis

Trento, provincial capital of Trentino, is a medium-sized city of around 120,000 inhabitants located along the valley of the Adige river, half-way between the Brenner pass and the Adriatic Sea. The main settlement hosts around 70% of the population and originates from the concentration of urban areas and infrastructures in the valley floor. The remaining 30% of the population lives in small villages spread across the large municipal territory (156 km^2). Agricultural areas, predominantly vineyards and orchards, occupy the few non-urbanized patches on the valley floor, and the sunny hillsides. The rest of the municipal area, up to an elevation of 2,180 m, is covered by forests (almost half of the total area). Natural protected areas account for more than 10 km^2, including 7 Natura2000 sites and 3 local reserves. Figure 4.2 provides an overview of the distribution of the main land uses in the city.

When in 2017 the municipal administration started the process for drafting a new urban plan, the regeneration of brownfields emerged as one of the main issues. Of the 13 ‘urban redevelopment areas’ identified by the previous plan, none has been converted during its implementation period. Most of the areas are former industrial sites or partially abandoned residential areas, ranging in size from 0.5 to 9.9 ha (Figure 4.2). The main problems are related to the costs (some of the areas area also contaminated and in need of remediation), the bureaucratic burden associated to redevelopment interventions, and the sometimes-contrasting interests of public administration and private owners. With few exceptions, the brownfields are close to the most populated parts of the city, i.e. the historical centre and the recent residential expansions to the North, thus their redevelopment would have an impact on a large part of the population. Considering the existing situation, it can be expected that only some of the brownfields will be converted to new industrial or residential areas in the next years. At the same time, for some of the areas, greening interventions can be advanced as a solution (maybe even a temporary one).

Accordingly, the study is aimed at supporting the decision about which of the existing brownfield could be converted into a new public green area, based on the expected benefits that the intervention would produce for the surrounding population. To this aim, a scenario representing the conversion to a public park was modeled for each brownfield and assessed in terms of provision of key ES. Through a comparison between the scenarios and the baseline condition, it is possible to quantify the expected benefits of each transformation, hence to compare the different scenarios and to rank the alternatives based on their performance.

4.2.2 Selection of key ecosystem services

Two key urban ES for Trento are used to assess brownfield redevelopment scenarios, namely microclimate regulation and recreation. The selection of microclimate regulation is linked to the growing concerns for summer heat waves, particularly intense in the city due to the low altitude and to the narrowness of the valley. As demonstrated by the 2003 event, Trento is more vulnerable to heat waves than other Italian cities (Conti et al., 2005).Heat wave effects combine with the urban heat island, particularly intense in the most urbanized and sealed part of the city (Giovannini et al., 2011), causing peaks in energy demand and posing serious threats to citizens' health and wellbeing. Considering the increased frequency and intensity of heat waves expected in the coming decades (Fischer and Schär, 2010), effective solutions to control the urban microclimate and provide cool areas for heat relief during the hot season are seen as one of the most pressing needs by citizens and administration.

The selection of recreation responds to a specific interest of city administration. One of the main aims of planning interventions in the last years has been to gain a more balanced distribution of public green areas over the city, thus providing equal opportunities for recreation and relaxation to all citizens. However, understanding if opportunities for nature-based recreation are equally distributed is not easy. Unlike other urban contexts, in Trento citizens benefit from the proximity to different typologies of green areas where they conduct a wide range of nature-based recreational activities. In addition to those commonly carried out in urban parks, typical day-to-day recreational activities in Trento include hiking, mountain-biking, skyrunning, and climbing in nearby forests and mountain areas. Hence, indicators based on the availability of and accessibility to public urban parks, though common in urban planning applications, are not enough to support planning actions. On the contrary, assessing recreation as an ecosystem service, considering different providing units and different levels of demand, could provide planners with useful information for achieving an equal distribution of recreational opportunities over the city.

The assessment of recreation is in line with the main planning objectives of the city administration. In the last years, the enhancement of public green areas has been targeted toward gaining a more balanced distribution over the city, hence providing equal opportunities to all citizens for recreation and relaxation. However, understanding if opportunities for nature-based recreation are equally distributed is not an easy task. In Trento, besides urban parks, citizens also benefit from the proximity to other typologies of green areas where they conduct a wide range of activities, including hiking, mountain-biking, skyrunning, and climbing. Indicators based on the availability of and accessibility to public urban parks are not enough to capture this variety. Assessing recreation as an ecosystem service, considering different providing units and different levels of demand, could provide planners with useful information for achieving an equal distribution of recreational opportunities over the city.

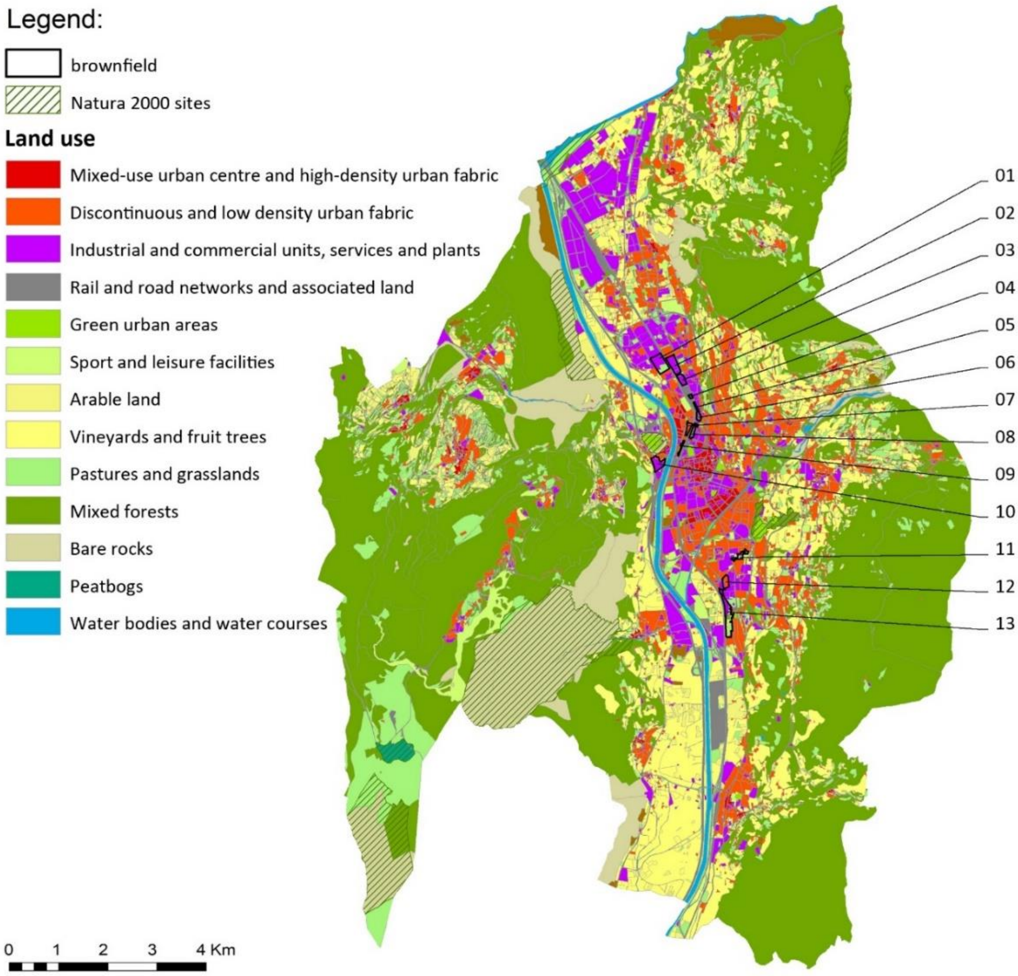

FIGURE 4.2*: Main land uses in Trento, Natura2000 sites, and the 13 brownfields identified by the urban plan as 'urban redevelopment areas'. (Source: webGIS database of the municipality and province of Trento, last accessed: December 2017).*

4.3 Methods

4.3.1 Mapping the cooling capacity and cooling effect of urban green infrastructure

The mapping and assessment of the cooling capacity and cooling effect of urban green infrastructure components was carried out through a method specifically designed to support planning and management decisions at the urban and sub-urban scale (Zardo et al., 2017). The method is based on the assessment of the main properties that affect the two ecosystem functions involved, namely shading and evapotranspiration. The three main properties are soil cover, canopy coverage, and size. Soil cover is classified into five categories, namely *water*, *grass*, *heterogeneous*, *bare soil*, and *sealed*. Since the cooling capacity only depends on the biophysical structure, contiguous areas with the same soil cover but different uses or properties are merged into a single polygon of soil cover. Canopy coverage is calculated as a percentage over each soil cover polygon, and classified into five classes (i.e., 0%-20%, 20%-40%, 40%-60%, 60%-80%, and 80%-100%). Size is estimated for each soil cover polygon and a threshold of 2 ha is applied to distinguish the different relative contribution of shading and evapotranspiration in *small* and *big* areas. The shape index of the polygon is also computed to account for the more intense effect of the surroundings, hence the lower cooling capacity, of areas between 2 and 10 ha when the shape index is higher than 6. These polygons are assimilated to polygons smaller than 2 ha.

Once the three properties are measured, a score can be assigned to each polygon based on the tables provided by Zardo et al. (2017) for three different climatic zones. Then, depending on the score, the polygon can be classified into one of the five classes of cooling capacity. For the present application, a slightly different version of the tables was used, with the range of scores scaled up to a maximum value of 172 and six final classes of cooling capacity, from A+ to E, as in Geneletti et al. (2016). Each class of cooling capacity can be associated to an expected temperature difference between the analysed area and another area in the same meteorological conditions with the lowest cooling capacity (i.e., sealed surface with no trees). Finally, the cooling effect produced on the surroundings can be mapped using different decay functions depending on the size of the polygon. The model assumes linear decay functions with a maximum distance of 100 m, 255 m, 1030 m, and 2000 m for polygons smaller than 2 ha, between 2 and 10 ha, between 10 and 50 ha, and bigger than 50 ha, respectively. Within these maximum distances, the same classes can be used to represent the cooling effect produced by the presence of green infrastructure components on the surrounding areas of the city. The direct cooling effect produced by tree shading can be accounted by computing a 5-meter buffer around canopies, which is assigned to the A class. Figure 4.3 summarizes through a flow chart the main steps of the model.

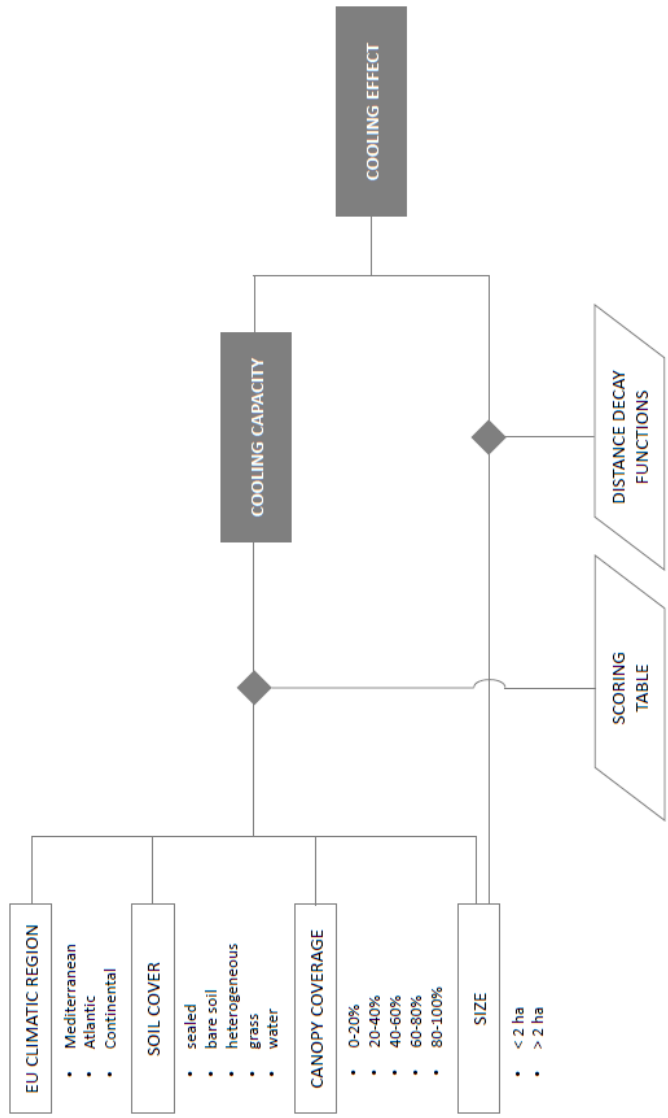

FIGURE 4.3: *Flow chart of the model for mapping and assessment of the cooling capacity and cooling effect of urban green infrastructure. For the scoring tables and the distance decay functions refer to* Geneletti et al. (2016) and Zardo et al. (2017).

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4.3.2 Mapping potential and opportunities for nature-based recreation: a local scale adaptation of the ESTIMAP recreation model

The mapping and assessment of the potential and opportunities for nature-based recreation in the city was carried out through the ESTIMAP recreation model. The model is part of a suite specifically developed by the Joint Research Centre of the European Commission to map a set of ES at the European scale. The key steps of the method are described in a report (Zulian et al., 2013), while results of the EU-wide application can be found in (Paracchini et al., 2014). Due to the relevance of the issue at the local scale, the model was later adopted in other contexts, mostly for city-wide assessments. Zulian et al. (2017) describe 7 local applications of the ESTIMAP recreation model carried out within the context of the OpenNESS H2020 project (http://www.openness-project.eu/, last accessed December 2017).

The ESTIMAP-recreation model is structured into three successive modules, each one producing an output that assesses nature-based recreation from a different perspective. The first module assesses the Recreation Potential (RP), i.e. the suitability of different areas to support nature-based recreational activities based on their intrinsic characteristics, independently from their actual or potential use. The RP is described by a raster map with relative values ranging from 0 (no recreation potential) to 1 (maximum recreation potential in the analysed area). The second module assesses the Recreation Opportunity Spectrum (ROS) by combining the RP with information about proximity, thus providing an assessment of the opportunities for recreational activities that are offered to the citizens. The ROS is described by a raster map classified into 9 categories resulting from the cross-tabulation of high/medium/low values of RP and high/medium/low values of proximity. The components that contribute to the values of RP and ROS are combined according to weights that must be assigned by the user. The third module assesses the demand by adding information about the spatial distribution of potential users. In the EU-wide application, the number of potential trips directed to each area was calculated based on the density of population within a defined maximum distance from areas with the highest values of ROS.

Within this general structure, adjustments can be done to adapt the model to local contexts. To this aim, Zulian et al. (2017) present a protocol based on the two successive steps of conceptual adaptation and structural adaptation. The step of conceptual adaptation requires framing the application of the model with respect to the specific policy question at hand, including issues related to the type of users and uses of the results, the scale of analysis, and the stakeholders that must be involved in the assessment. The step of structural adaptation refers to changes in the original model made to respond to the specific policy question detailed in the previous step. It requires: adapting the conceptual scheme in terms of number of components, combination of input data, scoring system, and weighting parameters; identifying and retrieving locally-relevant data, including the elicitation of weights from experts or stakeholders; and running the model and sharing results to get feedbacks, possibly feeding a further refinement of the conceptual scheme.

For the described application, the components of the different modules were adjusted to reflect the specificities of the context of Trento. Most of all, the adaptation was aimed at reflecting the local conditions in terms of different types of recreational activities and related natural settings, at providing practical information about what types of planning or management interventions are more needed, and at easing the weighting phase by maximizing the similarity of the elements gathered within the same component. Accordingly, in the final scheme, the RP module includes three components, namely natural features, urban green infrastructure, and land use, thus distinguishing urban and peri-urban green areas from prevalently natural and semi-natural areas outside the city. Moreover, 'proximity' is here defined as the availability of facilities and infrastructures that allow accessing and using green areas for nature-based recreational activities. Therefore, the ROS module includes, beyond RP, two distinct components for access-related and use-related facilities. Finally, population distribution is used to quantify the actual beneficiaries, based on the number of citizens living within a defined distance from areas with different ROS values. Figure 4.4 presents a flow chart of the ESTIMAP-recreation model adjusted for the application to Trento, while the data included in each component and the specific procedure adopted for assigning weights are described in the following Section 4.3.4.

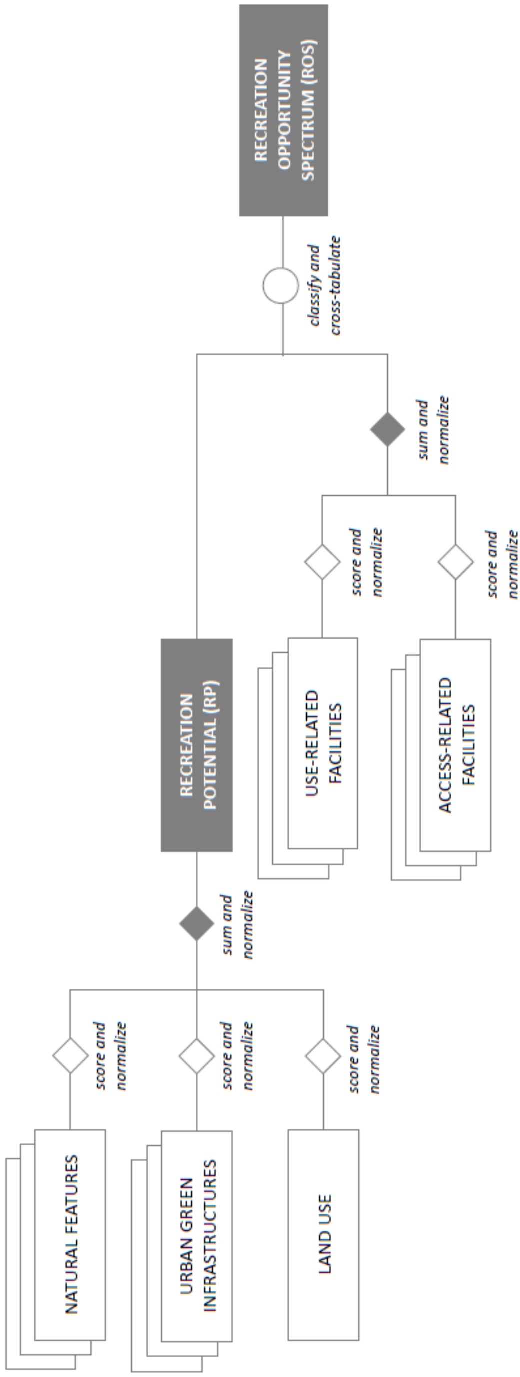


FIGURE 4.4*: Flow chart of the ESTIMAP-recreation model adjusted for the application to Trento. Modified after* Zulian et al. (2013).

4.3.3 Integrating ecosystem service assessments to evaluate planning scenarios

The two methods were applied to assess the benefits produced by the redevelopment of brownfields through greening interventions. The transformations of different brownfields have been considered as alternative planning scenarios, hence analysed independently. The result of the transformation was assumed to be, for each brownfield, a new urban park, intensely planted and open to public use. The assessment was based on a comparison of the 13 scenarios with the current condition (baseline). Similar indicators, based on the number of people affected by the transformation, were used to assess the two ES. For both ES, vulnerable people, defined as citizens' groups with a higher-than-average need for that specific ecosystem service, were identified and quantified as a sub-group of the total beneficiaries.

To assess the improvement in micro-climate regulation, new urban parks were modelled as areas covered by grass, with 80% to 100% canopy coverage (i.e., cooling capacity class equal to A+ for areas bigger than 2 ha and cooling capacity class equal to A for areas smaller than 2 ha). The map of the cooling effect was computed within the area of influence of each brownfield and compared with the baseline condition. Then, a change map was calculated by subtracting the two maps. The final indicator was defined as the number of affected residents weighted by the intensity of change (number of classes) and calculated through an overlay between the change map and population data. Young children (< 5 years old) and the elderly (> 65 year old) were selected as vulnerable groups, based on their higher sensitivity to heat stress (Basu and Samet, 2002; Kabisch et al., 2017; Kenny et al., 2010).

To assess the enhanced opportunities for nature-based recreation, brownfields were alternatively assigned to the land use class 'green urban areas', considering the same presence of infrastructures and facilities inside the new park as in other parks of the city with comparable dimension. People living within 300m from the new parks were considered as beneficiaries of the transformation (Kabisch et al., 2016; Stessens et al., 2017). Children and teenagers (< 20 years old) and the elderly (> 65 year old) were identified as vulnerable groups, based on the higher demand for close-to-home recreation and relaxation areas (Kabisch and Haase, 2014). Furthermore, thosee beneficiaries already served by high-level opportunities for nature-based recreation in the baseline scenario (i.e., living within 300 m from areas classified in the highest class of ROS), were identified and counted separately.

A multi-criteria analysis was used to combine the results of ES assessments. The two ES and the different categories of beneficiaries are used as criteria and sub-criteria for the analysis. Three illustrative combinations of weights are considered, corresponding to three different policy perspectives and related objectives, as detailed in Table 4.1. Values for each criterion and sub-criterion are normalized according to the maximum, and a 'weighted summation' approach is used to calculate the overall score of each alternative.

Spatial data were analysed and elaborated using the GIS software Q-GIS 2.18.9 (QGIS Development Team, 2017) and Grass GIS 7.2.1 (GRASS Development Team, 2017), while the multi-criteria analysis was conducted using the free version of the software Definite (SPINlab Vrije Universiteit Amsterdam, 2016).

TABLE 4.1: *The three illustrative perspectives and respective combinations of weights considered in the multi-criteria analysis to prioritize brownfield redevelopment scenarios.*

	perspective 1 "balanced"	**perspective 2** "cool air for the elderly"	**perspective 3** "every child needs a park"
Cooling	0.50	0.80	0.20
non-vulnerable	0.20	0.14	0.20
< 5 years old	0.40	0.29	0.40
> 65 years old	0.40	0.57	0.40
Recreation	0.50	0.20	0.80
non-vulnerable	0.20	0.20	0.14
served	-	-	0.20
not served	-	-	0.80
< 20 years old	0.40	0.40	0.57
served	-	-	0.20
not served	-	-	0.80
> 65 years old	0.40	0.40	0.29
served	-	-	0.20
not served	-	-	0.80

4.3.4 Materials and data

A land use land cover map released in 2017 as an updated baseline information for the new plan was provided by the municipality of Trento. The map, in vector format, is the result of the classification of high-resolution (10 cm) aerial photographs combined with other data (e.g., cadastral map), and follows the common classification of the Corine project. Other data were retrieved from the municipal database for the management of public green areas and trees, which provided detailed information about their location and typology (including data about species, age, and dimension), and about the presence of facilities in urban parks (e.g., benches, fountains, playgrounds, etc.). The municipality also provided the number of residents for each census tract broken down into 5-year age groups (last update: 31st December 2014), which were considered as homogeneously distributed on the surface covered by the footprint of residential building*.

* The land use land cover map can be downloaded at the following link http://www.comune.trento.it/Aree-tematiche/Cartografia/Download/Carta-uso-del-suolo-Open-Data2 while the municipal databse is accessible from the webGIS of the municipality http://webapps.comune.trento.it/mapaccel/?project=generale&view=verde (last accessed: February 2018).

To prepare the input data for the cooling assessment model, first the land use land cover map was completed and detailed, for specific areas, with information available from other sources (e.g., municipal database of public green areas, map of community gardens). Then, each land use land cover class was assigned to one of the five soil cover classes identified by the model, as shown in Table 4.2. Canopy coverage was mapped by combining the land use land cover map with the provincial and municipal maps of forested areas and the municipal database of public green areas and trees.

Input data in the different components of the ESTIMAP-recreation model were retrieved from multiple sources, including both institutional databases and Open Street Map (Open Street Map Contributors, 2017) (Table 4.3)**. The scores were elicited from a pool of experts selected with the collaboration of a municipal officer through an on-line questionnaire. The questionnaire was sent to 19 experts who had previously agreed to collaborate to the project, and 17 valid answers were collected within the deadline (December 2017). Respondents include personnel of different provincial (3) and municipal (7) departments with an interest in recreational areas and activities, including green space management, environment, planning, common goods, social services, sport, protected areas, and landscape; local practitioners (1); and academics from the University of Trento (3) and

** Data from the provincial plan and other data collected by the province of Trento, including those related to the General Plan of Public Water Uses (PGUAP) can be downloaded at the following link http://www.territorio.provincia.tn.it/portal/server.pt/community/sgc_-_geocatalogo/862/sgc_-_geocatalogo/32157 (last accessed: February 2018). Hiking trails are mapped by the Trentino Alpine Association (SAT) and can be accessed form the public webGIS https://trentino.webmapp.it/#/?map=11/46.0612/11.1357 (last accessed: February 2018).

other research centres (3) working on topics related to ES, urban green infrastructures, and urban planning. The experts were asked to assign to each element a score from 0 to 5, corresponding to its relevance in supporting or promoting nature-based recreational activities (direct assessment). The scores were then averaged, excluding the highest and the lowest score, and converted to a 0-to-1 scale. The final scores used to run the model are listed in Table 4.2 and Table 4.3.

TABLE 4.2: *Land use land cover classes of the municipal map, respective class of soil cover assigned for the cooling assessment, and score resulting from the questionnaire about recreation. Codes follow the Corine Land Cover classification (italic indicates partial correspondence with the official CLC classes). * Green urban areas and water courses are included in other components of the model; hence they are not assigned a score in the land use component. ** A score of 0.7 was assigned to community gardens.*

code	description	soil cover class	score
1.1.1	Mixed-use urban centre, continuous high-density urban fabric	sealed	0.6
1.1.2	Discontinuous urban fabric	sealed	0.6
1.1.2	Disc. low-density or sparse urban fabric	heterogeneous	0.7
1.2.1	Industrial units	sealed	0.2
1.2.1	Commercial units	sealed	0.4
1.2.1	Large areas for public and private services	sealed	0.4
1.2.1	Areas for technological systems and plants	sealed	0.2
1.2.2	Rail network and associated land	sealed	0.1
1.2.2	Road network and associated land	sealed	0.2
1.2.2	Parking areas	sealed	0.3
1.2.4	Airports	sealed	0.3
1.3.1	Mineral extraction sites	bare soil	0.3
1.3.2	Dump sites	sealed	0.1
1.3.3	Construction sites and other artificial areas	bare soil	0.1
1.4.1	Green urban areas	grass	*
1.4.2	Sport and leisure facilities	sealed	0.9
1.4.2	Sport and leisure facilities -ski areas	grass	0.9
2.1	Arable land	heterogeneous	0.4
2.2.1	Vineyards	grass	0.5
2.2.2	Fruit trees and berry plantations	grass	0.5
2.3.1	Pastures	grass	0.8
2.4.2	Complex cultivation patterns	heterogeneous	0.6**
3.1.3	Mixed forest	heterogeneous	0.9
3.2.1	Natural grasslands	grass	0.7
3.2.1	Other grasslands	grass	0.7
3.3.2	Bare rock	sealed	0.7
4.1.2	Peatbogs	grass	0.6
5.1.1	Water courses	water	*
5.1.2	Water bodies	water	0.9

TABLE 4.3: *Input data of the ESTIMAP-recreation model and respective scores resulting from the questionnaire.*

	source	spatial entity	score
Natural features			
local reserve	provincial database	point	0.8
Natura 2000 sites	provincial database	polygon	0.8
monumental tree	municipal database + OSM	point	0.7
mountain pass or saddle	OSM	point	0.7
mountain peak	OSM	point	0.8
karstic area	provincial plan	point	0.5
canyon	provincial plan	point	0.8
site of geomorphological interest / rock	provincial plan + OSM	point	0.7
paleontological site / cave	provincial plan	point	0.7
site of stratigraphic interest	provincial plan	point	0.6
spring	OSM	point	0.5
valuable landscapes	provincial plan	point	0.8
viewpoint	OSM	point	0.9
river areas with landscape value	PGUAP	polygon	0.8
river or water course - primary	land use map	polygon	0.8
river or water course - secondary	land use map	polygon	0.7
Urban parks			
> 2 ha	municipal database	polygon	1
> 0.5 ha	municipal database	polygon	0.9
< 0.5 ha	municipal database	polygon	0.8
historical garden	municipal database	polygon	0.7
Access-related facilities			
parking area	OSM	point	0.7
bus stop	municipal database	point	0.8
cycle path – local	municipal database	line	0.9
provincial road	municipal database	line	0.7
local road	municipal database	line	0.8
forest track	provincial database	line	0.6
Use-related facilities in non-urban context			
alpine hut	OSM	point	0.9
rock climbing route	OSM	point	0.8
picnic area	OSM	point	0.7
cycle path – long distance	provincial database	line	0.9
forest track	provincial database	line	0.7
hiking trail	SAT database	line	0.9
MTB track	OSM	line	0.8
Use-related facilities in urban parks			
playground	municipal database	point	0.9
sport field	municipal database	point	0.7
dog area	municipal database	point	0.7
benches and tables / picnic area	municipal database	point	0.7
water feature / fountain	municipal database	point	0.7

4.4 Results

4.4.1 Cooling capacity and cooling effect of urban green infrastructure in Trento

The left side of Figure 4.5 shows the map of the valley floor, the most urbanized part of Trento, classified according to the cooling capacity of the different areas. The highest class of cooling capacity mostly corresponds to woodlands and open spaces at the border of the urban settlement. Within the city, forest patches left behind by urban expansion, the main water courses, and the largest urban parks are characterized by the highest cooling capacity, corresponding to the highest expected temperature difference during hot days compared to the surroundings. Yellow and orange areas in the map, which prevail in the historical centre but also in the mixed-use expansion to the North, are the worst performing in terms of cooling capacity. Considering the cooling effect (Figure 4.5 – centre), the major part of the city benefits from the presence of the surrounding wooded hills and of the Adige river and its tributaries. The most disadvantaged areas are in the densest neighbourhoods close to the city centre, and within the northern suburb, where the mix of residential and industrial areas scarcely equipped with green infrastructure, as well as the presence of major transportation infrastructures, have a negative impact on the cooling performance. Quite interestingly, most of the brownfields are strategically located close to the centre of the settlement, in areas characterized by low availability of green infrastructure and scarcely benefitting from the cooling effect of the natural and semi-natural areas surrounding the city.

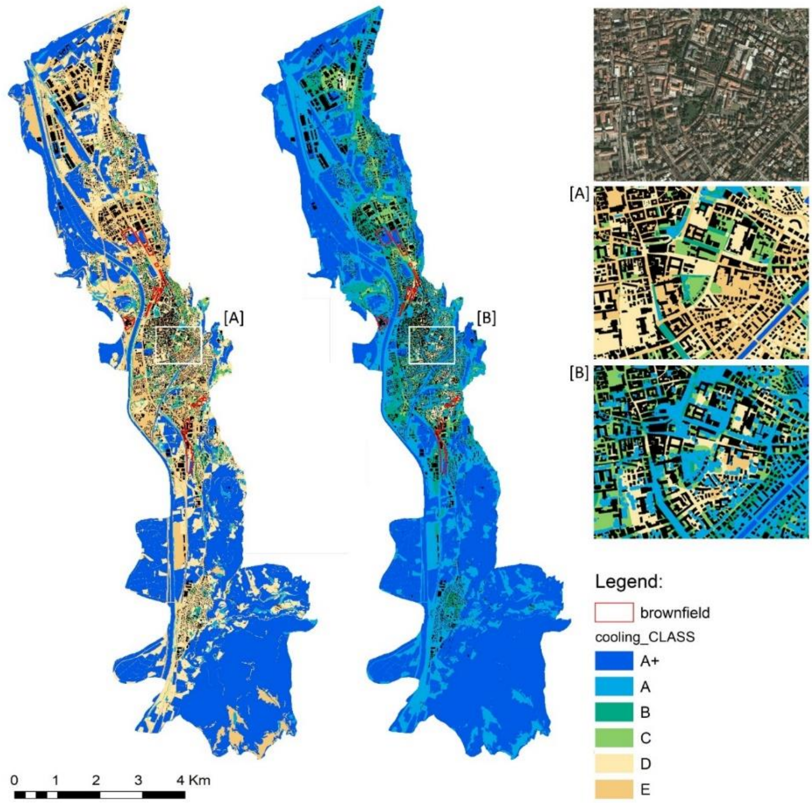


FIGURE 4.5: *Maps of the cooling capacity (left and [A] zoom) and of the cooling effect (center and [B] zoom) of urban green infrastructure in the most urbanized part of the city of Trento.*

4.4.2 Potential and opportunities for nature-based recreation in Trento

Figure 4.6 shows the map of the Recreation Potential (RP), expressed as a normalized value ranging from 0 (no RP) to 1 (maximum RP). A large part of the map is covered by values around 0.5. This is a consequence of the high scores received by most of the elements both in the ‘natural features’ and in the ‘urban green infrastructure’ components, probably due to the wide range of nature-based recreational activities carried out by the population of Trento. Since the scores of the three components of the model are summed up to obtain the total value of RP, the areas that reach the highest value are urban parks with relevant natural features, as in the case of two among the largest urban parks of the city that partially overlap with natural protected areas. Overall, excluding urban parks, the valley floor presents low values of RP. The river itself does not emerge, since its dimension and speed make it unsuitable to support water-based recreational activities. However, its banks receive a high score, particularly those recognized for their landscape value. The surrounding hills and mountains have an overall higher recreation potential compared to the urban area. This is mostly due to the presence of forests and of various attractive natural features that promote nature-based recreational activities.

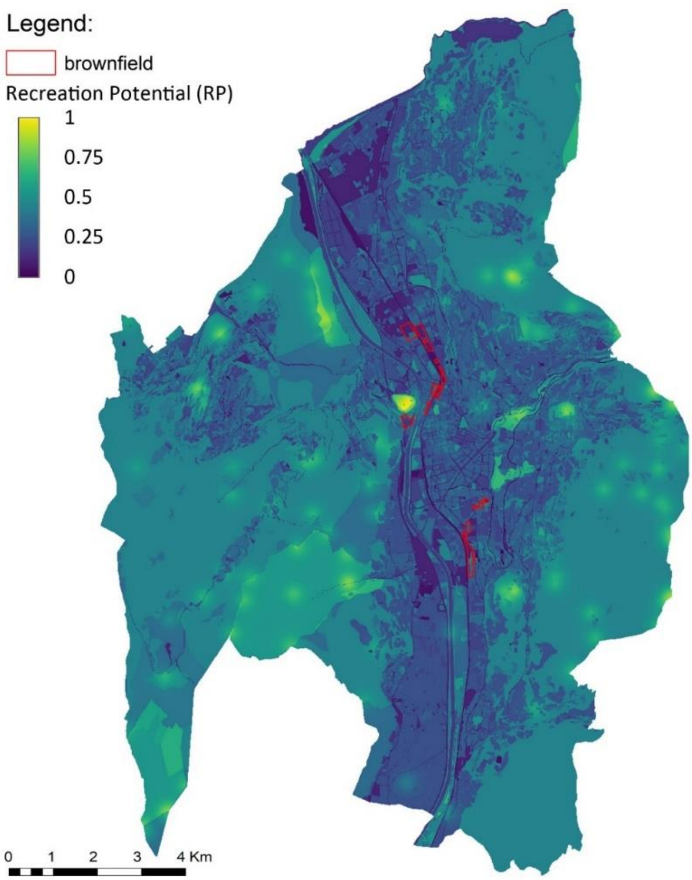


FIGURE 4.6: *Map of the Recreation Potential (RP) in Trento calculated through the ESTIMAP-recreation model.*

Figure 4.7 shows the map of the Recreation Opportunity Spectrum, which combines values of RP and proximity, here defined as the availability of facilities and infrastructures for accessing and using the areas. The main urban settlement in the valley floor, though mostly characterized by a low RP, presents the highest concentration of infrastructures and facilities. All urban parks are in the best classes of ROS. The same can be said for the river banks, which host one of the most important touristic cycle paths in Italy, used by Trento citizens for running, cycling, and skating. Accounting for the availability of infrastructures and facilities helps to discriminate the different opportunities offered by extra-urban areas, particularly forests. Areas with high proximity due to the presence of forest tracks, hiking trails, and facilities dedicated to specific activities such as climbing routes and MTB trails, emerge, especially close to the settlements. A quite different performance in terms of recreation opportunities characterizes the two sides of the valley. The East side is characterized, on average, by a higher proximity compared to the West side, where the settlements are sparser and the connections with the valley floor are more difficult. Considering the brownfields and their surroundings, all of them are in areas with high proximity. Some are close to existing urban parks, while other, e.g. the three in the northern part of the city, are far from any area with high RP, hence represent occasions for enhancing the condition of people that currently have no or very few close-to-home opportunities for nature-based recreation.

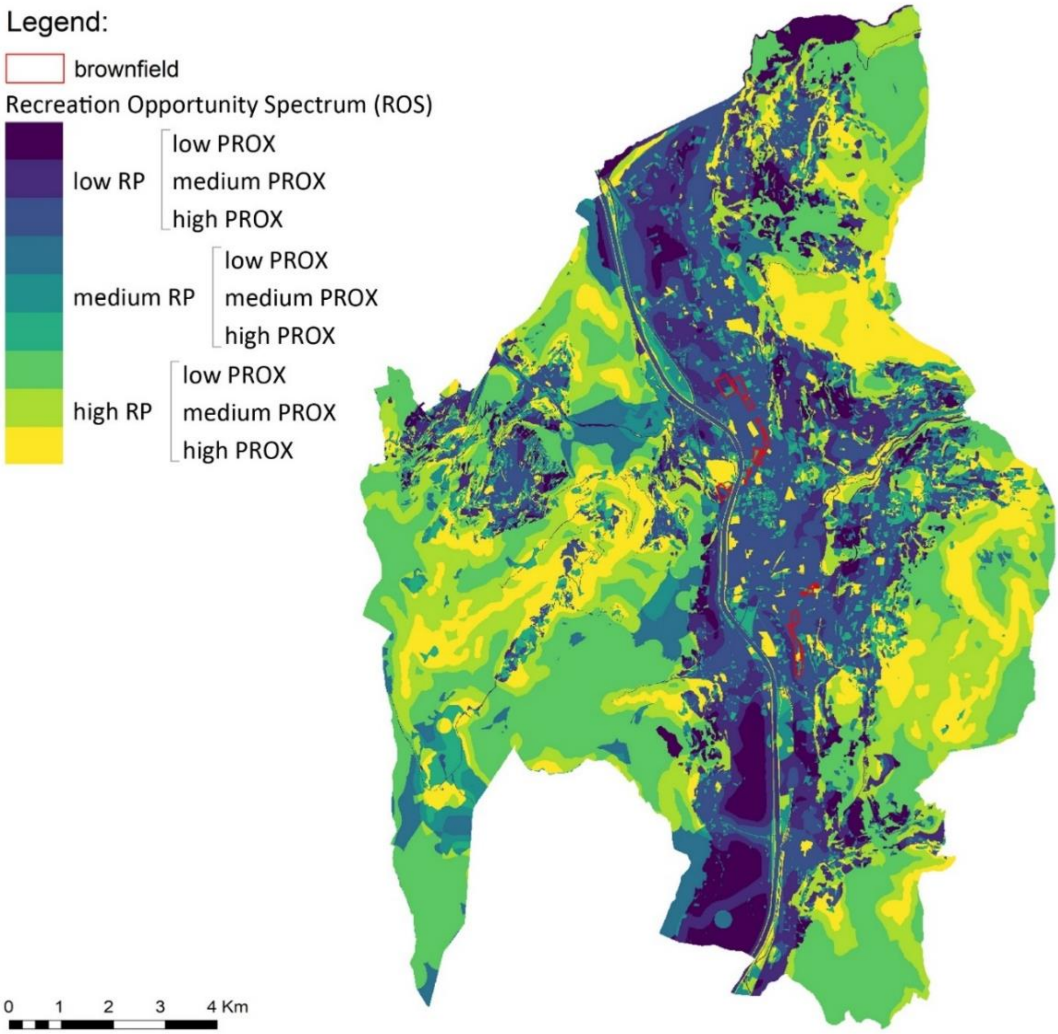


FIGURE 4.7*: Map of the Recreation Opportunity Spectrum (ROS) in Trento calculated through the ESTIMAP-recreation model.*

4.4.3 Assessment of brownfield redevelopment scenarios

An illustrative example of the maps resulting from the analysis of redevelopment scenarios is shown in Figure 4.8. The conversion of brownfield n.11 to a new intensely-planted urban park produces benefits in terms of both improved cooling effect on the surroundings and enhanced opportunities for close-to-home nature-based recreation. In this specific case, as it emerges from a comparison between the maps, the cooling effect shows a significant improvement, which positively affects many of the surrounding residents. In the present condition, most of the surrounding residents gain very little or no thermal benefit at all from the presence of green infrastructure, limited almost exclusively to single shading trees. In the redevelopment scenario, the improvement is noticeable in the major part of the area, especially in the neighbourhood to the North where some portions move from class E to class A of cooling effect. In terms of recreation opportunities, the new urban park would fall into the best class of ROS, with high Recreation Potential and high proximity. The map highlights also the possibility of connecting the new park to an adjacent open-air soccer field, already classified in the best class of ROS. Despite being already served by other parks and green areas close by, all the households included in the map would benefit from an additional space for recreation within walking distance from their location.

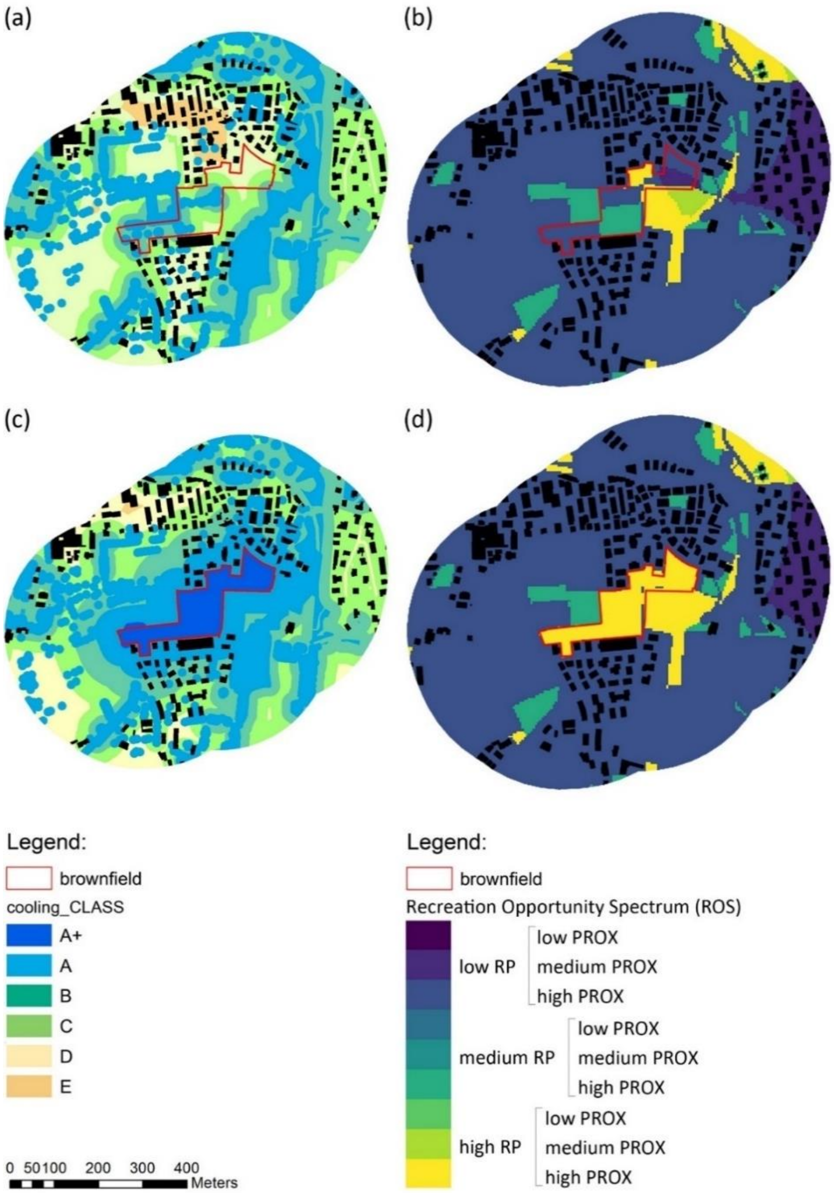


FIGURE 4.8*: Maps of the cooling effect and of the Recreation Opportunity Spectrum in the baseline ((a) and (b)) and in the redevelopment scenario ((c) and (d)) for brownfield n.11. The area shown in maps (a) and (c) considers the maximum distance potentially reached by the cooling effect generated by the brownfield. The area shown in maps (b) and (d) considers a maximum distance of 300 m from the brownfield and is used to identify potential beneficiaries of enhanced close-to-home opportunities for nature-based recreation.*

A comparison of the performance of the different redevelopment scenarios is made in Figure 4.9, where the brownfields are compared according to the number of potential beneficiaries produced by the transformation. Considering cooling, brownfield n.11 is by far the best performing, being a potentially large green area inside a heavily built-up and densely populated part of the city. The performance in terms of recreation is more balanced, with brownfields n.07 and n.08 producing the highest number of beneficiaries. However, only brownfields n.01, n.02, and n.03, if converted to new urban parks, would serve people that, at present, have no access to close-to-home nature-based recreational opportunities. It should also be noted that the ratio between total beneficiaries and specific vulnerable groups is not the same across scenarios. For example, the share of children and young people is higher in brownfield n.01 and n.02 compared to the others, while the share of people aged more than 65 is the highest in brownfield n.11. A final remark concerns an overall comparison between the number of beneficiaries of the two ES considered. Apart from the redevelopment of brownfield n.11, which is clearly an outlier, the enhancement of the cooling effect is likely to affect very few people, in the range of some hundreds for most scenarios. On the other hand, the number of people that would benefit from increased close-to-home opportunities for nature-based recreation are much more: around one thousand even in the worst-performing scenario (redevelopment of brownfield n.10).

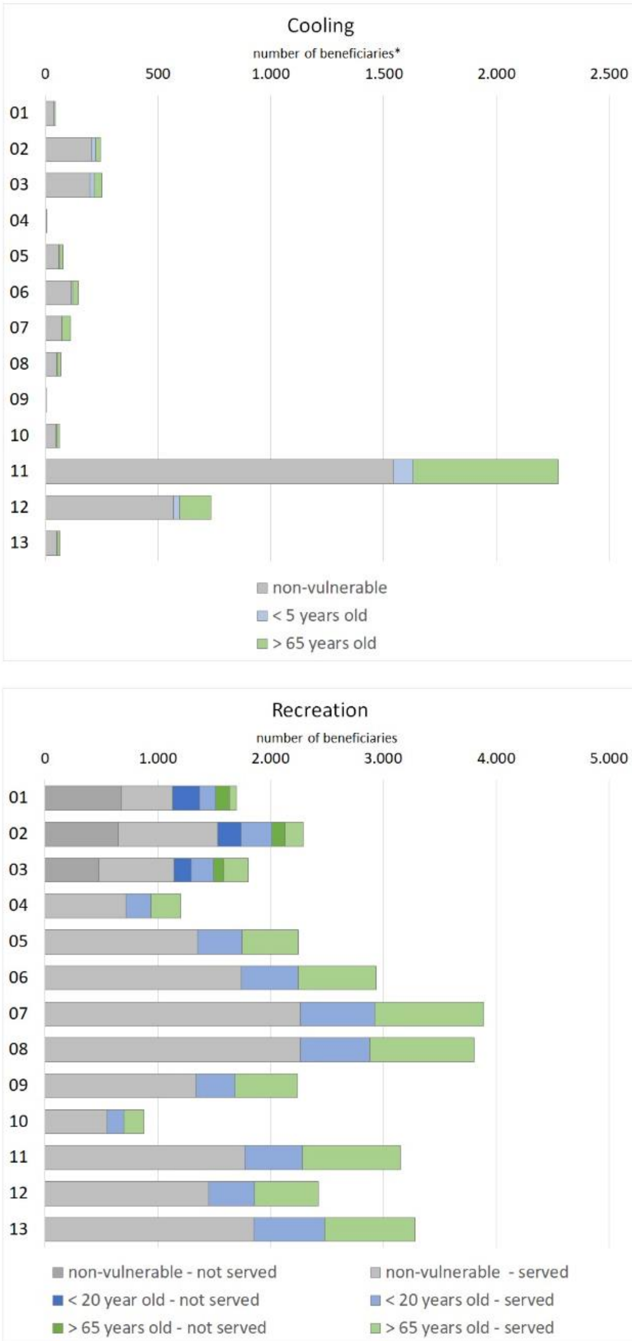


FIGURE 4.9: *Expected benefits produced by the different scenarios in terms of enhanced cooling effect by urban green infrastructure and improved opportunities for nature-based recreation: number of beneficiaries broken down into different vulnerability classes.*

The information about the number of residents benefitting from the increased provision of the two ES in the different scenarios were combined through a multi-criteria analysis. Three perspectives and related combinations of weights were simulated, producing the results presented in Figure 4.10. Assuming a ‘balanced’ perspective, with the same weight assigned to cooling and recreation and a double weight assigned to vulnerable compared to non-vulnerable groups, brownfield n.11 ranks first. The second perspective, which aims to improve the cooling effect in areas with a high share of old population, leads to the same first-ranking scenario. Although the other positions change between the two perspectives, all scenarios gain a very low score compared to brownfield n.11. The third perspective focuses on providing opportunities for recreation to people, especially children and teenagers, who are not served in the present condition. In this case, the final ranking changes significantly, and the first positions are occupied by the three brownfields (n.01, n.02, and n.03) located in the northern part of the city. In such neighbourhoods, the population is comparatively younger and the opportunities for recreation are scarcer. Overall, the three illustrative perspectives show how priorities change based on the relative importance attributed to the different ES and to the respective categories of beneficiaries (Figure 4.11). Moreover, a sensitivity analysis can be conducted on the assigned weights, to assess the stability of the ranking and the robustness of the results (Figure 4.12). A low sensitivity, as the one resulting in perspective 1 and 2 with respect to the weight of the ‘cooling’ criterion, ensures that the first-ranking alternative maintains its position even for large fluctuations of the weight. Figure 4.11 shows, on a map, the final ranking of the alternatives resulting from the different combinations of criteria and weights assumed in the three perspectives.

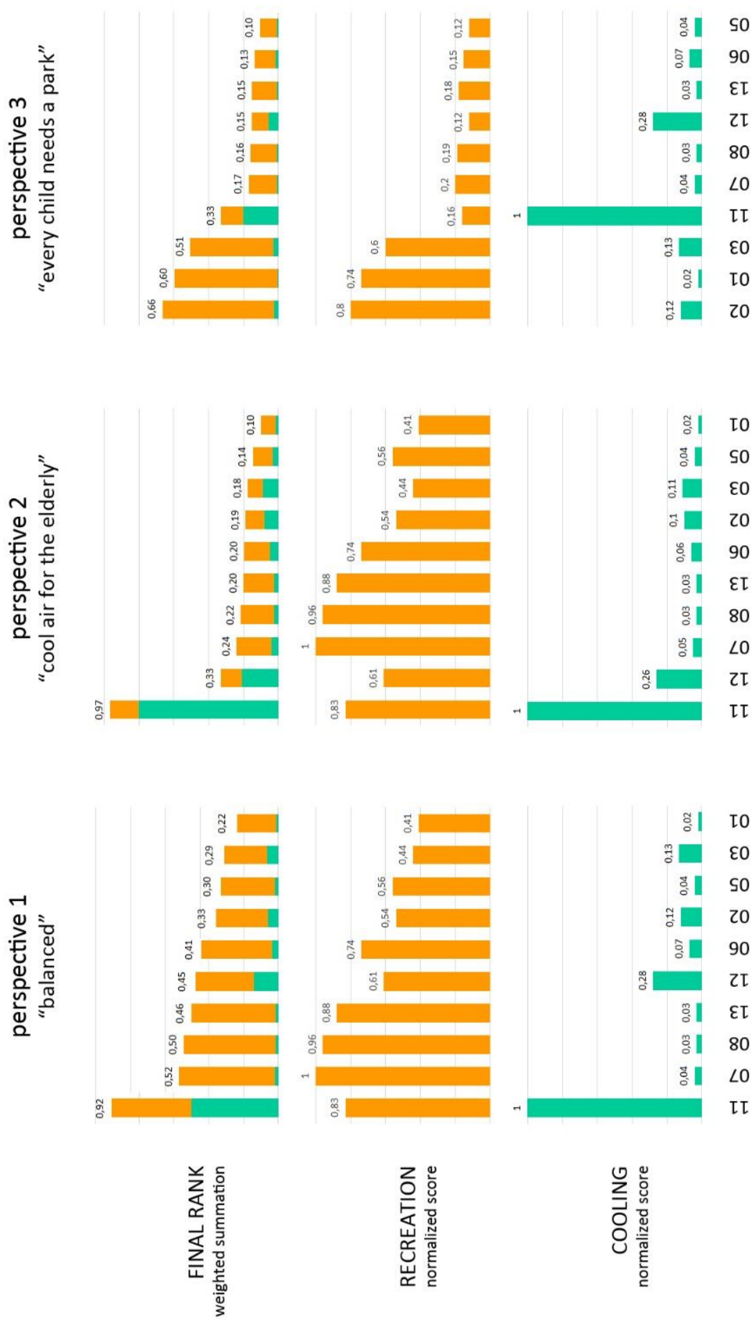


FIGURE 4.10*: Final rankings of the brownfield redevelopment scenarios according to three perspectives considered in the multi-criteria analysis. The weights assigned to the different ES and the different categories of beneficiaries are reported in* **TABLE 4.1***. Brownfields n.04, n.09, and n.10 are excluded from the multi-criteria analysis.*

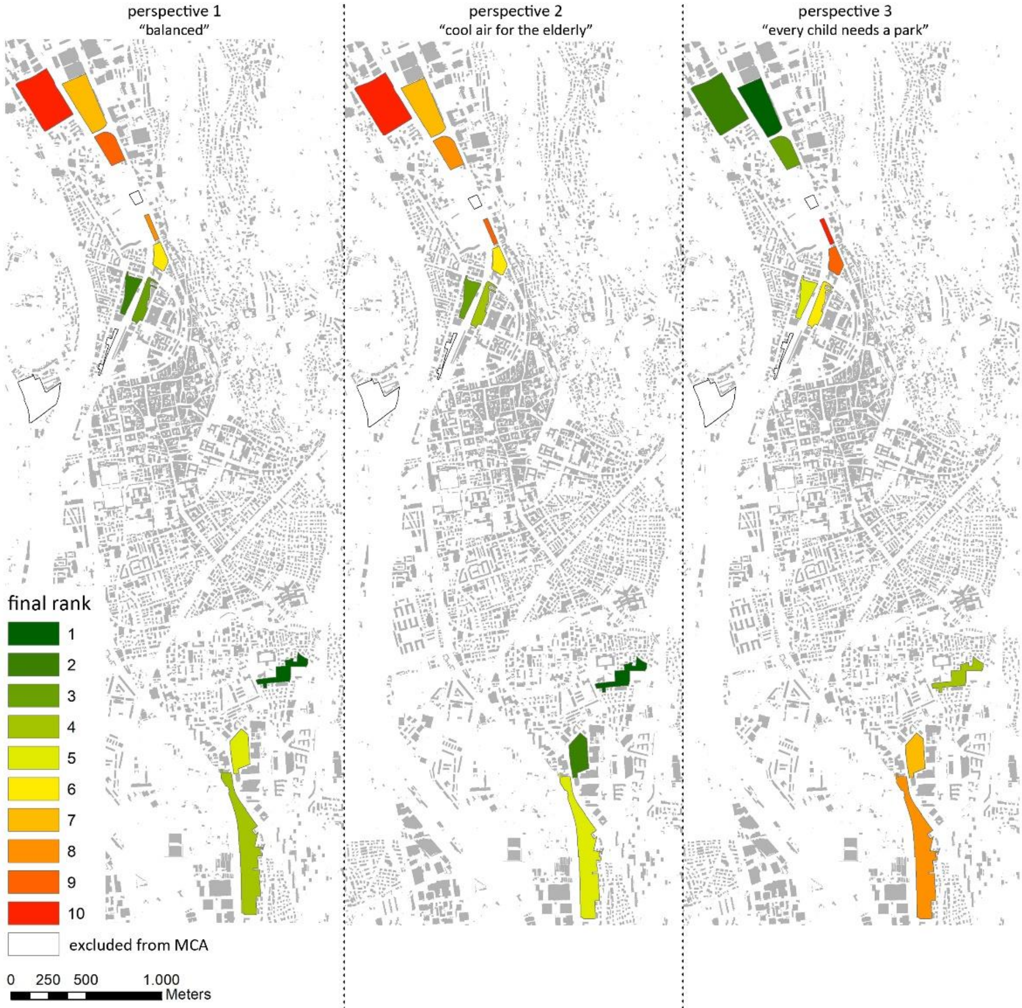


FIGURE 4.11: *Map of the priority level of brownfield redevelopment scenarios according to three perspectives considered in the multi-criteria analysis.*

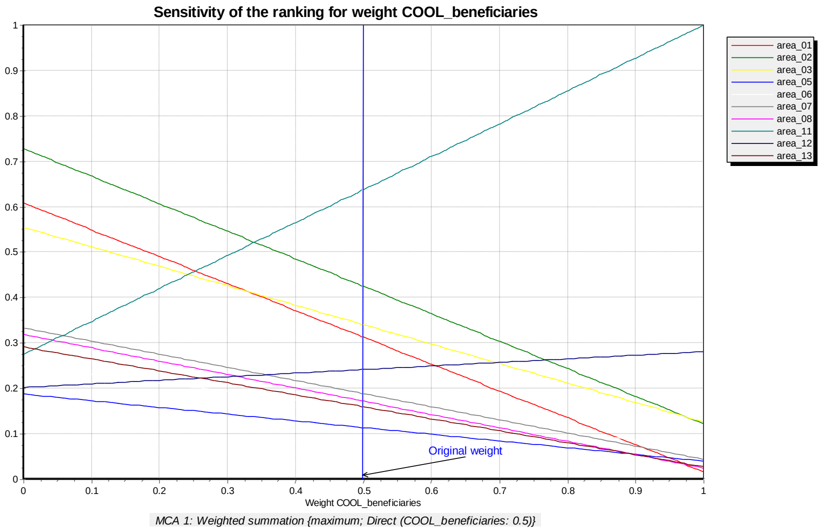


FIGURE 4.12: *Example of sensitivity analysis for the weight assigned to the 'cooling' criterion in perspective 3 'Every child needs a park'. The graph shows which weight should be assigned to the criterion to produce a change in the first-ranking alternative.*

4.5 Discussion

This chapter investigated one of the possile tasks that ES mapping and assessment can perform to support urban planning, i.e. the assessment of alternative planning scenarios (Barton et al., 2017). considered planning scenarios formulated as alternative sites where existing brownfields could be converted to new public green areas. The analysis considered different brownfields in the city of Trento that could be converted to new urban parks and assessed the expected effects of the transformations in terms of ES benefits. The presence of brownfields and abandoned areas is a key issue for today's cities, with strong economic and social implications (Nassauer and Raskin, 2014), hence their regeneration is promoted among the strategies for sustainable urban development (European Commission, 2016a). Recent studies have analysed how, depending on their actual conditions, brownfields are or may be turned - through interventions that range from simply changing the management of the areas to demolishing, de-paving, and regreening - into sources of ES for the urban population (Beames et al., 2018; Collier, 2014; Geneletti et al., 2016; Mathey et al., 2015;

McPhearson et al., 2013). The study focused specifically on the expected benefits of the interventions in terms of improved cooling effect by vegetation during the hot season and enhanced opportunities for nature-based recreation, thus addressing two among the most critical issues for citizens' wellbeing in Trento. Comparing the performance of the different scenarios allowed targeting planning actions toward the most desirable one, given a set of criteria and respective weights to consider in the assessment.

In the analysis, three alternative perspectives with related combinations of criteria and weights simulated three different decision-makers' orientations, corresponding to different planning objectives. In the case of perspective 1, a 'balanced' weighting was performed by assigning the same weight to the two ES. In the case of perspective 2 and 3, one ES received a weight significantly higher than the other, and specific vulnerable groups were identified as the main targets of policy interventions. The results clearly show how priorities change with changing policy goals (Grêt-Regamey et al., 2013; Kremer and Hamstead, 2016; Sanon et al., 2012). Put in relation to the results of Chapter 2, this finding highlights the need for a strategic approach to ES in planning and for the inclusion of explicit ES-related objectives. Simply providing ES knowledge as part of the information base of urban plans is not enough to guarantee that it is (usable and) used to guide decisions (Saarikoski et al., 2017). On the contrary, formulating objectives and targets for ES provision helps to identify the values against which the effectiveness of planning actions should be measured, hence also to clarify the possible role(s) of ES knowledge within the process.

Previous applications of multi-criteria analysis to the assessment of urban ES have mostly focused on trade-offs among different ES and how they can be minimized in the context of planning interventions (Grêt-Regamey et al., 2013; Sanon et al., 2012). Here, the study considered a case in which all scenarios are expected to improve the current condition and to generate benefits that decision-makers aim to maximize. This situation is not an unusual one in the context of ecosystem-based actions and nature-based solutions, often characterized by synergies rather than trade-offs among ES, and related multiple benefits for nature, society, and the economy (Albert et al., 2017; Demuzere et al., 2014; Geneletti and Zardo, 2016; Raymond et al., 2017). In the analysed case, potential trade-offs may be related to competing uses of the existing brownfields (Kain et al., 2016), and to other non-ES criteria, for example the cost of intervention (Koschke et al., 2012). Within this context, multi-criteria analysis provides a platform for integrating different information about multiple costs and benefits of planning scenarios (Saarikoski et al., 2016), and for balancing conservation and enhancement of green infrastructure with other objectives (Adem Esmail and Geneletti, 2018).

In the described application, multi-criteria analysis is used to combine results about two ES belonging to different categories: one regulating ES and one cultural ES. While most urban ES studies have focused on a single ES (Haase et al., 2014b), integrating multiple values and related indicators, especially across different ES categories, is still a challenge (Jacobs et al., 2016). Also in this respect, multi-criteria analysis appears as a useful tool to combine multiple value dimensions (Adem Esmail and Geneletti, 2018; Saarikoski et al., 2016). However, perhaps even more important in the context of decision-making is that indicators are meaningful and informative for who is responsible for the decision. From this perspective, the focus of ES assessment methods and practices on biophysical aspects is a limit to their relevance (Bagstad et al., 2014; Olander et al., 2018), especially in decision-making contexts where social and economic objectives prevail over ecological concerns. On the contrary, indicators based on beneficiaries, explicitly linking ES provision with changes in human wellbeing, are a promising way to integrate ES knowledge in decision-making processes (Geneletti et al., 2016; Olander et al., 2018) and to communicate ecological knowledge to planners and politicians primarily interested in enhancing citizens' wellbeing and quality of life (Schleyer et al., 2015).

Part of the challenge of integrating different ES assessments lays in finding common indicators to express benefits and associated values across the whole range of ES. So far, this has mostly been done through monetary units, whose popularity is probably also linked to this capability. However, several authors have already highlighted limitations and potential drawbacks of monetary valuation of ES (Saarikoski et al., 2016), to the point that the need for economic valuation in ES assessments has been defined as a 'misconception' and "an unnecessary barrier for both the science development side […] and the practitioner side" (Ruckelshaus et al., 2015). In the described application, different ES have been assessed through the same units of measurement, not based on monetary values, but based on the number of beneficiaries produced under the different planning scenarios. The results confirm the potential of 'benefit relevant indicators' (Olander et al., 2018), like those listed in the previous Chapter 3 (Table 3.7), to provide a common ground to assess multiple ES in a way that is relevant for decision-making (Olander et al., 2017).

Beneficiary-based or 'benefit relevant indicators' refer to the stage of the ES Cascade that describes how ES 'appropriation' (Spangenberg et al., 2014) generates benefits, i.e. contributes to specific aspects of wellbeing (Haines-Young and Potschin, 2010). Such indicators are not necessarily the result of socio-cultural methods aimed at eliciting preferences and values from stakeholders (Harrison et al., 2017), which may be difficult to integrate in planning processes. As shown for the case of cooling, simple beneficiary-based indicators can be obtained through the combination of biophysical modelling with information commonly available to planners, such as the distribution and level of demand of the actual and potential beneficiaries. What is needed, though still challenging, is to follow the whole 'production chain' of ES, from urban ecological structures and functions to ES benefits (Luederitz et al., 2015; Olander et al., 2018), which requires synthesizing multiple inputs into a true trans-disciplinary assessment (Jacobs et al., 2016; Potschin-Young et al., 2017).

The two methods adopted in the case study are specifically aimed at assessing urban ES for decision support (Zardo et al., 2017; Zulian et al., 2017). Accordingly, they work at the city scale and have the necessary resolution to capture the heterogeneity and fragmentation of urban green infrastructure (Gómez-Baggethun and Barton, 2013), and the limited dimension of the resulting service benefitting areas (see Chapter 3). However, not all ES assessment methods suitable for city-wide applications can be successfully adopted to compare planning scenarios. Assessing and comparing urban planning scenarios requires methods responsive to small changes in land uses (Kain et al., 2016) and able to measure variations in ES due to changes in management that may not be reflected by land use changes. The ESTIMAP-recreation model adjusted for the described application, with a component specifically devoted to assessing the presence of infrastructures and facilities, is a good example of how management interventions that affect ES provision can be taken into account even when land uses do not change.

However, both the methods and their application are characterized by some limitations that must be acknowledged. Due to the classification of soil cover and canopy coverage on which it is based, the model for assessing the cooling capacity and cooling effect of urban green infrastructure is sensitive to classification errors, and the different resolutions of input data may have produced inaccurate results particularly in private areas where detailed data were not available. The application of the ESTIMAP-recreation model was partly driven by the availability of spatially-explicit data, especially for what regards the 'natural features' and the 'use-related infrastructures and facilities' components. Data from Open Street Map (Open Street Map Contributors, 2017) allowed overcoming the lack of information in the municipal databases, but poses issues of completeness and reliability. Furthermore, the involvement of experts from different departments and sectors does not guarantee that citizens' needs and preferences are reflected in the assessment. A final limitation regards the use of population data to identify ES beneficiaries. Surrounding residents may represent only a part of the users of an area, and methods that take into account the real distribution of people (including non-residents and commuters) across the city and its variations during the day would represent a significant advancement in the quantification of ES beneficiaries.

4.6 Conclusions

The chapter explored the use of ES knowledge to support urban planning in the specific phase of the planning process where *decisions* among alternative scenarios are to be made. Essential in this phase is to account for the multiple ES that are affected by planning actions, considering changes triggered by planning decisions in both the supply of and the demand for ES (Langemeyer et al., 2016). To this aim, beneficiary-based indicators combined through multi-criteria analysis seem to be a promising methodology. Contrary to strictly biophysical measures and to monetary values, beneficiary-based indicators are coherent with planning objectives directed to pursue public interests and societal benefits (von Haaren and Albert, 2011), hence adequate to integrate ES knowledge in the assessment of planning actions (Olander et al., 2018). Multi-criteria analysis offers a platform to combine the results of multiple ES assessments with other relevant criteria, exploring different stakeholder perspectives and balancing competing interests (Adem Esmail and Geneletti, 2018; Saarikoski et al., 2016).

Nevertheless, ES assessment methods usable for planning and able to produce beneficiary-based indicators with the required level of detail are not common in the urban ES literature. While most methods focus on biophysical features, beneficiary-based indicators require a transdisciplinary effort that allows linking ecological values with social benefits (Potschin-Young et al., 2017). At this level, methods, data, and indicators from the urban planning discipline may provide a valuable contribution, for example to the spatial analysis of beneficiaries and the identification of specific vulnerable groups.

Overall, ES knowledge proved useful and usable to inform the urban planning process. However, the presented application was only an exercise, where ES benefits were the only criteria considered. While the ES approach may promote an enhancement of urban green infrastructure leading to an increase in ES beneficiaries and benefits, this does not guarantee that the proposed action is sustainable. Further criteria (e.g., economic aspects, equity in the distribution of benefits, consequences for ecosystem health) should be taken into account to assess the expected consequences of planning decisions, measuring their overall sustainability.

Chapter 5

Framing the ecosystem service approach in the context of European strategies for sustainable urban development*

5.1 Introduction

The objective of this chapter is to frame the integration of ecosystem services (ES) in urban planning in the wider context of European spatial strategies for sustainable urban development. The analyses presented in the previous chapters proved that the ES approach is a valuable tool to promote the conservation and enhancement of urban green infrastructure. Making urban planners and decision-makers aware of the links between ecological functions and human wellbeing not only supports the design and impact assessment of planning actions, but can also strengthen the inclusion of ES-related strategies among the guiding principles of the planning process. The 'green city' strategy aimed at enhancing urban ES and related benefits is acknowledged as a fundamental contribution to a more sustainable urban development, and many initiatives are ongoing, to promote its uptake in the urban

* This chapter is based on: Cortinovis, C., Haase, D., Zanon, B., Geneletti, D. (in review). Is urban spatial development on the right track? Comparing strategies and trends in the European Union. *Landscape and Urban Planning*.

planning processes of cities worldwide. However, urban green infrastructure is just one component of urban systems: even restricting the analysis only to the spatial aspects directly controlled by urban planning, i.e. the urban form and the spatial arrangement of land uses, other features of the urban systems emerge that affect their sustainability (Jabareen, 2006).

Several studies have analysed the relation between the spatial development of cities and their sustainability. Key spatial features of urban systems have been found to determine cities' performance in terms of mobility (Camagni et al., 2002), energy and resource efficiency (Alberti, 1999; Ewing, 2010), climate change mitigation and adaptation (Hamin and Gurran, 2009), and biodiversity and ES (Tratalos et al., 2007). Accordingly, strategies have been formulated to manage and regulate some of these features, including cities' territorial extension (e.g., 'no net land take' (Seto et al., 2011)), relation with surrounding rural and natural areas (e.g., green belts, green wedges (Amati and Taylor, 2010; Frey, 2000)), urban form (e.g., compact, polycentric (OECD, 2012; Parr, 2004)), and arrangement of land uses and activities (e.g., functional mix, density (Grant, 2002; Jabareen, 2006)). In the last decades, the implementation of these strategies in cities and urban regions across the world has allowed assessing their potential effectiveness across different contexts, and provided insights into adjustments and solutions applicable in different local conditions (see for example McCrea and Walters (2012); Millward (2006); Westerink et al. (2013)).

More recently, some of the spatial strategies for urban development have been included in policies at the international level. The *New Urban Agenda* adopted in 2016 represented a milestone along this process, advancing a set of spatial strategies for the first time agreed-upon at the global level: compactness, density, polycentrism, mixed use, and prioritization of urban renewal (UN General Assembly, 2016, §51-52). Within this context, the European Union (EU) is probably the most advanced case of formulation and application of common spatial strategies to cities with different historic backgrounds, planning traditions, economic and social conditions, as well as current and expected development trends (Commission of the European Communities, 1997; Nadin and Stead, 2008). In the last 25 years, under the overall objective of territorial cohesion, EU Member States have debated, among others, also the issue of urban spatial development (Faludi, 2010). The discussion has produced a series of agreed-upon resolutions, the most recent being the *Urban Agenda for the EU 'Pact of Amsterdam'* (European Commission, 2016a). Parallel to this process of negotiation among Member States, policies directly promoted by the European Commission have also defined and supported specific spatial strategies, mostly in relation to the implementation of sectoral policies under the direct competence of the EU (e.g., environment, energy, mobility) (Ravesteyn and Evers, 2004), which contributed to steer the principles of the *Urban Agenda for the EU* (Atkinson, 2001).

Today, the applicability of common spatial strategies to the large variety of conditions of cities worldwide is still debated (Watson, 2016), and divergent theoretical approaches as well as local barriers are expected to emerge in the implementation phase (Barnett and Parnell, 2016). To this respect, it must be notes that both the spatial strategies promoted at the global level by the *New Urban Agenda* and the strategies agreed-upon at the EU level are 'soft regulations' that do not rely on statutory land-use plans. Hence, their mainstreaming requires mobilizing the lower governance levels through joint visions, coordination, and cooperation (Dühr et al., 2007; Faludi, 2010). This non-prescriptive status, together with the diversity of conditions to which the strategies are directed, calls for a comparative approach in analysing their influence in the development of cities (Sykes, 2008). Moreover, little is known about how the strategies interact in the implementation phase. While in the last decade comparative studies have been carried out on a variety of topic, including population dynamics (Turok and Mykhnenko, 2007), land use development models (Kasanko et al., 2006), and the availability of green spaces and ES (Kabisch et al., 2016; Kabisch and Haase, 2013; Larondelle et al., 2014), to name just a few among those focusing on the EU, a systematic monitoring of the progresses in the multiple directions suggested by common spatial strategies is still lacking.

In this chapter, EU is used as a test-bed to observe the relation between the 'green city' strategy and other spatial strategies for sustainable urban development. The specific objectives are: first, to identify the main spatial strategies for sustainable urban development agreed-upon in the EU; second, to investigate the presence of synergies and trade-offs among them and to shed light on the context- and path-dependencies that may catalyse or hinder their successful implementation. This should help to understand whether and under what conditions the 'green city strategy' supported by the ES approach can be expected to reinforce other strategies, or rather to conflict with them, ultimately promoting a more informed uptake in urban planning and a correct monitoring of its implementation.

Since no information is available about the actual inclusion of the strategies in the urban plans of the cities, and even less is known about if and how the strategies have been pursued and implemented through planning actions, the proposed approach is to look at on-the-ground spatial development trends. To capture the relation among the strategies, the recent trends in the spatial development of a large sample of 175 EU cities are analysed and compared with the direction suggested by the strategies. The comparison highlights where and how often the directions suggested by different strategies have been successfully pursued together. This allows inferring potential synergies and trade-offs that, although not linked by strict causality, reveal elements to which planners must pay special attention.

Beyond this introduction, the chapter is structured into five sections. Section 5.2 identifies the main spatial strategies agreed-upon at EU level. Section 5.3 presents the key spatial features and the set of indicators selected to measure the coherence of cities' spatial development with each strategy. The results are presented in Section 5.4, considering the whole sample as well as specific categories of cities based on geographical location and population dynamics. Finally, Section 5.5 discusses methods and findings, and Section 5.6 draws from the study some key conclusions, including directions for future application and research.

5.2 Identifying European spatial strategies for sustainable urban development

To identify spatial strategies for sustainable urban development agreed-upon at EU level, relevant policy documents published since 1993, i.e. the year in which the EU replaced the European Community, were analysed. The selection process involved a a snowball search through references (Greenhalgh and Peacock, 2005), starting from the list of reference documents of the latest *Urban Agenda 'Pact of Amsterdam'* (European Commission, 2016a) and progressively integrating the list with other documents related to urban spatial planning. Since spatial planning encompass different sectoral policies, strategies may respond to multiple objectives, including protection of cultural and natural heritage, biodiversity conservation, social inclusion, reduction of air and water pollution, resilience to natural hazards, and climate change mitigation and adaptation (European Commission, 2011a; UN General Assembly, 2015). Therefore, the review considered policies on urban environment, resource use efficiency, green infrastructure, soil protection, and smart and inclusive growth, among others.

I limited the search to two types of documents, which capture the formulation of policies at the supra-national strategic level:

A. Documents agreed by Member States Ministers during informal meetings (bottom-up agreements on common strategies to pursue EU-wide);
B. Communications from the European Commission (top-down recommendations to Member States to adopt EU-relevant strategies in their internal policies).

The search resulted in 30 policy documents, 13 from group A (Table B.1) and 17 from group B (Table B.2), which were analysed through qualitative content analysis (Hsieh and Shannon, 2005). Spatial strategies explicitly referring to cities and urban areas and addressing either the urban form or the spatial arrangement of land uses were defined as relevant contents. The analysis followed two successive steps. First, the documents were analysed and a database was compiled with relevant contents. Second, recurring spatial strategies were identified as emerging categories and clustered the entries according to the strategy of reference. To ensure that no relevant content was omitted, a second-round keyword-based search through the documents was performed, using selected keywords associated to each strategy (Table B.3). The analysis identified the six main spatial strategies presented in Table 5.1: *green city*, *compact city*, *urban regeneration*, *functional mix*, *no land take*, and *high density*.

TABLE 5.1*: The six main spatial strategies for urban development promoted at EU level. The policy documents mentioning each strategy are shown in Figure 5.1.*

strategy	rationale and actions to implementation
Green city	The strategy aims at improving the quality of life and wellbeing of urban population. It requires increasing the quantity and quality of green areas within the city, including their accessibility.
Compact city	The strategy aims at reducing the negative boundary effects produced by urban areas on their rural and natural surroundings. It requires building in contiguity with existing urbanized areas, avoiding sprawling and sprinkling shapes, and minimizing the fragmentation of non-urban land caused by the enclosure of green patches within urban areas.
Urban regeneration	The strategy aims at increasing the quality and liveability of urban environment while reducing new expansion and land take. It requires directing growth pressures to already-urbanized areas through urban infill (inner-city developments) and the re-use of brownfields, greyfields, and abandoned sites.
Functional mix	The strategy aims at reducing travel needs and related environmental pressures while fostering attractiveness and social inclusion. It requires creating a mix of urban functions within each neighbourhood.
No land take	The strategy aims at halting the physical expansion of cities and the consequent loss of non-urban soil. It requires avoiding (or compensating) the expansion of urban areas at the expenses of rural and natural areas.
High density	The strategy aims at increasing the efficiency in the use and management of land, energy and materials. It requires increasing the concentration of population and activities.

Figure 5.1 indicates their occurrence in the analysed documents and their persistence through time. The scheme allows recognizing the

temporal extent of the strategies (e.g., *functional mix* is never mentioned after 2007), and distinguishing those supported for long time (e.g., *urban regeneration*) from those that were abandoned after few years (e.g., *high density*). Furthermore, it highlights the different ownership of the strategies, and the presence - or absence - of joint efforts in their promotion by the European Commission and EU Member States. For example, the *no land take* strategy has been supported almost exclusively by the European Commission, and is mentioned just once in documents agreed by Ministers of the Member States. G*reen city* and *compact city* strategies, in contrast, are common to both top-down and bottom-up policies.

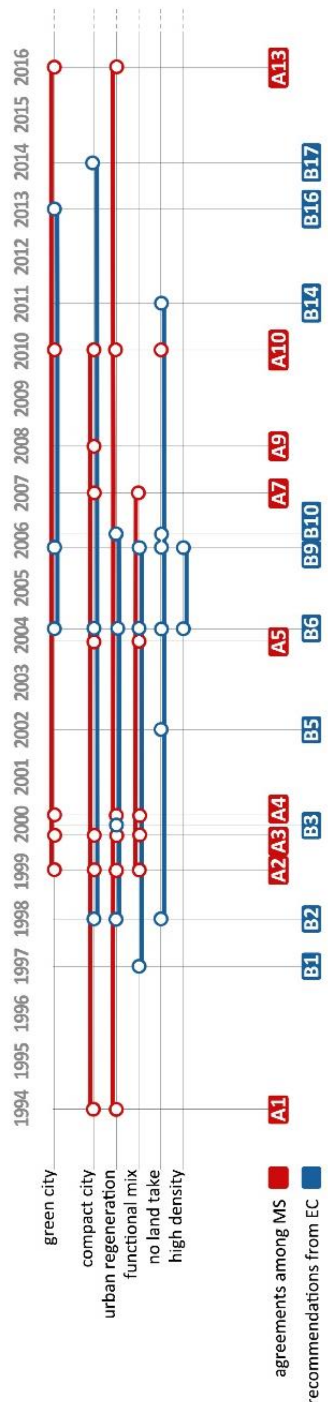


FIGURE 5.1: *Timeline recording the presence of the spatial strategies in the two groups of analyzed policy documents. Document details are reported in Appendix B.*

5.3 Analysing urban spatial development through the lens of the strategies

5.3.1 Methods and indicators

Progresses in the direction suggested by the strategies can be captured by measuring changes in relevant spatial features of cities over time (Seto and Fragkias, 2005; Grădinaru et al., 2017). Although some strategies also include qualitative aspects, the analysis focused on quantifying changes related to the urban form and the spatial arrangement of land uses. A number of indicators suitable to this purpose exist in the scientific literature, including indicators based on land uses and land covers, population, landscape metrics, and a combination of these (Clifton et al., 2008; Lowry and Lowry, 2014; Schwarz, 2010). To compare the observed spatial development of cities with the six spatial strategies identified in Section 5.2, three types of indicators were selected, namely: i) indicators related to the share of different land use and land cover (LULC) classes and population density, ii) landscape metrics (Uuemaa et al., 2009) related to urban form and spatial arrangement of LULC, and iii) land cover flows (EEA, 2006) that detail the amount of land involved in each type of LULC transition. All selected indicators can be calculated from publicly available data sets, as described in Section 5.3.2. Table 5.2 provides an overview of the indicators adopted in the study and the rationale for their use.

TABLE 5.2: *List of indicators adopted for the analysis. Indicator types: LP = LULC classes and population, LM = landscape metric, LCF = land cover flow. * For land cover flows, the cited reference is based on a different LULC classification system (Corine), thus the definition of each specific LCF may be different. Full details on the LULC classes used to quantify each LCF are provided in Appendix B.*

strategy	indicator and type	description and rationale	direction promoted by the strategy	methodological ref.
green city	urban green area - LP	The total amount of green area in the city.	increase	(Kabisch and Haase, 2013; Tratalos et al., 2007)
	per-capita urban green area - LP	The total amount of green areas divided by population. The indicator is consistent with a common formulation of targets for the strategy.	increase	(Kabisch and Haase, 2014)
	new urban green areas - LCF	The realization of new urban green areas either as urban expansions or through the conversion of already urbanized land.	maximize	(EEA, 2006)*
	loss of urban green areas - LCF	The conversion of existing urban green areas to other LULC. Low values of the indicator indicate conservation of green areas and protection of urban biodiversity.	minimize	(EEA, 2006)*
compact city	Edge Density (ED) - LM	The total length of urban edges per unit of urban area. It measures shape complexity and fragmentation. Compact shapes are characterized by low values of ED.	decrease	(Herold et al., 2003; Schwarz, 2010)
	new green fragments without use - LCF	Remnants of green/non-urbanized land surrounded by urban LULC. They represent non-urban land left behind during urbanization.	minimize	(EEA, 2006)*
urban regeneration	recycling of urban land - LCF	All LULC changes occurring between two urban classes. A high share of recycling compared with new urbanization indicates a focus on urban regeneration.	maximize	(EEA, 2006)*

strategy	**indicator and type**	**description and rationale**	**direction promoted by the strategy**	**methodological ref.**
urban regeneration *(continued)*	in-fill development and re-use of brownfields - LCF	LULC changes involving “land without current use”. A high rate of conversion of unused land indicates a progress toward urban regeneration.	maximize	(EEA, 2006)*
functional mix	Interspersion and Juxtaposition Index (IJI) - LM	The intermixing of different LULC classes within the landscape. Higher values of IJI within urban areas describe cities where different LULC classes are well-interspersed, i.e. tend to be equally adjacent to the other classes.	increase	(Griffith et al., 2000; Lowry and Lowry, 2014)
no land take	urban area - LP	The overall amount of urbanized land.	no change / decrease	(Kasanko et al., 2006)
	new urbanization - LCF	The expansion of urban area at the expenses of non-urban land. New urbanization should be limited and counterbalanced by the conversion of urban land to non-urban LULC.	minimize	(Kasanko et al., 2006; Schneider and Woodcock, 2008)
	conversion from urban to non-urban uses - LCF	The retreat of urban area in favour of agricultural uses, forests, or semi-natural areas. This flow should counterbalance new urbanizations.	maximize	(EEA, 2006)*
high density	urban density - LP	The number of inhabitants per unit of urban area. It is a widely-adopted indicator to measure the density of urban areas.	increase	(Hasse and Lathrop, 2003; Kasanko et al., 2006; Schneider and Woodcock, 2008)
	residential density - LP	The number of inhabitants per unit of residential area. It measures population concentration within residential areas.	increase	(Galster et al., 2001; Kasanko et al., 2006)
	residential densification - LCF	The sum of all the LULC flows that involve a change from a less-dense to a more-dense residential class.	maximize	(Broitman and Koomen, 2015)

The geographical information system ArcGIS10.0 was used to quantify the share of LULC classes and LULC flows on vector maps, while landscape metrics were computed on the respective 2.5 m-resolution raster maps using Fragstat4 (McGarigal et al., 2012). Descriptive statistics were used to measure the variability of the indicators across the sample. Then, considering one representative indicator for each strategy (Table 5.3), the overall performance of each city was assessed based on the number of strategies 'matched' by the observed development trend. Finally, clusters of cities were identified considering the specific set of strategies for which some progresses could be observed during the analysed period. A hierarchical clustering was performed using the *hclust* algorithm in the R stats package (R Core Team, 2014), adopting Ward's clustering criterion (*ward.D2* agglomeration method) and *asymmetric binary* distance measure. The level of aggregation was defined by imposing that all the cities in the same cluster have a homogeneous behaviour with respect to at least one strategy.

TABLE 5.3*: Representative indicators for each strategy and related thresholds considered in the assessment of the overall performance of cities.*

strategy	indicator i	positive change	no relevant change	negative change
green city	urban green area	Δi>1%	-1%<Δi≤1%	Δi≤-1%
compact city	Edge Density (ED)	Δi<-0.1	-0.1≤Δi<0.1	Δi≥0.1
urban regeneration	recycling of urban land/new urbanization	i>1.1	0.9<i≤1.1	i≤0.9
functional mix	Interspersion and Juxtaposition Index (IJI)	Δi>0.1	-0.1<Δi≤0.1	Δi≤-0.1
no land take	urban area	Δi<0.1%	0.1%≤Δi<1%	Δi≥1%
high density	urban density	Δi>1%	-1%<Δi≤1%	Δi≤-1%

5.3.2 Sample of cities and data

LULC data were derived from the Urban Atlas database, which provides comparable, high resolution maps of EU core cities and Functional Urban Areas with more than 100,000 inhabitants for 2006 and 2012 (http://land.copernicus.eu/local/urban-atlas). The analysis focused on core cities, which correspond to municipal administrative units, and combined spatial data with population data from the Eurostat Urban Audit (http://ec.europa.eu/eurostat/web/cities/data/database). The sample comprises 175 core cities, based on the availability of both LULC and population data in the two reference years (last accessed October 2016). The complete list of cities can be found in Table B.5.

To investigate spatial and structural patterns within the large sample, the cities were classified based on i) population dynamics, and ii) geographical location within Europe. Both classifications are common in comparative studies of European cities (Haase et al., 2013; Kasanko et al., 2006; Larondelle et al., 2014; Turok and Mykhnenko, 2007). Population dynamics allow discriminating between growing and shrinking cities. Shrinking cities are defined as cities where population decreased of more than 0.15% per annum in the analysed period. Conversely, growing cities are those where population grew of more than 0.15% per annum (Wolff et al., 2018). The sample comprises 96 growing cities, 55 shrinking cities, and 24 stable cities (i.e. characterized by a total population change between -0.9% and +0.9% between 2006 and 2012) (Figure 5.2-top). Geographical location considers the distribution of the respective countries in the four main EU regions, as defined by the official Thesaurus of the EU and the UN Statistics Division (EuroVoc, 2017; United Nations Statistics Division, 2017). The sample comprises 63 cities from Eastern countries, 9 from Northern countries, 28 from Southern countries, and 75 from Western countries (Figure 5.2-bottom).

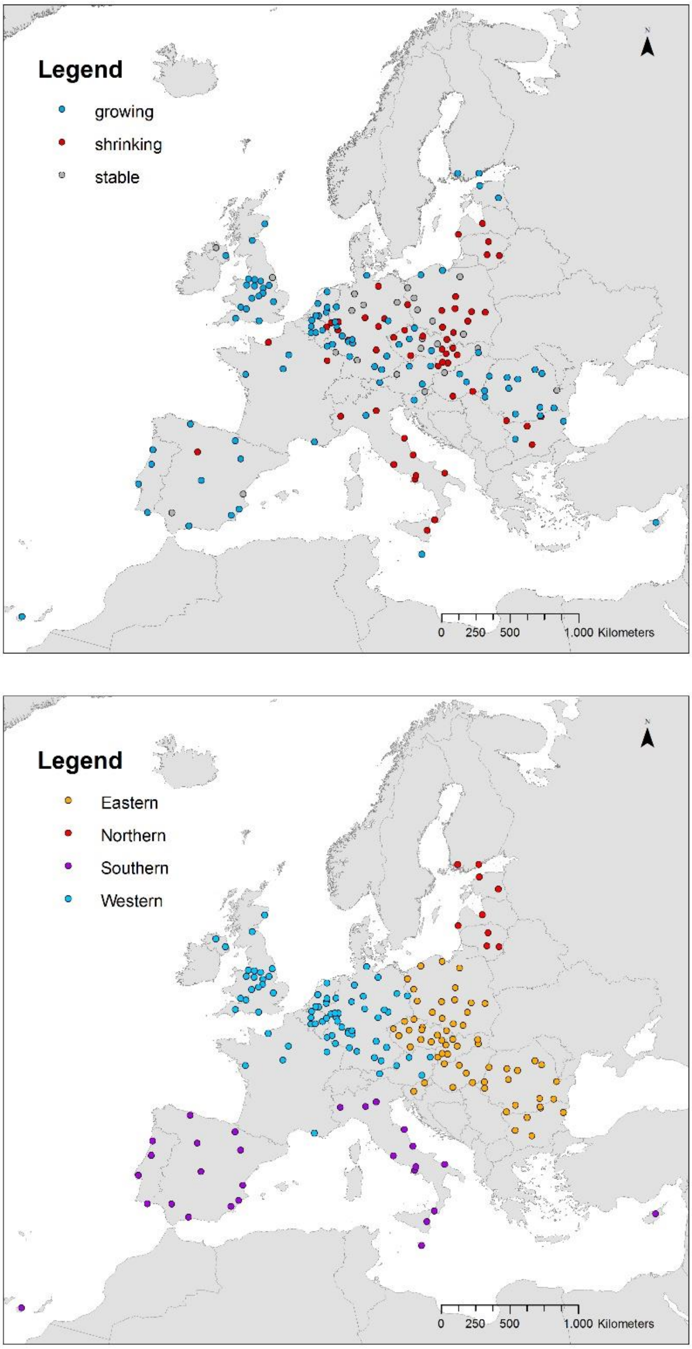


FIGURE 5.2: *Sample of cities broken down by population dynamic (top) and region (bottom). No city from Greece, Ireland, and Sweden is included in the sample due to the lack of population data in the Eurostat database.*

5.4 Results

5.4.1 Trends with respect to the individual strategies

Green city

In 2012, according to the Urban Atlas classification, urban green areas represented 4.0% of the total administrative area of the cities, and around 9% of the urbanized areas. Data are characterized by a large variability: the share of urban green areas varies between 0.2% (Faro – PT) and 20.6% (Karlovy Vary – CZ) of the total city area, equal to 1.1% and 41.5% of the urban area, respectively. Most analysed cities (*N=94*) experienced a reduction in urban green area, but changes range from -14.4% to +12.2% of the existing urban green area in 2006 (Figure 5.3b). The balance between new urban green areas and conversion of existing urban green areas is negative in all city categories except Southern cities, with an average per city increase of around 1.1 ha/year and a contemporary loss of around 1.4 ha/year over the whole sample (Figure 5.4b).

The average per-capita urban green area is also slightly decreasing from 22.8 m^2 in 2006 to 22.6 m^2 in 2012 (Figure 5.5). Values range from 2.2 m^2 in Piatra Neamț (RO) to 244.1 m^2 in Karlovy Vary (CZ), both in 2012. In 2006, shrinking cities already had, on average, 4.8 m^2 of urban green areas per inhabitant more than growing cities, and their divergent trends in the analysed period increased the gap. Northern cities have the largest availability of per-capita green space (39.5 m^2 and 41.6 m^2 the mean values in 2006 and 2012), but with differences in trend between growing cities in Estonia and Finland (decreasing) and shrinking cities in Latvia and Lithuania (increasing). Eastern and Western cities have similar distributions of per-capita urban green area, while the average value for Southern cities is the lowest, less than half compared to all the other regions (10.3 m^2 in 2006). Southern and Eastern cities do not show significant changes between the two reference years, while in Western cities the average value decreased of 0.7 m^2.

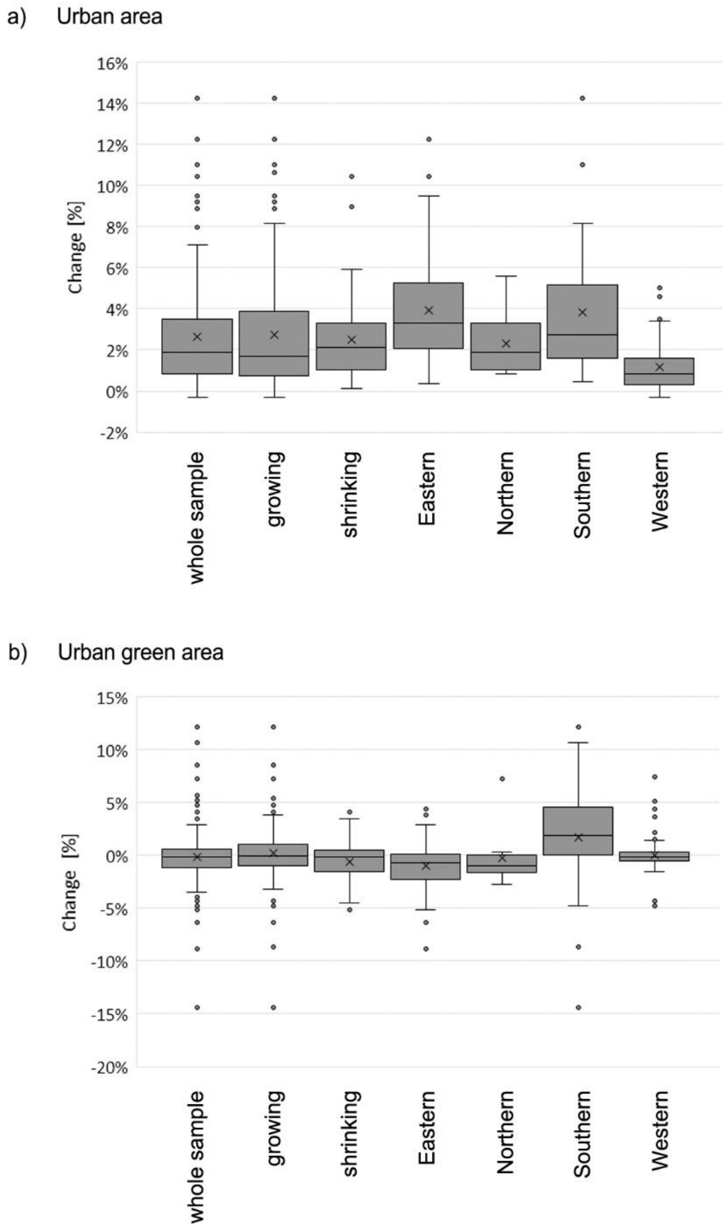
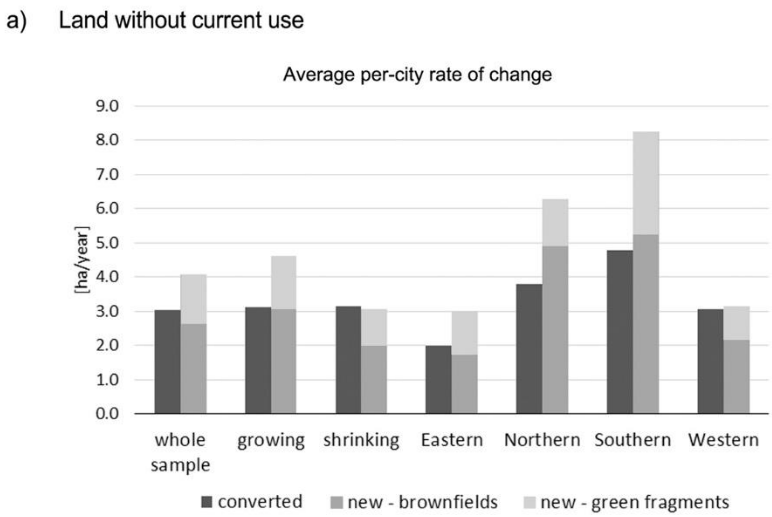


FIGURE 5.3*: Percental changes in a) urban area and b) urban green areas between 2006 and 2012.*

Urban regeneration

Recycling of urban land represents 41.4% of the total LULC flows in the analysed period: a value slightly lower than the share of new urbanization (around 50%) (Figure 5.6a). The predominance of new urbanization is more evident in shrinking cities, while recycling of urban land accounts for the majority of LULC flows in both Northern and Western cities. In Eastern cities, new urbanization doubles the area involved in urban recycling. By looking more in detail at the LULC classes involved, a quite differentiated picture emerges. The completion of construction sites represents the major part of recycling of urban land. Construction for economic uses (i.e., industrial, commercial, infrastructures, and leisure) is generally higher than construction for residential use (an average of +7%, with a peak of +20% in Southern cities), with the only exception of Western cities, where the two flows are almost balanced.

Around 20% of the areas classified as ‘land without current use’ in 2006 was converted to a new urban use during the analysed period. This flow accounts for both cases of in-fill development, when the conversion involved green fragments without use in urban areas, and brownfield redevelopment. The share of conversion is slightly higher in growing cities compared to shrinking cities, and in Western cities compared to the other regional groups. Most of this land was converted to economic uses (45.4%), while conversion to new residential areas was less frequent (28.5%). Only 3.4% was converted to new urban green areas (Figure 5.6c). However, the conversion is counterbalanced by the presence of new brownfields from the abandonment of previous urban uses, or even from the abandonment of constructions sites (Figure 5.6c): 25.6% of the total ‘land without current use’ in 2012 was produced in the previous six years. Considering the whole sample, new brownfields almost equal the areas without use converted during the same years (an average of 3.0 vs. 2.6 ha per city per year), and overcome them both in Northern and Southern cities (Figure 5.4a). Interestingly, shrinking cities are the only category showing a positive balance between re-use and creation of brownfields. Both the number of new brownfields and their extension are higher in growing cities compared to shrinking cities, and in Southern and Northern cities compared to Eastern and Western (Figure 5.8b).



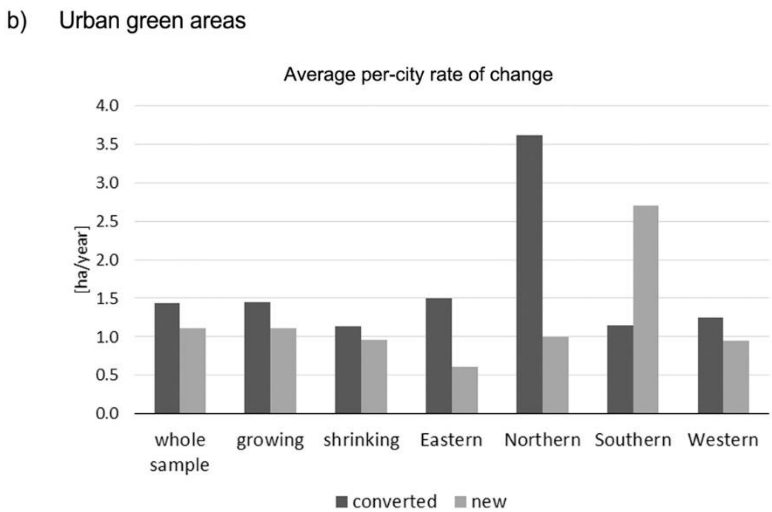


FIGURE 5.4: *Average per-city rate of change of selected LULC classes in the period 2006-2012: a) "land without current use", and b) urban green areas.*

Functional mix

Interspersion and Juxtaposition Index of urban classes increased in most cities (*N=117*), indicating a higher level of functional mix within urban area in 2012 compared to 2006. The same as for compactness, no significant difference emerges between growing and shrinking cities, while regional groups are characterized by a quite differentiated behaviour (Figure 5.7b). Here too, Southern cities show the most positive change and the highest variability, while most Northern cities are characterized by a decreasing functional mix.

No land take

Total urban area increased in almost all cities of the sample, excluding only the two cities of Belfast and Wrexham in the UK. Changes lower than 0.1% of the existing urban areas are measured in other three cities. The average increase is 2.6% of the existing urban area in 2006, but it reaches peaks of more than 10% in five cities, and is above 5% in other twenty cities. No significant difference exists between growing and shrinking cities, while, on a regional basis, only Western cities show a distinct behaviour characterized by a lower urban expansion. All cities with limited land take (< 0.35%) are in the Western region (Figure 5.3a). The expansion of urban area is for the largest part (around 95%) at the expenses of agricultural and semi-natural areas. In the whole sample, expansion in forested area account for less than 5% of the new urban areas, but it reaches a share of around 20% in Northern cities.

Land take is mostly driven by economic uses, which - excluding land take for new infrastructures - account for 38.1% of the new urban expansion. The residential sector is responsible for 22.0% (Figure 5.6b), mimicking the ratio between residential and economic uses found in the conversion of land without current use. Assuming the same ratio also for the future uses of construction sites, economic uses alone, without accounting for infrastructures, caused half of the total land take in the analysed period. The main difference between growing and shrinking cities lies in the relative weight of land take for infrastructures, almost two times more intense in the former compared to the latter (8.3% vs. 4.6%). Considering regional groups, Northern cities are the only case in which residential expansion overcomes new land take for economic uses.

The opposite flows, i.e. the conversion of urban land to agricultural, semi-natural, and natural areas, account for around 4% of the total LULC flows in the sample of cities (Figure 5.6a), and involved 0.1% of their total administrative areas (corresponding to 0.2% of the urbanized area). The largest part (72.5%) has been converted to agricultural uses, especially pastures and arable land, while around a quarter turned to semi-natural areas, prevalently to “herbaceous vegetation associations”.

Conversion to forest involved only 1.7% of the total area converted from urban to non-urban uses, but is particularly relevant in Southern cities, where it reaches 5%.

Hing density

Growing and shrinking cities are characterized by a clearly different trend in density during the analysed period (Figure 5.9). Growing cities show a various pattern, but most of them were subject to a density increase between 2006 and 2012. On the other hand, urban density decreased in almost all shrinking cities due to combined population loss and contemporary increase in built-up area. Residential density followed a similar pattern. Looking at mean values, urban density in 2006 was already higher in growing (43.9 p/ha) than in shrinking cities (39.7 p/ha), while the mean residential density in 2006 was almost the same for the two groups (104.2 vs 102.1 p/ha). Most of the cities in the Eastern, Southern, and Northern regions moved toward a less dense development model, while densification prevailed in Western cities. However, average values for the four regional groups are very different. The lowest values are found in Northern cities (33.3 p/ha 81.5 p/ha for urban and residential density in 2012, respectively), and the highest values in Southern cities (56.7 and 149.3 in 2006).

Residential densification accounts for 4.35% of the total LULC flows in the reference period (10.5% of LULC flows among urban classes). The share is slightly higher in growing than in shrinking cities (4.6% vs. 4.0%), but regional differences are more noteworthy. In Northern cities, residential densification represents 14.7% of the total LULC flows, while values for the other regional groups are much lower (5.9% in Western cities, 2.6% in Eastern cities, and 2.2% in Southern cities). The apparent contradiction between trends in density and residential densification within the Northern group is explained by the contrast between growing and shrinking cities already observed for green areas dynamics.

Table B.6 presents the main statistics of the indicators for the different categories of cities, and the maps of the cities in line with the direction suggested by the strategies.

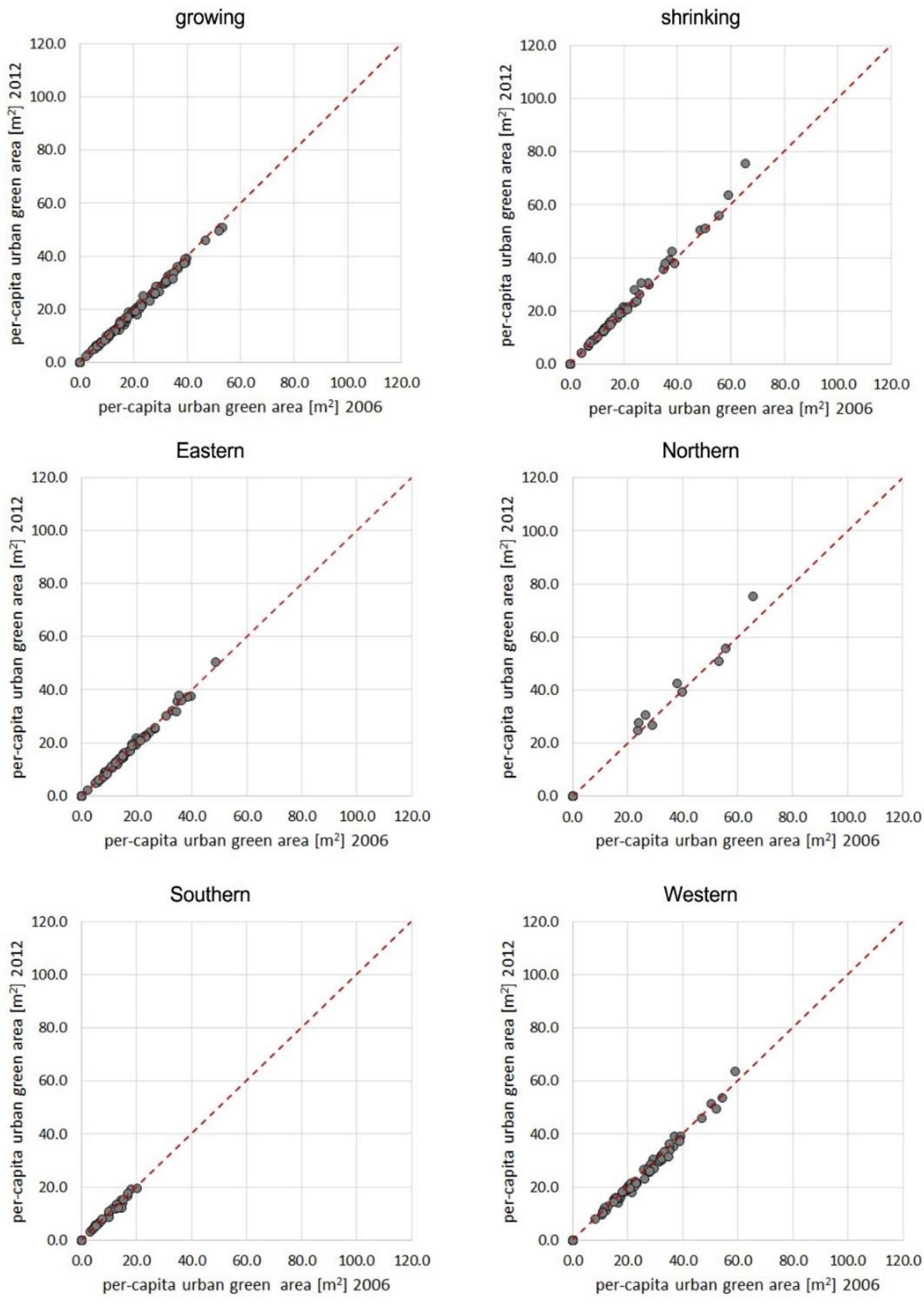
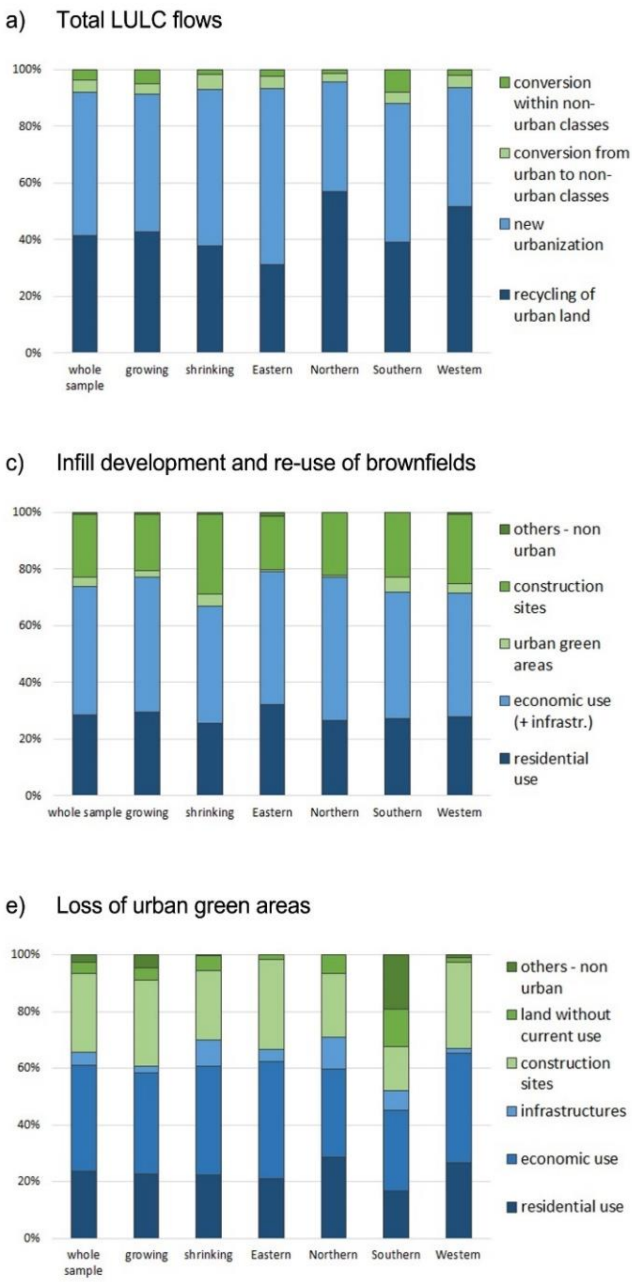


FIGURE 5.5*: Comparison among values of per-capita urban green area in the two reference years, grouped by city category. Values for Karlovy Vary are outliers and are not visible in the graphs (per-capita urban green area = 232.5 m^2 in 2006 and = 235.5 m^2 in 2012).*


a) Total LULC flows
100%
80%
60%
40%
20%
0%
whole sample
growing
shrinking
Eastern
Northern
Southern
Western
conversion within non-urban classes
conversion from urban to non-urban classes
new urbanization
recycling of urban land
c) Infill development and re-use of brownfields
100%
80%
60%
40%
20%
0%
whole sample
growing
shrinking
Eastern
Northern
Southern
Western
others - non urban
construction sites
urban green areas
economic use (+ infrastr.)
residential use
e) Loss of urban green areas
100%
80%
60%
40%
20%
0%
whole sample
growing
shrinking
Eastern
Northern
Southern
Western
others - non urban
land without current use
construction sites
infrastructures
economic use
residential use

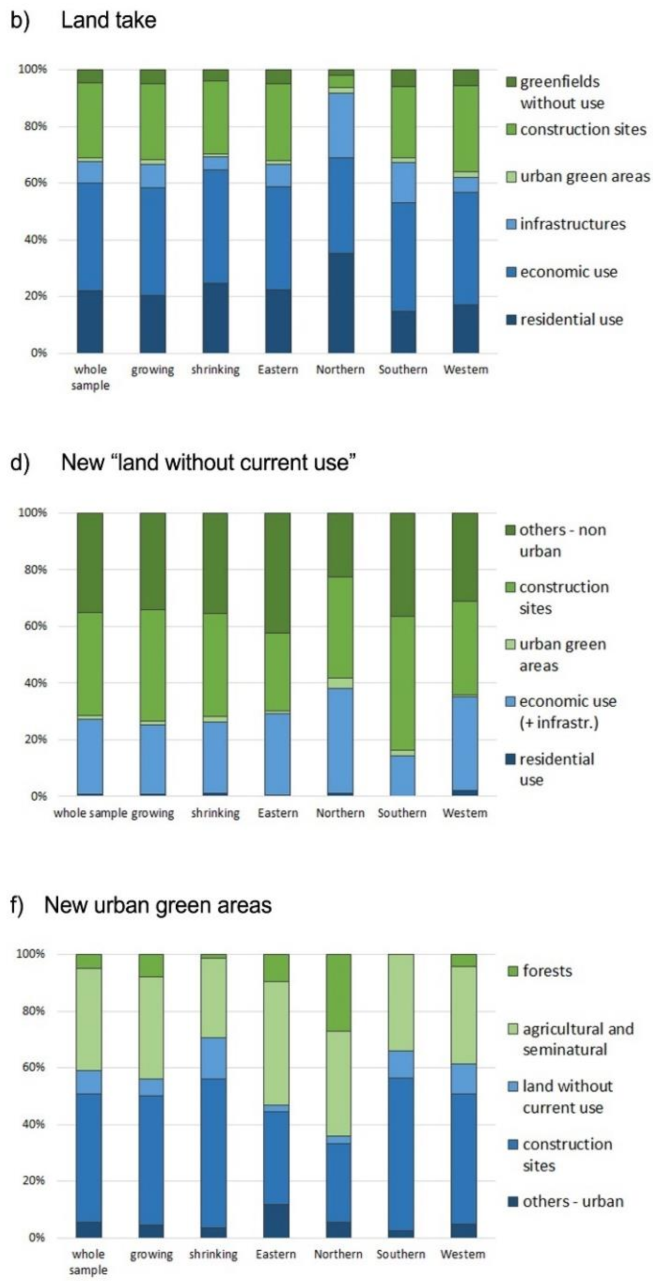


FIGURE 5.6: *Distribution of LULC flows during the analyzed period: a) total LULC flows considering urban and non-urban classes, b) land take by different LULC classes, c) conversion of 'land without current use' by new LULC class, d) new 'land without current use' by previous LULC class, e) conversion of urban green area by new LULC class, f) new urban green area by previous LULC class.*

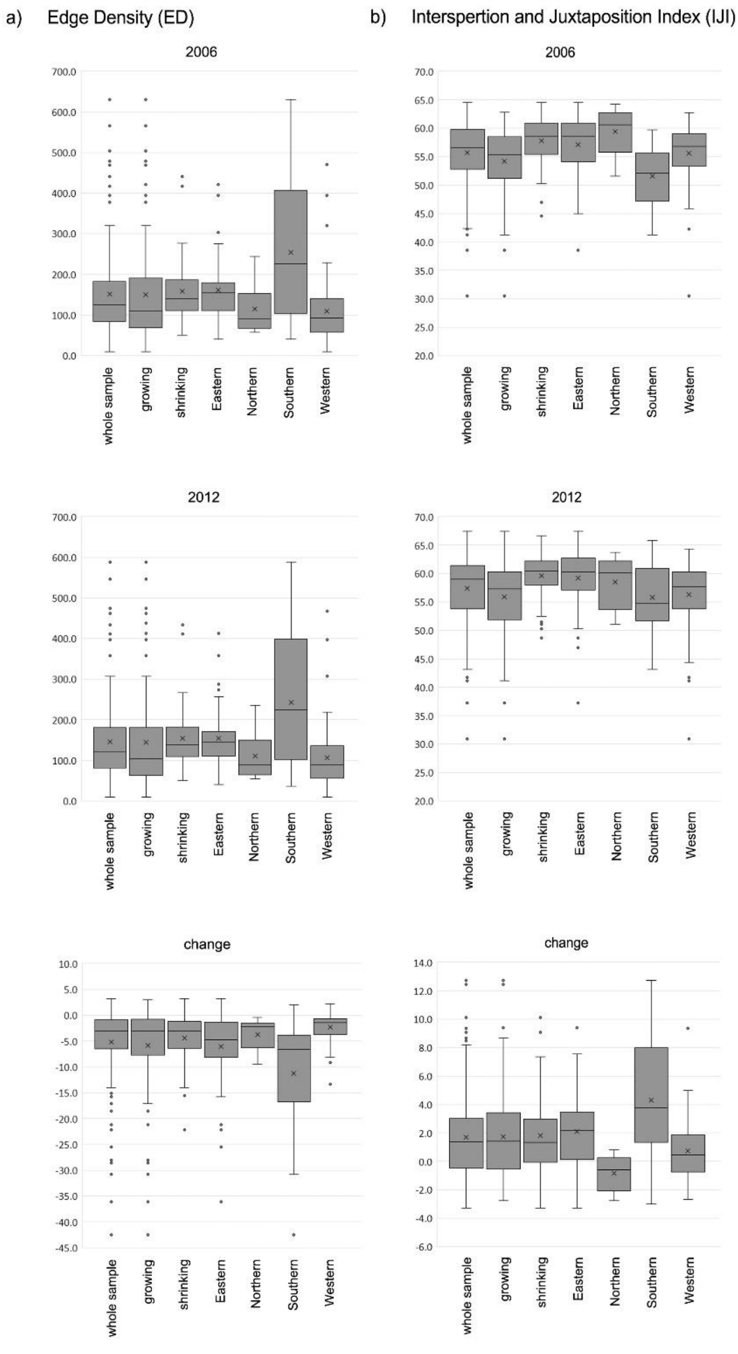


FIGURE 5.7: *Landscape Metrics in the two reference years and their changes during the analyzed period in the whole sample and for each category of cities: a) Urban Edge Density (ED), and b) Interspersion and Juxtaposition Index for urban LULC classes (IJI).*

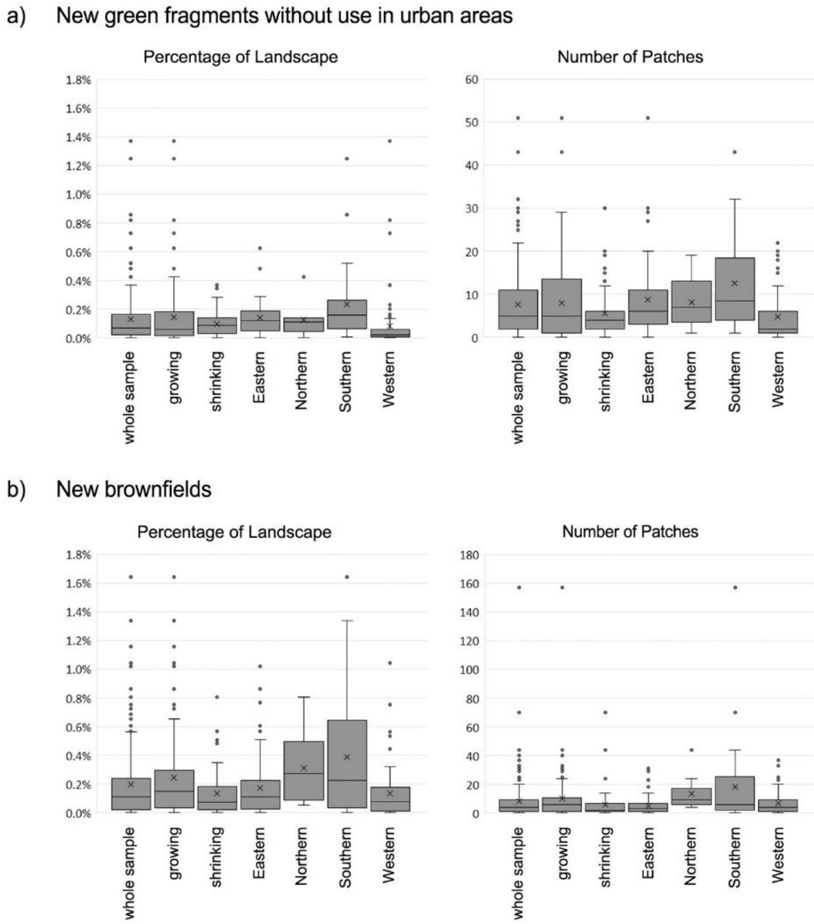


FIGURE 5.8: *Percentage of landscape and number of patches of a) new green fragments without use in urban areas, and b) new brownfields that were produced during the analyzed period.*

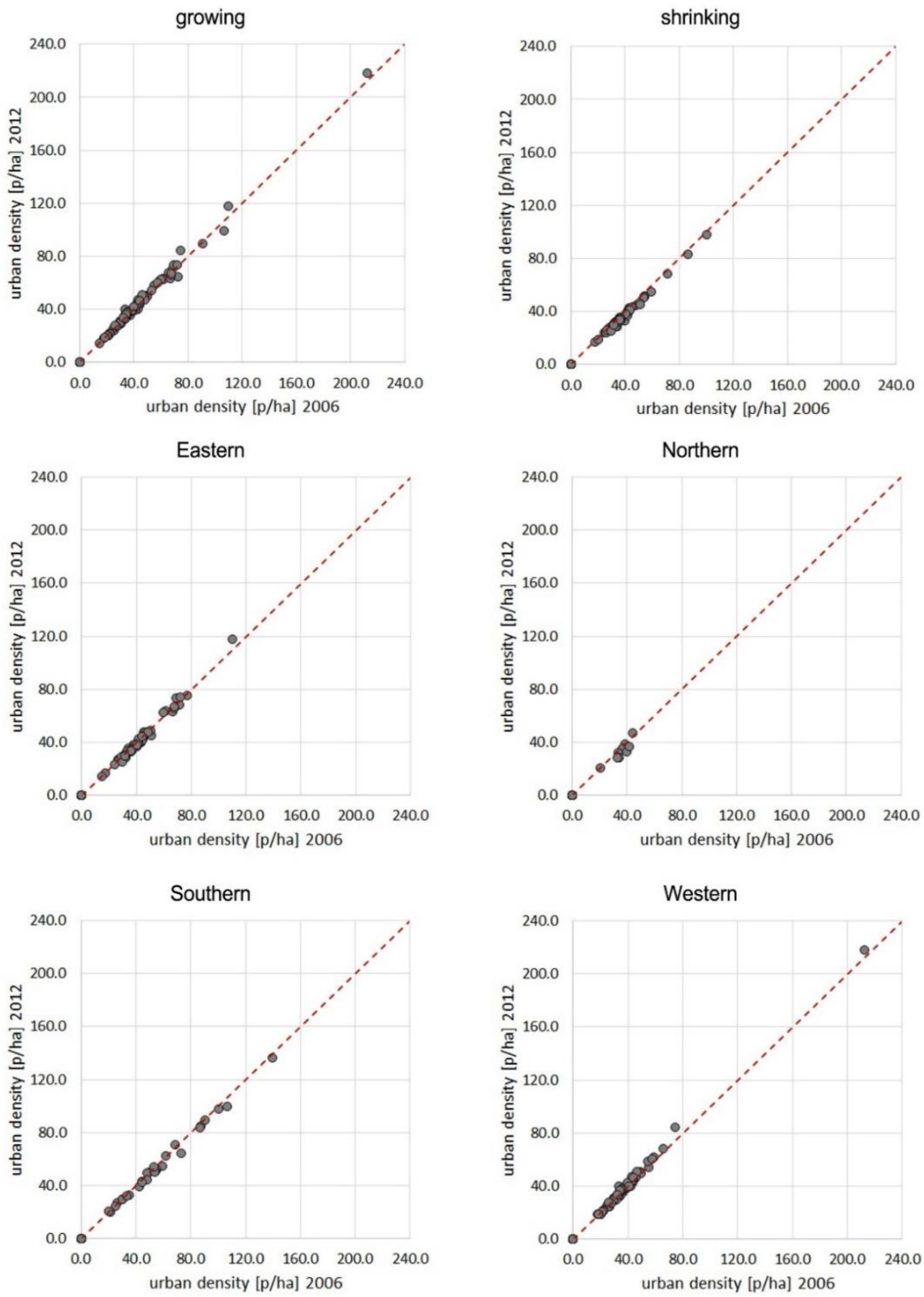


FIGURE 5.9: *Comparison among values of urban density in the two reference years, grouped by city category.*

5.4.2 Overall performance across cities

Table 5.4 provides an overview of the comparison between the spatial development trends observed in the different categories of cities and the six main spatial strategies. For almost all city categories, the development trend was coherent with the principles of *compact city* and *functional mix*, while no category achieved *no land take*. A mixed behaviour emerges with respect to *urban regeneration* and *high density*, with specific categories of cities moving in the directions suggested by the strategies. Overall, no city in the sample presents a spatial development trend coherent with all the six strategies (Figure 5.10). Only in two cities, namely Trier (DE) and Valletta (MT), the spatial development trend was coherent with five strategies, but both failed in halting land take. On the other hand, for two cities, namely Szeged (HU) and Žilina (SK), no coherence at all could be observed.

TABLE 5.4: *Comparison between the spatial development trends of different categories of cities and the direction suggested by the strategies. Results based on the average values of the representative indicators (Table 5.3) for each city category. Key: ↑ = positive change, ↓ = negative change, ⇔ = no relevant change.*

STRATEGY	whole sample	growing cities	shrinking cities	Eastern cities	Northern cities	Southern cities	Western cities
Green city	⇔	⇔	⇔	↓	⇔	↑	⇔
Compact city	↑	↑	↑	↑	↑	↑	↑
Urban regeneration	↓	↓	↓	↓	↑	↓	↑
Functional mix	↑	↑	↑	↑	↓	↑	↑
No land take	↓	↓	↓	↓	↓	↓	↓
High density	⇔	↑	↓	↓	↓	↓	↑

Southern and Western cities tend to be more virtuous than other regional groups, with 55% of them moving in the directions suggested by three or more strategies. On the other hand, the development trend of almost 80% of Eastern cities was coherent with a maximum of two strategies. Growing cities appear by far the most in line with EU-level strategies, with around 25% of them matching the principles of four or five strategies, compared to a share of less than 2% of shrinking cities.

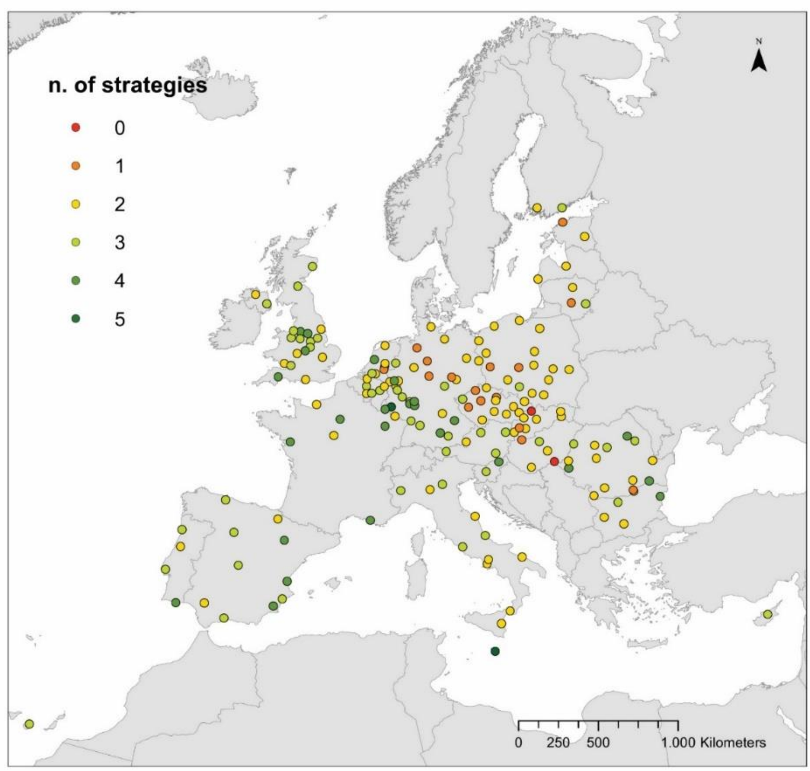


FIGURE 5.10*: Number of strategies 'matched' by the development trend of each of the analyzed cities. Cities were awarded 1 point for each positive change in the indicators, as defined in* **TABLE 5.3***. No city obtained the maximum score.*

Based on the specific set of spatial strategies matched by the development trend of each city, seven clusters were identified that describe different recurring types of spatial development. Figure 5.12 shows the cities included in each cluster and their profile with respect to the strategies. Finally, Figure 5.11 presents a cross-comparison of the changes toward the *green city* and the other analysed strategies. For each pair of strategies, the figure shows how often they are pursued together, i.e. how often the development trend of cities was coherent with both. For example, a more compact urban form and an enhanced functional mix were frequently observed in cities with an increased availability of urban green spaces. On the other hand, only 2% of the cities in the sample followed the directions of both the *green city* and the *no land take* strategies.

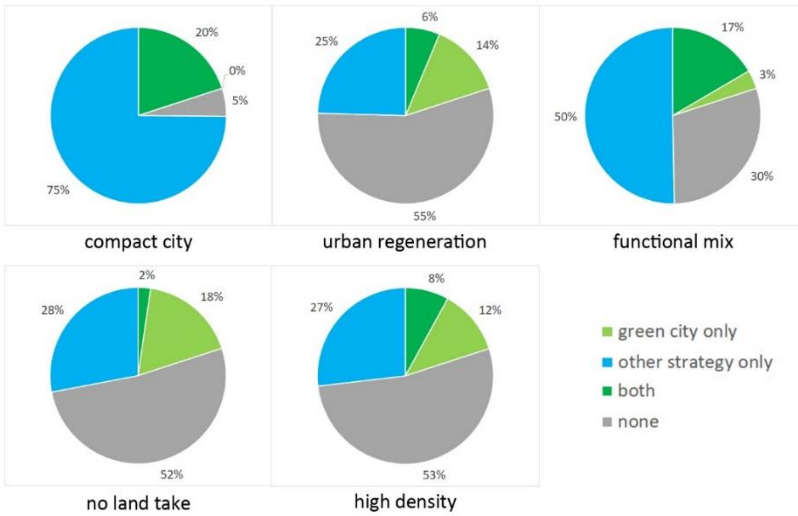


FIGURE 5.11: *Cross-comparison of the observed changes toward the 'green city' and the other strategies: frequency of occurrence of the different combinations in the whole sample of cities.*

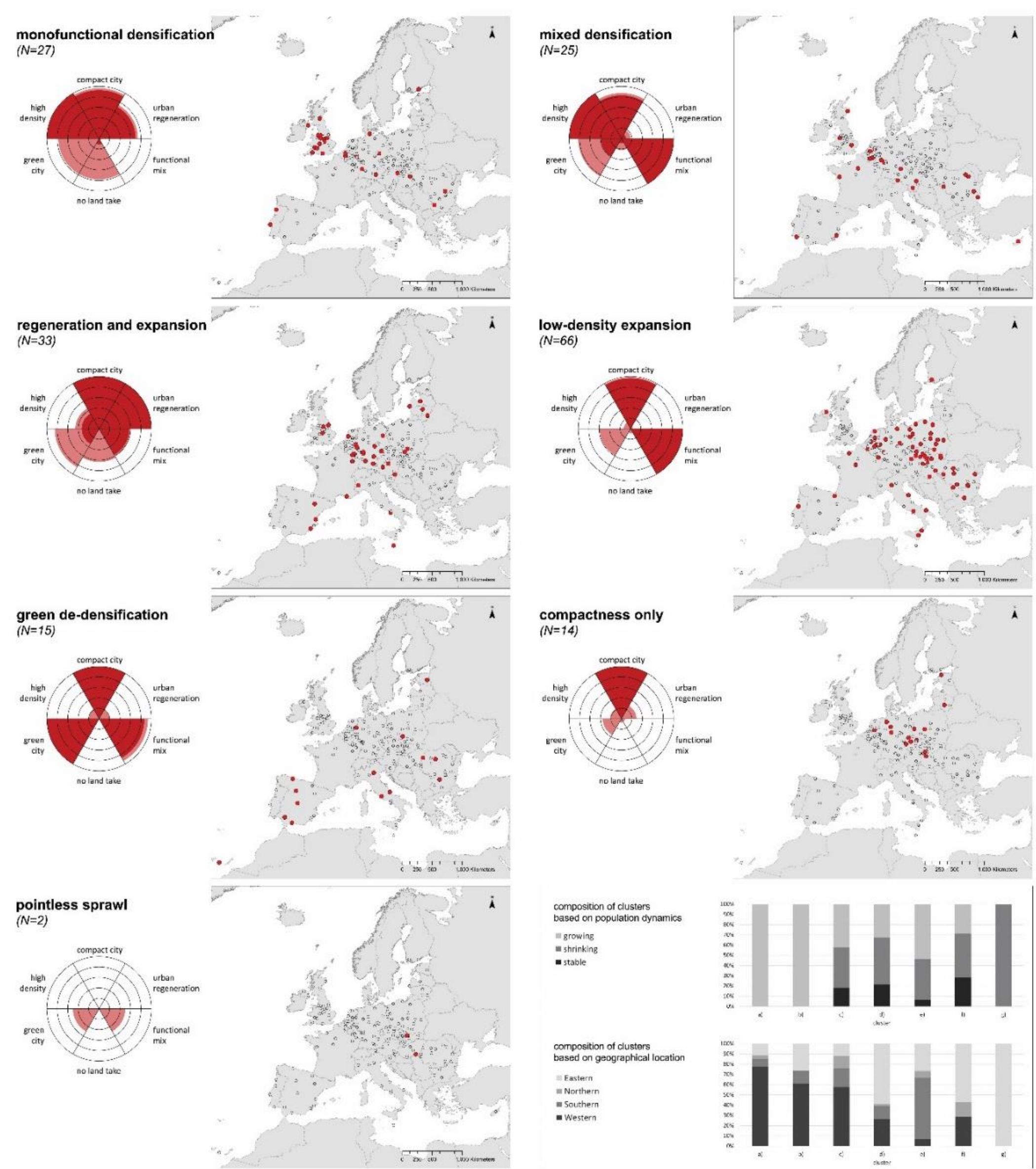


FIGURE 5.12*: Clusters of cities based on similar trends with respect to the analyzed spatial strategies. Wheel diagrams show the composition of each cluster based on the representative indicators in Table 5.3: dark red indicates cities with a positive change, light red cities with no relevant change. The box shows a cross-analysis of clusters and city categories based on population dynamics and geographical location.*

5.5 Discussion

5.5.1 About the findings

In this chapter, the spatial development trend of 175 European cities between 2006 and 2012 was investigated and compared with the six main spatial strategies agreed-upon at the EU level. Certain trends, both positive and negative, characterize almost homogeneously the whole sample: increasing compactness, growing functional mix, and urban expansion. Others are distinctive of specific categories, based either on population dynamics or on geographical location. Population dynamics often mirror economic dynamics (Turok and Mykhnenko, 2007), which can be interpreted as the underlying reason for some of the differences between growing and shrinking cities. At the same time, geographical location in different EU regions reflects different prevailing urban forms (Kasanko et al., 2006) as well as different planning systems and traditions (Commission of the European Communities, 1997), which are captured by some of the analysed indicators. These legacies shape the range of possible development trajectories for each city, and may determine the success or failure in the implementation of strategies and policies.

Despite all the directions suggested by the strategies aim at a more sustainable urban development, the results confirm that, in practice, transformative actions are not always able to target multiple strategies. No clear synergy emerges, for example, between urban regeneration and increase in green space availability or functional mix. Economic and residential uses largely prevail in the recovery of abandoned sites, whose conversion still mostly follows the logic of mono-functional zoning. Higher density, increasing compactness, and growing functional mix do not necessarily support each other, and a growing density alone is not sufficient to slow down land take when large areas remain vacant and de-sealing is not successfully pursued (Haase et al., 2014a; Nassauer and Raskin, 2014). If it is true that potential synergies exist in the implementation of the strategies, they must be consciously promoted and supported by policy-making. This also warns against simplistic solutions and single spatial strategies presented as a panacea for urban sustainability: each strategy has potential, place-specific drawbacks, which should be known and controlled (Westerink et al., 2013). For example, densification may hinder the implementation of climate change adaptation measures (Viguié and Hallegatte, 2012) and worsen urban environmental quality and wellbeing (Melia et al., 2011). Urban regeneration and the enhancement of urban green areas may result in negative social phenomena, triggering ‘eco-gentrification’ and social exclusion or displacement processes (Cameron, 2003; Wolch et al., 2014).

The results also highlight potential conflicts in the use of land that may arise from the implementation of the strategies. For example, only few cities succeeded in achieving higher density while enhancing green space availability: an evidence of the potential trade-off already discussed by several authors (Haaland and Konijnendijk van Den Bosch, 2015; Nilsson et al., 2014; Tratalos et al., 2007), with possible implications also in terms of environmental justice (Kabisch and Haase, 2014; Lin et al., 2015). The trade-off is even more evident between *green city* and *no land take*. Among the cities with a limited land take during the analysed period, only three - namely Darmstadt (DE), Maribor (SI), and Nancy (FR) - showed an increase in the amount of urban green areas. Three quarters of the new green areas derive from the conversion of non-urbanized land, in a kind of ‘open-into-green’ or ‘green-into-green’ change, and most of the cities that succeeded in becoming “greener”, particularly in the Northern and Eastern regions of Europe, achieved this goal through urban expansions.

Considering population dynamics, two paradoxes emerge from the non-linear behaviour of urban systems. Coherently with previous analysis of specific case studies (Couch et al., 2005), shrinking cities continued to expand, with land take mostly driven by economic uses, scarce reuse of urban voids, and a contemporary decline in green space availability This confirms the results by Haase, Kabisch, & Haase (2013) about the mismatch of population and urban area, and tells about a low-intensity development model with increasing per-capita living space. At the same time, contrary to what could be expected from a growing competition for land, growing cities were less efficient in their expansion, with strong effects in terms of fragmentation of non-urban land. Probably due to a larger availability of financial resources for urban expansion, the creation of new brownfields was even higher than in shrinking cities. These examples of unexpected behaviours question the validity of the simplistic assumptions at the basis of many models applied to predict urban spatial development (Batty, 2009). Causal networks, rather than causal chains, link indicators such as those adopted in this study (Niemeijer and de Groot, 2008), with multiple drivers, some of which outside the spatial planning domain (Seto et al., 2011), contributing to the observed results, and land use transitions connected by feedback loops to their same causes (Nuissl et al., 2009).

Finally, some considerations may be drawn about the applicability of the analysed spatial strategies to the diverse conditions of European cities. Despite the high share of land involved in LULC flows, spatial development in Eastern European cities is the most distant from the strategies elaborated at EU level, probably due to a combination of historical reasons and legacies, planning tradition, and socio-economic factors (Stanilov, 2007). On the opposite side, the spatial development of Western cities is the most coherent with the direction suggested by the strategies, confirming the East-West dichotomy already observed by Turok & Mykhnenko (2007) in a wider time-frame. Overall, the map of the number of strategies matched by the spatial development of each city questions the same objective of territorial cohesion under which the formulation of the strategies was initiated (Faludi, 2004). However, the cluster analysis proves that the region-based classification provides only a partial picture of the similarities and differences in the spatial development trends of European cities. Almost all clusters are crosscutting at least two regions, sometimes highlighting similarities in very distant cities. This brings back to the non-prescriptive status of the strategies and to the mechanisms of their dissemination (Faludi, 2010). National policies and, most of all, the commitment of city administrations determine the uptake of the strategies in planning processes and tools at the local level, hence their effective implementation. Clusters could therefore reflect– at least partly - the map of networking initiatives and cooperation programmes that, since the *Charter of European Cities and Town Towards Sustainability* (Aalborg Charter, 1994)have been the most relevant tool for building a joint vision of sustainable urban development among European cities (Dühr et al., 2007).

5.5.2 About the approach

The analysed sample covers one quarter of the almost 700 European cities with more than 100,000 inhabitants and can therefore be considered representative of the variety of conditions across the EU (Kabisch and Haase, 2013). Nevertheless, the results are affected by some uncertainties related to both classification accuracy (> 85% for urban classes) and resolution, which may lead to an underestimation of land cover classes consisting prevalently of small patches (e.g., urban green areas). It was hypothesized that no further changes happened between 2006 and 2012 than those observed through the comparison of the two reference maps: this may affect the quantification of LULC flows related to fast-changing classes (e.g., industrial areas to brownfields and vice-versa). Also, interventions that do not imply a LULC change could not be detected, which restricts the understanding of *urban regeneration* to the re-use of brownfields and the recycling of urban land. However, despite limitations, LULC flows involving in total around 2% of the land in the analysed cities can be considered representative of the main ongoing trends, and the use of a homogeneous, high-resolution database ensures robust and comparable results (Larondelle et al., 2014).

The spatial and temporal scale of analysis should be kept in mind to avoid misinterpretation of the results. Core cities, i.e. areas within city administrative boundaries, were chosen as spatial reference units. Since core cities are the main target of policy making at the city level, this allows assessing the effectiveness of policies and comparing, and possibly benchmarking, different cities based on a homogeneous institutional scale. Moreover, most international initiatives toward the implementation of spatial strategies are directed at mobilizing city administrations, hence this scale of analysis could help measuring the effectiveness of such international projects and agreements. Nevertheless, certain trends (e.g., compaction vs sprawl) and phenomena (e.g., urban expansion and peri-urbanization) may be only partially captured at this scale, and their full understanding would require a wider perspective considering the entire metropolitan or functional urban area. The same also applies to the limited temporal scale: despite capturing changes in LULC that will have, in any case, a long-lasting effect on urban spatial development, a window of time of six years offers only a partial view on long transformative processes, and is sensitive to the influence of contingent factors. However, drawbacks related to the temporal coverage of the data are compensated by their spatial resolution: at present, in Europe, longer time series are available only for the Corine database, which describes urban areas with a lower level of detail. Here, the use of Urban Atlas data, specifically aimed at differentiating among 17 urban LULC classes, allowed capturing fine-grain LULC flows such as densification processes and fragmentation of green areas. From this perspective, the study should be considered a first testing of the potential of Urban Atlas data for measuring a wide range of urban spatial development indicators.

A set of 15 indicators was adopted to understand the progress toward multiple spatial strategies based on changes in the urban form and the spatial arrangement of land uses. Although many strategies also address qualitative aspects, such as the accessibility and pleasantness of urban green spaces, or the liveability and quality of life of regenerated districts, this study focused on quantitative indicators measurable from LULC and population data. A more complete understanding of the progress toward certain strategies could be achieved by complementing the findings with qualitative indicators, e.g. citizens' opinion about the quality of life, which at present are available only for some cities in Europe (European Commission, 2016b). Following recommendations from the literature (Alberti, 1996; Cushman et al., 2008; Mega and Pedersen, 1998; Schwarz, 2010; United Nations, 2007), The adopted indicators are based on publicly available data, are easy to calculate, and derive from simple conceptual representations of urban systems (Table 5.2). To keep a straightforward relation with decision-making, combined and complex indicators were purposely avoided, although they could be considered more suitable to describe complex phenomena such as urban sprawl (e.g. Jaeger & Schwick (2014)). Instead, the combination of three types of indicators that provide multiple perspectives on the same aspects allowed preventing partial or misleading interpretations, as in the case of compactness (Angel et al., 2011). This combination proved to be useful to interpret urban spatial development. Also, LULC flows appears as a promising way to investigate spatial development trends not only in terms of accounting, as already done by the EEA (2006), but also making use of the spatially-explicit information to analyse specific parts of the city (e.g., peri-urban areas) or LULC classes (e.g., residential areas).

A final remark concerns the approach to assess the overall performance of cities through a representative indicator for each strategy. With the only exception of *no land take*, for which the European Commission set the target of zero net land take to be achieved within 2050 (European Commission, 2011b), no other strategy is expressed in quantitative terms. In the analysis, tentative thresholds that respect coherence across indicator types and significance in relation to the quantities involved were defined. However, the actual meaning of such thresholds when applied to the differentiated conditions of European cities, hence their potential to be used as targets for the strategies, is an issue that goes beyond the purpose of this study and was not further investigated.

5.6 Conclusions

This chapter focused on the EU as an advanced case of definition of spatial strategies for urban development at the international level. Beyond the 'green city' strategy supported by the ES approach, five other main spatial strategies promoted at the EU level were identified. The observation of the recent spatial development trends of a large sample of European cities through the lens of the strategies provided the first pan-European comparison between strategic planning policies and on-the-ground urban development. The sometimes-contrasting trends in the indicators highlight that, despite they all aim at sustainable urban development, the strategies may have drawbacks and even trade-off among each other. Hence the need for looking at urban spatial development from multiple perspectives and for monitoring changes in land use and urban form through multiple indicators. In this view, the list of indicators adopted in the study could be considered a first monitoring scheme that allows measuring the progresses of European cities towards the whole set of common spatial strategies.

A successful implementation of the strategies largely depends on local administrations and communities, hence the analysis considered simple indicators clearly linked with planning decisions. Two publicly-available EU-wide databases proved an useful source of information that allows a consistent measument of the indicators across EU cities.

Benchmarking and cross-city comparison can play a key role in promoting the uptake of the strategies, hence the availability of consistent, robust, and updated data is essential (Morais and Camanho, 2011). Furthermore, mapping clusters of cities that are following similar development trends may stimulate exchange and cooperation, the only tools through which an effective mainstreaming of non-prescriptive strategies may take place.

Quantitative analyses, such as the one here presented, can capture the spatial development trends of cities, but do not tell whether the observed trends are the result of a local uptake of the strategies. Further studies are needed to analyse the content of cities' plans and policies and to interview key stakeholders, in order to understand the actual extent of the integration in decision-making processes at the local scale. Combining the two information would set the basis for an overall assessment of the strategies, of the available pathways for their implementation, including their applicability across different contexts, and, ultimately, of their actual effectiveness in promoting sustainable urban development.

Chapter 6

Synthesis and conclusions

6.1 Summary of the main findings

The objective of the thesis was to explore the integration of ecosystem services (ES) in urban planning practices, based on the hypothesis that the ES approach, grounded on the growing ES science, can be a valuable support to improve decision-making toward sustainable and resilient cities. Contrary to other analyses, the thesis did not considered the ES approach as a decision-making process where ES represent the starting point and the central issue (Verburg et al., 2016): a quite rare case, especially in urban contexts. Rather, the interest was in understanding how, through the ES concept and approach, ecological knowledge can be integrated into decision-making processes that mainly pursue socio-economic goals (Schleyer et al., 2015; von Haaren and Albert, 2011).

'Integrating', as mentioned in the title, is therefore primarily referred to the interface between the ES science and urban planning. The ES science is seen as a provider of credible and relevant knowledge that can complement the set of information - a combination of both scientific and traditional/local knowledge - on which planning decisions are usually based (Ruckelshaus et al., 2015). From an operational point of view, the ES science offers a wide range of methods and tools that can be used by planners to analyse the current condition and to assess the expected impacts of planning decisions (Grêt-Regamey et al., 2017; Harrison et al., 2017). Particularly relevant for planning applications are spatially-explicit methods that allow mapping the distribution of ES and related benefits across the city (Martínez-Harms and Balvanera, 2012; Wolff et al., 2015).

Regarding urban planning, acknowledging that the ES science has devoted little attention to understanding the specificities of this decision-making process, one of the main efforts of the thesis has been to also characterize this side of the interface. Urban planning represents a specific category of decision-making processes addressing the spatial arrangement of land uses and functions in cities. In terms of contents, urban planning is characterized by specific goals, a range of possible actions, and a defined set of implementation tools. In terms of process, specific categories of decision-makers, stakeholders, and power relations among them characterize urban planning processes, with times and modes of interaction often partly regulated by law. As a discipline, urban planning integrates inputs from different fields and sectors into a well-established disciplinary knowledge, whose concepts, approaches, and methods should not be overlooked when considering the interface with other knowledge branches.

Given these premises, the thesis was structured around four specific objectives. Chapter 2 was dedicated to investigating the state of the art by reviewing a set of recent urban plans. The aim here was to understand what ES information is already used in current plans and how a further integration of the ES approach could improve planning decisions. Considering the contents rather than the terminology, a high number and a great variety of actions addressing urban ES emerged, with a corresponding large set of tools for implementation. Current plans acknowledge that many urban challenges, especially related to environmental problems, can be tackled through ecosystem-based approaches. Recreation and a set of key regulating ES (i.e. water flow regulation and runoff mitigation, air purification, urban temperature regulation, and noise reduction) are widely addressed. However, the information base that supports planning actions is poor, especially for what regards regulating ES. Available scientific knowledge is only partially transferred to planning practices, and usable methods for the

ES assessment of urban ES are still lacking. Overall, a pragmatic and heuristic approach to ES prevails. Moreover, while the supply of ES is enhanced through targeted actions bringing evident benefits, ES are rarely mentioned in the strategic part of the plans. Probably, this is also due to the lack of analysis of demand and beneficiaries, which makes it difficult to link ES targets and objectives to expected outcomes in terms of human wellbeing.

Chapter 3 was aimed at enhancing the usability of scientific findings on urban ES for urban planners, thus supporting their operationalization. Moving from the results of the review, the chapter focused on urban regulating ES and addressed two specific barriers that prevent their integration in planning processes: the complexity of their biophysical foundations and the lack of indicators to explicitly account for the demand. The chapter introduces an own framework built by combining, detailing, and adjusting two approaches with demonstrated applicability to planning contexts: the Cascade conceptual model (Potschin-Young et al., 2017) and the supply-demand approach for mapping ES (Burkhard et al., 2012). The framework breaks down the complex process of urban regulating ES provision and use, and describes the main elements involved and their interactions. Moreover, it identifies the two entry points and respective pathways through which planning decisions affect the provision of urban regulating ES and associated benefits in cities: either acting on the supply side, by improving urban green infrastructure availability and efficiency, or acting on the demand side, by enabling people to more effectively benefit from ES. The role of environmental conditions and their effects on both the supply and the demand are made explicit, and a 'parallel cascade' is drawn on the demand side to clarify the link between vulnerability to environmental conditions and demand for urban regulating ES.

The framework was applied as an 'organizing structure' to distil and systematize a wide but fragmented scientific evidence on seven urban regulating ES. Planning-relevant knowledge, methods, and indicators were collected and organized according to the main components of the framework, thus guiding planners in their operationalization. The gathered information allows a better understanding of the most relevant variables that determine the effects of planning actions, hence of the pathways that can be expected to produce the highest benefits. A key issue emerged from the overlaps among multiple urban regulating ES that were identified both in the supply side, due to the multifunctionality of urban green infrastructure components, and in the demand side, due to the vulnerability of ES beneficiaries to various environmental conditions. These overlaps represent opportunities for planners to exploit and promote synergies across different regulating ES. The spatial dimension also emerged as a fundamental factor to consider, thus confirming the crucial role of spatially-explicit indicators in informing planning decisions. Overall, the framework makes planners aware of the socio-ecological processes behind urban regulating ES and of the levers on which they can act to enhance their provision, ultimately supporting the design of effective planning actions and the assessment of their impacts on both the supply and the demand of urban regulating ES.

Chapter 4 explored the integration of the ES approach to support a real-life planning decision. Among the different roles that ES knowledge can play in the different stages of the planning process, the chapter focused on the use of ES assessments as criteria to evaluate future planning scenarios. This use poses specific requirements, including the capacity of assessing the consequences of small-scale planning interventions on multiple ES, considering variations in both the supply and the demand, and explicitly addressing potential trade-offs among different ES and competing land uses. The illustrative application addressed the prioritization of planning interventions in the city of Trento. Two ES of critical importance for the city, one regulating and one cultural ES, were assessed in the current condition and under the planning scenarios that foresee the conversion of alternative brownfield sites into new public green areas. The benefits of the interventions in terms of improved cooling effect by vegetation during the hot season and enhanced opportunities for nature-based recreation were quantified considering the number of expected beneficiaries broken down into different vulnerability classes, and then compared through a multi-criteria analysis. The application demonstrated the potential of beneficiary-based indicators, coherent with planning objectives, to integrate ES knowledge in the assessment of planning decisions. Multi-criteria analysis proved to be a useful platform for integrating different information about costs and benefits of planning scenarios, exploring diverse stakeholder perspectives, and balancing the trade-off between the enhancement of green infrastructure and other competing objectives.

Finally, Chapter 5 was aimed at framing the integration of ES in urban planning in the wider context of existing strategies for sustainable spatial development. The study focused on the European Union, arguably the most advanced case of definition of common spatial strategies to be pursued in cities with different historic backgrounds, planning traditions, economic and social conditions, as well as current and expected development trends. Through the analysis of policy documents, beyond the ‘green city’ strategy supported by the ES approach, other five spatial strategies were identified, namely, ‘compact city’, ‘urban regeneration’, ‘functional mix’, ‘no land take’, and ‘high density’. To capture potential synergies and trade-offs among the strategies, the recent development trends of 175 EU cities was analysed and compared with the strategies through a set of indicators based on land use-land cover data. A differentiated panorama emerged across the EU, with certain trends characterizing almost homogeneously the whole sample, and others limited to specific categories of cities. Contrasting trends in the indicators highlight that the strategies may have drawbacks and even trade-off among each other, as in the case of ‘green city’ and ‘no land take’. Thus, no single strategy can be considered a panacea for cities’ sustainability, and multiple perspectives, hence multiple indicators, are needed to assess true progresses towards more sustainable development models.

6.2 Challenges for future research and planning

6.2.1 How to approach the interface

As briefly presented in the Introduction (see Section 1.2), several challenges characterize the integration of ES in urban planning from both sides of the science-policy interface. The thesis explored some of them, highlighting what opportunities exist and what further steps still need to be taken in both the ES science and the planning domains. Indeed, the same approach that the thesis tried to apply appears still challenging for research, and this investigation provided only some insights on the results that can be achieved. Despite the explicit link with decision-making being among its distinctive traits since the beginning (Daily et al., 2009), so far, ES science has mostly approached the issue of integration from its disciplinary perspective. Planning practices have been mostly interpreted as 'empty boxes', with limited focus on the interaction with established planning processes and a scarce interest on how the planning discipline, with its wealth of knowledge, concepts, and methods, may contribute to the integration. On the contrary, the 'salience' or 'relevance' of ES knowledge, a key attribute to measure its capacity to inform planning decisions (Cash et al., 2003), depends on the specific needs that are not yet answered by existing knowledge and approaches within the planning discipline. In this respect, the case of recreation in Chapter 2 is exemplary: while ES science is struggling to develop appropriate indicators to measure cultural ES, urban planning seems already well-equipped to account for them.

Studies investigating the objectives and modes of use of ES knowledge (Mckenzie et al., 2014; Posner et al., 2016a) and exploring how planners and decision-makers perceive its relevance (Albert et al., 2014b; Beery et al., 2016) are a good starting point. However, very few have specifically focused on urban planning. Even the cases in which the integration of ES knowledge has been tested are often included in studies about decision-making processes in general (Dick et al., 2017; Dunford et al., 2017; Ruckelshaus et al., 2015), failing to capture the specificities of urban planning. This thesis tried to go further in characterizing the urban planning side of the interface, to understand its specific needs and the requirements that it poses to ES knowledge. This was done by reviewing a set of recent urban plans (Chapter 2), by conceptually framing the relationships between planning actions and ES (Chapter 3), and by testing the application of ES assessments in the specific stage of the planning process where decisions among alternative scenarios are to be made (Chapter 4). The results highlight shortcomings in how the current planning practices address ES and suggest some entry points along the planning process where ES knowledge could be successfully integrated. However, the cases that could be analysed are limited in number and geographical coverage, and urban planning processes partially vary across countries in terms of objectives, stakeholders, tools, and possible actions, while different planning traditions still influence how planners approach certain issues and the interface with other disciplines. More studies are needed to understand how this variability affect the knowledge needs and the requirements that urban planning processes present to ES science.

Possible contributions of the urban planning discipline to the ES science also emerged from the thesis. Indeed, the urban planning discipline is accustomed to addressing the overlaps between the socio-economic and the ecological spheres, precisely where ES originate. While ES studies are still unbalanced towards ecological analyses, methods and approaches already in use by planners may support a better focus on the demand side of ES. This includes the identification of actual and potential beneficiaries, the quantification of ES benefits, and the inclusion of concerns related to equity in ES provision, as it emerged from both Chapter 3 and Chapter 4. Another key contribution from planning is represented by the implementation tools that could support the operationalization of ES knowledge and approach. Apart from ‘payment-for-ES’ schemes, the ES science has developed very few models for implementation. On the contrary, as demonstrated in Chapter 2, urban planners are equipped with a large toolbox where the right tools to implement ES-informed decisions and to secure and enhance the provision of ES in cities can certainly be found, including regulations, building standards, financial and non-financial incentives, among others (see Chapter 2). Overall, a strong potential can be expected to develop from a further integration - and cross-fertilization - of ES science and urban planning, but efforts are still needed to frame this integration within a more balanced relation.

6.2.2 Multiple levels of integration

‘Integration’ is a keyword for ES science. Addressing ES requires integrating multiple fields of knowledge, and multiple levels of integration must be fulfilled before ES knowledge is perceived as relevant for urban planning processes. First, the different values associated to ES must be integrated. Most ES methods and analyses are focused on quantifying biophysical variables, whose knowledge is needed to characterize the ecosystem functions and processes that support ES supply. However, biophysical indicators are not sufficient to communicate ES values in a meaningful way. This is especially true for decision-making processes driven by social concerns, as also suggested by the absence of almost any biophysical quantification in the information base of the reviewed urban plans (Chapter 2). Other indicators are needed to describe ES across the different stages of the Cascade. Some of these indicators are presented in Chapter 3, but they are only few and illustrative, and their applicability to real-life decision-making processes is still to be tested by planners and decision-makers. At the same time, scientific advancements produced by transdisciplinary efforts are needed to define appropriate and effective methods that allow combining the results of biophysical, socio-cultural, and economic methods, thus accounting for the multiple values of ES.

Actually, as far as strategic decisions are concerned, economic indicators seem not so relevant from the urban planning perspective, although possible uses for awareness-rising and in the cost-benefit analysis of planning actions may be envisioned. As demonstrated by the application described in Chapter 4, indicators describing the social values of ES by accounting for their beneficiaries and the different levels of demand that they express have the capacity to reflect different planning objectives and stakeholders' perspectives, hence to inform and support planning decisions. However, within the ES science, social methods to analyse the demand for ES are only few, and several authors acknowledge that their development deserves more attention (Chan et al., 2016; Haase et al., 2014b; La Rosa et al., 2015). ES science could take advantage of approaches commonly adopted by urban planning (e.g., in the spatial analysis of population, in the identification of specific target groups, and in the elicitation of citizens' preferences and opinions), and further develop their use, today almost exclusively limited to the analysis of the recreational value of green spaces (see Chapter 2). This probably represents the most promising field of cross-fertilization between ES science and the urban planning discipline. On the other hand, some shortcomings also emerge in biophysical methods. Urban contexts, characterized by heterogeneity and fragmentation of green infrastructure components, define specific requirements to biophysical methods in terms of accuracy and resolution. At the same time, their integration in planning processes determines a limited availability of data, as well as resource constraints (e.g., in terms of time, costs, and expertise), often stricter compared to scientific applications. Chapter 4 tested two methods specifically designed (or adjusted, as in the case of ESTIMAP-recreation) to support urban planning applications. But this is not a common case, and biophysical methods to assess many urban ES require further efforts to be tailored to urban planning needs.

The second level of integration that appears necessary to make ES information relevant for urban planning is the integration across different ES. Urban green infrastructure components act as providing units of multiple ES (see Chapter 3), hence planning actions can be expected to produce effects on a bundle of ES. This is not only the case of planning actions specifically aimed at enhancing ES provision (i.e., ecosystem-based actions and nature-based solutions) or directly affecting urban green infrastructure. As shown by the framework presented in Chapter 3 for urban regulating ES, but equally valid for the other urban ES, also planning actions affecting the distribution of beneficiaries, the environmental conditions of the city, or the accessibility to certain areas have indirect effects on urban ES and related benefits. While the multifunctionality of urban green infrastructure generates potential synergies, which are indicated as one of the main strengths of ecosystem-based actions, looking at the whole range of ES affected by planning actions may reveal unexpected and undesired outcomes.

However, accounting for multiple ES is still a challenge for ES assessments. From the scientific point of view, even if robust and usable methods are available to quantify each ES (which is not always the case), finding common indicators able to support an overall assessment is not an easy task. ES assessment must thus be pushed towards the last stages of the ES cascade, where benefits and associated values are expressed. Monetary indicators offer a solution, but it should not be considered the only one. In fact, beneficiary-based and 'benefit-relevant' indicators (Olander et al., 2018) appear as promising ways to respond to the need. Chapter 3 provides a list of indicators, alternative to monetary valuations, that can be adopted to account for demand and benefits of urban regulating ES. Moreover, Chapter 4 experimented with a quantification of ES benefits based on the analysis of population distribution and on the identification of vulnerable groups. Such indicators have a clear meaning for planning and allow comparing quantitative results related to different ES. But further research is needed to test their usability and capacity to inform real-life planning processes, investigating their strengths and weaknesses in relation to different decision-making contexts.

Combining values associated to different ES is also a difficult task. A methodological support is provided by multi-criteria analysis techniques, which offer a platform for combining multiple value dimensions, integrating stakeholders' opinions along with technical inputs. Multi-criteria analysis allows exploring different stakeholders' perspectives and balancing competing interests to find an agreed-upon solution. Urban planning would benefit from adopting such methodologies, which enhance participation and transparency, ultimately strengthening the ownership of the results. However, a pre-requisite is the identification of clear ES-related objectives, rarely done in current planning practices, as shown in Chapter 2. Increasing ES supply is not equal to increasing the number of ES beneficiaries, which again is not the equal to maximising ES benefits produced by planning actions. Consequently, only a clear definition of planning objectives can set the basis for the design of effective planning actions. Furthermore, formulating objectives and targets for ES provision helps to identify the values against which the effectiveness of planning actions should be measured, hence to clarify the possible role(s) of ES knowledge within the planning process.

6.3 Concluding remarks

Going back to the original meaning of integration in the title, the thesis provided some insights on both the 'what' and the 'how' of such integration. What ES knowledge and information could support the design and assessment of planning actions, ultimately producing better decisions, was explored in Chapter 2, with reference to the level of integration in current planning practices, and in Chapter 3, where planning-relevant knowledge on urban regulating ES was distilled from a wide scientific literature. Chapter 3 also tried to clarify how such integration may take place, by identifying the entry points for planning to affect urban ES, hence providing a conceptual guidance on the role of ES knowledge in steering planning actions. The operational side of the integration was supported by providing a set of illustrative indicators that describe all the elements involved in the provision of urban regulating ES and related benefits (Chapter 3), and by presenting the results of the use of ES assessments to inform a real-life planning decision (Chapter 4).

However, the crucial issue is the ‘what for’ of such integration. Although the effectiveness of the ES concept and approach in innovating the current planning practices is still largely to be demonstrated, and real-world case studies are needed to test on the ground the benefits of their uptake, expectations are many. The ES approach certainly has the potential for promoting sustainable planning strategies related to the conservation and enhancement of urban green infrastructure. Nevertheless, such strategies are not the only ones in the panorama of spatial strategies for sustainable urban development, and they should not be expected to produce, alone, more sustainable and resilient cities. Understanding potential synergies and trade-offs among planning strategies, together with the specific local conditions that may foster or hamper their implementation, is necessary to ensure that expectations are met. Little is known about these complex issues, which require looking at the development trends of different cities under multiple perspectives, as tentatively done in Chapter 5. Further research is needed to compare the ground-truth of spatial development with the planning strategies that are pursued, considering the actions and the tools through which they are implemented.

A growing demand for ES knowledge to be integrated in urban planning practices is determined by the strong support that ecosystem-based actions and nature-based solutions are receiving. As it emerged from the presented exploration, the ES approach, providing a holistic framework that describes the multiple relations between ecosystems and human wellbeing, offers to urban planning much more than solutions. Within this framework, objectives that account for the complex interactions between the ecological and the socio-economic spheres can be set, and decisions assessed based on their expected long-term consequences. Urban planning plays a key role in coordinating different sectoral policies and bridging multiple institutional scales (Rozas-Vásquez et al., 2018). Provided that local and short-term perspectives are overcome, and that the complexity of ES-related decisions (Jax et al., 2017) and the consequent need for vertical and horizontal integration (Schleyer et al., 2015) are acknowledged, urban planning can become the starting point from which ES knowledge permeates other decision-making processes. While urbanization is one of the major threats to ES worldwide, promoting the ES approach through urban planning may seem a paradox. But it is also a great opportunity to make human development truly sustainable.

References

Aalborg Charter, 1994. Charter of European Cities & Towns Towards Sustainability.
Abson, D.J., von Wehrden, H., Baumgärtner, S., …, Walmsley, D., 2014. Ecosystem services as a boundary object for sustainability. Ecol. Econ. 103, 29–37. doi:10.1016/j.ecolecon.2014.04.012
Adem Esmail, B., Geneletti, D., 2018. Multi-criteria decision analysis for nature conservation: A review of 20 years of applications. Methods Ecol. Evol. 2018, 42–53. doi:10.1111/2041-210X.12899
Adem Esmail, B., Geneletti, D., 2017. Design and impact assessment of watershed investments: An approach based on ecosystem services and boundary work. Environ. Impact Assess. Rev. 62, 1–13. doi:10.1016/j.eiar.2016.08.001
Adem Esmail, B., Geneletti, D., Albert, C., 2017. Boundary work for implementing adaptive management: A water sector application. Sci. Total Environ. 593–594, 274–285. doi:10.1016/j.scitotenv.2017.03.121
Albert, C., Aronson, J., Fürst, C., Opdam, P., 2014a. Integrating ecosystem services in landscape planning: requirements, approaches, and impacts. Landsc. Ecol. 29, 1277–1285. doi:10.1007/s10980-014-0085-0
Albert, C., Galler, C., Hermes, J., Neuendorf, F., von Haaren, C., Lovett, A., 2015. Applying ecosystem services indicators in landscape planning and management: The ES-in-Planning framework. Ecol. Indic. 61, 100–113. doi:10.1016/j.ecolind.2015.03.029
Albert, C., Hauck, J., Buhr, N., von Haaren, C., 2014b. What ecosystem services information do users want? Investigating interests and requirements among landscape and regional planners in Germany. Landsc. Ecol. 1–13. doi:10.1007/s10980-014-9990-5
Albert, C., Spangenberg, J.H., Schroeter, B., 2017. Nature-based solutions: criteria. Nature 543, 315. doi:10.1038/543315b
Alberti, M., 1999. Urban patterns and environmental performance: What Do We Know? J. Plan. Educ. Res. 19, 151–163. doi:10.1177/0739456X9901900205
Alberti, M., 1996. Measuring urban sustainability. Environ. Impact Assess. Rev. 16, 381–424. doi:10.1016/S0195-9255(96)00083-2
Amati, M., Taylor, L., 2010. From Green Belts to Green Infrastructure. Plan. Pract. Res. 25, 143–155. doi:10.1080/02697451003740122
Andersson, E., McPhearson, T., Kremer, P., Haase, D., Tuvendal, M., Wurster, D., 2015. Scale and context dependence of ecosystem service providing units. Ecosyst. Serv. 12, 157–164. doi:10.1016/j.ecoser.2014.08.001
Angel, S., Parent, J., Civco, D.L., Blei, A.M., 2011. Making room for a planet of cities. Cambridge MA. doi:10.4337/9781849808057.00023
Atkinson, R., 2001. The Emerging "Urban Agenda" and the European Spatial Development Perspective: Towards an EU Urban Policy? Eur. Plan. Stud. 9, 385–406. doi:10.1080/09654310120037630
Aylor, D., 1972. Noise Reduction by Vegetation and Ground. J. Acoust. Soc. Am. 51, 197. doi:10.1121/1.1912830
Bagstad, K.J., Johnson, G.W., Voigt, B., Villa, F., 2013. Spatial dynamics of ecosystem service flows: A comprehensive approach to quantifying actual services. Ecosyst. Serv. 4, 117–125. doi:10.1016/j.ecoser.2012.07.012
Bagstad, K.J., Villa, F., Batker, D., Harrison-Cox, J., Voigt, B., Johnson, G.W., 2014. From theoretical to actual ecosystem services: Mapping beneficiaries and spatial flows in ecosystem service assessments. Ecol. Soc. 19. doi:10.5751/ES-06523-190264

Baker, I., Peterson, A., Brown, G., McAlpine, C., 2012. Local government response to the impacts of climate change: An evaluation of local climate adaptation plans. Landsc. Urban Plan. 107, 127–136. doi:10.1016/j.landurbplan.2012.05.009
Barbier, E.B., Koch, E.W., Silliman, B.R., ..., Reed, D.J., 2008. Coastal Ecosystem – Based Ecological Functions and Values. Science (80-.). 319, 321–323. doi:10.1126/science.1150349
Barbosa, A.E., Fernandes, J.N., David, L.M., 2012. Key issues for sustainable urban stormwater management. Water Res. 46, 6787–6798. doi:10.1016/j.watres.2012.05.029
Barbosa, O., Tratalos, J.A., Armsworth, P.R., ..., Gaston, K.J., 2007. Who benefits from access to green space? A case study from Sheffield, UK. Landsc. Urban Plan. 83, 187–195. doi:10.1016/j.landurbplan.2007.04.004
Barnett, C., Parnell, S., 2016. Ideas, implementation and indicators: epistemologies of the post-2015 urban agenda. Environ. Urban. 28, 87–98. doi:10.1177/0956247815621473
Baró, F., Chaparro, L., Gómez-Baggethun, E., Langemeyer, J., 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
Baró, F., Haase, D., Gómez-Baggethun, E., Frantzeskaki, N., 2015. Mismatches between ecosystem services supply and demand in urban areas: A quantitative assessment in five European cities. Ecol. Indic. 55, 146–158. doi:10.1016/j.ecolind.2015.03.013
Baró, F., Palomo, I., Zulian, G., Vizcaino, P., Haase, D., Gómez-Baggethun, E., 2016. Mapping ecosystem service capacity, flow and demand for landscape and urban planning: A case study in the Barcelona metropolitan region. Land use policy 57, 405–417. doi:10.1016/j.landusepol.2016.06.006
Bartesaghi Koc, C., Osmond, P., Peters, A., 2016. Towards a comprehensive green infrastructure typology: a systematic review of approaches, methods and typologies. Urban Ecosyst. 1–21. doi:10.1007/s11252-016-0578-5
Barton, D.N., Kelemen, E., Dick, J., ..., Lapola, D.M., 2017. (Dis) integrated valuation – Assessing the information gaps in ecosystem service appraisals for governance support. Ecosyst. Serv. doi:https://doi.org/10.1016/j.ecoser.2017.10.021
Bassett, E., Shandas, V., 2010. Innovation and Climate Action Planning. J. Am. Plan. Assoc. 76, 435–450. doi:10.1080/01944363.2010.509703
Bastian, O., Haase, D., Grunewald, K., 2012. Ecosystem properties, potentials and services – The EPPS conceptual framework and an urban application example. Ecol. Indic. 21, 7–16. doi:10.1016/j.ecolind.2011.03.014
Basu, R., Samet, J.M., 2002. Relation between elevated ambient temperature and mortality: A review of the epidemiologic evidence. Epidemiol. Rev. 24, 190–202. doi:10.1093/epirev/mxf007
Batty, M., 2009. Urban Modeling, in: Thrift, N., Kitchin, R. (Eds.), International Encyclopedia of Human Geography. Elsevier, Oxford, UK, pp. 51–58. doi:10.1016/B978-008044910-4.01092-0
Bauler, T., Pipart, N., 2013. Ecosystem Services in Belgian Environmental Policy Making: Expectations and Challenges Linked to the Conceptualization and Valuation of Ecosystem Services, Ecosystem Services: Global Issues, Local Practices. Elsevier. doi:10.1016/B978-0-12-419964-4.00012-3
Baumgardner, D., Varela, S., Escobedo, F.J., Chacalo, A., Ochoa, C., 2012. The role of a peri-urban forest on air quality improvement in the Mexico City megalopolis. Environ. Pollut. 163, 174–83. doi:10.1016/j.envpol.2011.12.016
Baynham, M., Stevens, M.R., 2013. Are we planning effectively for climate change? An evaluation of official community plans in British Columbia. J. Environ.

Plan. Manag. 57, 557–587. doi:10.1080/09640568.2012.756805
Beames, A., Broekx, S., Schneidewind, U., ..., Seuntjens, P., 2018. Amenity proximity analysis for sustainable brownfield redevelopment planning. Landsc. Urban Plan. 171, 68–79. doi:10.1016/j.landurbplan.2017.12.003
Beery, T., Stålhammar, S., Jönsson, K.I., ..., Schubert, P., 2016. Perceptions of the ecosystem services concept: Opportunities and challenges in the Swedish municipal context. Ecosyst. Serv. 17, 123–130. doi:10.1016/j.ecoser.2015.12.002
Bennett, E.M., Chaplin-Kramer, R., 2016. Science for the sustainable use of ecosystem services. F1000Research 5, 2622. doi:10.12688/f1000research.9470.1
Berke, P.R., Conroy, M.M., 2000. Are We Planning for Sustainable Development? J. Am. Plan. Assoc. 66, 21–33. doi:10.1080/01944360008976081
Berke, P.R., Godschalk, D., 2009. Searching for the Good Plan: A Meta-Analysis of Plan Quality Studies. J. Plan. Lit. 23, 227–240. doi:10.1177/0885412208327014
Blackwell, M.S.A., Maltby, E., 2006. How To Use Floodplains for Flood Risk Reduction.
Bodnaruk, E.W., Kroll, C.N., Yang, Y., Hirabayashi, S., Nowak, D.J., Endreny, T.A., 2017. Where to plant urban trees? A spatially explicit methodology to explore ecosystem service tradeoffs. Landsc. Urban Plan. 157, 457–467. doi:10.1016/j.landurbplan.2016.08.016
Bolund, P., Hunhammar, S., 1999. Ecosystem services in urban areas. Ecol. Econ. 29, 293–301. doi:10.1016/S0921-8009(99)00013-0
Botzat, A., Fischer, L.K., Kowarik, I., 2016. Unexploited opportunities in understanding liveable and biodiverse cities. A review on urban biodiversity perception and valuation. Glob. Environ. Chang. 39, 220–233. doi:10.1016/j.gloenvcha.2016.04.008
Bouwma, I., Schleyer, C., Primmer, E., Winkler, K.J., Berry, P., Young, J., Carmen, E., Špulerová, J., Bezák, P., Preda, E., Vadineanu, A., 2017. Adoption of the ecosystem services concept in EU policies. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.02.014
Bowler, D.E., Buyung-Ali, L.M., Knight, T.M., Pullin, A.S., 2010. A systematic review of evidence for the added benefits to health of exposure to natural environments. BMC Public Health 10, 456. doi:10.1186/1471-2458-10-456
Braat, L., de Groot, R., 2012. The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosyst. Serv. 1, 4–15. doi:10.1016/j.ecoser.2012.07.011
Bragagnolo, C., Geneletti, D., Fischer, T.B., 2012. Cumulative effects in SEA of spatial plans – evidence from Italy and England. Impact Assess. Proj. Apprais. 30, 100–110. doi:10.1080/14615517.2012.677522
Braquinho, C., Cvejić, R., Eler, K., ..., Železnikar, Š., 2017. A typology of urban green spaces, ecosystem provisioning services and demands.
Breaux, A., Farber, S., Day, J., 1995. Using Natural Coastal Wetlands Systems for Wastewater Treatment: An Economic Benefit Analysis. J. Environ. Manage. 44, 285–291. doi:10.1006/jema.1995.0046
Brink, E., Aalders, T., Ádám, D., ..., Wamsler, C., 2016. Cascades of green: A review of ecosystem-based adaptation in urban areas. Glob. Environ. Chang. 36, 111–123. doi:10.1016/j.gloenvcha.2015.11.003
Brody, S.D., 2003. Measuring the Effects of Stakeholder Participation on the Quality of Local Plans Based on the Principles of Collaborative Ecosystem Management. J. Plan. Educ. Res. 22, 407–419. doi:10.1177/0739456X03022004007
Brody, S.D., Highfield, W., Carrasco, V., 2004. Measuring the collective planning

capabilities of local jurisdictions to manage ecological systems in southern Florida. Landsc. Urban Plan. 69, 33–50. doi:10.1016/j.landurbplan.2003.09.002
Broitman, D., Koomen, E., 2015. Residential density change: Densification and urban expansion. Comput. Environ. Urban Syst. 54, 32–46. doi:10.1016/j.compenvurbsys.2015.05.006
Burkhard, B., Crossman, N.D., Nedkov, S., Petz, K., Alkemade, R., 2013. Mapping and modelling ecosystem services for science, policy and practice. Ecosyst. Serv. 4, 1–3. doi:10.1016/j.ecoser.2013.04.005
Burkhard, B., Kandziora, M., Hou, Y., Müller, F., 2014. Ecosystem service potentials, flows and demands-concepts for spatial localisation, indication and quantification. Landsc. Online 34, 1–32. doi:10.3097/LO.201434
Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecol. Indic. 21, 17–29. doi:10.1016/j.ecolind.2011.06.019
Burkhard, B., Maes, J., 2017. Mapping Ecosystem Services, Advanced Books. doi:10.3897/ab.e12837
Burkhard, B., Santos-Martin, F., Nedkov, S., Maes, J., 2018. An operational framework for integrated Mapping and Assessment of Ecosystems and their Services (MAES). One Ecosyst. 3, e22831. doi:10.3897/oneeco.3.e22831
Cadenasso, M.L., Pickett, S.T.A., Schwarz, K., 2007. Spatial Heterogeneity in Urban Ecosystems: Reconceptualizing Land Cover and a Framework for Classification. Front. Ecol. Environ. 5, 80–88.
Camagni, R., Gibelli, M.C., Rigamonti, P., 2002. Urban mobility and urban form: The social and environmental costs of different patterns of urban expansion. Ecol. Econ. 40, 199–216. doi:10.1016/S0921-8009(01)00254-3
Cameron, S., 2003. Gentrification, Housing Redifferentiation and Urban Regeneration: "Going for Growth" in Newcastle upon Tyne. Urban Stud. 40, 2367–2382. doi:10.1080/0042098032000136110
Cao, X., Onishi, A., Chen, J., Imura, H., 2010. Quantifying the cool island intensity of urban parks using ASTER and IKONOS data. Landsc. Urban Plan. 96, 224–231. doi:10.1016/j.landurbplan.2010.03.008
Carleton, J.N., Grizzard, T.J., Godrej, A.N., Post, H.E., 2001. Factors affecting the performance of stormwater treatment wetlands. Water Res. 35, 1552–1562. doi:10.1016/S0043-1354(00)00416-4
Carpenter, S.R., Mooney, H.A., Agard, J., ..., Whyte, A., 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proc. Natl. Acad. Sci. 106, 1305–1312. doi:10.1073/pnas.0808772106
Cash, D.W., Clark, W.C., Alcock, F., ..., Mitchell, R.B., 2003. Knowledge systems for sustainable development. Proc. Natl. Acad. Sci. U. S. A. 100, 8086–8091. doi:10.1073/pnas.1231332100
CBD, 2011. Strategic Plan for Biodiversity 2011–2020, Including Aichi Biodiversity Targets. Montreal.
Chan, K.M.A., Balvanera, P., Benessaiah, K., ..., Turner, N., 2016. Why protect nature? Rethinking values and the environment. Proc. Natl. Acad. Sci. 113, 1462–1465. doi:10.1073/pnas.1525002113
Chan, K.M.A., Guerry, A.D., Balvanera, P., ..., Hannahs, N., 2012. Where are Cultural and Social in Ecosystem Services? A Framework for Constructive Engagement. Bioscience 62, 744–756. doi:10.1525/bio.2012.62.8.7
Clar, M.L., Barfield, B.J., O'Connor, T.P., 2004a. Stormwater Best Management Practice Design Guide: Volume 1 General Considerations. Cincinnati, OH.
Clar, M.L., Barfield, B.J., O'Connor, T.P., 2004b. Stormwater Best Management Practice Design Guide: Volume 2 Vegetative Biofilters. Cincinnati, OH.
Clifton, K., Ewing, R., Knaap, G.-J., Song, Y., 2008. Quantitative analysis of urban form: a multidisciplinary review. J. Urban. Int. Res. Placemaking Urban

Sustain. 1, 17–45. doi:10.1080/17549170801903496
Collier, M.J., 2014. Novel ecosystems and the emergence of cultural ecosystem services. Ecosyst. Serv. 9, 166–169. doi:10.1016/j.ecoser.2014.06.002
Collier, M.J., Nedović-Budić, Z., Aerts, J., ..., Verburg, P., 2013. Transitioning to resilience and sustainability in urban communities. Cities 32. doi:10.1016/j.cities.2013.03.010
Commission of the European Communities, 1997. The EU Compendium of Spatial Planning Systems and Policies. Office for Official Publications of the European Communities, Luxembourg.
Conti, S., Meli, P., Minelli, G., ..., Perini, L., 2005. Epidemiologic study of mortality during the Summer 2003 heat wave in Italy. Environ. Res. 98, 390–399. doi:10.1016/j.envres.2004.10.009
Costanza, R., D'Age, R., de Groot, R., ..., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.
Couch, C., Karecha, J., Nuissl, H., Rink, D., 2005. Decline and sprawl: an evolving type of urban development – observed in Liverpool and Leipzig. Eur. Plan. Stud. 13, 117–136. doi:10.1080/0965431042000312433
Coutts, A.M., Tapper, N.J., Beringer, J., Loughnan, M., Demuzere, M., 2012. Watering our cities: The capacity for Water Sensitive Urban Design to support urban cooling and improve human thermal comfort in the Australian context. Prog. Phys. Geogr. 37, 2–28. doi:10.1177/0309133312461032
Cowling, R.M., Egoh, B., Knight, A.T., ..., Wilhelm-Rechman, A., 2008. An operational model for mainstreaming ecosystem services for implementation. Proc. Natl. Acad. Sci. 105, 9483–9488. doi:10.1073/pnas.0706559105
Crouzat, E., Mouchet, M.A., Gos, P., Byczek, C., Lavorel, S., 2014. An interdisciplinary methodological guide for quantifying associations between ecosystem services ´ ne. Glob. Environ. Chang. 28, 298–308. doi:10.1016/j.gloenvcha.2014.07.012
Curtis, P.S., Wang, X., 1998. A meta-analysis of elevated CO2 effects on woody plant mass, form, and physiology. Oecologia 113, 299–313.
Cushman, S.A., McGarigal, K., Neel, M.C., 2008. Parsimony in landscape metrics: Strength, universality, and consistency. Ecol. Indic. 8, 691–703. doi:10.1016/j.ecolind.2007.12.002
D'Ippoliti, D., Michelozzi, P., Marino, C., ..., Perucci, C.A., 2010. The impact of heat waves on mortality in 9 European cities: results from the EuroHEAT project. Environ. Heal. 9, 37. doi:10.1186/1476-069X-9-37
Daily, G.C., 1997. Nature's Services. Societal Dependence On Natural Ecosystems. Island Press, Washington, DC.
Daily, G.C., Polasky, S., Goldstein, J., ..., Shallenberger, R., 2009. Ecosystem services in decision making: Time to deliver. Front. Ecol. Environ. 7, 21–28. doi:10.1890/080025
Danielsen, F., Sørensen, M.K., Olwig, M.F., ..., Suryadiputra, N., 2005. The Asian tsunami: a protective role for coastal vegetation. Science (80-.). 310, 643. doi:10.1126/science.1118387
Das, S., Vincent, J.R., 2009. Mangroves protected villages and reduced death toll during Indian super cyclone. Proc. Natl. Acad. Sci. 106, 7357–7360. doi:10.1073/pnas.0810440106
Davies, H.J., Doick, K.J., Hudson, M.D., Schreckenberg, K., 2017. Challenges for tree officers to enhance the provision of regulating ecosystem services from urban forests. Environ. Res. 156, 97–107. doi:10.1016/j.envres.2017.03.020
Davies, Z.G., Edmondson, J.L., Heinemeyer, A., Leake, J.R., Gaston, K.J., 2011. Mapping an urban ecosystem service: Quantifying above-ground carbon storage at a city-wide scale. J. Appl. Ecol. 48, 1125–1134. doi:10.1111/j.1365-2664.2011.02021.x

de Groot, R., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010a. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Complex. 7, 260–272. doi:10.1016/j.ecocom.2009.10.006
de Groot, R., Fisher, B., Christie, M., ..., Ring, I., 2010b. Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation, in: Kumar, P. (Ed.), The Economics of Ecosystems and Biodiversity (TEEB): Ecological and Economic Foundations. Earthscan, London, pp. 4–90. doi:10.4324/9781849775489
Demuzere, M., Orru, K., Heidrich, O., ..., Faehnle, M., 2014. Mitigating and adapting to climate change: Multi-functional and multi-scale assessment of green urban infrastructure. J. Environ. Manage. 146, 107–115. doi:10.1016/j.jenvman.2014.07.025
Derkzen, M.L., van Teeffelen, A.J.A., Verburg, P.H., 2015. REVIEW: Quantifying urban ecosystem services based on high-resolution data of urban green space: an assessment for Rotterdam, the Netherlands. J. Appl. Ecol. 52, 1020–1032. doi:10.1111/1365-2664.12469
Díaz, S., Demissew, S., Carabias, J., ..., Zlatanova, D., 2015. The IPBES Conceptual Framework - connecting nature and people. Curr. Opin. Environ. Sustain. 14, 1–16. doi:10.1016/j.cosust.2014.11.002
Dick, J., Turkelboom, F., Woods, H., ..., Zulian, G., 2017. Stakeholders' perspectives on the operationalisation of the ecosystem service concept: Results from 27 case studies. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.09.015
Dobbs, C., Escobedo, F.J., Zipperer, W.C., 2011. A framework for developing urban forest ecosystem services and goods indicators. Landsc. Urban Plan. 99, 196–206. doi:10.1016/j.landurbplan.2010.11.004
Dühr, S., Stead, D., Zonneveld, W., 2007. The Europeanization of spatial planning through territorial cooperation. Plan. Pract. Res. 22, 291–307. doi:10.1080/02697450701688245
Dunford, R., Harrison, P., Smith, A., ..., Yli-Pelkonen, V., 2017. Integrating methods for ecosystem service assessment: Experiences from real world situations. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.10.014
EEA, 2012. Urban adaptation to climate change in Europe. Challenges and opportunities for cities together with supportive national and European policies. Copenhagen. doi:10.2800/41895
EEA, 2006. Land accounts for Europe 1990–2000: Towards integrated land and ecosystem accounting, EEA Report. Copenhagen.
Ehrlich, P.R., Ehrlich, A.H., 1981. Extinction: the causes and consequences of the disappearance of species. Random House Inc., New York.
Elmqvist, T., Gómez-Baggethun, E., Langemeyer, J., 2016. Ecosystem Services provided by urban green infrastructure, in: Potschin, M., Haines-Young, R., Fish, R., Turner, R.K. (Eds.), Routledge Handbook of Ecosystem Services. Routledge, pp. 452–464.
Elmqvist, T., Setälä, H., Handel, S.N., ..., de Groot, R., 2015. Benefits of restoring ecosystem services in urban areas. Curr. Opin. Environ. Sustain. 14, 101–108. doi:10.1016/j.cosust.2015.05.001
Elo, S., Kyngäs, H., 2008. The qualitative content analysis process. J. Adv. Nurs. 62, 107–115. doi:10.1111/j.1365-2648.2007.04569.x
Ernstson, H., 2013. The social production of ecosystem services: A framework for studying environmental justice and ecological complexity in urbanized landscapes. Landsc. Urban Plan. 109, 7–17. doi:10.1016/j.landurbplan.2012.10.005
Escobedo, F.J., Kroeger, T., Wagner, J.E., 2011. Urban forests and pollution mitigation: analyzing ecosystem services and disservices. Environ. Pollut. 159,

2078–2087. doi:10.1016/j.envpol.2011.01.010
Escobedo, F.J., Luley, C.J., Bond, J., Staudhammer, C., Bartel, C., 2009. Hurricane debris and damage assessment for Florida urban forests. Arboric. Urban For. 35, 100–106.
Escobedo, F.J., Nowak, D.J., 2009. Spatial heterogeneity and air pollution removal by an urban forest. Landsc. Urban Plan. 90, 102–110. doi:10.1016/j.landurbplan.2008.10.021
European Commission, 2016a. Urban Agenda for the EU "Pact of Amsterdam." Amsterdam.
European Commission, 2016b. Quality of Life in European Cities 2015 - Flash Eurobarometer 419. Publications Office of the European Union, Luxembourg. doi:10.2776/870421
European Commission, 2015. Towards an EU Research and Innovation policy agenda for Nature-Based Solutions & Re-Naturing Cities. Final Report of the Horizon2020 expert group on nature-based solutions and re-naturing cities. Brussels. doi:10.2777/765301
European Commission, 2011a. Cities of tomorrow - Challenges, visions, ways forward. Publications Office of the European Union, Luxembourg. doi:10.2776/41803
European Commission, 2011b. Roadmap to a Resource Efficient Europe (COM/2011/0571 final). Brussels.
European Commission, 2010. Our life insurance, our natural capital: an EU biodiversity strategy to 2020 (COM/2011/0244 final). Brussels.
European Commission, 2006. Halting the loss of biodiversity by 2010 – and beyond – Sustaining ecosystem services for human well-being (COM/2006/0216 final). Brussels.
European Union, 2008. Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe.
European Union, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Off. J. Eur. Parliam. doi:10.1039/ap9842100196
EuroVoc, 2017. 7206 Europe [WWW Document]. Multiling. Thes. Eur. Union. URL http://eurovoc.europa.eu/ (accessed 5.1.17).
Ewing, R., 2010. The Impact of Urban Form on U.S. Residential Energy Use. Hous. Policy Debate 19, 37–41. doi:10.1080/10511482.2008.9521624
Faehnle, M., Söderman, T., Schulman, H., Lehvävirta, S., 2014. Scale-sensitive integration of ecosystem services in urban planning. GeoJournal 411–425. doi:10.1007/s10708-014-9560-z
Faludi, A., 2010. Centenary paper: European spatial planning: past, present and future. Town Plan. Rev. 81, 1–22. doi:10.3828/tpr.2009.21
Faludi, A., 2004. Spatial planning traditions in Europe: their role in the ESDP process. Int. Plan. Stud. 9, 155–172. doi:10.1080/1356347042000311758
Fang, C.-F., Ling, D.-L., 2003. Investigation of the noise reduction provided by tree belts. Landsc. Urban Plan. 63, 187–195. doi:10.1016/S0169-2046(02)00190-1
Farrugia, S., Hudson, M.D., McCulloch, L., 2013. An evaluation of flood control and urban cooling ecosystem services delivered by urban green infrastructure. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 9, 136–145. doi:10.1080/21513732.2013.782342
Fewtrell, L., Kay, D., Ashley, R., 2008. Flooding and health - an evaluation of the health impacts of urban pluvial flooding in the UK, Health impact assessment for sustainable water management. IWA Publishing, London.
Fischer, E.M., Schär, C., 2010. Consistent geographical patterns of changes in high-impact European heatwaves. Nat. Geosci. 3, 398–403. doi:10.1038/ngeo866
Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem

services for decision making. Ecol. Econ. 68, 643–653. doi:10.1016/j.ecolecon.2008.09.014
Frantzeskaki, N., Kabisch, N., McPhearson, T., 2016. Advancing urban environmental governance: Understanding theories, practices and processes shaping urban sustainability and resilience. Environ. Sci. Policy 62, 1–6. doi:10.1016/j.envsci.2016.05.008
Frey, H.W., 2000. Not green belts but green wedges: the precarious relationship between city and country. URBAN Des. Int. 5, 13–25. doi:10.1057/palgrave.udi.9000003
Fürst, C., Luque, S., Geneletti, D., 2017. Nexus thinking – how ecosystem services can contribute to enhancing the cross-scale and cross-sectoral coherence between land use, spatial planning and policy-making. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 13, 412–421. doi:10.1080/21513732.2017.1396257
Galler, C., Albert, C., von Haaren, C., 2016. From regional environmental planning to implementation: Paths and challenges of integrating ecosystem services. Ecosyst. Serv. 18, 118–129. doi:10.1016/j.ecoser.2016.02.031
Galster, G., Hanson, R., Ratcliffe, M.R., Wolman, H., Coleman, S., Freihage, J., 2001. Wrestling Sprawl to the Ground: Defining and measuring an elusive concept. Hous. Policy Debate 12, 681–717. doi:10.1080/10511482.2001.9521426
García-Nieto, A.P., Garcìa-Llorente, M., Iniesta-Arandia, I., Martín-López, B., 2013. Mapping forest ecosystem services: From providing units to beneficiaries. Ecosyst. Serv. 4, 126–138. doi:10.1016/j.ecoser.2013.03.003
Geijzendorffer, I.R., Roche, P.K., 2014. The relevant scales of ecosystem services demand. Ecosyst. Serv. 10, 49–51. doi:10.1016/j.ecoser.2014.09.002
Geneletti, D., 2016. Handbook on Biodiversity and Ecosystem Services in Impact Assessment. Edward Elgar Publishing. doi:10.4337/9781783478996
Geneletti, D., 2015. A Conceptual Approach to Promote the Integration of Ecosystem Services in Strategic Environmental Assessment. J. Environ. Assess. Policy Manag. 17, 1550035. doi:10.1142/S1464333215500350
Geneletti, D., 2013. Ecosystem services in environmental impact assessment and strategic environmental assessment. Environ. Impact Assess. Rev. 40, 1–2. doi:10.1016/j.eiar.2013.02.005
Geneletti, D., 2011. Reasons and options for integrating ecosystem services in strategic environmental assessment of spatial planning. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 7, 143–149. doi:10.1080/21513732.2011.617711
Geneletti, D., La Rosa, D., Spyra, M., Cortinovis, C., 2017. A review of approaches and challenges for sustainable planning in urban peripheries. Landsc. Urban Plan. 165, 231–243. doi:10.1016/j.landurbplan.2017.01.013
Geneletti, D., Zardo, L., 2016. Ecosystem-based adaptation in cities: An analysis of European urban climate adaptation plans. Land use policy 50, 38–47. doi:10.1016/j.landusepol.2015.09.003
Geneletti, D., Zardo, L., Cortinovis, C., 2016. Promoting nature-based solutions for climate adaptation in cities through impact assessment, in: Geneletti, D. (Ed.), Handbook on Biodiversity and Ecosystem Services in Impact Assessment. Edward Elgar Publishing, pp. 428–452. doi:10.4337/9781783478996.00025
Giffinger, R., Fertner, C., 2007. City-ranking of European medium-sized cities. Cent. Reg. Sci. Vienna UT 1–12.
Giovannini, L., Zardi, D., de Franceschi, M., 2011. Analysis of the urban thermal fingerprint of the city of Trento in the Alps. J. Appl. Meteorol. Climatol. 50, 1145–1162. doi:10.1175/2010JAMC2613.1
Gómez-Baggethun, E., Barton, D.N., 2013. Classifying and valuing ecosystem services for urban planning. Ecol. Econ. 86, 235–245. doi:10.1016/j.ecolecon.2012.08.019
Grant, J., 2002. Mixed Use in Theory and Practice: Canadian Experience with

Implementing a Planning Principle. J. Am. Plan. Assoc. 68, 71–84. doi:10.1080/01944360208977192
GRASS Development Team, 2017. GRASS GIS.
Greenhalgh, T., Peacock, R., 2005. Effectiveness and efficiency of search methods in systematic reviews of complex evidence: audit of primary sources. BMJ 331, 1064–5. doi:10.1136/bmj.38636.593461.68
Grêt-Regamey, A., Celio, E., Klein, T.M., Wissen Hayek, U., 2013. Understanding ecosystem services trade-offs with interactive procedural modeling for sustainable urban planning. Landsc. Urban Plan. 109, 107–116. doi:10.1016/j.landurbplan.2012.10.011
Grêt-Regamey, A., Sirén, E., Brunner, S.H., Weibel, B., 2017. Review of decision support tools to operationalize the ecosystem services concept. Ecosyst. Serv. 26, 306–315. doi:10.1016/j.ecoser.2016.10.012
Griffith, J.A., Martinko, E.A., Price, K.P., 2000. Landscape structure analysis of Kansas at three scales. Landsc. Urban Plan. 52, 45–61. doi:10.1016/S0169-2046(00)00112-2
Guerry, A.D., Polasky, S., Lubchenco, J., ..., Vira, B., 2015. Natural capital and ecosystem services informing decisions: From promise to practice. Proc. Natl. Acad. Sci. 112, 7348–7355. doi:10.1073/pnas.1503751112
Haaland, C., Konijnendijk van Den Bosch, C.C., 2015. Challenges and strategies for urban green-space planning in cities undergoing densification: A review. Urban For. Urban Green. 14, 760–771. doi:10.1016/j.ufug.2015.07.009
Haase, D., Haase, A., Rink, D., 2014a. Conceptualizing the nexus between urban shrinkage and ecosystem services. Landsc. Urban Plan. 132, 159–169. doi:10.1016/j.landurbplan.2014.09.003
Haase, D., Kabisch, N., Haase, A., 2013. Endless Urban Growth? On the Mismatch of Population, Household and Urban Land Area Growth and Its Effects on the Urban Debate. PLoS One 8, 1–8. doi:10.1371/journal.pone.0066531
Haase, D., Larondelle, N., Andersson, E., ..., Elmqvist, T., 2014b. A quantitative review of urban ecosystem service assessments: Concepts, models, and implementation. Ambio 43, 413–433. doi:10.1007/s13280-014-0504-0
Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being., in: Raffaelli, D., Frid, C. (Eds.), Ecosystems Ecology: A New Synthesis. Cambridge University Press, pp. 110–139. doi:10.1017/CBO9780511750458
Hamin, E.M., Gurran, N., 2009. Urban form and climate change: Balancing adaptation and mitigation in the U.S. and Australia. Habitat Int. 33, 238–245. doi:10.1016/j.habitatint.2008.10.005
Hansen, R., Frantzeskaki, N., McPhearson, T., ..., Pauleit, S., 2015. The uptake of the ecosystem services concept in planning discourses of European and American cities. Ecosyst. Serv. 12, 228–246. doi:10.1016/j.ecoser.2014.11.013
Hansen, R., Pauleit, S., 2014. From multifunctionality to multiple ecosystem services? A conceptual framework for multifunctionality in green infrastructure planning for Urban Areas. Ambio 43, 516–529. doi:10.1007/s13280-014-0510-2
Harrison, P.A., Dunford, R., Barton, D.N., ..., Zulian, G., 2017. Selecting methods for ecosystem service assessment: A decision tree approach. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.09.016
Hasse, J.E., Lathrop, R.G., 2003. Land resource impact indicators of urban sprawl. Appl. Geogr. 23, 159–175. doi:10.1016/j.apgeog.2003.08.002
Hauck, J., Görg, C., Varjopuro, R., Ratamäki, O., Jax, K., 2013a. Benefits and limitations of the ecosystem services concept in environmental policy and decision making: Some stakeholder perspectives. Environ. Sci. Policy 25, 13–21. doi:10.1016/j.envsci.2012.08.001
Hauck, J., Gorg, C., Varjopuro, R., Ratamaki, O., Maes, J., Wittmer, H., Jax, K.,

2013b. “Maps have an air of authority”: Potential benefits and challenges of ecosystem service maps at different levels of decision making. Ecosyst. Serv. 4, 25–32. doi:10.1016/j.ecoser.2012.11.003
Hauck, J., Schweppe-Kraft, B., Albert, C., ..., Grêt-Regamey, A., 2013c. The Promise of the Ecosystem Services Concept for Planning and Decision-Making. GAIA 22, 232–236.
Heidrich, O., Dawson, R.J., Reckien, D., Walsh, C.L., 2013. Assessment of the climate preparedness of 30 urban areas in the UK. Clim. Change 120, 771–784. doi:10.1007/s10584-013-0846-9
Hemond, H.F., Benoit, J., 1988. Cumulative impacts on water quality functions of wetlands. Environ. Manage. 12, 639–653. doi:10.1007/BF01867542
Herold, M., Goldstein, N.C., Clarke, K.C., 2003. The spatiotemporal form of urban growth: Measurement, analysis and modeling. Remote Sens. Environ. 86, 286–302. doi:10.1016/S0034-4257(03)00075-0
Holt, A.R., Mears, M., Maltby, L., Warren, P.H., 2015. Understanding spatial patterns in the production of multiple urban ecosystem services. Ecosyst. Serv. 16, 33–46. doi:10.1016/j.ecoser.2015.08.007
Honrado, J.P., Vieira, C., Soares, C., ..., Partidário, M.R., 2013. Can we infer about ecosystem services from EIA and SEA practice? A framework for analysis and examples from Portugal. Environ. Impact Assess. Rev. 40, 14–24. doi:10.1016/j.eiar.2012.12.002
Howard, E., 1902. Garden Cities of To-morrow, in: Howard, E., Osborn, F.J., Munford, L. (Eds.), Garden Cities of To-Morrow. Reprinted Ed. with a Preface by F.J. Osborn. With an Introductory Essay by L. Mumford. Faber & Faber, 1951, London.
Hsieh, H.-F., Shannon, S.E., 2005. Three approaches to qualitative content analysis. Qual. Health Res. 15, 1277–1288. doi:10.1177/1049732305276687
Huang, Y.J., Akbari, H., Taha, H., 1990. The wind-shielding and shading effects of trees on residential heating and cooling requirements.
Iacob, O., Rowan, J.S., Brown, I., Ellis, C., 2014. Evaluating wider benefits of natural flood management strategies: an ecosystem-based adaptation perspective. Hydrol. Res. 45, 774–787. doi:10.2166/nh.2014.184
IPBES, 2017. Update on the classification of nature’s contributions to people by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Bonn.
Jabareen, Y.R., 2006. Sustainable Urban Forms: Their Typologies, Models, and Concepts. J. Plan. Educ. Res. 26, 38–52. doi:10.1177/0739456X05285119
Jacobs, S., Dendoncker, N., Martín-López, B., ..., Washbourn, C.-L., 2016. A new valuation school: Integrating diverse values of nature in resource and land use decisions. Ecosyst. Serv. 22, 213–220. doi:10.1016/j.ecoser.2016.11.007
Jacobs, S., Martín-López, B., Barton, D.N., ..., Smith, R., 2017. The means determine the end - Pursuing integrated valuation in practice. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.07.011
Jaeger, J.A.G., Schwick, C., 2014. Improving the measurement of urban sprawl: Weighted Urban Proliferation (WUP) and its application to Switzerland. Ecol. Indic. 38, 294–308. doi:10.1016/j.ecolind.2013.11.022
Jahn, M., 2015. Economics of extreme weather events: Terminology and regional impact models. Weather Clim. Extrem. 10, 29–39. doi:10.1016/j.wace.2015.08.005
Jax, K., Furman, E., Saarikoski, H., ..., Watt, A.D., 2017. Handling a messy world: Lessons learned when trying to make the ecosystem services concept operational. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.08.001
Jo, H.-K., McPherson, E.G., 1995. Carbon Storage and Flux in Urban Residential Greenspace. J. Environ. Manage. 45, 109–133. doi:10.1006/jema.1995.0062

Jochner, S., Markevych, I., Beck, I., Traidl-Hoffmann, C., Heinrich, J., Menzel, A., 2015. The effects of short- and long-term air pollutants on plant phenology and leaf characteristics. Environ. Pollut. 206, 382–389. doi:10.1016/j.envpol.2015.07.040

Jones, H.P., Hole, D.G., Zavaleta, E.S., 2012. Harnessing nature to help people adapt to climate change. Nat. Clim. Chang. 2, 504–509. doi:10.1038/nclimate1463

Jopke, C., Kreyling, J., Maes, J., Koellner, T., 2015. Interactions among ecosystem services across Europe: Bagplots and cumulative correlation coefficients reveal synergies, trade-offs, and regional patterns. Ecol. Indic. 49, 46–52. doi:10.1016/j.ecolind.2014.09.037

Kabisch, N., 2015. Ecosystem service implementation and governance challenges in urban green space planning—The case of Berlin, Germany. Land use policy 42, 557–567. doi:10.1016/j.landusepol.2014.09.005

Kabisch, N., Haase, D., 2014. Green justice or just green? Provision of urban green spaces in Berlin, Germany. Landsc. Urban Plan. 122, 129–139. doi:10.1016/j.landurbplan.2013.11.016

Kabisch, N., Haase, D., 2013. Green spaces of European cities revisited for 1990-2006. Landsc. Urban Plan. 110, 113–122. doi:10.1016/j.landurbplan.2012.10.017

Kabisch, N., Qureshi, S., Haase, D., 2015. Human - environment interactions in urban green spaces - A systematic review of contemporary issues and prospects for future research. Environ. Impact Assess. Rev. 50, 25–34. doi:doi.org/10.1016/j.eiar.2014.08.007

Kabisch, N., Strohbach, M.W., Haase, D., Kronenberg, J., 2016. Urban green space availability in European cities. Ecol. Indic. 70, 586–596. doi:10.1016/j.ecolind.2016.02.029

Kabisch, N., van den Bosch, M., Lafortezza, R., 2017. The health benefits of nature-based solutions to urbanization challenges for children and the elderly – A systematic review. Environ. Res. 159, 362–373. doi:10.1016/j.envres.2017.08.004

Kain, J.-H., Larondelle, N., Haase, D., Kaczorowska, A., 2016. Exploring local consequences of two land-use alternatives for the supply of urban ecosystem services in Stockholm year 2050. Ecol. Indic. 70, 615–629. doi:10.1016/j.ecolind.2016.02.062

Kandziora, M., Burkhard, B., Müller, F., 2013. Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators—A theoretical matrix exercise. Ecol. Indic. 28, 54–78. doi:10.1016/j.ecolind.2012.09.006

Kasanko, M., Barredo, J.I., Lavalle, C., ..., Brezger, A., 2006. Are European cities becoming dispersed?. A comparative analysis of 15 European urban areas. Landsc. Urban Plan. 77, 111–130. doi:10.1016/j.landurbplan.2005.02.003

Kates, R.W., Clark, W.C., Corell, R.W., ..., Svedin, U., 2001. Sustainability Science. Science (80-.). 292, 641–642. doi:10.1126/science.1059386

Kemkes, R.J., Farley, J., Koliba, C.J., 2010. Determining when payments are an effective policy approach to ecosystem service provision. Ecol. Econ. 69, 2069–2074. doi:10.1016/j.ecolecon.2009.11.032

Kenny, G.P., Yardley, J., Brown, C., Sigal, R.J., Jay, O., 2010. Heat stress in older individuals and patients with common chronic diseases. Cmaj 182, 1053–1060. doi:10.1503/cmaj.081050

Knight, R.L., Landres, P. (Eds.), 2013. Stewardship across boundaries. Island Press.

Koschke, L., Fürst, C., Frank, S., Makeschin, F., 2012. A multi-criteria approach for an integrated land-cover-based assessment of ecosystem services provision to support landscape planning. Ecol. Indic. 21, 54–66. doi:10.1016/j.ecolind.2011.12.010

Kremer, P., Hamstead, Z.A., 2016. The value of urban ecosystem services in New

York City: A spatially explicit multicriteria analysis of landscape scale valuation scenarios. Environ. Sci. Policy 62, 57–68. doi:10.1016/j.envsci.2016.04.012
Kremer, P., Hamstead, Z., Haase, D., ..., Elmqvist, T., 2016. Key insights for the future of urban ecosystem services research. Ecol. Soc. 21, 29. doi:10.5751/ES-08445-210229
Kumar, P., Geneletti, D., 2015. How are climate change concerns addressed by spatial plans? An evaluation framework, and an application to Indian cities. Land use policy 42, 210–226. doi:10.1016/j.landusepol.2014.07.016
La Notte, A., D’Amato, D., Mäkinen, H., ..., Crossman, N.D., 2017. Ecosystem services classification: A systems ecology perspective of the cascade framework. Ecol. Indic. 74, 1–11. doi:10.1016/j.ecolind.2016.11.030
La Rosa, D., 2014. Accessibility to greenspaces: GIS based indicators for sustainable planning in a dense urban context. Ecol. Indic. 42, 122–134. doi:10.1016/j.ecolind.2013.11.011
La Rosa, D., Spyra, M., Inostroza, L., 2015. Indicators of Cultural Ecosystem Services for urban planning: A review. Ecol. Indic. 61, 74–89. doi:10.1016/j.ecolind.2015.04.028
Landers, D.H., Nahlik, A.M., 2013. Final Ecosystem Goods and Services Classification System (FEGS-CS) 108. doi:EPA/600/R-13/ORD-004914
Langemeyer, J., Gómez-Baggethun, E., Haase, D., Scheuer, S., Elmqvist, T., 2016. Bridging the gap between ecosystem service assessments and land-use planning through Multi-Criteria Decision Analysis (MCDA). Environ. Sci. Policy 62, 45–56. doi:10.1016/j.envsci.2016.02.013
Langemeyer, J., Palomo, I., Baraibar, S., Gómez-Baggethun, E., 2018. Participatory multi-criteria decision aid: Operationalizing an integrated assessment of ecosystem services. Ecosyst. Serv. 30, 49–60. doi:10.1016/j.ecoser.2018.01.012
Larondelle, N., Haase, D., Kabisch, N., 2014. Mapping the diversity of regulating ecosystem services in European cities. Glob. Environ. Chang. 26, 119–129. doi:10.1016/j.gloenvcha.2014.04.008
Larondelle, N., Lauf, S., 2016. Balancing demand and supply of multiple urban ecosystem services on different spatial scales. Ecosyst. Serv. 22, 18–31. doi:10.1016/j.ecoser.2016.09.008
Laurans, Y., Rankovic, A., Billé, R., Pirard, R., Mermet, L., 2013. Use of ecosystem services economic valuation for decision making: Questioning a literature blindspot. J. Environ. Manage. 119, 208–219. doi:10.1016/j.jenvman.2013.01.008
Lin, B., Meyers, J., Barnett, G., 2015. Understanding the potential loss and inequities of green space distribution with urban densification. Urban For. Urban Green. 14, 952–958. doi:10.1016/j.ufug.2015.09.003
Liquete, C., Zulian, G., Delgado, I., Stips, A., Maes, J., 2013. Assessment of coastal protection as an ecosystem service in Europe. Ecol. Indic. 30, 205–217. doi:10.1016/j.ecolind.2013.02.013
Liu, W., Chen, W., Peng, C., 2014. Assessing the effectiveness of green infrastructures on urban flooding reduction: A community scale study. Ecol. Modell. 291, 6–14. doi:10.1016/j.ecolmodel.2014.07.012
Lowry, J.H., Lowry, M.B., 2014. Comparing spatial metrics that quantify urban form. Comput. Environ. Urban Syst. 44, 59–67. doi:10.1016/j.compenvurbsys.2013.11.005
Luck, G.W., Daily, G.C., Ehrlich, P.R., 2003. Population diversity and ecosystem services. Trends Ecol. Evol. 18, 331–336. doi:10.1016/S0169-5347(03)00100-9
Luederitz, C., Brink, E., Gralla, F., ..., von Wehrden, H., 2015. A review of urban ecosystem services: six key challenges for future research. Ecosyst. Serv. 14,

98–112. doi:10.1016/j.ecoser.2015.05.001
Lyles, W., Stevens, M.R., 2014. Plan Quality Evaluation 1994-2012: Growth and Contributions, Limitations, and New Directions. J. Plan. Educ. Res. 34, 433–450. doi:10.1177/0739456X14549752
MA, 2005. Ecosystems and Human Well-being: Synthesis. A report of the Millenium Ecosystem Assessement. Island Press, Washington, DC.
Mace, G.M., 2016. Ecosystem services: where is the discipline heading?, in: Potschin, M., Haines-Young, R., Fish, R., Turner, R.K. (Eds.), Routledge Handbook of Ecosystem Services. Routledge, pp. 602–606.
Mace, G.M., 2014. Whose conservation? Science (80-.). 345, 1558–1560. doi:10.1126/science.1254704
Maczka, K., Matczak, P., Pietrzyk-Kaszyńska, A., ..., Grodzińska-Jurczak, M., 2016. Application of the ecosystem services concept in environmental policy-A systematic empirical analysis of national level policy documents in Poland. Ecol. Econ. 128, 169–176. doi:10.1016/j.ecolecon.2016.04.023
Maes, J., Egoh, B., Willemen, L., ..., Bidoglio, G., 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst. Serv. 1, 31–39. doi:10.1016/j.ecoser.2012.06.004
Maes, J., Liquete, C., Teller, A., ..., Lavalle, C., 2016. An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. Ecosyst. Serv. 17, 14–23. doi:10.1016/j.ecoser.2015.10.023
Maes, J., Teller, A., Erhard, M., ..., Santos-Martín, F., 2014. Mapping and assessment of ecosystems and their services. Indicators for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020. doi:10.2779/75203
Maes, J., Teller, A., Erhard, M., ..., Bidoglio, G., 2013. Mapping and Assessment of Ecosystems and their Services. An Analytical Framework for Ecosystem Assessments Under Action 5 of the EU Biodiversity Strategy to 2020. Luxembourg. doi:10.2779/12398
Makri, A., Stilianakis, N.I., 2008. Vulnerability to air pollution health effects. Int. J. Hyg. Environ. Health 211, 326–336. doi:10.1016/j.ijheh.2007.06.005
Martínez-Harms, M.J., Balvanera, P., 2012. Methods for mapping ecosystem service supply: a review. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 8, 17–25. doi:10.1080/21513732.2012.663792
Mascarenhas, A., Ramos, T.B., Haase, D., Santos, R., 2015. Ecosystem services in spatial planning and strategic environmental assessment—A European and Portuguese profile. Land use policy 48, 158–169. doi:10.1016/j.landusepol.2015.05.012
Mascarenhas, A., Ramos, T.B., Haase, D., Santos, R., 2014. Integration of ecosystem services in spatial planning: a survey on regional planners' views. Landsc. Ecol. 1–14. doi:10.1007/s10980-014-0012-4
Mastrangelo, G., Fedeli, U., Visentin, C., Milan, G., Fadda, E., Spolaore, P., 2007. Pattern and determinants of hospitalization during heat waves: an ecologic study. BMC Public Health 7, 200. doi:10.1186/1471-2458-7-200
Mathey, J., Rößler, S., Banse, J., Lehmann, I., Bräuer, A., 2015. Brownfields As an Element of Green Infrastructure for Implementing Ecosystem Services into Urban Areas. J. Urban Plan. Dev. 4015001, 1–13. doi:10.1061/(ASCE)UP.1943-5444.0000275.
Matzdorf, B., Meyer, C., 2014. The relevance of the ecosystem services framework for developed countries' environmental policies: A comparative case study of the US and EU. Land use policy 38, 509–521. doi:10.1016/j.landusepol.2013.12.011
McCrea, R., Walters, P., 2012. Impacts of Urban Consolidation on Urban Liveability: Comparing an Inner and Outer Suburb in Brisbane, Australia. Housing, Theory Soc. 29, 190–206. doi:10.1080/14036096.2011.641261

McDonough, K., Hutchinson, S., Moore, T., Hutchinson, J.M.S., 2017. Analysis of publication trends in ecosystem services research. Ecosyst. Serv. 25, 82–88. doi:10.1016/j.ecoser.2017.03.022
McGarigal, K., Cushman, S.A., Ene, E., 2012. FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical and Continuous Maps.
Mckenzie, E., Posner, S., Tillmann, P., Bernhardt, J.R., Howard, K., Rosenthal, A., 2014. Understanding the use of ecosystem service knowledge in decision making: Lessons from international experiences of spatial planning. Environ. Plan. C Gov. Policy 32, 320–340. doi:10.1068/c12292j
McPhearson, T., Andersson, E., Elmqvist, T., Frantzeskaki, N., 2015. Resilience of and through urban ecosystem services. Ecosyst. Serv. 12, 152–156. doi:10.1016/j.ecoser.2014.07.012
McPhearson, T., Hamstead, Z.A., Kremer, P., 2014. Urban ecosystem services for resilience planning and management in New York City. Ambio 43, 502–515. doi:10.1007/s13280-014-0509-8
McPhearson, T., Kremer, P., Hamstead, Z.A., 2013. Mapping ecosystem services in New York City: Applying a social-ecological approach in urban vacant land. Ecosyst. Serv. 5, 11–26. doi:10.1016/j.ecoser.2013.06.005
Mega, V., Pedersen, J., 1998. Urban Sustainability Indicators. Office for Official Publications of the European Communities, Luxembourg.
Melia, S., Parkhurst, G., Barton, H., 2011. The paradox of intensification. Transp. Policy 18, 46–52. doi:10.1016/j.tranpol.2010.05.007
Millward, H., 2006. Urban containment strategies: A case-study appraisal of plans and policies in Japanese, British, and Canadian cities. Land use policy 23, 473–485. doi:10.1016/j.landusepol.2005.02.004
Mindell, J., Joffe, M., 2004. Predicted health impacts of urban air quality management. J. Epidemiol. Community Heal. 58, 103–113.
Morais, P., Camanho, A.S., 2011. Evaluation of performance of European cities with the aim to promote quality of life improvements. Omega 39, 398–409. doi:10.1016/j.omega.2010.09.003
Morani, A., Nowak, D.J., Hirabayashi, S., Calfapietra, C., 2011. How to select the best tree planting locations to enhance air pollution removal in the MillionTreesNYC initiative. Environ. Pollut. 159, 1040–1047. doi:10.1016/j.envpol.2010.11.022
Müller, F., Burkhard, B., 2012. The indicator side of ecosystem services. Ecosyst. Serv. 1, 26–30. doi:10.1016/j.ecoser.2012.06.001
Müller, N., Ignatieva, M., Nilon, C.H., Werner, P., Zipperer, W.C., 2013. Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities. doi:10.1007/978-94-007-7088-1
Nadin, V., Stead, D., 2008. European spatial planning systems, social models and learning. Disp 172, 35–47. doi:10.1080/02513625.2008.10557001
Nahuelhual, L., Laterra, P., Villarino, S., ..., Burgos, N., 2015. Mapping of ecosystem services: Missing links between purposes and procedures. Ecosyst. Serv. 13, 162–172. doi:10.1016/j.ecoser.2015.03.005
Narayan, S., Beck, M.W., Reguero, B.G., ..., Burks-Copes, K.A., 2016. The effectiveness, costs and coastal protection benefits of natural and nature-based defences. PLoS One 11, 1–17. doi:10.1371/journal.pone.0154735
Nassauer, J.I., Raskin, J., 2014. Urban vacancy and land use legacies: A frontier for urban ecological research, design, and planning. Landsc. Urban Plan. 125, 245–253. doi:10.1016/j.landurbplan.2013.10.008
Nedkov, S., Burkhard, B., 2012. Flood regulating ecosystem services - Mapping supply and demand, in the Etropole municipality, Bulgaria. Ecol. Indic. 21, 67–79. doi:10.1016/j.ecolind.2011.06.022
Niemeijer, D., de Groot, R., 2008. Framing environmental indicators: Moving from

causal chains to causal networks. Environ. Dev. Sustain. 10, 89–106. doi:10.1007/s10668-006-9040-9
Niemelä, J., Saarela, S.-R., Söderman, T., Kopperoinen, L., Yli-Pelkonen, V., Väre, S., Kotze, D.J., 2010. Using the ecosystem services approach for better planning and conservation of urban green spaces: A Finland case study. Biodivers. Conserv. 19, 3225–3243. doi:10.1007/s10531-010-9888-8
Nilsson, K., Nielsen, T.S., Aalbers, C., ..., Zasada, I., 2014. Strategies for Sustainable Urban Development and Urban-Rural Linkages. Eur. J. Spat. Dev. 1–26.
Nisbet, T.R., Thomas, H., 2006. The role of woodland in flood control: a landscape perspective, in: Water and the Landscape: The Landscape Ecology of Freshwater Ecosystems. Proceedings of the 14th Annual IALE (UK) Conference. pp. 118–125. doi:10.1017/CBO9781107415324.004
Norton, R.K., 2008. Using content analysis to evaluate local master plans and zoning codes. Land use policy 25, 432–454. doi:10.1016/j.landusepol.2007.10.006
Nowak, D.J., Crane, D.E., Stevens, J.C., 2006. Air pollution removal by urban trees and shrubs in the United States. Urban For. Urban Green. 4, 115–123. doi:10.1016/j.ufug.2006.01.007
Nowak, D.J., Crane, D.E., Stevens, J.C., Hoehn, R.E., Walton, J.T., Bond, J., 2008. A Ground-Based Method of Assessing Urban Forest Structure and Ecosystem Services. Arboric. Urban For. 34, 347–358. doi:10.1039/b712015j
Nowak, D.J., Greenfield, E.J., Hoehn, R.E., Lapoint, E., 2013. Carbon storage and sequestration by trees in urban and community areas of the United States. Environ. Pollut. 178, 229–236. doi:10.1016/j.envpol.2013.03.019
Nuissl, H., Haase, D., Lanzendorf, M., Wittmer, H., 2009. Environmental impact assessment of urban land use transitions-A context-sensitive approach. Land use policy 26, 414–424. doi:10.1016/j.landusepol.2008.05.006
OECD, 2012. Compact City Policies: A Comparative Assessment, OECD Green. ed, OECD Green Growth Studies. OECD Publishing. doi:10.1787/9789264167865-en
Olander, L.P., Johnston, R.J., Tallis, H., ..., Palmer, M., 2018. Benefit relevant indicators: Ecosystem services measures that link ecological and social outcomes. Ecol. Indic. 85, 1262–1272. doi:10.1016/j.ecolind.2017.12.001
Olander, L.P., Polasky, S., Kagan, J.S., ..., Yoskowitz, D., 2017. So you want your research to be relevant? Building the bridge between ecosystem services research and practice. Ecosyst. Serv. 26, 170–182. doi:10.1016/j.ecoser.2017.06.003
Olsen, A.S., Zhou, Q., Linde, J.J., Arnbjerg-Nielsen, K., 2015. Comparing methods of calculating expected annual damage in urban pluvial flood risk assessments. Water (Switzerland) 7, 255–270. doi:10.3390/w7010255
Open Street Map Contributors, 2017. Planet dump retrieved from https://planet.osm.org.
Palo, T., Lagercrantz, K., Bramryd, T., ..., Ekelund, N., 2016. Priority areas in municipality planning: ecosystem services, environmental impact assessments and research areas. One Ecosyst. 1, e9869. doi:10.3897/oneeco.1.e9869
Palomo, I., Martín-López, B., Potschin, M., Haines-Young, R., Montes, C., 2013. National Parks, buffer zones and surrounding lands: Mapping ecosystem service flows. Ecosyst. Serv. 4, 104–116. doi:10.1016/j.ecoser.2012.09.001
Paracchini, M.L., Zulian, G., Kopperoinen, L., ..., Bidoglio, G., 2014. Mapping cultural ecosystem services: A framework to assess the potential for outdoor recreation across the EU. Ecol. Indic. 45, 371–385. doi:10.1016/j.ecolind.2014.04.018
Parr, J., 2004. The Polycentric Urban Region: A Closer Inspection. Reg. Stud. 38, 231–240. doi:10.1080/003434042000211114
Patz, J.A., Frumkin, H., Holloway, T., Vimont, D.J., Haines, A., 2014. Climate

Change. Challenges and Opportunities for Global Health. JAMA 312, 1565. doi:10.1001/jama.2014.13186
Persson, C., 2013. Deliberation or doctrine? Land use and spatial planning for sustainable development in Sweden. Land use policy 34, 301–313. doi:10.1016/j.landusepol.2013.04.007
Piwowarczyk, J., Kronenberg, J., Dereniowska, M.A., 2013. Marine ecosystem services in urban areas: Do the strategic documents of Polish coastal municipalities reflect their importance? Landsc. Urban Plan. 109, 85–93. doi:10.1016/j.landurbplan.2012.10.009
Polasky, S., Nelson, E., Camm, J., ..., Tobalske, C., 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biol. Conserv. 141, 1505–1524. doi:10.1016/j.biocon.2008.03.022
Posner, S., Getz, C., Ricketts, T., 2016a. Evaluating the impact of ecosystem service assessments on decision-makers. Environ. Sci. Policy 64, 30–37. doi:10.1016/j.envsci.2016.06.003
Posner, S., McKenzie, E., Ricketts, T.H., 2016b. Policy impacts of ecosystem services knowledge. Proc. Natl. Acad. Sci. 113, 1760–1765. doi:10.1073/pnas.1502452113
Potschin-Young, M., Haines-Young, R., Görg, C., Heink, U., Jax, K., Schleyer, C., 2017. Understanding the role of conceptual frameworks: Reading the ecosystem service cascade. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.05.015
Potschin, M., Haines-Young, R., Fish, R., Turner, R.K., 2016. Ecosystem Services in the Twenty-first Century, in: Potschin, M., Haines-Young, R., Fish, R., Turner, R.K. (Eds.), Routledge Handbook of Ecosystem Services. Routledge, pp. 1–9.
Pouyat, R. V., Yesilonis, I.D., Nowak, D.J., 2006. Carbon storage by urban soils in the United States. J. Environ. Qual. 35, 1566–1575. doi:Doi 10.2134/Jeq2005.0215
Prüss-Üstün, A., Wolf, J., Corvalán, C.F., Bos, R., Neira, M.P., 2016. Preventing disease through healthy environments. A global assessment of the burden of disease from environmental risk.
Pulighe, G., Fava, F., Lupia, F., 2016. Insights and opportunities from mapping ecosystem services of urban green spaces and potentials in planning. Ecosyst. Serv. 22, 1–10. doi:10.1016/j.ecoser.2016.09.004
QGIS Development Team, 2017. Quantum GIS.
R Core Team, 2014. R: A language and environment for statistical computing.
Rall, E.L., Kabisch, N., Hansen, R., 2015. A comparative exploration of uptake and potential application of ecosystem services in urban planning. Ecosyst. Serv. 16, 230–242. doi:10.1016/j.ecoser.2015.10.005
Ravesteyn, N. Van, Evers, D., 2004. Unseen Europe : a survey of EU politics and its impact on spatial development in the Netherlands.
Raymond, C.M., Frantzeskaki, N., Kabisch, N., ..., Calfapietra, C., 2017. A framework for assessing and implementing the co-benefits of nature-based solutions in urban areas. Environ. Sci. Policy 77, 15–24. doi:10.1016/j.envsci.2017.07.008
Revi, A., Satterthwaite, D.E., Aragón-Durand, F., ..., White, L.L. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 535–612. doi:10.1017/CBO9781107415379.013
Reyers, B., Biggs, R., Cumming, G.S., Elmqvist, T., Hejnowicz, A.P., Polasky, S., 2013. Getting the measure of ecosystem services: A social-ecological approach. Front. Ecol. Environ. 11, 268–273. doi:10.1890/120144
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
Rosenthal, A., Verutes, G., McKenzie, E., ..., Vogl, A.L., 2015. Process matters: a framework for conducting decision-relevant assessments of ecosystem services. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 11, 190–204. doi:10.1080/21513732.2014.966149
Rozas-Vásquez, D., Fürst, C., Geneletti, D., Almendra, O., 2018. Integration of ecosystem services in strategic environmental assessment across spatial planning scales. Land use policy 71, 303–310. doi:10.1016/j.landusepol.2017.12.015
Ruckelshaus, M., McKenzie, E., Tallis, H., ..., Bernhardt, J.R., 2015. Notes from the field: Lessons learned from using ecosystem service approaches to inform real-world decisions. Ecol. Econ. 115, 11–21. doi:10.1016/j.ecolecon.2013.07.009
Saarikoski, H., Mustajoki, J., Barton, D.N., ..., Santos, R., 2016. Multi-Criteria Decision Analysis and Cost-Benefit Analysis: Comparing alternative frameworks for integrated valuation of ecosystem services. Ecosyst. Serv. 0–1. doi:10.1016/j.ecoser.2016.10.014
Saarikoski, H., Primmer, E., Saarela, ..., Young, J., 2017. Institutional challenges in putting ecosystem service knowledge in practice. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.07.019
Saaroni, H., Ziv, B., 2003. The impact of a small lake on heat stress in a Mediterranean urban park: the case of Tel Aviv, Israel. Int. J. Biometeorol. 47, 156–165. doi:10.1007/s00484-003-0161-7
Sanon, S., Hein, T., Douven, W., Winkler, P., 2012. Quantifying ecosystem service trade-offs: The case of an urban floodplain in Vienna, Austria. J. Environ. Manage. 111, 159–172. doi:10.1016/j.jenvman.2012.06.008
Schleyer, C., Görg, C., Hauck, J., Winkler, K.J., 2015. Opportunities and challenges for mainstreaming the ecosystem services concept in the multi-level policy-making within the EU. Ecosyst. Serv. 16, 174–181. doi:10.1016/j.ecoser.2015.10.014
Schmidt, K., Sachse, R., Walz, A., 2016. Current role of social benefits in ecosystem service assessments. Landsc. Urban Plan. 149, 49–64. doi:10.1016/j.landurbplan.2016.01.005
Schneider, A., Woodcock, C.E., 2008. Compact, Dispersed, Fragmented, Extensive? A Comparison of Urban Growth in Twenty-five Global Cities using Remotely Sensed Data, Pattern Metrics and Census Information. Urban Stud. 45, 659–692. doi:10.1177/0042098007087340
Schröter, M., Albert, C., Marques, A., ..., Bonn, A., 2016. National Ecosystem Assessments in Europe: A Review. Bioscience 66, 813–828. doi:10.1093/biosci/biw101
Schröter, M., Remme, R.P., Hein, L., 2012. How and where to map supply and demand of ecosystem services for policy-relevant outcomes? Ecol. Indic. 23, 220–221. doi:10.1016/j.ecolind.2012.03.025
Schröter, M., van der Zanden, E.H., Van Oudenhoven, A.P.E., ..., Opdam, P., 2014. Ecosystem Services as a Contested Concept: A Synthesis of Critique and Counter-Arguments. Conserv. Lett. 7, 514–523. doi:10.1111/conl.12091
Schwarz, N., 2010. Urban form revisited-Selecting indicators for characterising European cities. Landsc. Urban Plan. 96, 29–47. doi:10.1016/j.landurbplan.2010.01.007
Science for Environment Policy, 2015. Indicators for sustainable cities. Bristol. doi:10.2779/61700
Seto, K.C., Fragkias, M., 2005. Quantifying spatiotemporal patterns of urban land-use change in four cities of China with time series landscape metrics. Landsc. Ecol.

20, 871–888. doi:10.1007/s10980-005-5238-8
Seto, K.C., Fragkias, M., Güneralp, B., Reilly, M.K., 2011. A Meta-Analysis of Global Urban Land Expansion. PLoS One 6, e23777. doi:10.1371/journal.pone.0023777
Shashua-Bar, L., Hoffman, M.E., 2000. Vegetation as a climatic component in the design of an urban street. Energy Build. 31, 221–235. doi:10.1016/S0378-7788(99)00018-3
Silvennoinen, S., Taka, M., Yli-Pelkonen, V., Koivusalo, H., Ollikainen, M., Setälä, H., 2017. Monetary value of urban green space as an ecosystem service provider: A case study of urban runoff management in Finland. Ecosyst. Serv. 28, 17–27. doi:10.1016/j.ecoser.2017.09.013
Slootweg, R., 2015. Ecosystem services in SEA: are we missing the point of a simple concept? Impact Assess. Proj. Apprais. 5517, 1–8. doi:10.1080/14615517.2015.1096039
Spake, R., Lasseur, R., Crouzat, E., ..., Eigenbrod, F., 2017. Unpacking ecosystem service bundles: Towards predictive mapping of synergies and trade-offs between ecosystem services. Glob. Environ. Chang. 47, 37–50. doi:10.1016/j.gloenvcha.2017.08.004
Spangenberg, J.H., von Haaren, C., Settele, J., 2014. The ecosystem service cascade: Further developing the metaphor. Integrating societal processes to accommodate social processes and planning, and the case of bioenergy. Ecol. Econ. 104, 22–32. doi:10.1016/j.ecolecon.2014.04.025
Spanò, M., Gentile, F., Davies, C., Lafortezza, R., 2017. The DPSIR framework in support of green infrastructure planning: A case study in Southern Italy. Land use policy 61, 242–250. doi:10.1016/j.landusepol.2016.10.051
SPINlab Vrije Universiteit Amsterdam, 2016. Definite.
Stanilov, K., 2007. The Post-Socialist City, The Post-Socialist City: Urban Form and Space Transformations in Central and Eastern Europe after Socialism, The GeoJournal Library. Springer Netherlands, Dordrecht. doi:10.1007/978-1-4020-6053-3
Steiner, F., 2016. Opportunities for urban ecology in community and regional planning. J. Urban Ecol. 2, juv004. doi:10.1093/jue/juv004
Steiner, F., 2008. The Living Landscape. An Ecological Approach to Landscape Planning. Island Press, Washington, DC.
Stessens, P., Khan, A.Z., Huysmans, M., Canters, F., 2017. Analysing urban green space accessibility and quality: A GIS-based model as spatial decision support for urban ecosystem services in Brussels. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.10.016
Strohbach, M.W., Haase, D., 2012. Above-ground carbon storage by urban trees in Leipzig, Germany: Analysis of patterns in a European city. Landsc. Urban Plan. 104, 95–104. doi:10.1016/j.landurbplan.2011.10.001
Sun, R., Chen, A., Chen, L., Lü, Y., 2012. Cooling effects of wetlands in an urban region: The case of Beijing. Ecol. Indic. 20, 57–64. doi:10.1016/j.ecolind.2012.02.006
Sutherland, W.J., Gardner, T., Bogich, T.L., ..., Dicks, L. V., 2014. Solution Scanning as a Key Policy Tool: Identifying Management Interventions to Help Maintain and Enhance Regulating Ecosystem Services. Ecol. Soc. 19, art3. doi:10.5751/ES-06082-190203
Sykes, O., 2008. The Importance of Context and Comparison in the Study of European Spatial Planning. Eur. Plan. Stud. 16, 537–555. doi:10.1080/09654310801983464
Syrbe, R.U., Walz, U., 2012. Spatial indicators for the assessment of ecosystem services: Providing, benefiting and connecting areas and landscape metrics. Ecol. Indic. 21, 80–88. doi:10.1016/j.ecolind.2012.02.013

Tallis, M., Taylor, G., Sinnett, D., Freer-Smith, P., 2011. Estimating the removal of atmospheric particulate pollution by the urban tree canopy of London, under current and future environments. Landsc. Urban Plan. 103, 129–138. doi:10.1016/j.landurbplan.2011.07.003

Tan, J., Zheng, Y., Tang, X., ..., Chen, H., 2010. The urban heat island and its impact on heat waves and human health in Shanghai. Int. J. Biometeorol. 54, 75–84. doi:10.1007/s00484-009-0256-x

Tang, Z., Brody, S.D., Quinn, C., Chang, L., Wei, T., 2010. Moving from agenda to action: evaluating local climate change action plans. J. Environ. Plan. Manag. 53, 41–62. doi:10.1080/09640560903399772

TEEB, 2010a. The Economics of Ecosystems and Biodiversity: Mainstreaming the economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB.

TEEB, 2010b. The Economics of Ecosystem and Biodiversity for local and regional policy makers.

ten Veldhuis, J.A.E., Clemens, F.H.L.R., 2010. Flood risk modelling based on tangible and intangible urban flood damage quantification. Water Sci. Technol. 62, 189. doi:10.2166/wst.2010.243

Tiwary, A., Sinnett, D., Peachey, C., ..., Hutchings, T.R., 2009. An integrated tool to assess the role of new planting in PM10 capture and the human health benefits: a case study in London. Environ. Pollut. 157, 2645–53. doi:10.1016/j.envpol.2009.05.005

Tratalos, J.A., Fuller, R.A., Warren, P.H., Davies, R.G., Gaston, K.J., 2007. Urban form, biodiversity potential and ecosystem services. Landsc. Urban Plan. 83, 308–317. doi:10.1016/j.landurbplan.2007.05.003

Trinomics B.V., 2016. Supporting the Implementation of Green Infrastructure. Final report. Rotterdam. doi:10.2779/781371

Turner, B.L., Kasperson, R.E., Matson, P.A., ..., Schiller, A., 2003. A framework for vulnerability analysis in sustainability science. Proc. Natl. Acad. Sci. U. S. A. 100, 8074–8079. doi:10.1073/pnas.1231335100

Turok, I., Mykhnenko, V., 2007. The trajectories of European cities, 1960-2005. Cities 24, 165–182. doi:10.1016/j.cities.2007.01.007

Tzoulas, K., Korpela, K., Venn, S., ..., James, P., 2007. Promoting ecosystem and human health in urban areas using Green Infrastructure: A literature review. Landsc. Urban Plan. 81, 167–178. doi:10.1016/j.landurbplan.2007.02.001

UN, 1992. Agenda 21. Rio de Janeiro. doi:10.1007/s11671-008-9208-3

UN, 1987. Our Common Future: Report of the World Commission on Environment and Development. doi:10.1080/07488008808408783

UN-Habitat, 2009. Planning Sustainable Cities - Global Report on Human Settlements 2009, United Nations Human Settlements Programme. doi:10.1007/s13398-014-0173-7.2

UN General Assembly, 2016. New Urban Agenda - Quito Declaration on Sustainable Cities and Human Settlements for All.

UN General Assembly, 2015. Transforming our world: the 2030 Agenda for Sustainable Development. New York.

United Nations, 2007. Indicators of Sustainable Development: Guidelines and Methodologies. New York. doi:10.1016/j.cirpj.2010.03.002

United Nations Statistics Division, 2017. Standard country or area codes for statistical use (M49) [WWW Document]. URL https://unstats.un.org/unsd/methodology/m49/ (accessed 5.1.17).

US EPA, 2011. Summary of State Stormwater Standards.

Uuemaa, E., Antrop, M., Marja, R., Roosaare, J., Mander, Ü., 2009. Landscape Metrics and Indices: An Overview of Their Use in Landscape Research. Living Rev. Landsc. Res. 3, 1–28. doi:10.12942/lrlr-2009-1

van den Bosch, M., Sang, O., 2017. Urban natural environments as nature-based solutions for improved public health – A systematic review of reviews. Environ. Res. 158, 373–384. doi:10.1016/j.envres.2017.05.040
Van Renterghem, T., 2014. Guidelines for optimizing road traffic noise shielding by non-deep tree belts. Ecol. Eng. 69, 276–286. doi:10.1016/j.ecoleng.2014.04.029
Van Renterghem, T., Botteldooren, D., Verheyen, K., 2012. Road traffic noise shielding by vegetation belts of limited depth. J. Sound Vib. 331, 2404–2425. doi:10.1016/j.jsv.2012.01.006
Van Renterghem, T., Forssén, J., Attenborough, K., Jean, P., Defrance, J., Hornikx, M., Kang, J., 2015. Using natural means to reduce surface transport noise during propagation outdoors. Appl. Acoust. 92, 86–101. doi:10.1016/j.apacoust.2015.01.004
Vandewalle, M., Sykes, M.T., Harrison, P.A., ..., Zobel, M., 2013. Review paper on concepts of dynamic ecosystems and their services 2014.
Vauramo, S., Setälä, H., 2011. Decomposition of labile and recalcitrant litter types under different plant communities in urban soils. Urban Ecosyst. 14, 59–70. doi:10.1007/s11252-010-0140-9
Veisten, K., Smyrnova, Y., Klæboe, R., Hornikx, M., Mosslemi, M., Kang, J., 2012. Valuation of green walls and green roofs as soundscape measures: Including monetised amenity values together with noise-attenuation values in a cost-benefit analysis of a green wall affecting courtyards. Int. J. Environ. Res. Public Health 9, 3770–3778. doi:10.3390/ijerph9113770
Verburg, R., Selnes, T., Verweij, P., 2016. Governing ecosystem services: National and local lessons from policy appraisal and implementation. Ecosyst. Serv. 18, 186–197. doi:10.1016/j.ecoser.2016.03.006
Vienneau, D., Perez, L., Schindler, C., ..., Röösli, M., 2015. Years of life lost and morbidity cases attributable to transportation noise and air pollution: A comparative health risk assessment for Switzerland in 2010. Int. J. Hyg. Environ. Health 218, 514–21. doi:10.1016/j.ijheh.2015.05.003
Viguié, V., Hallegatte, S., 2012. Trade-offs and synergies in urban climate policies. Nat. Clim. Chang. 2, 334–337. doi:10.1038/nclimate1434
Villamagna, A.M., Angermeier, P.L., Bennett, E.M., 2013. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecol. Complex. 15, 114–121. doi:10.1016/j.ecocom.2013.07.004
von Haaren, C., Albert, C., 2011. Integrating ecosystem services and environmental planning: Limitations and synergies. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 7, 150–167. doi:10.1080/21513732.2011.616534
Watson, V., 2016. Locating planning in the New Urban Agenda of the urban sustainable development goal. Plan. Theory 15, 435–448. doi:10.1177/1473095216660786
Weber, F., Kowarik, I., Säumel, I., 2014. Herbaceous plants as filters: immobilization of particulates along urban street corridors. Environ. Pollut. 186, 234–40. doi:10.1016/j.envpol.2013.12.011
Wei, Y.M., Fan, Y., Lu, C., Tsai, H.T., 2004. The assessment of vulnerability to natural disasters in China by using the DEA method. Environ. Impact Assess. Rev. 24, 427–439. doi:10.1016/j.eiar.2003.12.003
Westerink, J., Haase, D., Bauer, A., Ravetz, J., Jarrige, F., Aalbers, C., 2013. Dealing with Sustainability Trade-Offs of the Compact City in Peri-Urban Planning Across European City Regions. Eur. Plan. Stud. 21, 473–497. doi:10.1080/09654313.2012.722927
WHO, 2009. Night noise guidelines for Europe. Copenhagen. doi:10.1093/ejechocard/jer095

Wilkinson, C., Saarne, T., Peterson, G.D., Colding, J., 2013. Strategic spatial planning and the ecosystem services concept - An historical exploration. Ecol. Soc. 18, 37. doi:10.5751/ES-05368-180137

Wittig, V.E., Ainsworth, E.A., Naidu, S.L., Karnosky, D.F., Long, S.P., 2009. Quantifying the impact of current and future tropospheric ozone on tree biomass, growth, physiology and biochemistry: A quantitative meta-analysis. Glob. Chang. Biol. 15, 396–424. doi:10.1111/j.1365-2486.2008.01774.x

Wolch, J.R., Byrne, J., Newell, J.P., 2014. Urban green space, public health, and environmental justice: The challenge of making cities "just green enough." Landsc. Urban Plan. 125, 234–244. doi:10.1016/j.landurbplan.2014.01.017

Wolff, M., Haase, D., Haase, A., 2018. Compact or spread? A quantitative spatial model of urban areas in Europe since 1990. PLoS One 13, e0192326. doi:10.1371/journal.pone.0192326

Wolff, S., Schulp, C.J.E., Verburg, P.H., 2015. Mapping ecosystem services demand: A review of current research and future perspectives. Ecol. Indic. 55, 159–171. doi:10.1016/j.ecolind.2015.03.016

Woodruff, S.C., BenDor, T.K., 2016. Ecosystem services in urban planning: Comparative paradigms and guidelines for high quality plans. Landsc. Urban Plan. 152, 90–100. doi:10.1016/j.landurbplan.2016.04.003

Xiao, Q., McPherson, E.G., 2002. Rainfall interception by Santa Monica's municipal urban forest. Urban Ecosyst. 6, 291–302. doi:10.1023/B:UECO.0000004828.05143.67

Yang, L., Zhang, L., Li, Y., Wu, S., 2015. Water-related ecosystem services provided by urban green space: A case study in Yixing city (china). Landsc. Urban Plan. 136, 40–51. doi:10.1016/j.landurbplan.2014.11.016

Yanosky, T.M., 1982. Effects of flooding upon woody vegetation along parts of the Potomac River flood plain (Geological Survey professional paper n.1206). Washington.

Young, R.F., 2013. Mainstreaming urban ecosystem services: A national survey of municipal foresters. Urban Ecosyst. 16, 703–722. doi:10.1007/s11252-013-0287-2

Zardo, L., Geneletti, D., Pérez-Soba, M., Van Eupen, M., 2017. Estimating the cooling capacity of green infrastructures to support urban planning. Ecosyst. Serv. 26, 225–235. doi:10.1016/j.ecoser.2017.06.016

Zhao, C., Sander, H.A., Bond-Lamberty, B., 2015. Quantifying and mapping the supply of and demand for carbon storage and sequestration service from urban trees. PLoS One 10. doi:10.1371/journal.pone.0136392

Zheng, D., Ducey, M.J., Heath, L.S., 2013. Assessing net carbon sequestration on urban and community forests of northern New England, USA. Urban For. Urban Green. 12, 61–68. doi:10.1016/j.ufug.2012.10.003

Zulian, G., Paracchini, M.L., Maes, J., Liquete Garcia, M.D.C., 2013. ESTIMAP: Ecosystem services mapping at European scale. doi:10.2788/64713 (print); 10.2788/64369 (online)

Zulian, G., Stange, E., Woods, H., ..., Viinikka, A., 2017. Practical application of spatial ecosystem service models to aid decision support. Ecosyst. Serv. doi:10.1016/j.ecoser.2017.11.005

Appendices

Appendix A: Supplementary material to Chapter 2

TABLE A.1: *Coding table used for reviewing the inclusion of ES-related contents in urban plans.*

			urban ecosystem services								
			A	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**
			Food supply	**Water flow regulation and runoff mitigation**	**Urban temperature regulation**	**Noise reduction**	**Air purification**	**Moderation of environmental extremes**	**Waste treatment**	**Climate regulation**	**Recreation**
plan components	**b**	**information base**	plan content bA1	plan content bB1	plan content bC1	plan content bD1	plan content bE1	plan content bF1	plan content bG1	plan content bH1	plan content bI1
			plan content bA2	plan content bB2	plan content bC2	plan content bD2	plan content bE2	plan content bF2	plan content bG2	plan content bH2	plan content bI2
			plan content bAn	plan content bBn	plan content bCn	plan content bDn	plan content bEn	plan content bFn	plan content bGn	plan content bHn	plan content bIn
	o	**vision and objectives**	plan content oA1	plan content oB1	plan content oC1	plan content oD1	plan content oE1	plan content oF1	plan content oG1	plan content oH1	plan content oI1
			plan content oA2	plan content oB2	plan content oC2	plan content oD2	plan content oE2	plan content oF2	plan content oG2	plan content oH2	plan content oI2
			plan content oAn	plan content oBn	plan content oCn	plan content oDn	plan content oEn	plan content oFn	plan content oGn	plan content oHn	plan content oIn
	a	**actions**	plan content aA1	plan content aB1	plan content aC1	plan content aD1	plan content aE1	plan content aF1	plan content aG1	plan content aH1	plan content aI1
			plan content aA2	plan content aB2	plan content aC2	plan content aD2	plan content aE2	plan content aF2	plan content aG2	plan content aH2	plan content aI2
			plan content aAn	plan content aBn	plan content aCn	plan content aDn	plan content aEn	plan content aFn	plan content aGn	plan content aHn	plan content aIn

TABLE A.2*: Scoring table for the assessment of ES inclusion in urban plans.*

		urban ecosystem services								
		A	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**
		Food supply	**Water flow regulation and runoff mitigation**	**Urban temperature regulation**	**Noise reduction**	**Air purification**	**Moderation of environmental extremes**	**Waste treatment**	**Climate regulation**	**Recreation**
plan components	**information base**	*plan content* bA1	*plan content* bB1	plan content bC1	plan content bD1	plan content bE1	plan content bF1	plan content bG1	plan content bH1	plan content bI1
	vision and objectives	plan content bA2	plan content bB2	plan content bC2	plan content bD2	plan content bE2	plan content bF2	plan content bG2	plan content bH2	plan content bI2
	actions	plan content bAn	plan content bBn	plan content bCn	plan content bDn	plan content bEn	plan content bFn	plan content bGn	plan content bHn	plan content bIn

FIGURE A.1: *Overview of the number of ES considered in the reviewed urban plans broken down per plan component. See Table A.3 for plan ID.*

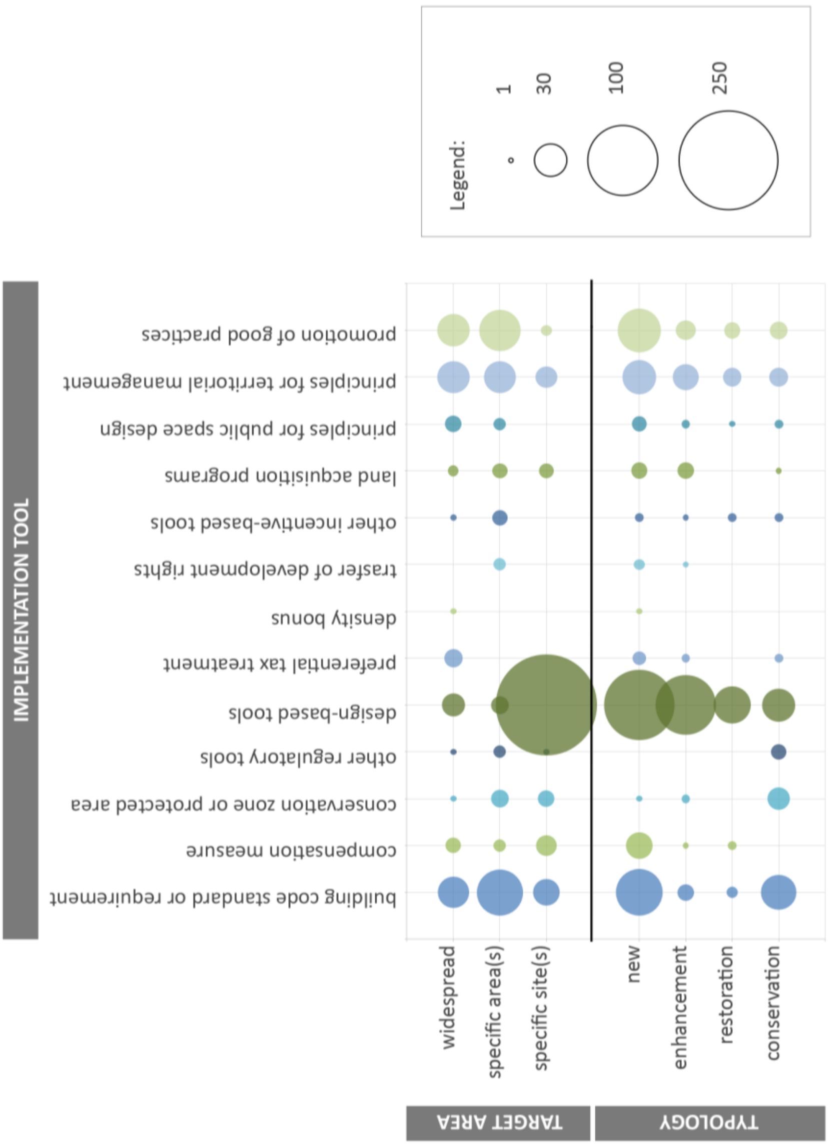


FIGURE A.2: *Frequency of occurrence of the different combinations of implementation tool/target area and implementation tool/typology in the whole sample of planning actions.*

TABLE A.3: *Sample of urban plans.*

ID	city	year of approval	population 1/1/2014 *	city area [km^2]
P01	Ascoli P.	2014	50,079	158.02
P02	Benevento	2012	60,770	130.84
P03	Brescia	2012	193,599	90.34
P04	Como	2013	84,834	37.12
P05	Cremona	2013	71,184	70.49
P06	Genoa	2014	596,958	240.29
P07	Lecco	2014	48,131	45.14
P08	Mantua	2012	48,588	63.81
P09	Milan	2012	1,324,169	181.67
P10	Padua	2014	209,678	93.03
P11	Pavia	2013	71,297	63.24
P12	Piacenza	2014	102,404	118.24
P13	Prato	2013	191,268	97.35
P14	Rovigo	2012	52,099	108.81
P15	Savona	2012	61,761	65.32
P16	Treviso	2013	83,145	55.58
P17	Trieste	2014	204,849	85.11
P18	Udine	2012	99,528	57.17
P19	Varese	2014	80,927	54.84
P20	Venice	2014	264,534	415.90
P21	Vercelli	2012	46,992	79.78
P22	Vibo V.	2014	33,675	46.57

Appendix B: Supplementary material to Chapter 5

TABLE B.1: *EU policy documents analysed. Group A: policy documents agreed by EU Member States Ministers.*

ID	Place and date	Title	Meeting and signatories
A1	Leipzig, September 1994	*European Spatial Planning*	Informal Council of Spatial Planning Ministers
A2	Potsdam, May 1999	*European Spatial Development Perspective – Towards Balanced and Sustainable Development of the Territory of the European Union*	Informal Council of Ministers responsible for Spatial Planning
A3	Hanover, September 2000	*Guiding Principles for Sustainable Spatial Development of the European Continent*	12th Session of the European Conference of Ministers responsible for Regional Planning
A4	Lille, November 2000	*Lille Action Programme*	Informal Council of Ministers responsible for urban affairs
A5	Rotterdam, November 2004	*Urban Acquis*	Informal Council of Ministers responsible for territorial cohesion
A6	Bristol, December 2005	*Bristol Accord*	Informal Council of Ministers on sustainable communities
A7	Leipzig, May 2007	*Leipzig Charter on sustainable European cities*	Informal Council Meeting of Ministers on urban development
A8	Leipzig, May 2007	*Territorial Agenda of the EU - Towards a More Competitive and Sustainable Europe of Diverse Regions*	the Informal Council of Ministers responsible for spatial planning and urban development
A9	Marseille, November 2008	*Marseille Declaration*	Informal Ministerial Meeting of Ministers responsible for urban development

TABLE B.1 *(continued)*

ID	Place and date	Title	Meeting and signatories
A10	Toledo, June 2010	*Toledo Declaration*	Informal Council Meeting of Ministers on urban development
A11	Gödöllő, May 2011	*Territorial agenda of the EU 2020*	Informal Ministerial Meeting of Ministers responsible for Spatial Planning and Territorial Development
A12	Riga, June 2015	*Declaration of Ministers towards the EU Urban Agenda*	Informal meeting of EU ministers responsible for Territorial Cohesion and Urban Matters
A13	Amsterdam, May 2016	*Urban Agenda for the EU 'Pact of Amsterdam'*	Informal Meeting of EU Ministers Responsible for Urban Matters

TABLE B.2: *EU policy documents analysed. Group B: communications from the European Commission.*

ID	Ref. number	Title
B1	COM(1997)197	*Towards an urban agenda in the European Union*
B2	COM(1998)605	*Sustainable Urban Development in the European Union: A Framework for Action*
B3	COM(2000)1100	*Communication laying down guidelines for a Community initiative concerning economic and social regeneration of cities and of neighbourhoods in crisis in order to promote sustainable urban development (URBAN II)*
B4	COM(2001)0264	*A Sustainable Europe for a Better World: A European Union Strategy for Sustainable Development (Commission's proposal to the Gothenburg European Council)*
B5	COM(2002)179	*Towards a Thematic Strategy for Soil Protection*
B6	COM(2004)60	*Towards a thematic strategy on the urban environment*
B7	COM(2005)0658	*Communication on the review of the Sustainable Development Strategy - A platform for action*

ID	Ref. number	Title
B8	COM(2005)670	*Thematic Strategy on the sustainable use of natural resources*
B9	COM(2005)0718	*Thematic Strategy on the Urban Environment*
B10	COM(2006)231	*Thematic Strategy for Soil Protection*
B11	COM(2008)0616	*Green Paper on Territorial Cohesion: Turning territorial diversity into strength*
B12	COM(2009)490	*Action Plan on Urban Mobility*
B13	COM(2010)2020	*EUROPE 2020: A strategy for smart, sustainable and inclusive growth*
B14	COM(2011)571	*Roadmap to a Resource Efficient Europe*
B15	COM(2012)042	*Proposal for a Decision of the European Parliament and of the Council on accounting rules and action plans on greenhouse gas emissions and removals resulting from activities related to land use, land use change and forestry*
B16	COM(2013)249	*Green Infrastructure (GI) — Enhancing Europe's Natural Capital*
B17	COM(2014)0490	*Communication on the urban dimension of EU policies – key features of an EU urban agenda*

TABLE B.3*: Strategies and related keywords.*

STRATEGY	KEYWORDS
compact city	compact(ness), concentrated (urban) development, expansion
urban regeneration	regeneration, revitalization, reconstruction, rehabilitation, brownfields, gap/neglected/abandoned areas/sites, inner-city development
functional mix	mix, mixture
no land take	land take, land/soil consumption, soil sealing
green city	green areas/spaces, natural areas/heritage, (urban) nature, (urban) biodiversity, (urban) ecosystems, green infrastructure, protection, conservation, renaturing, greening
high density	density, densification

Definition of land cover flows

The definition of land cover flows (LCF) is inspired by the system of land accounting developed by the European Environment Agency on the basis of the Corine Land Cover classification (EEA, 2006). A similar system was developed for the Urban Atlas database, taking into account the higher level of detail in the definition of urban LULC classes, hence the possibility of identifying a higher number of transition typologies. The following tables detail and provide a visual representation of the LULC classes involved in each type of LCF analysed in the paper (Figure 5.6). Rows represent LULC classes in the 2006 Urban Atlas database while columns are LULC classes in the 2012 database, as listed below (Table B.4). Crossed cells are LCF non-present in the analysis. An asterisk in the title or legend item identifies indicators listed in Table 5.2.

Table B.4: *Land use-land cover classes in the two editions of the Urban Atlas (UA) database.*

<table>
<tr><th></th><th>UA 2006</th><th>UA 2012</th></tr>
<tr><td>Continuous Urban Fabric</td><td>11100</td><td>11100</td></tr>
<tr><td>Discontinuous Dense Urban Fabric</td><td>11210</td><td>11210</td></tr>
<tr><td>Discontinuous Medium Density Urban Fabric</td><td>11220</td><td>11220</td></tr>
<tr><td>Discontinuous Low Density Urban Fabric</td><td>11230</td><td>11230</td></tr>
<tr><td>Discontinuous Very Low Density Urban Fabric</td><td>11240</td><td>11240</td></tr>
<tr><td>Isolated structures</td><td>11300</td><td>11300</td></tr>
<tr><td>Industrial, commercial, public, military and private units</td><td>12100</td><td>12100</td></tr>
<tr><td>Fast transit roads and associated land</td><td>12210</td><td>12210</td></tr>
<tr><td>Other roads and associated land</td><td>12220</td><td>12220</td></tr>
<tr><td>Railways and associated land</td><td>12230</td><td>12230</td></tr>
<tr><td>Port areas</td><td>12300</td><td>12300</td></tr>
<tr><td>Airports</td><td>12400</td><td>12400</td></tr>
<tr><td>Mineral extraction and dump sites</td><td>13100</td><td>13100</td></tr>
<tr><td>Construction sites</td><td>13300</td><td>13300</td></tr>
<tr><td>Land without current use</td><td>13400</td><td>13400</td></tr>
<tr><td>Green urban areas</td><td>14100</td><td>14100</td></tr>
<tr><td>Sports and leisure facilities</td><td>14200</td><td>14200</td></tr>
<tr><td>Arable land (annual crops)</td><td rowspan="5">20000</td><td>21000</td></tr>
<tr><td>Permanent crops</td><td>22000</td></tr>
<tr><td>Pastures</td><td>23000</td></tr>
<tr><td>Complex and mixed cultivation</td><td>24000</td></tr>
<tr><td>Orchards</td><td>25000</td></tr>
<tr><td>Forests</td><td>30000</td><td>31000</td></tr>
<tr><td>Herbaceous vegetation associations</td><td rowspan="3">20000</td><td>32000</td></tr>
<tr><td>Open spaces with little or no vegetation</td><td>33000</td></tr>
<tr><td>Wetlands</td><td>40000</td></tr>
<tr><td>Water</td><td>50000</td><td>50000</td></tr>
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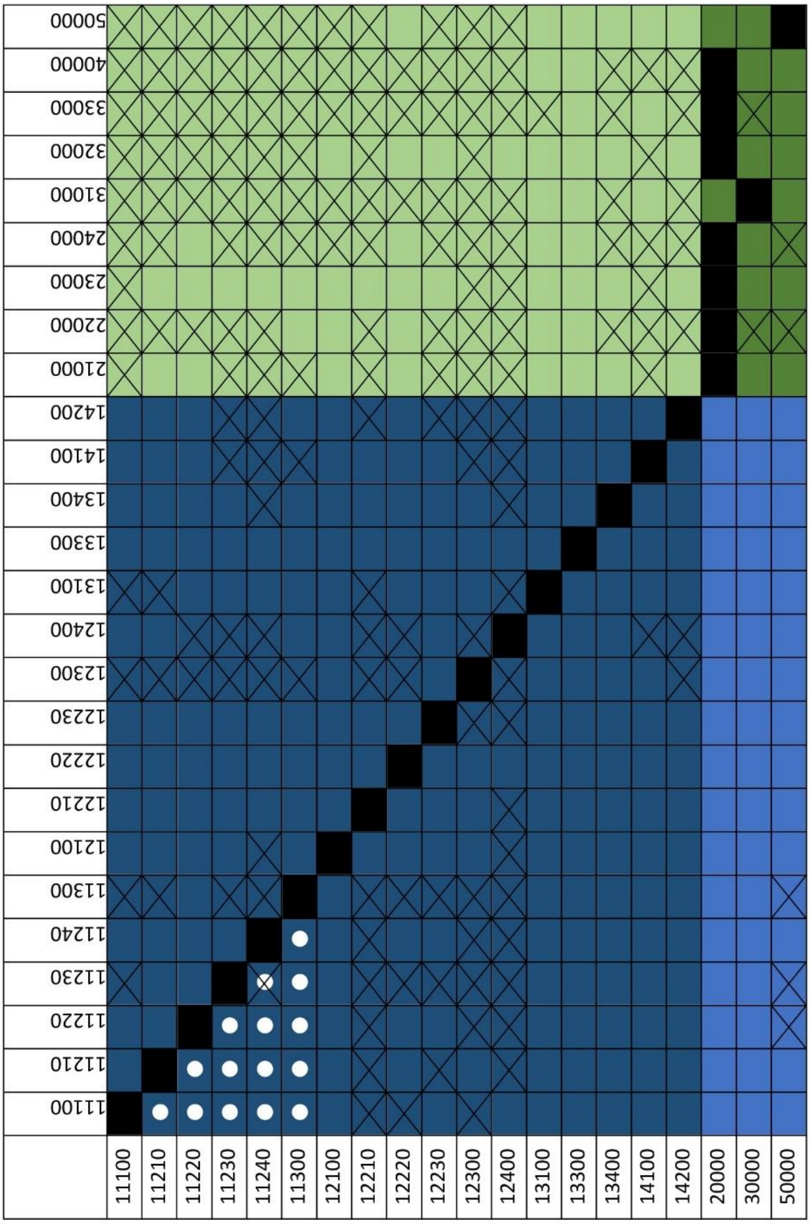
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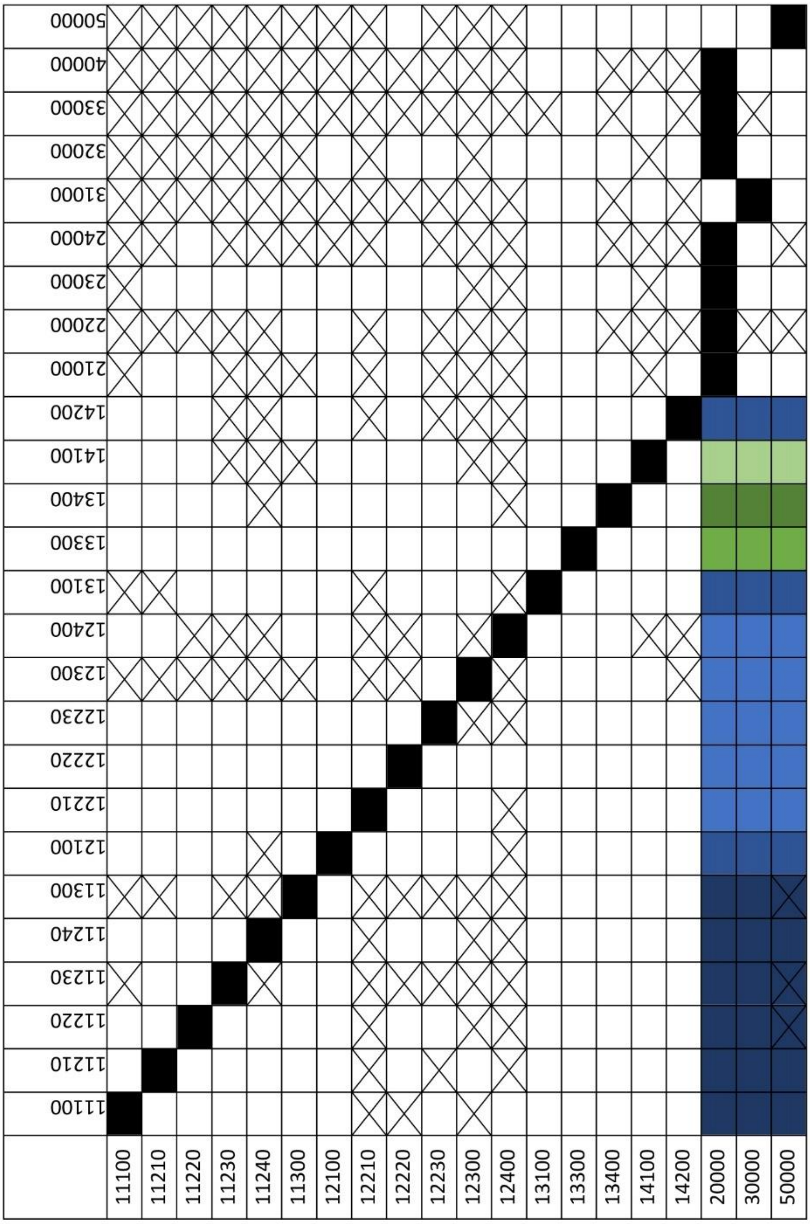
Total LULC flows (Figure 5.6a)

Legend

- conversion within non-urban uses
- conversion from urban to non-urban uses*
- new urbanization*
- recycling of urban land*
- residential densification*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32000
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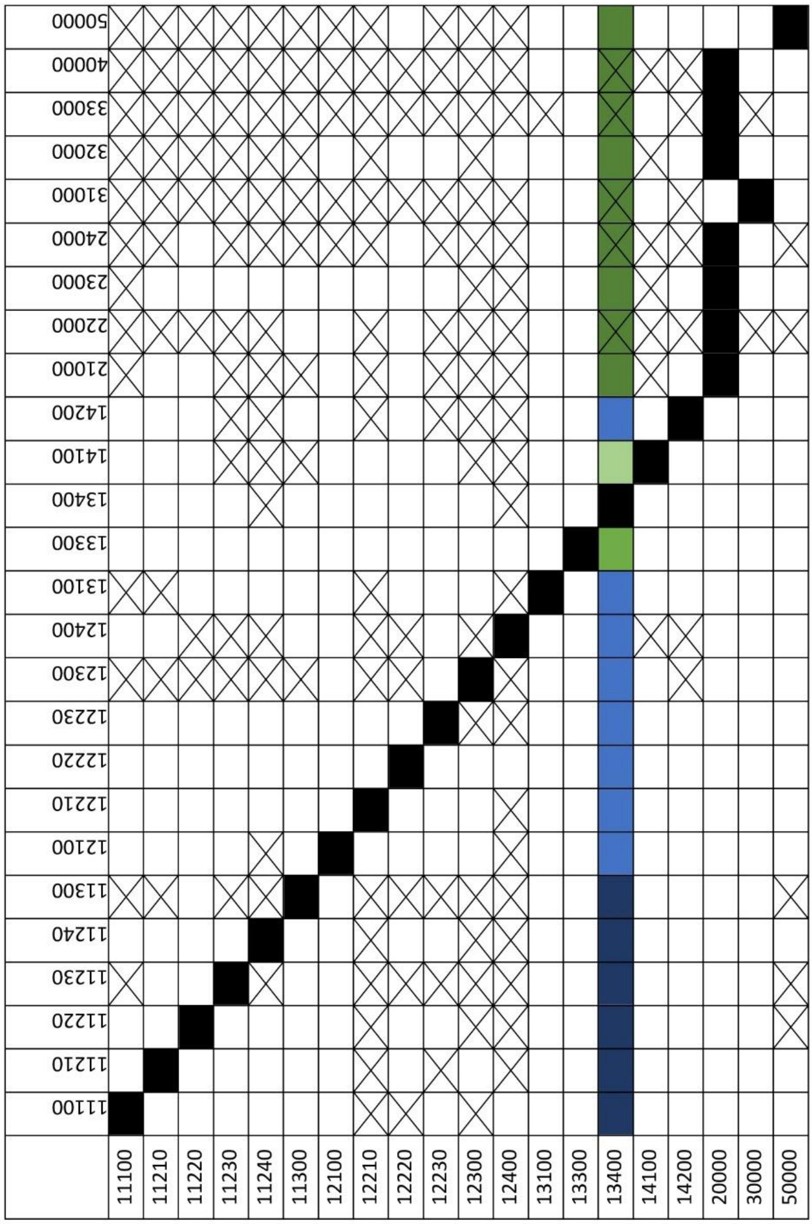
Land take (Figure 5.6b)

Legend

conversion to…

- green fragments without use*
- construction sites
- urban green areas
- infrastructures
- economic use
- residential use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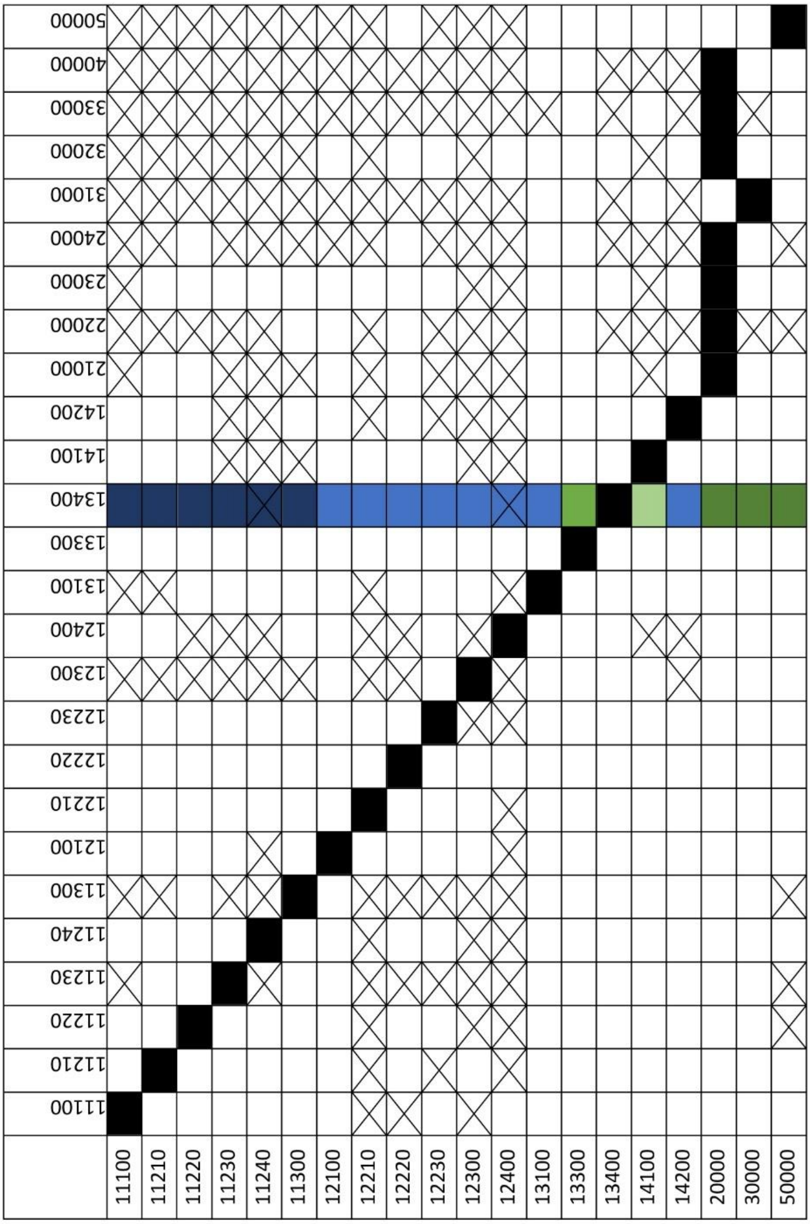
Infill development and re-use of brownfields* (Figure 5.6c)

Legend

conversion to…

- others - non-urban
- construction sites
- urban green areas
- economic use + infrastructures
- residential use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21000
22000
23000
24000
31000
32000
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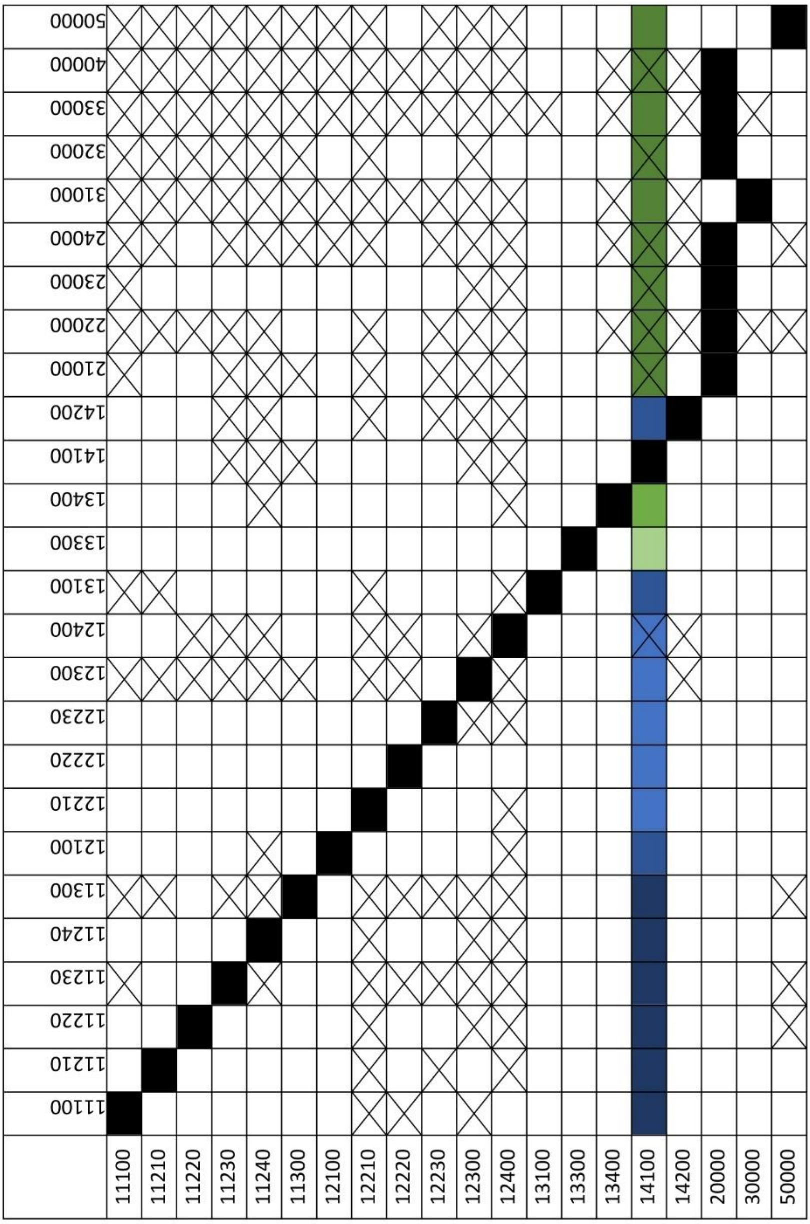
40000
50000
20000
30000

New 'land without current use' (Figure 5.6d)

Legend

conversion from…

- others - non-urban
- construction sites
- urban green areas
- economic use + infrastructures
- residential use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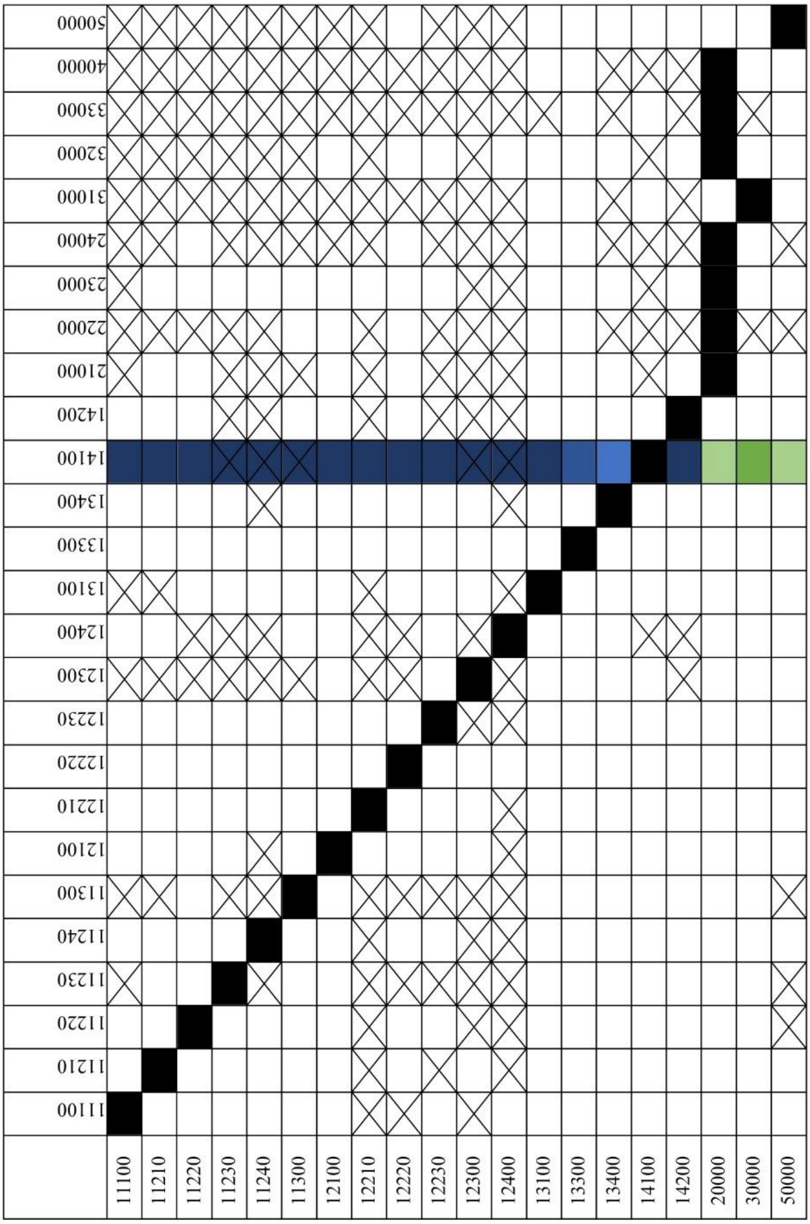
Loss of urban green areas* (Figure 5.6e)

Legend

conversion to…

- others – non-urban
- land without current use
- construction sites
- infrastructures
- economic use
- residential use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24000
31000
32000
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40000

New urban green areas* (Figure 5.6f)

Legend

conversion from…

- forests
- agricultural and semi-natural
- land without current use
- construction sites
- others - urban

TABLE B.5: *List of European cities analysed and main features considered in the study. The clusters refer to Figure 5.12 and are defined as follow: a) monofunctional densification, b) mixed densification, c) regeneration and expansion, d) low-density expansion, e) green de-densification, f) compactness only, g) pointless sprawl. * Population data not available for the analysed period (01/01/2007-01/01/2013). Data for the period 02/02/2006-01/01/2012 were used instead.*

Code	Country	City	Pop. dynamic	Region	N. of strategies	Cluster
AT001	AT	Wien	growing	W	3	a
AT002	AT	Graz	growing	W	3	b
AT003	AT	Linz	growing	W	3	c
AT004	AT	Salzburg	stable	W	2	c
AT005	AT	Innsbruck	growing	W	3	b
BE001	BE	Bruxelles / Brussel	growing	W	3	a
BE002	BE	Antwerpen	growing	W	2	a
BE004	BE	Charleroi	growing	W	2	d
BE005	BE	Liège	growing	W	3	b
BE007	BE	Namur	growing	W	3	b
BG001	BG	Sofia	growing	E	2	a
BG002	BG	Plovdiv	shrinking	E	2	d
BG003	BG	Varna	growing	E	4	b
BG005	BG	Pleven	shrinking	E	3	e
BG006	BG	Ruse	shrinking	E	2	d
BG007	BG	Vidin	shrinking	E	2	d
CY001	CY	Lefkosia	growing	S	3	b
CZ001	CZ	Praha	growing	E	1	f
CZ002	CZ	Brno	growing	E	2	d
CZ003	CZ	Ostrava	shrinking	E	2	c
CZ004	CZ	Plzen	growing	E	1	f
CZ005	CZ	Ústí nad Labem	shrinking	E	1	f
CZ006	CZ	Olomouc	stable	E	2	d
CZ007	CZ	Liberec	growing	E	2	d
CZ008	CZ	České Budějovice	shrinking	E	2	d
CZ009	CZ	Hradec Králové	shrinking	E	1	f
CZ010	CZ	Pardubice	stable	E	2	d
CZ011	CZ	Zlín	shrinking	E	2	d
CZ013	CZ	Karlovy Vary	shrinking	E	3	c
CZ014	CZ	Jihlava	stable	E	2	d
DE001	DE	Berlin	stable	W	2	c
DE003	DE	München	growing	W	3	a
DE004	DE	Köln	growing	W	2	a
DE005	DE	Frankfurt am Main	growing	W	4	c
DE007	DE	Stuttgart	stable	W	3	c

Code	Country	City	Pop. dynamic	Region	N. of strategies	Cluster
DE008	DE	Leipzig	growing	W	2	a
DE011	DE	Düsseldorf	growing	W	4	c
DE012	DE	Bremen	stable	W	1	f
DE013	DE	Hannover	stable	W	1	f
DE014	DE	Nürnberg	shrinking	W	2	c
DE017	DE	Bielefeld	stable	W	2	d
DE018	DE	Halle an der Saale	shrinking	W	1	d
DE020	DE	Wiesbaden	shrinking	W	2	d
DE021	DE	Göttingen	shrinking	W	1	f
DE025	DE	Darmstadt	growing	W	4	c
DE026	DE	Trier	growing	W	5	c
DE028	DE	Regensburg	growing	W	4	b
DE029	DE	Frankfurt (Oder)	shrinking	W	2	d
DE030	DE	Weimar	shrinking	W	3	c
DE031	DE	Schwerin	shrinking	W	2	d
DE033	DE	Augsburg	growing	W	4	c
DE034	DE	Bonn	shrinking	W	3	c
DE035	DE	Karlsruhe	growing	W	3	a
DE036	DE	Mönchengladbach	shrinking	W	2	d
DE037	DE	Mainz	growing	W	4	b
DE039	DE	Kiel	growing	W	2	a
DE040	DE	Saarbrücken	stable	W	2	d
DE042	DE	Koblenz	growing	W	3	b
DE546	DE	Wuppertal	shrinking	W	2	d
EE001	EE	Tallinn	growing	N	1	f
EE002	EE	Tartu	growing	N	2	e
ES003	ES	Valencia	stable	S	4	c
ES004	ES	Seville	stable	S	2	e
ES005	ES	Zaragoza	growing	S	4	c
ES006	ES	Málaga	growing	S	3	e
ES007	ES	Murcia	growing	S	4	c
ES008	ES	Las Palmas	growing	S	3	e
ES009	ES	Valladolid	shrinking	S	3	e
ES013	ES	Oviedo	growing	S	3	e
ES014	ES	Pamplona / Iruña	growing	S	2	d
ES016	ES	Toledo	growing	S	3	e
ES021	ES	Alicante / Alacant	growing	S	3	b
FI001	FI	Helsinki	growing	N	3	a
FI003	FI	Turku	growing	N	2	d
FR001	FR	Paris*	growing	W	4	b
FR008	FR	Nantes*	growing	W	4	b
FR012	FR	Le Havre*	shrinking	W	2	d

Code	Country	City	Pop. dynamic	Region	N. of strategies	Cluster
FR016	FR	Nancy*	shrinking	W	4	c
FR019	FR	Orléans*	growing	W	2	d
FR203	FR	Marseille*	growing	W	4	c
HU001	HU	Budapest	growing	E	3	a
HU004	HU	Pécs	shrinking	E	2	d
HU006	HU	Szeged	shrinking	E	0	g
HU007	HU	Győr	stable	E	1	f
HU008	HU	Kecskemét	growing	E	2	d
IT001	IT	Roma	shrinking	S	3	e
IT003	IT	Napoli	shrinking	S	2	c
IT004	IT	Torino	shrinking	S	3	c
IT008	IT	Bari	shrinking	S	2	d
IT010	IT	Catania	shrinking	S	2	d
IT012	IT	Verona	shrinking	S	3	e
IT013	IT	Cremona	growing	S	2	d
IT017	IT	Ancona	shrinking	S	2	d
IT019	IT	Pescara	shrinking	S	3	e
IT021	IT	Caserta	shrinking	S	2	d
IT025	IT	Reggio di Calabria	shrinking	S	2	d
LT001	LT	Vilnius	shrinking	N	3	c
LT002	LT	Kaunas	shrinking	N	1	f
LT003	LT	Panevėžys	shrinking	N	2	c
LU001	LU	Luxembourg*	growing	W	4	c
LV001	LT	Riga	shrinking	N	2	c
LV002	LT	Liepāja	shrinking	N	2	c
MT001	MT	Valletta	growing	S	5	c
NL002	NL	Amsterdam	growing	W	4	c
NL006	NL	Tilburg	growing	W	2	d
NL008	NL	Enschede	growing	W	3	e
NL009	NL	Arnhem	growing	W	4	b
NL010	NL	Heerlen	shrinking	W	2	d
NL012	NL	Breda	growing	W	3	b
NL013	NL	Nijmegen	growing	W	1	f
NL014	NL	Apeldoorn	growing	W	2	d
NL015	NL	Leeuwarden	growing	W	2	d
PL002	PL	Łódź	shrinking	E	2	d
PL003	PL	Kraków	stable	E	2	d
PL004	PL	Wrocław	stable	E	2	d
PL006	PL	Gdańsk	growing	E	2	d
PL007	PL	Szczecin	stable	E	2	d
PL009	PL	Lublin	shrinking	E	2	d
PL010	PL	Katowice	shrinking	E	2	c

Code	Country	City	Pop. dynamic	Region	N. of strategies	Cluster
PL012	PL	Olsztyn	shrinking	E	2	d
PL014	PL	Opole	stable	E	2	d
PL016	PL	Gorzów Wielkopolski	shrinking	E	3	e
PL017	PL	Zielona Góra	stable	E	2	d
PL018	PL	Radom	stable	E	1	f
PL025	PL	Płock	shrinking	E	2	d
PL026	PL	Kalisz	shrinking	E	2	d
PL027	PL	Koszalin	shrinking	E	1	f
PL028	PL	Lisboa	growing	E	2	d
PT001	PT	Porto	growing	S	3	a
PT002	PT	Coimbra	growing	S	3	a
PT005	PT	Faro	growing	S	2	d
PT009	PT	București	growing	S	4	b
RO001	RO	Cluj-Napoca	growing	E	2	a
RO002	RO	Timișoara	growing	E	2	d
RO003	RO	Craiova	growing	E	4	b
RO004	RO	Brăila	growing	E	2	d
RO005	RO	Oradea	stable	E	2	d
RO006	RO	Bacău	growing	E	3	e
RO007	RO	Arad	growing	E	3	b
RO008	RO	Târgu Mureș	growing	E	2	d
RO010	RO	Piatra Neamț	growing	E	3	e
RO011	RO	Călărași	growing	E	4	b
RO012	RO	Giurgiu	growing	E	4	b
RO013	RO	Alba Iulia	growing	E	1	d
RO014	RO	Ljubljana	growing	E	2	d
SI001	SI	Maribor*	growing	E	3	b
SI002	SI	Bratislava	stable	E	4	c
SK001	SK	Košice	shrinking	E	2	d
SK002	SK	Banská Bystrica	growing	E	2	d
SK003	SK	Nitra	shrinking	E	2	d
SK004	SK	Prešov	shrinking	E	2	d
SK005	SK	Žilina	stable	E	2	d
SK006	SK	Trnava	shrinking	E	0	g
SK007	SK	Trenčín	shrinking	E	1	f
SK008	SK	Liverpool	shrinking	E	2	d
UK006	UK	Edinburgh	growing	W	3	a
UK007	UK	Manchester	growing	W	3	a
UK008	UK	Cardiff	growing	W	4	a
UK009	UK	Sheffield	growing	W	2	a
UK010	UK	Bristol	growing	W	4	c
UK011	UK	Belfast	growing	W	3	a

Code	Country	City	Pop. dynamic	Region	N. of strategies	Cluster
UK012	UK	Leicester	growing	W	3	a
UK014	UK	Derry	growing	W	3	a
UK015	UK	Aberdeen	stable	W	2	d
UK016	UK	Cambridge	growing	W	3	b
UK017	UK	Exeter	growing	W	2	b
UK018	UK	Lincoln	growing	W	4	a
UK019	UK	Wrexham	growing	W	3	a
UK022	UK	Portsmouth	growing	W	3	b
UK023	UK	Worcester	growing	W	2	a
UK024	UK	Coventry	growing	W	2	a
UK025	UK	Kingston upon Hull	growing	W	4	c
UK026	UK	Stoke-on-Trent	stable	W	2	c
UK027	UK	Nottingham	growing	W	3	a
UK029	UK	Olsztyn	growing	W	3	a

Table B.6: *Main statistics for the indicators in the different categories of European cities.*

a) Edge Density (ED)

	all		**growing**		**shrinking**	
	2006	2012	2006	2012	2006	2012
mean	151.2	146.0	149.9	144.0	158.5	154.2
st. dev.	109.2	104.6	127.3	121.0	77.7	75.6
max	630.7	588.2	630.7	588.2	440.2	434.2
min	9.0	9.0	9.0	9.0	50.3	49.5
range	621.8	579.2	621.8	579.2	389.9	384.7

	E		**N**		**S**		**W**	
	2006	2012	2006	2012	2006	2012	2006	2012
mean	160.8	154.7	114.4	110.6	253.9	242.6	109.1	106.8
st. dev.	71.3	67.3	61.4	59.9	172.6	164.9	80.4	79.3
max	420.4	413.1	243.4	235.2	630.7	588.2	470.1	467.1
min	40.3	40.0	56.7	54.7	40.9	36.3	9.0	9.0
range	380.1	373.0	186.7	180.5	589.8	551.9	461.1	458.1

b) Interspersion and Juxtaposition Index (IJI)

	all		growing		shrinking	
	2006	2012	2006	2012	2006	2012
mean	55.7	57.4	54.2	55.9	57.8	59.6
st. dev.	5.6	6.0	5.8	6.5	4.2	3.9
max	64.5	67.4	62.8	67.4	64.5	66.6
min	30.5	30.9	30.5	30.9	44.6	48.7
range	33.9	36.5	32.3	36.5	19.9	17.9

	E		N		S		W	
	2006	2012	2006	2012	2006	2012	2006	2012
mean	57.1	59.2	59.4	58.5	51.5	55.8	55.5	56.3
st. dev.	5.2	5.6	4.2	4.6	5.0	6.2	5.4	6.0
max	64.5	67.4	64.3	63.7	59.7	65.9	62.7	64.3
min	38.5	37.2	51.6	51.1	41.3	43.1	30.5	30.9
range	25.9	30.2	12.6	12.5	18.4	22.7	32.1	33.4

c) Urban Area (% of city area)

	all		growing		shrinking	
	2006	2012	2006	2012	2006	2012
mean	46.8	47.8	50.3	51.3	41.2	42.1
st. dev.	20.0	20.0	22.4	22.4	14.8	14.8
max	97.6	97.6	97.6	97.6	82.0	82.6
min	13.2	13.7	13.2	13.8	17.6	17.9
range	84.4	83.9	84.4	83.8	64.4	64.7

	E		N		S		W	
	2006	2012	2006	2012	2006	2012	2006	2012
mean	37.0	38.4	54.8	56.1	42.3	43.6	55.7	56.3
st. dev.	12.6	12.8	12.4	13.0	23.9	24.2	20.1	20.1
max	73.9	75.5	68.2	72.0	86.7	88.0	97.6	97.6
min	13.2	13.8	34.3	35.7	13.4	14.8	13.6	13.7
range	60.7	61.7	34.0	36.4	73.3	73.2	84.0	83.9

d) Urban green areas (% of city area)

	all		growing		shrinking	
	2006	2012	2006	2012	2006	2012
mean	4.2	4.2	4.5	4.5	3.7	3.7
st. dev.	3.4	3.4	3.5	3.5	3.5	3.5
max	20.6	20.6	16.7	16.7	20.6	20.6
min	0.2	0.2	0.2	0.2	0.3	0.3
range	20.4	20.4	16.6	16.5	20.3	20.3

	E		N		S		W	
	2006	2012	2006	2012	2006	2012	2006	2012
mean	2.9	2.8	7.6	7.6	2.7	2.7	5.5	5.5
st. dev.	2.8	2.8	3.5	3.5	2.8	2.7	3.3	3.3
max	20.6	20.6	14.9	14.7	11.2	9.6	16.7	16.7
min	0.3	0.3	3.7	3.7	0.2	0.2	0.4	0.4
range	20.3	20.3	11.2	11.1	11.0	9.4	16.3	16.3

e) Per-capita urban green area (m^2)

	all		growing		shrinking	
	2006	2012	2006	2012	2006	2012
mean	22.8	22.6	20.9	20.0	25.7	26.7
st. dev.	20.4	21.0	11.5	11.1	32.3	33.5
max	237.8	244.1	53.2	50.9	237.8	244.1
min	2.2	2.2	2.2	2.2	4.0	4.2
range	235.6	241.9	50.9	48.7	233.8	239.9

	E		N		S		W	
	2006	2012	2006	2012	2006	2012	2006	2012
mean	21.1	21.1	39.5	41.6	10.3	10.4	26.8	26.1
st. dev.	29.4	30.2	15.4	16.7	4.9	4.8	10.6	10.8
max	237.8	244.1	65.5	75.4	20.3	19.6	59.1	63.6
min	2.2	2.2	23.6	25.0	3.0	3.1	8.1	7.8
range	235.6	241.9	41.9	50.4	17.2	16.5	51.1	55.7

f) Population density (p/ha)

	all		growing		shrinking	
	2006	2012	2006	2012	2006	2012
mean	42.7	42.3	43.9	44.8	39.7	37.3
st. dev.	22.0	22.4	24.6	25.2	14.4	14.1
max	212.4	218.1	212.4	218.1	100.5	98.1
min	14.6	14.0	14.6	14.0	17.5	17.0
range	197.8	204.1	197.8	204.1	83.0	81.1

	E		N		S		W	
	2006	2012	2006	2012	2006	2012	2006	2012
mean	42.3	41.2	35.9	33.3	56.7	54.9	38.6	39.5
st. dev.	16.0	17.0	6.9	7.5	28.8	27.9	22.8	23.9
max	109.8	117.9	44.3	46.9	139.4	136.5	212.4	218.1
min	14.6	14.0	20.6	20.3	20.2	19.9	18.0	18.5
range	95.3	103.9	23.7	26.5	119.2	116.5	194.4	199.7

g) Residential density (p/ha)

	all		growing		shrinking	
	2006	2012	2006	2012	2006	2012
mean	104.8	103.6	104.2	106.3	102.1	95.8
st. dev.	57.1	57.4	60.7	61.9	42.6	41.0
max	468.6	481.4	468.6	481.4	238.1	231.8
min	26.6	26.3	26.6	26.3	50.5	47.2
range	442.0	455.1	442.0	455.1	187.6	184.6

	E		N		S		W	
	2006	2012	2006	2012	2006	2012	2006	2012
mean	104.8	101.6	89.0	81.5	149.3	146.0	90.1	92.1
st. dev.	38.5	39.2	22.4	20.5	87.1	85.8	51.1	53.3
max	228.6	244.4	126.5	112.4	357.7	352.8	468.6	481.4
min	26.6	26.3	43.4	42.7	39.9	38.4	30.4	31.3
range	202.0	218.1	83.1	69.8	317.8	314.4	438.2	450.2

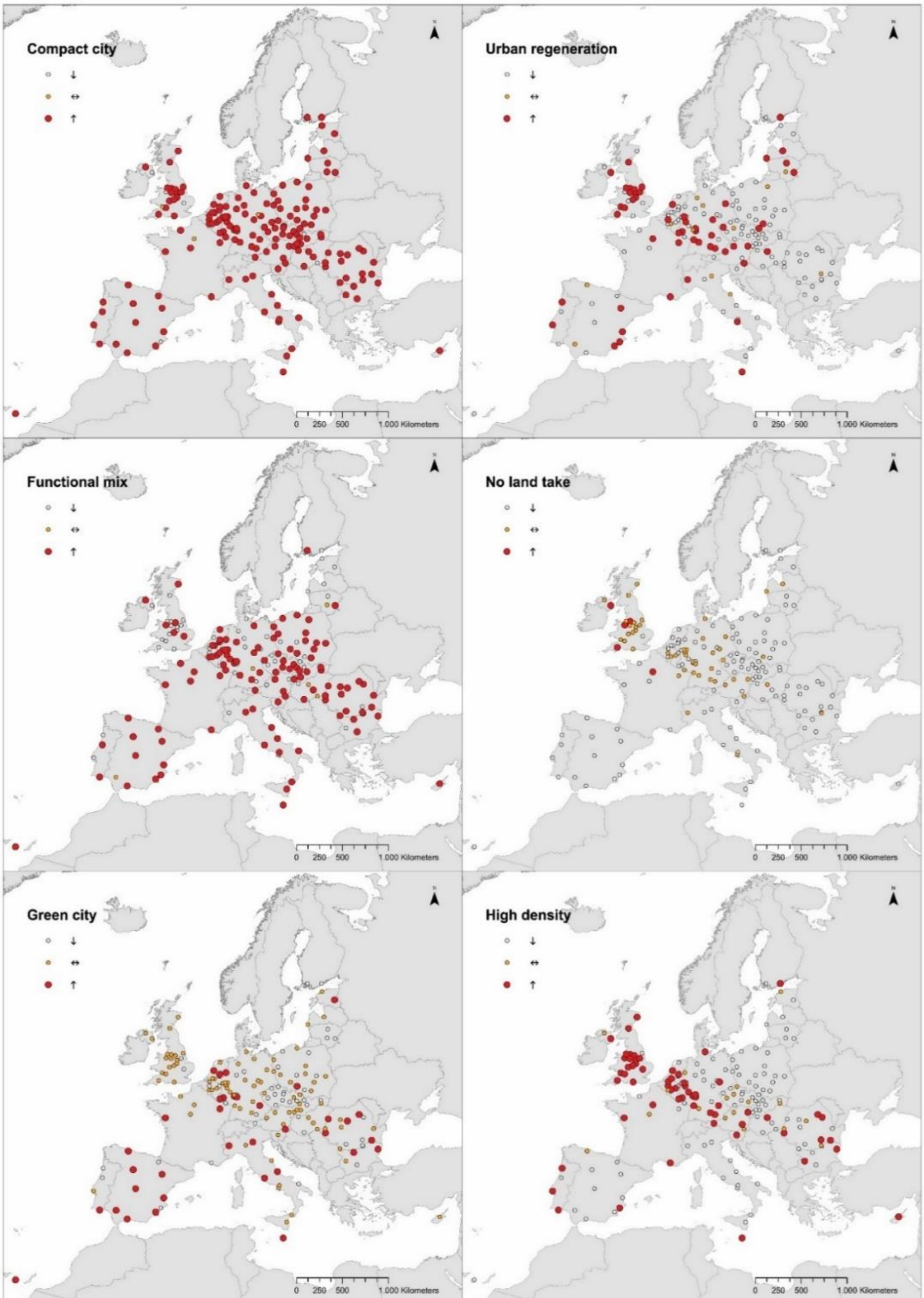

FIGURE B.1*: Comparison between the six main EU-level strategies and the observed spatial development trend in the sample of cities. Thresholds for the indicators as defined in Table 5.3.*

While ecosystem service knowledge has demonstrated to enhance decision-making at different levels, successfully managing the interface between science and policies is still a challenge. The thesis focuses on cities, and aims to explore the integration of ecosystem services in urban planning processes and tools. A preliminary review of recent urban plans reveals shortcomings in current practices and the potential benefits of a further integration.
At the conceptual level, the problem of integration is addressed by building a framework that shows the entry points and pathways through which planning actions affect the supply and demand of urban ecosystem services. The framework is applied to systematise a fragmented scientific evidence, and to select a set of indicators that planners can use to assess the impacts of planning decisions.
At the operational level, the integration of ecosystem service knowledge is tested in a real-life planning context dealing with the prioritization of brownfield regeneration scenarios in the city of Trento (Italy). Alternative scenarios are assessed based on the beneficiaries of two key ecosystem services, namely microclimate regulation and nature-based recreation, hence compared through a multi-criteria analysis that allows exploring multiple perspectives and balancing competing interests.
The last part of the thesis frames the integration of ecosystem services in urban planning in the wider context of spatial strategies for sustainable urban development. The six main spatial strategies agreed-upon at the European level, including the 'green city' strategy supported by the ecosystem service approach, are compared with the recent development trends of 175 European cities. The results reveal factors that may hinder the implementation of the strategies, as well as potential conflicts and trade-offs that should be carefully considered when aiming at a truly-sustainable urban development.

Chiara Cortinovis holds a MSc in Building Engineering/Architecture (2011) from the Polytechnic University of Milan, Italy. After a few years of experience as planning consultant for regional and local administrations, she dedicated to investigating the interface between science and policies. Her main interest is on how scientific knowledge can support planners and decision-makers in promoting more sustainable, livable, and resilient cities. Her current research explores the integration of ecosystem services in planning practices and tools, and the implementation of green infrastructure strategies at the urban scale.