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Characterisation, biophysical modelling and monetary valuation of urban nature-based solutions as a support tool for urban planning and landscape design.

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Par

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Characterisation, biophysical modelling and monetary valuation of urban nature-based solutions as a support tool for urban planning and landscape design.

Sous la direction de :
Guido SONNEMANN et Davide GENELETTI

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A differenza delle città ideali rappresentate nei quadri rinascimentali, completamente edificate e senza la benché minima presenza di un singolo filo d'erba, le città del futuro dovranno essere completamente ricoperte dalle piante.

Stefano Mancuso, *La Pianta del Mondo*, 2020 (1a ed.), Editori Gius. Laterza & Figli Spa, p. 44

Unlike the ideal cities represented in Renaissance paintings, completely built and without the slightest presence of a single blade of grass, the cities of the future will have to be completely covered with plants.

(English translation provided by Javier Babí Almenar)

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The work of this PhD thesis has been conducted at the Luxembourg Institute of Science & Technology (LIST), the University of Bordeaux and the University of Trento with the financial support of the H2020 Research and Innovation Project Nature4Cities (Grant No: 730468). Many people supported me on the development of this PhD research, whom I would like to thank briefly in the following lines.

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GLOSSARY

Acronym	Full name
A/C	Air Conditioning
AQI	Air Quality Index
AS	Asia
AW	Average Weighted
Aw	Topical Savannah with Dry Winter
BC	Betweenness centrality
CAQI	Common Air Quality Index
CELAC	Community of Latin American and Caribbean States
Cfa	Temperate without Dry Season and with Hot Summer
Cfb	Temperate without Dry Season and without Hot Summer
CICES	Common International Classification of Ecosystem Services
CONTAG	Contagion index
CORE	Core area
Csa	Mediterranean Hot Summer
CV	Coefficient of Variation
Dbh	Diameter at Breast Height
DSS	Decision Support System
EAQI	European Air Quality Index
EC	European Commission
ECC	Equivalent connected area
ED	Edge density
ENN	Euclidean nearest-neighbour distance
ES	Ecosystem Service(s)
EU	European Union
GDP	Gross Domestic Product
GNI	Gross National Income
HI	High Income
IIC	Integral index of connectivity
LA	Community of Latin American and Caribbean States
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCSA	Life Cycle Sustainability Assessment
LIST	Luxembourg Institute of Science & Technology
LMA	Large Metropolitan Area
LMI	Low Medium Income
LPI	Largest patch index
LTER	Long-term Ecological Research
IUCN	International Union for Conservation of Nature
LULCC	Land Use Land Cover Classes
MA	Metropolitan Area

MAES	Mapping and Assessment of Ecosystems and their Services
MC	Medium City
MEA	Millennium Ecosystem Assessment
MESH	Effective mesh size
MLavg	Measures by Landscape - average
MLmax	Measures by Landscape - maximum
MLmin	Measures by Landscape - minimum
MP	Measures by Pixel
NBS	Nature-based Solutions
NESCS	USA National Ecosystem Services Classification System
NLSI	Normalised landscape shape index
PC	Probability of connectivity
PD	Patch density
PI	Public Institution
PLAND	Percent land area
RA	Rest of America
S	Scenarios
SA	Scientific Articles
SC	Small City
SD	Species Distribution
SHAPE	Shape Average Weighted & Shape Coefficient of Variation
SHDI	Shannon's diversity index
SUDS	Sustainable Urban Drainage Systems
SWMM	Storm water management model
TEEB	The Economics of Ecosystems and Biodiversity
UC	Urban Challenge
USC	Urban Sub-Challenge
UK NEA	United Kingdom National Ecosystem Assessment
UMI	Upper Medium Income
UN	United Nations
UP	Urban Planning Reports
USLE	Universal Soil Loss Equation
VSC	Very Small City
WFD	Water Framework Directive
WG	Working Group
WWQM	Wetlands and Water Quality Model

Executive Summary:

The recognition of nature in the resolution of societal challenges has been growing in relevance. This recognition has been associated with the development of new concepts from science and policy such as natural capital, ecosystem services, green infrastructure, and more recently Nature-Based Solutions (NBS). NBS intends to address societal challenges in an effective and adaptive form providing economic, social, and environmental benefits. The overall aim of this PhD thesis is to develop an environmental and economic assessment of NBS for highly urbanised territories based on rationales and models underpinning ecosystem services, urban/landscape ecology, and life cycle thinking approaches. This combined evaluation approach would help to better understand if NBS are cost-effective or not. The aim is developed according to four specific objectives.

The first objective corresponds to the characterisation of NBS in relation to urban contexts and the problematics that they can help to address or mitigate. To achieve this objective a critical review on the study of the relationship between NBS, ecosystem services (ES) and urban challenges (UC) was developed. As a main output, a graph of plausible cause-effect relationships between NBS, ES and UC is obtained. The graph represents a first step to support sustainable urban planning, moving from problems (i.e. urban challenges) to actions (i.e. NBS) to resolutions (i.e. ES).

The second objective corresponds to the definition of an adequate set of biophysical and monetary assessment methods and indicators to evaluate the value of NBS in urbanised contexts. To achieve this objective, a review of existing methods on ecosystem services valuation, life cycle cost analysis and life-cycle assessment are developed. The review takes into account specific constraints such as easiness to use and availability of data. At the end, potential methods and indicators were selected, which will be later integrated in the combined assessment framework.

The third objective corresponds to the design of a combined assessment framework integrating methods from life cycle assessment, landscape/urban ecology and ecosystem services that quantifies the environmental and economic value of NBS informing about the cost-effectiveness of its entire life cycle. To achieve this objective, a conceptual framework is developed. From it, a system dynamics model of ecosystem (dis)services is developed and coupled with a life cycle assessment method. The combined evaluation is tested with a relevant NBS type (i.e. urban forest) in a case study in the metropolitan area of Madrid.

The fourth objective is the development of a decision support (DSS) tool that integrates the assessment framework as part of iterative design processes in urban planning and landscape design. The DSS intends to enhance the interrelation between *science*, *policy* and *planning/design*. To achieve this objective a user-friendly web-based prototype DSS on NBS, called NBenefit$®, is developed. The prototype DSS provides the user a simple form of quantifying the provision of multiple ES and costs over the entire life cycle (implementation, operational life, and end-of-life) of NBS.

This thesis contributed to the characterisation of NBS and its environmental and economic assessment to inform urban planning and landscape design processes, allowing decisions that are more informed.

Résumé:

La reconnaissance de la nature dans la résolution des défis de société est de plus en plus pertinente. Cette reconnaissance a été associé au développement de nouveaux concepts issus de la science et de la politique tels que le capital naturel, les services écosystémiques, les infrastructures vertes et plus récemment les solutions fondées sur la nature (NBS). NBS a l'intention de résoudre des problématiques the développement durable sous une forme efficace et adaptative offrant des avantages économiques, sociaux et environnementaux. L'objectif général de cette thèse de doctorat est de développer une évaluation environnementale et économique des NBS pour les territoires fortement urbanisés basée sur les approches des services écosystémiques, l'écologie urbaine / paysagère et l'analyse du cycle de vie. Cette approche d'évaluation combinée aiderait à comprendre si les NBS sont coût efficient ou non. L'objective général est développé selon quatre objectifs spécifiques.

Le premier objectif spécifique correspond à la caractérisation des NBS par rapport aux contextes urbains et ses problématiques qu'ils peuvent aider à résoudre ou atténuer. Pour atteindre cet objectif, une revue critique de la littérature scientifique sur la relation entre les NBS, les services écosystémiques (ES) et les problématiques urbaines (UC) a été développée. Le résultat principal est un graphique des relations de cause à effet plausibles entre NBS, ES et UC. Le graphique représente une première étape pour soutenir la planification urbaine durable, passant des problématiques urbaines aux actions (NBS) aux résolutions (ES).

Le deuxième objectif spécifique correspond à la définition d'un ensemble adéquat de méthodes d'évaluation et indicateurs pour estimer la performance biophysique et monétaire des NBS, en particulier dans les zones urbaines. Pour atteindre cet objectif, une revue des méthodes existantes sur l'évaluation des services écosystémiques et l'analyse du cycle de vie et l'analyse du cout de vie. La revue prend en compte des contraintes spécifiques telles que la facilité d'utilisation et la disponibilité des données. À la fin, des méthodes et des indicateurs spécifiques ont été sélectionnés en fonction.

Le troisième objectif spécifique correspond à la conception d'un cadre d'évaluation combiné intégrant des méthodes d'analyse du cycle de vie, d'écologie du paysage et écologie urbaine et des services écosystémiques. Ces méthodes quantifient les valeurs des NBS en unités biophysiques et monétaires en informant sur leur coût efficience. Pour atteindre cet objectif spécifique, un cadre conceptuel est développé. À partir de là, un modèle de dynamique de système des services et mauvais écosystémiques est développé et couplé à une méthode d'évaluation du cycle de vie. L'évaluation combinée est testée avec un type de NBS pertinent (une forêt urbaine) dans un projet d'espace vert à l'intérieur de la région métropolitaine de Madrid.

Le quatrième objectif spécifique correspond au développement d'un outil d'aide à la décision (DSS) en ligne qui intègre le cadre d'évaluation les processus de conception itératifs dans la planification urbaine et aménagement paysagère. Le DSS vise à améliorer l'interrelation entre la science, la politique et la planification et dessin urbaine. Pour atteindre cet objectif, un prototype online DSS pour informer sur les NBS, appelé NBenefit$®, est développé. Le prototype fournit à l'utilisateur une forme simple de quantification de plusieurs ES et des coûts pour tout le cycle de vie des NBS.

Cette thèse a contribué à la caractérisation du NBS et à son évaluation environnementale et économique pour meilleure les processus de planification urbaine et d'aménagement du paysage, permettant une décision plus éclairée.

Riassunto:

La natura sta rivestendo un ruolo sempre più importante nella risoluzione delle sfide globali della società umana. Il riconoscimento di questo ruolo è legato allo sviluppo di nuovi concetti provenienti dalla scienza e dalla politica, come il capitale naturale, i servizi ecosistemici, le infrastrutture verdi e, più recentemente, le soluzioni basate sulla natura (NBS). Le NBS contribuiscono ad affrontare le sfide della società fornendo benefici economici, sociali e ambientali in una forma efficace e adattabile. Lo scopo della ricerca in questa tesi di dottorato è quello di sviluppare una valutazione ambientale ed economica delle NBS per territori altamente urbanizzati basata su logiche e modelli che hanno alla base i servizi ecosistemici, l'ecologia urbana e del paesaggio e approcci basati sul concetto di *life cycle*. Questo quadro di valutazione combinato mira a facilitare la comprensione del rapporto costo-efficacia delle NBS e del loro contributo allo sviluppo sostenibile e basato sul concetto di resilienza. Tale scopo della ricerca è stato sviluppato secondo quattro obiettivi specifici.

Il primo obiettivo è stato di caratterizzare le NBS in relazione ai contesti urbani e alle problematiche che queste possono permettere di affrontare. Per raggiungere questo obiettivo è stata redatta una revisione critica della letteratura relativamente allo studio della relazione tra NBS, servizi ecosistemici (ES) e sfide urbane (UC). Un grafico delle relazioni plausibili di causa-effetto tra NBS, ES e UC è stato ottenuto come risultato principale. Tale mappatura rappresenta un primo step a supporto della pianificazione urbana sostenibile, facilitando il passaggio dai problemi (es. UC) alle azioni (es. NBS), fino alle risoluzioni (es. ES).

Il secondo obiettivo è stato di definire un insieme di metodi e indicatori di valutazione biofisica e monetaria adeguati per la stima del valore ambientale ed economico delle NBS in contesti urbanizzati. Per raggiungere questo obiettivo è stata effettuata una revisione dei metodi esistenti sulla valutazione dei servizi ecosistemici e sull'analisi dei costi ambientali ed economici del ciclo di vita. La revisione ha tenuto conto di vincoli metodologici specifici come la facilità d'uso e la disponibilità dei dati. I metodi e gli indicatori selezionati attraverso quest'analisi sono stati successivamente integrati nel quadro di valutazione combinato.

Il terzo obiettivo ha riguardato la progettazione vera e propria del quadro di valutazione combinato, che integra metodi di valutazione del ciclo di vita, ecologia del paesaggio / urbana e servizi ecosistemici, e quantifica infine il valore ambientale ed economico della NBS fornendo informazioni sulla sua efficacia in termini di costi lungo il ciclo di vita. Per raggiungere questo obiettivo è stato innanzitutto sviluppato un modello concettuale. A partire da esso è stato poi creato un modello di dinamica dei sistemi per calcolare il valore dei servizi (e disservizi) ecosistemici, integrato infine con un approccio di valutazione *life cycle*. Il funzionamento di questo quadro di valutazione combinata è stato testato modellizzando un tipo specifico di NBS (foresta urbana) e applicandolo a un caso di studio nell'area metropolitana di Madrid (Spagna).

Il quarto e ultimo obiettivo è stato di sviluppare uno strumento di supporto decisionale (DSS) capace d'integrare il quadro di valutazione proposto nella tesi all'interno di processi iterativi di progettazione nell'ambito della pianificazione urbana e del paesaggio. Questo DSS intende migliorare l'interrelazione tra scienza, politica e pianificazione / progettazione. Per raggiungere tale obiettivo è stato sviluppato NBenefit$®, un prototipo di DSS online di facile utilizzo per la valutazione delle NBS. NBenefit$® fornisce all'utente la possibilità di quantificare in maniera semplice i molteplici ES e costi (internalizzati o meno) legati alle NBS durante il loro intero ciclo di vita (implementazione, fasi operative e di fine vita).

In conclusione, questa tesi ha contribuito ad arricchire la conoscenza sugli strumenti di caratterizzazione delle NBS e sulla loro valutazione ambientale ed economica, in maniera tale da fornire maggiori elementi a supporto dei processi di pianificazione urbana e progettazione del paesaggio, e della presa di decisione sulla sostenibilità degli interventi basati su NBS.

Preface:

Working framework of the PhD thesis

The PhD student conducted this research inside the H2020 research project Nature4Cities[1], as part of the team of the Luxembourg Institute of Science and Technology (LIST). One of the main objectives of Nature4Cities is the development of a platform that integrates a set of assessment tools to assess the contribution of NBS to urban resilience and sustainability. The main responsibility of the PhD student was the development of an economic impact assessment tool, based on system dynamics, for the evaluation of nature-based solutions.

Besides Nature4Cities, the PhD research has been developed as a partnership between LIST (SUSTAIN Research Unit), the University of Bordeaux (CyVi Research Group) and the University of Trento (PLANES Research Group). Both universities do not form part of the consortium of Nature4Cities.

This PhD thesis presents the work developed by Javier Babí Almenar (the PhD student) and it is independent of (and might not be completely aligned with) Nature4Cities outputs. Some chapters are published or about to be submitted as collaborative peer-reviewed papers (or reports) with other researchers and under the oversight of the academic supervisors. In all these papers, the PhD student was the person responsible for the conceptualisation and development of the methodology, the calculation and discussion of the results, and the writing of the paper. Description of the specific roles of collaborators in chapters published (or about to be submitted) as a paper are provided in the first page of each chapter.

Peer review publications result of the PhD thesis

This PhD thesis is based on the following papers, which have either been published as peer-reviewed papers or are in preparation to be submitted:

[1] Complete name: *Nature-Based Solutions for re-naturing cities - knowledge diffusion and decision support platform through new collaborative models*; Research & Innovation Project H2020/SCC-03-2016.

· Babí Almenar, J., Rugani, B., Geneletti, D., Brewer, T. 2018. Integration of ecosystem services into a spatial planning framework based on a landscape ecology perspective. *Landscape Ecology,* 33(12): 2047-59.

· Babí Almenar, J., Bolowich, A., Elliot, T., Sonnemann, G., Geneletti, D., Rugani, B. 2019. Assessing habitat loss, fragmentation and ecological connectivity in Luxembourg to support spatial planning. *Landscape and Urban Planning,* 189: 335-351.

· Babí Almenar, J., Elliot, T., Rugani, B., Bodenan, P., Navarrete, T., Sonnemann, G., Geneletti, D. 2021. Nexus between nature-based solutions, ecosystem services and urban challenges. *Land Use Policy,* 100C, 104898.

· Babí Almenar, J., Petucco, C., Elliot, T., Sonnemann, G., Geneletti, D., Rugani, B. (In Preparation). A modelling framework to assess the costs and benefits of nature-based solutions: an application to urban forests. *Targeted journal: Ecosystem Services.*

· Babí Almenar, J., Navarrete, T., Petucco, C., Rugani, B. (In Preparation). A user-friendly decision support system to aid the planning and design of cost-effective nature-based solutions. *Targeted journal: Environmental Modelling & Software.*

Peer review publications outside the PhD thesis developed during the doctoral studies

Outside the scope of this PhD thesis, the following papers have been published as peer-review papers or are currently in review. These papers are the result of collaborations in other research projects and the supervision of an MSc student (last paper included) doing the daily work to define the basis for a new line of research at LIST:

· Elliot, T., Babí Almenar, J., Niza, S., Proença, V., Rugani, B. 2019. Pathways to modelling ecosystem services within an urban metabolism framework. *Sustainability,* 11(10) 2766.

Elliot, T., Bertrand, A., Babí Almenar, J., Petucco, C., Proença, V., Rugani, B. 2019. Spatial optimisation of urban ecosystem services through integrated participatory and multi-objective integer linear programming. *Ecological Modelling*, 4 108774.

· Katsou, E., Nika, C. E., Buehler, D., Marić, B., Megyesi, B., Mino, E., Babí Almenar, J. et al. 2020. Transformation tools enabling the implementation of nature-based solutions for creating a resourceful circular city. *Blue-Green Systems*, 2(1):186-211.

· Elliot, T., Rugani, B, Babí Almenar, J. 2020. Impacts of policy on urban energy metabolism at tackling climate change: the case of Lisbon. *Journal of Cleaner Production, 123510*

· Elliot, T., Babí Almenar, J., Rugani, B. 2020. Modelling the relationships between urban land cover change and local climate regulation to estimate urban heat island effect. *Urban For. & Urban Green.,* 50: 126650

- Elliot, T., Torres-Matallana, J.A., Goldstein, B., Babí Almenar, J. Gómez-Baggethun, E., Proença, V., Rugani, B. (In Review). An expanded framing of ecosystem services is needed for a sustainable urban future. *Scientific Reports.*

- Correa Hackenhaar, I., Babí Almenar, J., Elliot, T., Rugani, B. (In Review). A spatio-temporally differentiated product system modelling framework for consequential life cycle assessment. *Journal of Cleaner Production.*

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Chapter 1

Introduction†

1.1. Background

1.1.1. Sustainability, resilience, and their interrelation

Sustainability, or sustainable development, is usually understood as the development that fulfils the needs of the present without jeopardising the ability of future generations to achieve their own needs (Bruntland, 1987). As part of this concept, three dimensions (social, economic, and environmental) are considered, which are seen as interdependent and mutually reinforcing components of sustainable development (Kuhlman and Farrington, 2010). From an economic perspective, sustainability is related to the ability of non-decreasing the capacity of capita utility along time; which is composed of *produced, natural, human, and social* capital (Dietz and Neumayer, 2006). Whereas the environmental perspective emphasizes the relevance of thermodynamic principles, especially the irreversibility of increasing entropy as a consequence of development growth, which should tend to zero to move toward sustainability (Pulselli *et al.*, 2008). It also acknowledges the intrinsic value of different types of natural capital and the services they provide, which will affect the needs of future generations if they are lost. The social perspective introduces quality of life (or social well-being) as part of the needs of the present and future generations, and reinforces the social intrinsic value of natural capital (Pulselli *et al.*, 2008).

There are two main approaches to sustainability, weak sustainability and strong sustainability. Weak sustainability is aligned with neoclassical economy perspective. It states

† Chapter 1 (Section 1.2) is partially based on:

1. Babí Almenar, J., Rugani, B., Geneletti, D. & Brewer, T. Integration of ecosystem services into a conceptual spatial planning framework based on a landscape ecology perspective. *Landsc. Ecol.* 33, (2018).
2. Babí Almenar, J., Almenar, J.B., Elliot, T., Rugani, B., Bodenan, P., Gutierrez, T.N., Sonnemann, G. and Geneletti, D. Nexus between nature-based solutions, ecosystem services and urban challenges. *Land use policy* 100, 104898 (2021).

Roles of other authors in both papers (the second paper was published including a CReditT authorship contribution statement):
Benedetto Rugani, Davide Geneletti, Guido Sonnemann, and Tim Brewer were academic supervisors of those papers.
Thomas Elliot, and Philippe Bodénan contributed to the review and editing of paper 2.
Tomas Navarrete Gutierrez contributed to the training on the use of a graph-based software and helped with the visualisation of results.

that if the total capital is maintained above zero the natural capital can be exploited until its limits, since it is completely substitutable and only enough produced capital (of any type) should be generated to compensate future generations (Dietz and Neumayer, 2006; Pulselli *et al.*, 2008). Instead, strong sustainability is aligned with ecological economics perspective. It does not accept the substitution of natural capital by other types of capital (Dietz and Neumayer, 2006). It also acknowledges uncertainty and ignorance regarding the function of socio-ecological systems requiring application of the precautionary principle of Heisenber (Pulselli *et al.*, 2008).

The concept of resilience is usually tracked back to the ecological studies of Holling (1973) discussing the existence of multiple stability domains, and their relation to ecological processes, disturbances, and heterogeneity of temporal and spatial scales (Folke, 2006). Holling defined resilience as the capacity of an ecosystem to persist changes without moving to another stability domain (Holling, 1973). In later works, he differentiated two types of resilience: *engineering* resilience and *dynamic* resilience (Holling, 1996). The former refers to the capacity of the system to return to its previous state and the latter to the maintenance of basic functions of the system during disturbances (Meerow and Newell, 2016). Nowadays, resilience is usually described as the capacity of a system to absorb disturbances and re-organise during changes, meanwhile maintaining basic functional, structural, identity characteristics, and feedbacks (Walker *et al.*, 2004).

For some scholars, the concept of resilience, in a broad sense, can be understood as a component of sustainability (Pierce, Budd and Lovrich Jr, 2011; Childers *et al.*, 2015). Some authors consider both concepts as overlapping and treat them interchangeably, despite the advice of resilient scholars (Meerow, Newell and Stults, 2016; Elmqvist *et al.*, 2019). In certain cases, the two concepts could be conflicting with each other (Korhonen and Seager, 2008; Cascio, 2009; Meerow and Newell, 2016). This is why few scholars advocate differentiation of both paradigms, stating their connections and dissimilarities (Meerow and Newell, 2016).

Sustainability and resilience paradigms provide a holistic view (social, economic, environmental) and a strong focus on how to address changes in socio-ecological systems (Manyena, 2006; Bocchini *et al.*, 2014; Lizarralde *et al.*, 2015). However, sustainability is usually identified with strategies that look to enhance the efficiency of systems to achieve an optimal state and which aim to reduce negative impacts to prevent changes towards undesirable states (Bocchini *et al.*, 2014; Lizarralde *et al.*, 2015; Meerow and Newell, 2016). Instead, resilience aims for adaptation, resistance, transformation and recovery to face changes (Lizarralde *et al.*, 2015). Resilience emphasizes more the dynamic aspect of systems and the importance of functional redundancy (diversification) and connectedness as a key for rapid recovery and adaptation to uncertain changes, which might be reduced by fostering an increased efficiency (Korhonen and Snäkin, 2015; Meerow and Newell, 2016; Elmqvist, 2017). Consequently, resilience might be in conflict with traditional sustainability goals (Meerow, Newell and Stults, 2016), if the need to maintain redundancy of functions and connectedness are not explicitly stated. Hence, it is still not clear to whether resilience can be considered a factor of sustainability or not and to which extent the two concepts can be combined.

1.1.2. The relevance of sustainability and resilience for urbanised systems

In the last decades, the sustainability of cities and their role as enablers and barriers of global sustainability have become a main concern of researchers and policy makers (Wolfram and Frantzeskaki, 2016). From an environmental perspective, this emphasis is understandable taking into account that cities are responsible for more than 70% of world's greenhouse gas emissions and 80% of the world's energy consumption (UNDP, 2016). In addition, urban population accounts for more than 53% of the world's population (United Nations, 2018). In the case of the European Union, it accounts for 73% of the population, which will overpass the 80% by 2050 (Eurostat, 2015). The increase in urban population also explains the raising interest on urban sustainability from a social and economic perspective as well as on the resilience of urban areas to better face and adapt to unknown changes. This major concern is translated into the Sustainable Development Goals proposed by UN (2015). In fact, the eleventh goal (make cities inclusive, safe, resilient and sustainable) is specifically focused on cities, even when most of the other sustainability goals could also be applied to urban contexts.

The growth of urban population is expected to continue during the following decades together with the increasing contribution of urban systems to greenhouse gas emissions, other environmental pollutants, energy, material, and food consumption. Consequently, innovative solutions are required to face existing and emergent urban challenges (UC) related to the loss of ecosystem services, climate change effects, natural risks or human vulnerability to them, increase of public health issues, and a general loss of quality of life. According to Bettencourt et al (2007) the present situation demands for a sustainability transition, stabilisation of population, and enhancement of living standards, balancing human needs and the environmental carrying capacity.

In this sense, several authors state that cities could have the potential to remediate part of their own environmental impacts (sustainability) and increase their adaptive capacity (resilience) by:

i) minimising consumption of natural spaces and optimising urban form to protect ecosystems and human health (Wu, 2010);
ii) enhancing ecosystem services supply in urban contexts (Pincetl 2012; Pataki et al. 2011; Elmqvist et al. 2015); and
iii) using better their own and external natural resources, by optimising the urban metabolism through the inclusion of a life cycle thinking approach (Kennedy, Pincetl and Bunje, 2011; Pincetl, 2012).

As a consequence, the recognition of nature in the resolution of societal challenges (related to sustainability and resilience) has been growing in relevance. Besides ecosystem services and natural capital, this increasing interest has been also associated with the development of new concepts such as green infrastructure (Wang and Banzhaf, 2018), ecosystem-based approaches (Wamsler *et al.*, 2016), ecological engineering (Mitsch, 2012) and more recently nature-based solutions (NBS) (Raymond *et al.*, 2017)

1.1.3. Landscape ecology and urban ecology and their role for sustainable and resilient urbanised systems

In order to minimise consumption of natural spaces and optimise urban form, it is necessary to understand how the current landscape structure and changes on it influences different social and ecological processes. For this optimisation, we might need to look outside of urban areas and understand how they influence their hinterland to avoid missing important interactions (Wu, 2008). For example, through the study of habitats, landscape fragmentation, and ecological connectivity, we might better understand potential impacts of urban development on biodiversity conservation. In this sense, approaches and methods from landscape ecology could be instrumental, since the discipline studies the relationship between spatial patterns and ecological processes (Turner, 2005), being biodiversity conservation one of its key topics (Wu *et al.*, 2013).

In the same sense, to approach the optimisation of inner urban spaces from an urban ecology perspective might be quite useful. Under urban ecology, urban systems are usually studied from two paradigms, the ecology *in* cities and the ecology *of* cities (Coelho and Ruth, 2006; Wu, 2008, 2014; Breuste *et al.*, 2013; Alberti, 2016). The former focuses on studying distribution, diversity and changes in natural urban ecosystems or specific patches of them (Wu, 2014; McPhearson, Haase, *et al.*, 2016). Instead, the latter focuses on understanding the entire urban system as an ecosystem itself (Childers *et al.*, 2015; Pickett *et al.*, 2016).

Methods and approaches from ecology *in* cities could be useful to understand how urban climate, hydrology, soils, flora, fauna affect the ecological value of specific urban ecosystems and the ecosystem services derived from them (Coelho and Ruth, 2006; McPhearson, Pickett, *et al.*, 2016; Zhou, Pickett and Cadenasso, 2016). Instead, methods and approaches from ecology *of* cities and landscape ecology could provide a better understanding of urban areas as landscapes, seeing them and their hinterland as heterogeneous open spatially nested hierarchical patch systems (Wu 2014). As a result, ecology *in* cities can inform biodiversity conservation, landscape design and natural resource management of urban ecosystems (McPhearson, Pickett, *et al.*, 2016; Zhou, Pickett and Cadenasso, 2016). Meanwhile landscape ecology and ecology *of* cities can contribute to biodiversity conservation, urban planning and urban management works (Wu, 2014; Zhou, Pickett and Cadenasso, 2016). In both cases, methods and approaches can support the enhancement of the sustainability and resilience of urbanised contexts.

On the frontline of landscape ecology and urban ecology, scholars are pushing towards more integrated approaches. In landscape ecology, researchers advocate the expansion of the landscape ecology paradigm from *pattern:process* to *pattern:process:design* (Nassauer and Opdam, 2008). In other words, they stress the need for a more applied focus to make the discipline a suitable basis for the resolution of urban design/planning issues (Bastian, 2001; Termorshuizen and Opdam, 2009; Opdam, 2010). In the case of urban ecology, there is a movement to integrate ecology *in* cities and ecology *of* cities under a broader approach sometimes named ecology *for* cities (Childers *et al.*, 2015; McPhearson, Pickett, *et al.*, 2016). A movement towards ecology *for* cities would require expanding the field and interact with more disciplines. Fields such as industrial ecology or some of its approaches (i.e. urban metabolism, life cycle thinking) to study urban systems might help to provide more comprehensive urban ecology frameworks (McPhearson, Pickett, *et al.*, 2016). Like in the case of landscape ecology, a further integration with design sciences is also stressed. However, integration of ecological scientific knowledge into the design and planning of urban contexts is still in its infancy (Leitao and Ahern, 2002; Wu, 2010). Hence, to put ecological knowledge into practice, and enhance the sustainability and resilience of urbanised contexts, mutual learning and further collaboration between design/planning sciences and ecological sciences is necessary (Nassauer and Opdam, 2008; Childers *et al.*, 2015).

1.1.4. Ecosystem services and their role for sustainable and resilient urbanised systems

The Millennium Ecosystem Assessment (Alcamo et al., 2003; MEA, 2005) promoted the concept of ecosystem services (ES) in the policy agenda. It highlighted the capacity of ES to relate nature-human interactions and to show the relevance of nature for a sustainable maintenance of the main components of human well-being such as health, basic materials, and security. ES is already a central subject in the conservation biology and environmental science disciplines (de Groot, Wilson and Boumans, 2002; Wallace, 2007; Busch *et al.*, 2012). As part of a further collaboration between ecological sciences and design/planning sciences is starting to be more integrated in urban planning (Harris and Tewdwr-Jones, 2010; Geneletti, 2015; Cortinovis, 2017). However, while the integration of ES in urban planning is accelerating, key aspects on ES are still not agreed, standardized or advance enough, such as:

i) The classification of ES and a precise definition of its basic concepts (ecosystem functions, services, benefits and goods);
ii) The most appropriate units for ES accounting (Limburg et al., 2002; Balmford et al., 2008), or if social, biophysical and monetary units should be always used;
iii) The adequacy of ES as a stand-alone concept to fully inform practitioners and policy-makers without skewing interpretations (Potschin *et al.*, 2015; Schaubroeck, 2017).

1.1.4.1. Ecosystem services and their classification

Several ES classification systems exist (Haines-Young & Potschin, 2017). Some well-known examples are the Millennium Ecosystem Assessment classification (MEA) (MEA, 2005), The Economics of Ecosystems and Biodiversity (TEEB) (McVittie and Hussain, 2013), the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2018), the UK National Ecosystem Assessment (UKNEA, 2011), and the USA National Ecosystem Services Classification System (EPA, 2015). The diversity in classifications are justified by differences in frameworks, disciplinary approaches, and definition of the basic concepts (EPA, 2015; Haines-Young and Potschin, 2017). Moreover, they are also designed to better suit different purposes (Haines-Young & Potschin, 2014; Haines-Young & Potschin, 2017; Heink, Hauck, Jax, & Sukopp, 2016). However, the development of a common and rigorous ES classification and a clear differentiation of the basic concepts would improve the operationalisation of ES assessments.

Among the proposed classification systems, CICES has been already extensively used by scientists and policy makers to define and map ES indicators (Haines-Young and Potschin, 2017; La Notte et al., 2017). Moreover, this classification framework was initially proposed by the European Environment Agency and developed for the System of Integrated Environmental and Economic Accounting , and it is currently employed within the Mapping and Assessment of Ecosystem Services (MAES) reports (Haines-Young & Potschin, 2014; Haines-Young & Potschin, 2017). A straightforward comparison between CICES and the MEA and TEEB classifications is explicitly provided in the last version of CICES (v5.1), which can help to harmonise results from different studies.

For a clear differentiation of ecosystem function, services, goods and benefits, La Notte et al (2017) proposed a re-interpretation of the “cascade model” of Haines-Young and Potschin (2010). They applied it to CICES to enhance operationalisation of the ES categorisation. According to their work, an ecosystem function is defined as the set of interactions among components (biotic and abiotic) of ecosystems or biophysical structures, which may affect one or more ES. ES are defined as flows (e.g. carbon sequestration, water purification) generated by ecosystems, as a result of ecological processes and exchanges of information (e.g. genetic information, visual appreciation of natural features). Goods are represented by countable mass units and marketable resources (e.g. amount of biomass extracted from forest ecosystems, or fish resources) and the benefits as the contribution of these goods to a positive change in human well-being (e.g. availability of cleaner air or water).

1.1.4.2. The most appropriate units for ES accounting

Regarding the most appropriate ES unit, several studies translate ES into monetary units or integrate them into systems of economic accounting (Boyd and Banzhaf, 2007; Balmford *et al.*, 2008; de Groot *et al.*, 2010b; UKNEA, 2011, 2013; SEEA, 2012). These studies address issues, such as double counting, trade-offs, and the establishment of economic values for ES without markets (Balmford et al., 2008; Fisher et al., 2009; SEEA, 2012). Success in solving these problems can allow the use of comprehensive cost-benefit analyses during the decision making process (Busch *et al.*, 2012). It will also permit the establishment of clear relationships between economic activities and ecosystem functioning (Haines-Young, Potschin, & Kienast, 2012). However, according to several authors monetary valuation is limited with respect to its assumptions and scope and only partially expresses the value of ES (Limburg *et al.*, 2002; de Groot *et al.*, 2010a; Seppelt *et al.*, 2011).

The inclusion of ecological and socio-cultural values is recommended as part of ES valuations (de Groot *et al.*, 2010a) integrating the three dimensions considered in sustainability. However, a common unit for ES accounting of ecological and social values is still missing. Firstly, ES vary in typology, beneficiaries (at the level of society and biodiversity components) and scale of interest, making it difficult to develop a harmonized assessment framework based on common reference metrics. Secondly, while some ES can easily be quantified (e.g. cultivated terrestrial plants for nutrition purposes) and their values monitored over time, others (e.g. maintenance of nursery populations and habitats) are more difficult to value. Therefore, the use of multiple metrics based on *a priori* identification and definition of the ES beneficiaries is usually required (Kumar and Kumar, 2008; Chan et al., 2012; Martín-López et al., 2014; Scholte et al., 2015). Nevertheless, for ecological values biophysical capacity units are usually mentioned (Castro et al., 2014; Martín-López et al., 2014). Public perception is often used (Brown, 2013; Scholte et al., 2015) to support the development of indices to express social values (Sherrouse et al., 2011), such as Quality of Life indices (Fleury-Bahi et al., 2012; Hassine et al., 2014).

1.1.4.3. The adequacy of ES as a stand-alone concept

Environmental studies that are based only on the quantification and assessment of ES might suffer a limited understanding by stakeholders, especially those with a non-technical background. In fact, this issue was identified by Davies *et al.* (2017) as one of the main constraints of applying an ES approach to the management of urban forests. In addition, only assessing ES might offer a partial view or skew valuations, ignoring costs related to ecosystems (Lyytimäki and Sipilä, 2009). This potential skewness opens the question as to whether the use of additional concepts could mitigate limitations of ES as a concept. In a certain way, this issue is linked to the discourses on urban ecology and landscape ecology asking for a further integration of other disciplines, and therefore concepts.

To avoid a potential skewness in ES assessments, some authors propose accounting also for ecosystem disservices (i.e. nuisances, biological hazards, and geophysical hazards) derived from natural features (Shackleton et al., 2016; Schaubroeck, 2017). Some of these disservices (environmental, social or economic) might not occur once natural features are established. It might be that they occur during implementation or restoration phases of natural features or at the end of their life cycle. Methods from approaches such as life cycle thinking might be useful to assess disservices, since they are focused on quantifying flows, stocks and emissions over the entire life cycle of systems and products as well as their derived social, economic and environmental impacts. Hence, the accounting of both positive and negative impacts will allow a more comprehensive understanding of the net benefits derived from natural features.

1.1.5. Life cycle thinking and its role for sustainable and resilient urbanised systems

Life cycle thinking considers the impacts (environmental, social and/or economic) of a product or a system through its entire life cycle, going beyond the exclusive consideration of its use phase. Initially, life cycle thinking led to the development of life cycle assessment method (LCA) focused on environmental impacts. After the LCA method, international standards and guides for applying it were rapidly developed (e.g. ISO 14040, 1997; ISO 14049, 2000; ISO 14044, 2006). In the 1980s the economic dimension was incorporated introducing Life Cycle Cost Analysis (LCC) method, for which a recent guide exist from SETAC (Swarr *et al.*, 2011) as well as standards for specific sectors like buildings and constructed assets (ISO 15686, 2017). In the last decade, social life cycle assessment (S-LCA) methods have also been developed as well as guides for them (Benoît and Mazijn, 2009), which are currently being updated (i.e. v3 Draft UNEP S-LCA, open consultation held on February 2020). Recently, these three methods have been combined in the Life Cycle Sustainability Assessment (LCSA) framework (Valdivia, S. *et al.*, 2011; Valdivia *et al.*, 2013) to provide more holistic sustainability assessments. On the frontline of life cycle thinking, several scholars are focusing their efforts in the advancement of the LCSA approach (Zamagni, 2012; Sala, Farioli and Zamagni, 2013; Zamagni, Pesonen and Swarr, 2013). Others on the application of LCSA to urban sustainability assessments (Albertí *et al.*, 2017; Petit-Boix *et al.*, 2017). Among the emergent works, there are not works that have developed a practical integration of LCSA and ES for urban sustainability assessments.

1.1.5.1. Life Cycle Assessment

LCA is used to assess a broad range of environmental impacts of products, services, and entire systems considering all the life stages and the burden shifting between stages and types of environmental impacts (Bjørn *et al.*, 2018). The method makes use of quantitative data, combined with qualitative one, to calculate the input and outputs required by processes at different life cycle stages, aiding to understand the amount of resources and energy consumed as well as emissions generated (Tassielli, Renzulli and Notarnicola, 2016; Bjørn *et al.*, 2018). Four phases are defined as part of LCA studies (ISO 14040): goal and scope, life cycle inventory analysis, life cycle impact assessment, and interpretation. These phases are also applied in a similar way to LCC and S-LCA (Moltesen *et al.*, 2018; Rödger, Kjær and Pagoropoulos, 2018).

The first phase identifies the reason of the study, life cycle stages considered (e.g. from cradle to grave, from cradle to gate), the system boundaries, main assumptions, the intended audience and use of the results, the functions of the product or system assessed, and the functional unit. The functional unit permits a quantitative definition of the functions assessed, providing a reference to normalise the inputs and outputs accounted (Tassielli, Renzulli and Notarnicola, 2016). In addition, this phase also defines the allocation procedures for the environmental burden shifting when by-products are generated. During the definition of the system boundaries, two systems can be defined: the foreground system (main processes of interest) and the background system (auxiliary processes).

The life cycle inventory phase identifies the data for inputs and outputs and the calculation procedures. The collection of data could follow i) a process-based approach (use of inventories such as EcoInvent), ii) a “top-down” approach (use of extended economic input output tables) or, iii) a hybrid approach (Gemechu *et al.*, 2016). In addition, the inventory could be framed considering an attributional or a consequential approach. The former is focus on understanding the main sources of environmental impacts in a product system, without considering impacts derived from (in)direct effects on market interactions or the broader system (Vázquez-Rowe *et al.*, 2013). The latter, expands the product system to include market interactions, and therefore identifies and models processes in the background system induced by alterations made in the foreground system (JRC-IES, 2010).

The life cycle impact assessment phase uses the life cycle inventory results for doing the environmental assessment. This phase is composed by five steps: selection of impact categories, classification, characterisation, normalisation and weighting. Only the first three are mandatory. In the selection step, impact categories and their associated indicators are selected. The impact assessment can be done at midpoint or endpoint level. At midpoint level substance flows are grouped in categories that have the potential to contribute to similar environmental effects (e.g. global warming potential), whilst at end point level they are grouped in areas of protection (e.g. human health) affected by several environmental effects, representing the final damage to the environment (Rosenbaum *et al.*, 2018). In the classification step, the life cycle inventory data is assigned to each of the impact categories selected based on their potential contribution to types (categories) of environmental impacts. In the characterisation step, the contribution of each substance flow to each impact category is evaluated by making use of characterisation factors that convert the impact of substances to the common indicator used to represent each impact category selected (ISO 14044, 2006). In the normalisation step, the relative magnitude of the values for each impact category is normalised according to a relevant reference. Instead, in the weighting step, the values for each category are multiplied by a weight in relation to their importance, which permits an aggregation of all the impact results.

The interpretation phase identifies significant issues based on the previous two phases, evaluates completeness, sensitivity, uncertainty and consistency checks (Tassielli, Renzulli and

Notarnicola, 2016). In addition, in this phase a conclusion states limitations of the study and recommendations for decision makers.

There are recent advances in applying LCA to assess sustainability of urbanised contexts. For example, some scholars are working on the adaption of the methods to support land planning assessments (Loiseau *et al.*, 2013, 2014, 2018; Nitschelm *et al.*, 2016). Additionally, ES and the human impacts on ES are also starting to be integrated as part of LCA conceptual works and LCA studies (Arbault *et al.*, 2014; Othoniel *et al.*, 2016; Alejandre, van Bodegom and Guinée, 2019; Rugani *et al.*, 2019). Despite the recent advances, ES integration into LCA is still developed at a conceptual level.

1.1.5.2. Social Life Cycle Assessment

Social Life Cycle Assessment methods look to assess the social and socio-economic impacts of products or systems on social actors (e.g. workers, users, local communities) over their life cycle, and how they affect human well-being (Weidema 2006; UNEP 2009; Jorgensen et al 2008; Moltesen et al 2018). Since well-being includes physical and mental aspects, it is complex to assess as well as to gather data for, requiring objective and subjective indicators (King et al 2013, Moltesen et al 2018). For example, the perception of impacts is variable and dependent on the type of stakeholder (Jorgensen et al 2010), what makes difficult the aggregation of impacts across life cycle stages (Moltesen et al 2018). As another example, only to know individual processes would not permit the assessment of well-being's impact, since they also depend on other factors such as behaviour of a company towards stakeholders (Jorgensen et al 2008, Moltesen et al 2018). This impedes the use of generic databases as main sources, even if several social indicators can be roughly informed by them (Bathel 2004, Schmidt et al 2004, Weidema 2006). Despite a less advance methodological development compared to LCA, there are already few initial works applying social life cycle to components of urbanised contexts (Mcconville, 2006; Opher, Shapira and Friedler, 2017).

1.1.5.3. Life Cycle Cost Analysis

Life Cycle Cost Analysis (LCC) provides a better understanding of the lifetime costs of products and systems, and it has been widely used to assess infrastructures and building assets (Goh and Sun, 2016). Three major variants of the method exist: conventional LCC (includes only financial costs), environmental LCC (more aligned with LCA includes part of the external costs), and social LCC (integrates social external costs). The selection of one or another depends on the purpose of the study (Rödger, Kjær and Pagoropoulos, 2018).

LCC can consider internal and external costs, transforming them into monetary flows, and specifying the actor (e.g. manufacturer, user, government) assuming them (Swarr *et al.*, 2011).

Internal costs are usually financial cost (i.e. costs from good and services for which a market exist) and include the direct costs of the actors. Instead, external costs include changes consequence of a business transaction or its side effects (e.g. increased level of pollutants), which are not accounted in the price of the product (Belussi and Barozzi, 2015; Rödger, Kjær and Pagoropoulos, 2018). In many cases, external costs correspond to externalities (i.e. cost/benefits from goods and services non-traded in markets). To calculate the value of future costs (e.g. maintenance, management, mitigation measures) aggregation and discount rates are usually applied taking as a reference the net present value (Belussi and Barozzi, 2015). However, the definition of the discount rates incorporates subjectivity and can lead to over or under estimation of costs. Differently to LCA there is no need for characterisation factors as part of conventional LCC, since all the costs are calculated in monetary units (Swarr *et al.*, 2011).

The use of LCA and LCC can be combined, even if some limitations still exist (e.g. lack of harmonised cost inventories across sectors), being possible to define shared system boundaries and use identical functional unit, helping to avoid double counting of impacts in physical and monetary units (Swarr *et al.*, 2011). In fact, the use of LCC alone or in combination with LCA has been already applied to assess novel urban ecosystems such as green roofs and green walls (Wong *et al.*, 2003; Perini and Rosasco, 2013; Vineyard *et al.*, 2015). Moreover, several studies helped to internalise costs or benefits provided by ES or related to them such as air quality improvement, reduction of urban heat island effect, provision of recreational services, and improvement of aesthetics (Bianchini and Hewage, 2012; Perini and Rosasco, 2013; Belussi and Barozzi, 2015). Hence, conceptual basis for an integration of ES, LCA, LCC and its application to new nature-related concepts such as nature-based solutions are already on place.

1.1.6. Nature-based solutions and their role for sustainable and resilient urbanised systems

Initially, the concept of nature-based solutions (NBS) was developed and used by policy makers (Cohen-Shacham *et al.*, 2016), appearing as part of a World Bank Report (2008), and IUCN reports (2009, 2010, and 2013). However, NBS has soon become one of the concepts in the interface between science and design, but also between science and policy (Nesshöver *et al.*, 2017). The European Commission (EC) might have triggered this change by including NBS in their research and innovation agenda (European Commission, 2015). It created an Expert Group on NBS for developing a common understanding of the concept among scientists, policy makers, and practitioners (Faivre *et al.*, 2017). It also funded several Horizon 2020 projects (e.g. Nature4Cities, UNALAB, NAIAD) where practitioners (mainly from urban design and planning), scientists, and policy makers work together.

The two main definitions of NBS are still coming from policy sectors, but scholars are already intervening in their modification (Potschin *et al.*, 2016; Pauleit *et al.*, 2017). The International Union for Conservation of Nature understands NBS as *actions to protect,* *manage, and restore (create) natural or modified ecosystems* (Cohen-Shacham *et al.*, 2016). Instead, EC defines NBS as *living solutions inspired by, continuously supported by and using nature* (European Commission, 2016). Additionally, an expert report for the EC, further supported by several scholars (Faivre et al. 2017; Raymond et al. 2017; Dorst et al. 2019), emphasises that NBS are able to address multiple societal challenges (of sustainability and resilience) simultaneously, as well as to provide additional co-benefits (European Commission, 2015). Faivre et al (2017) state that NBS can increase social innovation in cities and aid in their sustainable transition, by promoting new models of planning, governance, finance, and business. For EC and IUCN, the purpose of NBS is quite similar, to address societal challenges in an effective and adaptive form, providing human well-being and biodiversity benefits (IUCN) or economic, social, and environmental benefits (EC). In this sense, NBS relates directly to the concept of sustainability and its three dimensions (Nesshöver et al 2017).

EC, IUCN and many scholars also relate NBS to the concept of ecosystem service (ES) and natural capital (Eggermont *et al.*, 2015; European Commission, 2015; Cohen-Shacham *et al.*, 2016; Potschin *et al.*, 2016; Maes and Jacobs, 2017; Nesshover *et al.*, 2017). For example, Eggermont et al. (2015) classify three types of NBS: i) better use of ecosystems; ii) sustainable and multifunctional management of ecosystems; and iii) design and management of new ecosystems. These types of NBS are organised based on their contribution to an increased supply of ES (i.e. linking ES and NBS) and the level of engineering to be applied to ecosystems to achieve this supply. As implied by IUCN and EC, and stated by other authors (Albert *et al.*, 2019; Bush and Doyon, 2019), NBS contain natural capital stocks or are actions to maintain and enhance the flow of ES. Then, NBS might optimise provision of multiple ES, requiring low physical resources to do it. This combination makes it an effective and credible solution to address sustainability and resilience challenges (Eggermont *et al.*, 2015; Kabisch *et al.*, 2016; Faivre *et al.*, 2017; Raymond *et al.*, 2017)

In terms of conceptualisation, several authors propose framing NBS as an umbrella concept under which other ecological concepts such as ecological engineering, ecosystem-based approaches, green infrastructure or ecological restoration could be integrated (Cohen-Shacham *et al.*, 2016; Nesshover *et al.*, 2017; Pauleit *et al.*, 2017; Dorst *et al.*, 2019). Compared to these other ecological concepts, NBS emphasise the value of nature to address societal challenges (Kabisch et al. 2016), the connection to policy and the relevance of implementation aspects (Pauleit et al. 2017). However, NBS is still a very open concept (Potschin *et al.*, 2016) and this vagueness hampers its mainstreaming as a resilient and sustainable solution for urban planning strategies and interventions (Dorst *et al.*, 2019). Thus, to facilitate its operationalisation, the concept itself and the added value of NBS compared to other solutions needs to be easily understood by practitioners and policy makers.

In practical terms, urban design and planning practitioners and policy makers need further studies relating specific urban NBS to particular benefits (e.g. Cortinovis & Geneletti, 2019; Frantzeskaki et al., 2019; Keeler et al., 2019). For this, first they need a clear NBS classification shared among different professionals. They also need to know when specific NBS are not suitable due to the specificity of the context (Albert *et al.*, 2019), or when NBS are not enough as a stand-alone solution, e.g. to address social related UC (Haase et al., 2017; Kotsila et al., 2020). Further research also needs to show which attributes of urban NBS affect the supply of specific ES, and therefore their effectiveness, to better consider those attributes during planning and design processes. There are previous systematic reviews on factors influencing ES supply (e.g. Bordt & Saner, 2019; Smith et al., 2017). However, these reviews are mainly focused on spatial levels such as entire ecosystem or landscape mosaics, rural contexts, and in general factors (e.g. population dynamics), which might not help to define individual NBS placed in urbanised contexts.

1.2. Research problem, research questions and hypothesis

1.2.1. Research Problem

In order to better understand the contribution of NBS to the sustainability and resilience of urbanised contexts, it is necessary to assess its contribution to the optimisation of urbanised systems (i.e. minimisation of negative environmental and socio-economic impacts and maximisation of positive ones) as well as to their capacity to resist and/or adapt to changes. In addition, as a concept, NBS requires a better operationalisation and an adaptation to urbanised contexts before researchers are able to assess its contribution to urban sustainability and resilience. As a starting point, a better definition and classification of urban NBS is necessary. They should be done in a form that is suitable for scientists, design and planning professionals as well as policy makers. It is also necessary to understand with more detail the causal relationships between different types of NBS, ES and the urban societal challenges that they are supposed to address. Moreover, in order to better plan and design NBS, it is relevant to know how the attributes of NBS and "the contexts" where they are placed influence their provision of ES.

Due to the multiple dimensions (environmental, economic and social) of NBS benefits, it is difficult to approach their assessment from a unique disciplinary field. Thus, the use of multiple approaches and methods seems a necessary step to overcome partial views and offer a more complete assessment. As introduced in previous sections, on one side urban ecology and landscape ecology could provide a better understanding in terms of biodiversity conservation and the value of NBS, once implemented, in the form of their supply of ES. On the other side, life cycle thinking could better inform about negative impacts derived from NBS. It could help to inform on the use of energy and materials, and associated emissions, required for their implementation, and the waste generated during their end-of-life. The adequate combination of methods from multiple disciplinary fields requires a previous harmonization in order to reduce misalignments between them. Hence, it is necessary to do this combination through the development of integrated or comprehensive modelling frameworks that can be used to assess the sustainability and resilience of NBS in an urbanised context.

1.2.2. Research Questions

There are two main research questions derived from the research problem:

Question one: Are nature-based solutions suitable interventions to enhance the sustainability and resilience of urban contexts and their hinterland?

In order to answer this question, we also need to understand i) if a relation between NBS, and the sustainability and resilience of urbanised contexts exist ii); and in which form and under which conditions NBS are related to the sustainability and resilience of urbanised contexts.

Question two: Does a combination of ecosystem services, landscape ecology, urban ecology, and life cycle thinking methods help to move towards a practical and coherent assessment of the contribution of NBS to the sustainability and resilience of urbanised contexts?

In order to answer this question, it is necessary to understand i) which is the contribution of individual methods from these disciplinary fields; ii) if the methods related to these disciplinary fields can be interrelated and integrated; ii) and which could be the most adequate modelling framework for their integration.

1.2.3. Hypotheses

To address the research problem and questions, the following hypothesis are defined as the basis of this research, which this thesis will try to interrogate and validate:

Hypothesis 1: Nature-based solutions can be considered living solutions or actions applied on them, which act as an umbrella for other nature-related concepts and have the potential to address societal challenges of sustainability and resilience by enhancing ecosystem services flows derived from natural capital.

Hypothesis 2: The use of a landscape ecology approach and a combination of its techniques can overcome major limitations of the rest of the approaches regarding the consideration of the spatial configuration.

Hypothesis 3: The ecosystem approach and life cycle thinking are compatible and can be integrated to offer a more complete understanding of positive and negative externalities derived from NBS.

From these hypotheses derives the following **thesis statement**:

NBS are adequate mechanisms for enhancing sustainability and resilience of urbanised contexts and the combination of methods from landscape ecology, urban ecology, ecosystem services, and life cycle thinking can help to understand to which extent and how effective is their contribution to sustainability and resilience.

1.3. Aim and Specific Objectives

The overall aim of this PhD thesis is to develop a coherent and practical methodological procedure for the environmental and economic assessment of NBS in urbanised contexts based on rationales and models underpinning landscape ecology, urban ecology, ecosystem services and life cycle thinking. This aim is achieved according to the following specific objectives:

1. Characterisation of NBS in relation to urbanised contexts and the problematics that they can help to address or mitigate. This objective is achieved through the following tasks:

- Identification and classification of UC, ES, NBS and their relationships based on the current scientific, policy and urban planning literature;
- Identification of the key attributes of NBS (e.g. soil properties, height of vegetation) and their contexts that influence social and biophysical processes, which lead to the generation of specific ES relevant for mitigating or addressing UC;
- Identification of similarities and differences in the UC, ES and NBS emphasised across urban studies based on their socio-economic and environmental conditions.

2. Definition of an adequate set of biophysical and monetary methods and indicators to assess the value of NBS in urbanised contexts. This objective is achieved through the following tasks:

- Identification of methods from ecosystem service valuation, life cycle thinking and landscape and urban ecology;
- Identification of adequate indicators to represent outputs in the form of cost and benefits in biophysical and monetary units.

3. Development of a modelling framework that combines methods from life cycle thinking, urban/landscape ecology, and ecosystem services to assess the contribution of NBS to urban sustainability and resilience. This objective is achieved through the following tasks:

- Conceptualisation of modelling frameworks that through the combination of individual methods reduce their specific limitations;
- Development of the conceptual modelling frameworks in the form of detailed methodological procedures with the potential of being generalised and applied to assess multiple NBS in different urbanised contexts;
- Testing the detailed methodological procedures for specific NBS and case studies.

4. Integration of the modelling framework as part of a decision support system to support sustainable and resilient urban planning and landscape design actions. This objective is achieved through the following tasks:

- Definition of a workflow to relate the conceptual modelling framework (and detailed methodological procedures) to a generalised simple to use decision support system;
- Definition of the structure of the graphical user interface and interactions with the user to ensure low computer-literacy requirements in the decision support system.

1.4. Thesis Outline

The thesis is organised into seven Chapters. As summarised in Figure 1.1, Chapter 2 to 6 address the four objectives that contribute to fulfil the aim of this thesis.

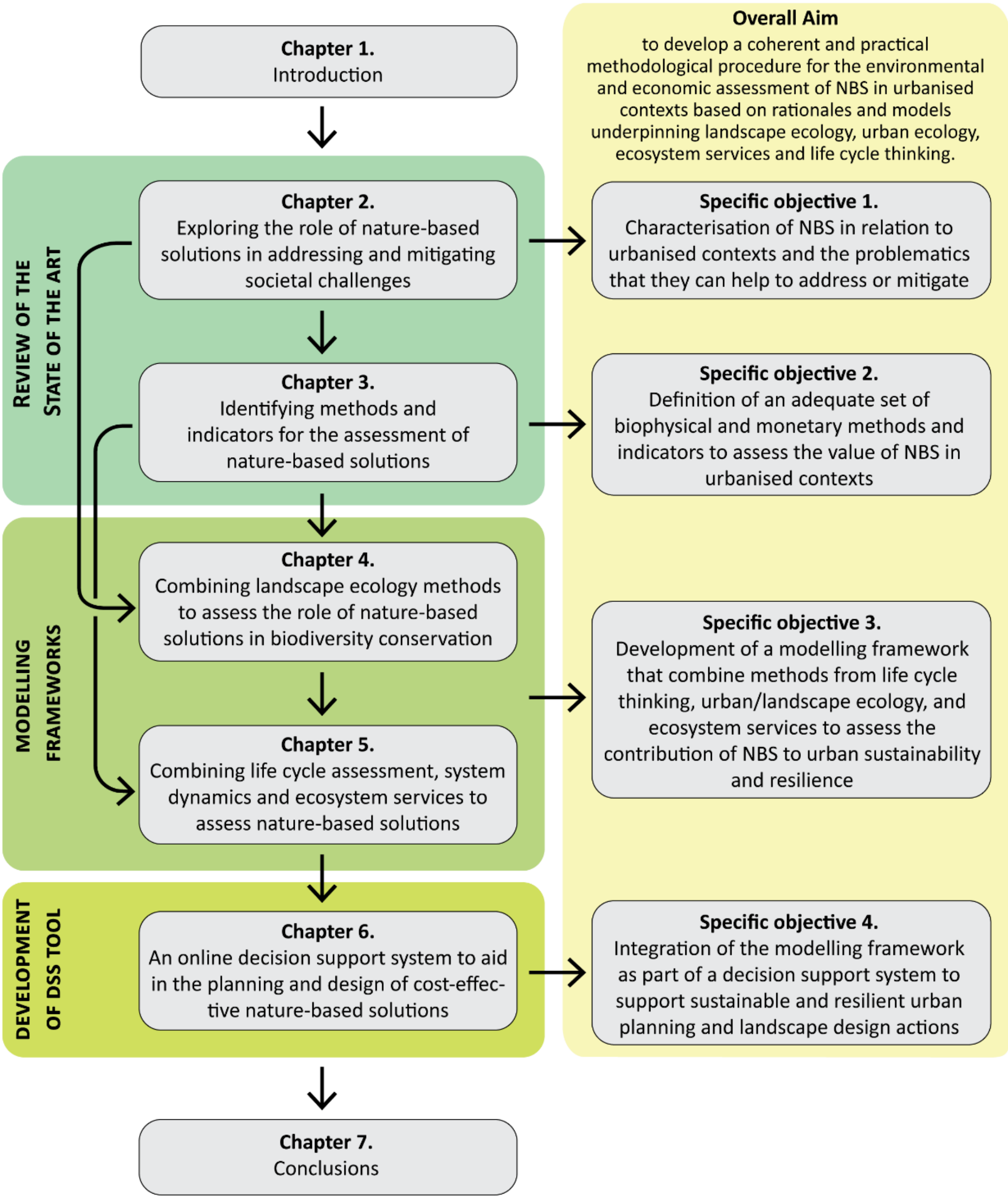

Figure 1.1. Visual outline summarising the structure of the PhD thesis

Chapter 2 identifies qualitative relationships between UC, ES and NBS, discuss if they are causal or not, and how they can be affected by urban context conditions. The identification of nexuses and the discernment of plausible causal relationships is guided by a conceptual framework and supported on a literature review. Before identifying relationships, to develop a more detailed classification of UC and NBS was necessary. As a main output, a graph of plausible cause-effect relationships between UC, ES and NBS is obtained.

Chapter 3 reviews current methods and indicators (indices) to assess positive and negative externalities (i.e. non-marketable goods, services, and impacts) derived from NBS in urbanised contexts. Specifically, the review focuses on methods from ES valuation, LCA and LCC. As a main output, the review identifies suitable indicators to evaluate positive externalities (benefits) and negative externalities (costs) derived from NBS in biophysical and monetary units.

Chapter 4 selects and tests a combination of landscape ecology methods to assess the influence of the composition and configuration of natural land covers (representing NBS) for biodiversity conservation. Specifically, the methods and metrics used assess changes in habitat loss, landscape fragmentation and ecological connectivity to understand potential impacts on biodiversity conservation. Luxembourg is used as a case study and changes are tested at urban region level, since the evaluation of potential effects of habitat loss and fragmentation on biodiversity require consideration of a broad spatial extent.

Chapter 5 develops a combined evaluation framework integrating methods from LCA, ES, and urban ecology to assess the environmental and economic value of urban NBS in biophysical and monetary units. Positive and negative externalities and financial benefits and costs (i.e. marketable goods, services, and impacts) are considered. To test the framework a specific model for the assessment of urban forests is developed and applied on a case study in the metropolitan area of Madrid (Spain).

Chapter 6 develops a prototype of an online decision support system (DSS) on NBS, called NBenefit$®, which integrates the evaluation framework set up in Chapter 5. NBenefit$ intends to make easy the integration of NBS assessment tools during early stages of urban planning and landscape design processes. The prototype DSS presents a simple form of quantifying the provision of multiple ES and costs over the entire life cycle (implementation, operational life, and end-of-life) of NBS. The results for 48 archetypes of urban forest and three hypothetical case studies are used to illustrate the DSS.

Finally, Chapter 7 provides a recapitulation of the thesis. General and specific conclusions will be given in this final chapter, including limitations and strengths of the research approach. Future perspectives are also introduced at the end of this chapter.

Chapter 2

Exploring the role of nature-based solutions in addressing societal challenges[‡]

2.1. Introduction

This chapter aims at reducing the conceptual vagueness of nature-based solutions (NBS) and to adapt the concept better for urbanised contexts, before its contribution to sustainability and resilience can be assessed. More specifically, the chapter proposes a more detailed classification of urban NBS and relates it to existing ecosystem classifications. It also identifies relationships between types of NBS, ecosystem services (ES) and societal challenges of urbanised contexts (named urban challenges in the rest of the thesis). Since there is no detailed agreed classification of urban challenges (UC), which is necessary before relationships with other elements can be drawn, the chapter also develops a prototype classification of UC. As part of the study of relationships, this chapter identifies similarities and differences in the UC emphasised depending on specific local conditions. Similarly, it also identifies key attributes of NBS and their contexts influencing social and biophysical processes that lead to the generation of ES that could be relevant for addressing or mitigating UC.

In this chapter and for the rest of the thesis, UC are always considered in terms of urban problematics related to sustainability and resilience. Regarding sustainability, for this thesis UC include all factors that limit the capacity of urbanised contexts to protect and conserve the environment, minimise environmental impacts and enhance resource-efficiency, human health, social inclusiveness and equality, as well as harness the productivity of local economies and value-added activities (United Nations, 2017). In terms of resilience, for this thesis UC relate to those factors that limit the capacity of urbanised contexts (including their inhabitants, institutions and inner systems) to resist and adapt to environmental, social or economic chronic stresses, and acute shocks (Meerow et al. 2016; Marron Institute of Urban Management, 2018).

[‡] Chapter 2 is partially based on:

Babí Almenar, J., Almenar, J.B., Elliot, T., Rugani, B., Bodenan, P., Gutierrez, T.N., Sonnemann, G. and Geneletti, D. Nexus between nature-based solutions, ecosystem services and urban challenges. *Land use policy* 100, 104898 (2021).

Roles of other authors (The paper was published including a CReditT authorship contribution statement):

In many cases, UC for sustainability and UC for resilience (hereafter referred generically to as UC) overlap and also share limiting factors. The nature of these limiting factors can be:

- biophysical (e.g. a lack of woody vegetation can contribute to the presence of heat islands);
- technological (e.g. insufficient technological development for achieving universal access to certain goods or services);
- human-social (e.g. the current human, institutional or social structure act as barriers for adapting to new situations); or
- financial (e.g. limited amount of economic resources restricts access to certain products).

UC can therefore be determined by many types of limiting factor, which need to be understood before strategies and interventions (including NBS or not) can be developed to mitigate or address those UC. Framing such strategies and interventions with specific solutions, requires a good understanding of the local environmental, social and economic conditions of urban contexts. Its acquisition would provide information about the suitability of the proposed solutions and the adequacy of their transfer and replicability in other urban contexts.

This chapter aims first to identify qualitative links (named nexuses in the rest of the chapter) between UC, ES and NBS; then it discusses if these nexuses are plausible causal relationships (i.e. if NBS can help or not to overcome limiting factors underlying UC through the supply of ES), and how these relationships can be affected by urban context conditions. The UC-ES-NBS nexuses are disclosed through a two-step systematic review plus a complementary non-systematic review. A posterior critical analysis of the collected evidence helps to assess which nexuses can be considered plausible causal relationships. We refer to "plausible" (i.e. likely) causal relationships according to a precautionary principle, because through a literature review a causal relationship cannot be confirmed with full certainty.

The identification of nexuses through the literature review as well as the discernment of plausible causal relationships was guided by the conceptual framework presented in Section 2.2. Section 2.3 describes the methodological procedure of the literature review that supported a detailed prototype classification of UC and NBS and the critical analysis of their plausible causal relationships through ES. Section 2.4 describes and discusses the results obtained from the literature, the discernment of plausible causal relationships UC-ES-NBS, presenting them as graph-based network. Finally, conclusions are drawn in Section 2.5.

2.2. Conceptual framework

The proposed framework aims to relate UC, ES and NBS in a simple form (see graphic outline in Figure 2.1). First, specific nexuses are identified through their reiterated occurrence in the empirical results of the works included in the different literature reviews. Then, further analysis is required to understand if those nexuses (UC-ES and ES-NBS) are in fact plausible causal relationships.

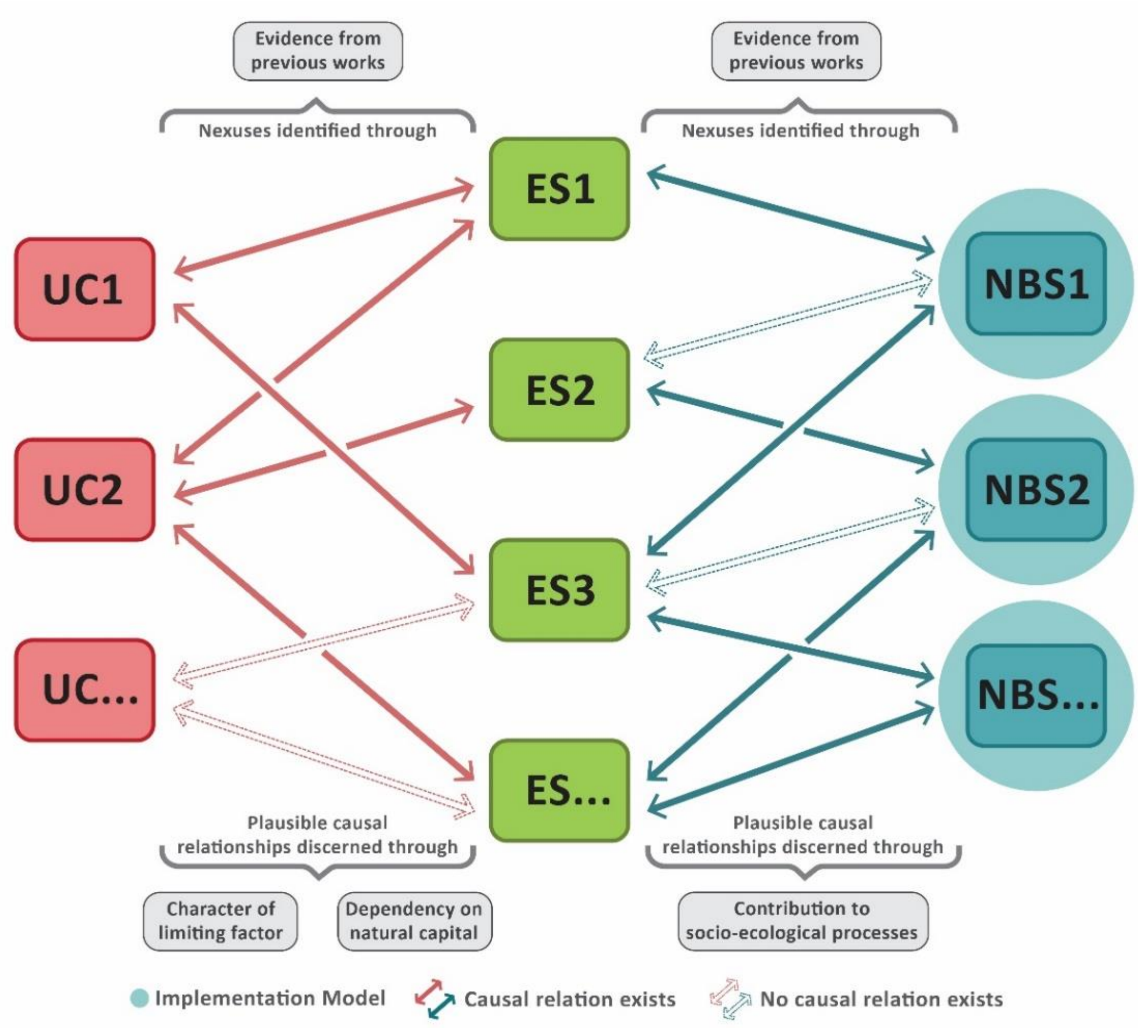

Figure 2.1.Conceptual diagram for the identification of UC-ES-NBS nexuses and plausible causal relationships, making explicit the factors, attributes and processes that define the latter; UC=Urban Challenge(s); ES=Ecosystem Service(s); NBS=Nature-based solution

Regarding the UC-ES nexuses, not all UC are rooted in biophysical limiting factors or could be mitigated with natural capital, and consequently their mitigation might not be achieved by increasing ES provision. In these cases, plausible causal relationships between UC-ES do not exist or they are not relevant enough to be acknowledged.

In terms of ES-NBS nexuses, each NBS provides only a specific set of ES. Then, a plausible causal relationship between NBS and ES does not exist when the NBS does not contribute to the socio-ecological processes (e.g. tree transpiration) that generate the specific ES (e.g. *regulation of hydrological cycle and water flow*)Following the three types of NBS of Eggermont *et al.*, (2015), for NBS that are biophysical structures (i.e. existing or novel ecosystems), the contribution occurs if the abiotic and biotic attributes of the NBS are involved in the socio-ecological processes. In the case of NBS as actions applied on ecosystems (e.g. management and restoration actions), the contribution occurs if these actions modify the attributes of the ecosystems involved in the socio-ecological processes. The extent to which a particular ES-NBS causal relationship not only exists, but it is relevant enough to be acknowledged also depends on the role of the abovementioned attributes in the performance of the socio-ecological processes. Besides the NBS itself, their implementation model (i.e. the combination of governance, business, and financial models under which the NBS is planned, developed and managed) might also influence its capacity to provide ES, and mitigate UC.

2.3. Methods

The method is composed of three main steps, namely a two-step systematic literature review on UC and ES[§], a complementary non-systematic review and *a posteriori* integrated analysis of the data (Figure 2.2).

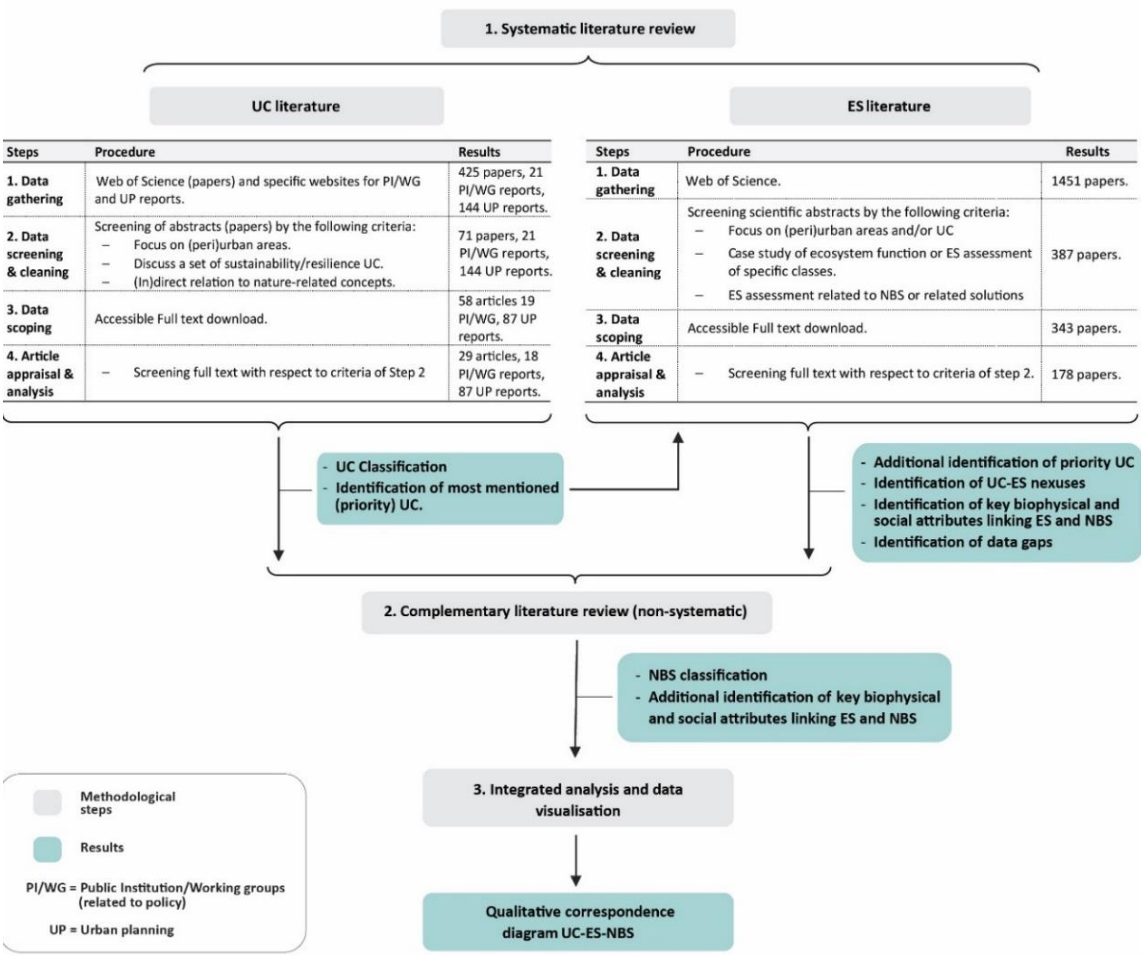

Figure 2.2. Methodological steps of the literature review including the criteria that the selected documents fulfilled to be kept in the review.

The systematic review adapts the review protocol of Luederitz et al. (2015) and Brink et al. (2016). Operationally, the review was conducted by:

§ The systematic literature review on ES of this chapter was also performed to fulfil the aim of Chapter 3. The results are split into two chapters because they relate to different aims. However, the collection of data on ES studies for both chapters was done simultaneously.

i) identifying an initial list of articles based on a broad search string that encompasses UC and ES topics;
ii) preselecting the articles if their abstracts meet specific criteria (as reported in Figure 2.2, Data screening & cleaning);
iii) selecting the articles and conducting a critical analysis if their complete text fulfils the screening criteria (see Figure 2.2, Article appraisal & Analysis).

2.3.1. Systematic literature review

For the two-step systematic literature review, the search of peer-review papers was limited to the last 20 years, from 1998 to early 2019. The concept of NBS and similar concepts (e.g. green infrastructure), as well as the study of ES in regard to them, are very recent, making it unnecessary to account for a longer time period. In addition, UC evolve over time, hence limiting the temporal extent of the search ensures that only currently relevant challenges are included in the analysis.

The papers were retrieved from Web of Science at the end of February 2019 using the search strings included in Figure 2.3. The screening phase (see the criteria in Figure 2.2) was performed to retain only papers describing an assessment of ES or ecosystem functions in urban contexts.

UC literature review

TS=("urban challeng*" OR "urban priorit*" OR "urban sustainability issu*" OR "urban resilence issu" OR "urban strateg*") Refined by: DOCUMENT TYPES: (ARTICLE OR REVIEW) AND LANGUAGES: (ENGLISH) Timespan=1998-2018
Indexes=SCI EXPANDED, SSCI, A&HCI; ESCI

ES literature review

TS=(("urban*") AND ("assess*") AND (“ecosystem servic*” OR “landscape servic*” OR (“ecosystem functio*” OR “landscape functio*” OR “ecosystem structur*” OR “landscape structur*” OR “nature-based solution*”)) Refined by: DOCUMENT TYPES: (ARTICLE OR REVIEW) AND LANGUAGES: (ENGLISH) Timespan=1998-2018
Indexes=SCI EXPANDED, SSCI, A&HCI; ESCI

Figure 2.3. Search string for the UC and ES literature review. UC = Urban Challenges; ES = Ecosystem Services.

The literature review of UC for sustainability and resilience included peer-reviewed papers, reports from international public institutions (e.g. United Nations, FAO) and local urban planning documents, thus integrating science, policy and local urban planning perspectives. The collection of policy reports and urban planning documents was completed in February 2019. The policy reports were selected from a pre-established list of well-known international institutions and related initiatives (see Annex 2.1 for the complete list). The local urban planning documents were reports selected from various global inter-city scale initiatives, such as Emerging and Sustainable Cities (Inter-American Development Bank), 100

Resilient Cities Initiative (Rockefeller Centre) and the C40 Cities. These initiatives include a comprehensive list of cities around the world that prove to be active in sustainable and resilient urban planning (see Annex 2.1 for the complete list). In the case of public institutions and local urban planning reports, the stages of data gathering, screening and scoping were done simultaneously.

The lack of a recognised exhaustive classification of UC in the literature made it necessary to develop an original one based on existing frameworks. The UC, and their sub-challenges, obtained from the selected literature were organised by combining the classifications of UC proposed in the Emerging and Sustainable Cities Initiative, the Reference Framework for Sustainable Cities and the EKLIPSE report (RFSC 2016; Raymond et al. 2017; IADB 2019). These classifications complement each other well and are already used in Latin America and Europe. New sub-challenges mentioned in the papers reviewed and not considered in the original classifications were also incorporated into the UC classification (See Annex 2.2 for a description of the classification). In the reviewed literature, multiple terms were used for the same UC or sub-challenge. This made a harmonisation of the terms necessary as part of the development of the classification system. In addition, when a paper or document referred to a UC or a sub-challenge using a vague terminology, and there was no definition and/or clear description to help its classification, this non-distinguishable UC or sub-challenge was disregarded from the review.

The papers of the UC review were analysed making use of nine categories that included the type of UC, type of source (science, policy, local urban planning) and contextual attributes that characterise specific socio-economic and environmental conditions. The contextual attributes included location, continent, climatic conditions, average elevation, population (size of urban areas), population density and gross national income (GNI) classes of the referenced cities (see Annex 2.3 for details on each category). Specific classes were defined for each of the contextual attributes (further description in Section 3.3).

The papers of the ES review included only peer-reviewed papers. The ES classification of reference used in this review is CICES v5.1. It categorises ES in three main sections: “provisioning services”, “regulation and maintenance services” and “cultural services” (Haines-Young and Potschin, 2018). CICES is recognised internationally, it has a more detailed classification, especially for cultural ES that are relevant in cities, and it is used in the initiative Mapping and Assessment of Ecosystems and their Services (MAES) of the EC. The authors acknowledge the existence of other ES classification systems (e.g. NESCS, TEEB), which all have their strengths and special attributes, but a choice formulated on the basis of a detailed comparison was considered out of the scope in the present paper. Further information on the differences and complementary features of ES classification systems can be found in La Notte et al. (2017) and McDonough et al. (2017).

The ES papers were analysed based on 15 categories, incorporating the nine used in the UC literature (see Annex 2.3. for detail on each category). The additional categories included ES sections, ES classes, key social and biophysical attributes, key social and ecological processes, types of NBS or similar solutions. The analysis helped to identify the most frequent UC mentioned in the ES literature, the most frequent ES classes and the NBS that were investigated more in urban areas. It also helped to justify relationships between i) specific UC and ES classes, ii) specific ES classes and social and biophysical processes; and iii) these processes and attributes of some types of NBS (those with a biophysical structure) or the urban contexts where they are placed.

2.3.2. Non-systematic literature review

A complementary non-systematic review was performed to help establish the classification of NBS types. It was also used to fill gaps in the identification of social and biophysical processes and attributes involved in the generation of ES already identified. This was necessary because many papers from the systematic literature review did not clearly identify the factors influencing the supply of specific ES classes, as it was also raised by Luederitz et al. (2015) in their review.

The non-systematic review was supported by land management and ecological restoration handbooks, papers on NBS types and their assessment (e.g. Xing et al. 2017a) and handbooks of ES process-based models (process-based models as defined by Santos-Martin et al. 2018). The handbooks on land management techniques (Morgan, 2013; Triest, Stiers and Van Onsem, 2016) and types of restoration ecology interventions (Hobbs, Higgs, & Harris, 2009; van Andel & Aronson, 2012) complemented the identification of NBS that were less studied in the urban ES studies. The papers and ES process-based model handbooks were identified making use of a snow-balling approach (Badampudi, Wohlin and Petersen, 2015), starting from the references of the systematic literature review. Only papers clearly stating the biophysical and social attributes influencing the supply of specific ES classes were included.

The classification of NBS types modifies the one proposed by IUCN (Cohen-Shacham *et al.*, 2016). In fact, the classification of IUCN is also an adaptation of the three NBS types of Eggermont et al (2015). The classification also makes use of three categories (i.e. ecosystem types, dominant media and spatial levels), as summarised in Figure 2.4. In terms of conceptualisation, NBS is assumed as an umbrella concept for other ecological concepts, and the definitions of IUCN and EC are respected. Therefore, NBS included in the classification are actions applied to enhance living solutions or which are formed of them that protect, sustainably manage, restore or create (natural, modified or novel) ecosystems (Cohen-Sacham 2016, European Commission 2016).

In this Chapter, and for the rest of the PhD thesis, NBS Type 1 are considered solutions that permit not only a better use, but also a better management (i.e. non-physical modifications) of existing natural or naturalistic ecosystems. Making better use of an ecosystem implies a change in its management or in the management of surrounding ecosystems (indirect change). It could also imply changes in how the resources obtained from the ecosystem are exploited. In this sense, it was considered not possible to distinguish between “a better use” and “a better management” as two different NBS types. With this modification NBS Type 2 include only solutions and procedures to restore ecosystems. These are further differentiated into reclamation and restoration categories. Following the adaptation of IUCN, NBS Type 3 are maintained as solutions that involve creating novel ecosystems. They also include solutions that involve an extensive (i.e. a large percentage of area) and intensive (i.e. high degree) modifications of existing ecosystems. An example of the latter case would be the case of converting a highly artificialized urban green area into a highly naturalised one.

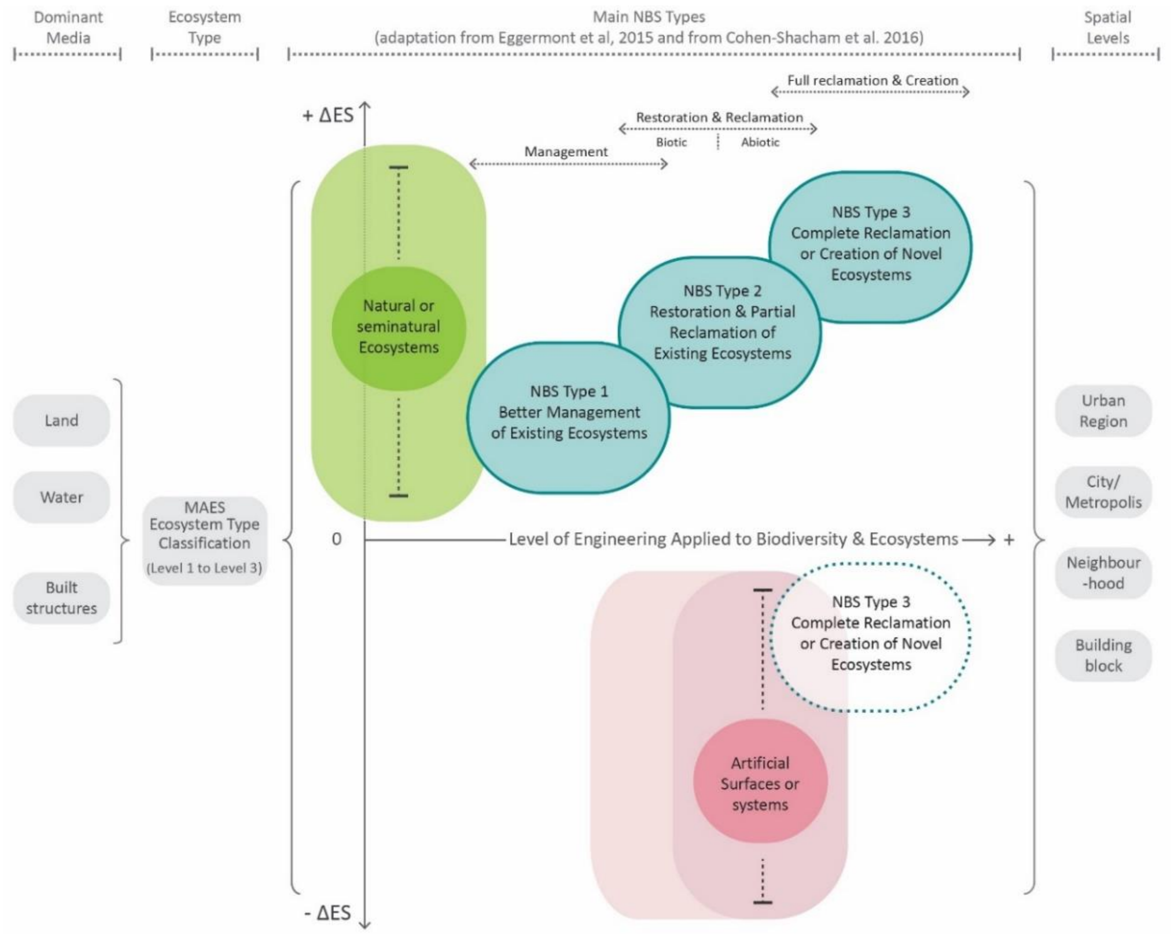

Figure 2.4. Conceptualisation of NBS types. Elements and their positioning in an X-Y diagram built on the framework of Eggermont et al. (2015) and (Cohen-Shacham et al., 2016). NBS = Nature-based solutions.

Regarding the categories, the three types of NBS refer to actions applied to ecosystems in one way or another, thus necessitating the organisation of NBS according to ecosystem types.

The ecosystem type classification of the MAES initiative (Maes *et al.*, 2013) was selected as the most appropriate one because of its detailed categorical resolution and correspondence with EUNIS and CORINE classifications (Moss, 2014; Büttner *et al.*, 2017). MAES ecosystem type classification is also the one used in Europe and promoted by the European Commission for ES assessment. Using the MAES classification can therefore facilitate in the future an exchange of information and harmonisation with other studies on ES and NBS. The identification of ecosystem types can also help to understand the dominant media (on the left in Figure 2.4) per each ecosystem type, which may constrain the specific NBS that can be implemented. References to (semi)natural and artificial ecosystems were included in the conceptualisation of NBS types (in the centre of Figure 2.4) as auxiliary elements. These ecosystems are the biophysical support on which NBS Types 1, 2 and 3 can be developed and define the initial ES supply that should be enhanced (see the y axis in Figure 2.4) by the implementation of NBS.

Finally, to make the classification relevant for urban planners and decision-makers, urban NBS also need to be organised according to the spatial level at which they should be implemented (on the right in Figure 2.4). The spatial level indicates the range of required space for each specific NBS, and consequently its adequacy for different types of urban strategies and interventions.

2.3.3. Integrated analysis and visualisation of the outputs

The outputs from the UC and ES literature review were analysed making use of the categories described in Section 2.3.1. The contextual attributes of the case studies were analysed in order to understand the similarities and differences in the UC, ES and NBS depending on the specific socio-economic and environmental conditions of their urban contexts. First, the location of the case studies was used to georeference them. Second, data associated with the remaining contextual attributes (e.g. population, climate) were collected making use of existing databases. Data on population and population density were extracted from Angel et al. (2011), which provide an informational database for 3,646 urban agglomerations worldwide. For urban agglomerations not included in Angel et al. (2011), the data were collected from databases found one by one on specific municipal, metropolitan or regional websites. The updated Köppen-Geiger climate classification world map (Kottek *et al.*, 2006) was used to assign regional climatic classes to each case study. The one-kilometre resolution map of the GLOBE project (Hastings and Dunbar, 1993) was applied to differentiate urban areas with low-lying elevation. Third, per each contextual attribute, qualitative classes were established to make easier the differentiation between urban contexts with similar contextual conditions. In terms of urban size, the cities were classified according to their population making use of OECD (2019) classes. Regarding inhabitants' income capacity, the GNI classes proposed by the World Bank (2019) were used.

The visualisation of the data for each UC with respect to the type of document and urban contextual conditions was done using an ordinal ranking approach supported on conditional tables. The same procedure was used for the visualisation of the most widely studied ES and NBS, UC-ES nexuses and ES-NBS nexuses (only links appearing more than three times were kept in the visualisation), but with a graph-based approach. These graphs are constructed in Gephi (Bastian, Heymann and Jacomy, 2009), a graph-based software allowing visualisation of networks. For the discernment of UC-ES plausible causal relationships, the nature of the limiting factors of the UC were analysed as well as if they could be overcome with increased natural capital (as stated in Figure 2.1). Similarly, for the discernment of ES-NBS plausible causal relationships, the conditions of the framework in Figure 2.1 were followed based on the identified social and biophysical attributes and socio-ecological processes. The names of these attributes and processes were harmonised to avoid repetitions and overlaps, and the related information was populated in a table. In the case of attributes and processes, the nexuses were not ranked because the objective was only to identify them and not to illustrate the most acknowledged ones. Finally, a qualitative correspondence diagram was constructed to depict potential plausible causal relationships among the most frequently mentioned UC, ES and NBS. The diagram also includes the association of UC with specific urban contextual classes and the attributes and processes linking ES and NBS.

2.4. Results & Discussion

The two-step systematic review examined 304 documents, 178 from the ES review and 126 from the UC review (further details in Figure 2.2). These documents included 366 case studies (i.e. several papers have more than one case study). As Figure 2.5 shows, the highest number of cases are located in Europe (162), America (109) and Asia (56), while only few documents investigated case studies in Africa (14) and Oceania (7). Among the selected documents, some also focused on the global context (17). In terms of the ES review, most of the case studies are from North America, Europe and Asia, which is consistent with the results of precedent reviews (Haase *et al.*, 2014; Luederitz *et al.*, 2015; Dobbs *et al.*, 2019; Keeler *et al.*, 2019). No difference emerged in the UC, ES, and NBS addressed in the studies based on average elevation and population density. In addition, due to the relatively low number of African and Oceanian case studies, these were not taken into consideration when looking for similarities and differences between continents. Likewise, only four climatic classes (Tropical Savannah with dry winter (Aw); Temperate without dry season and with hot summer (Cfa); Temperate without dry season and with warm summer (Cfb) and Mediterranean hot summer (Csa)) and three GNI classes (high income, upper-medium income and lower medium income) were considered, since the number of studies for other classes was negligible.

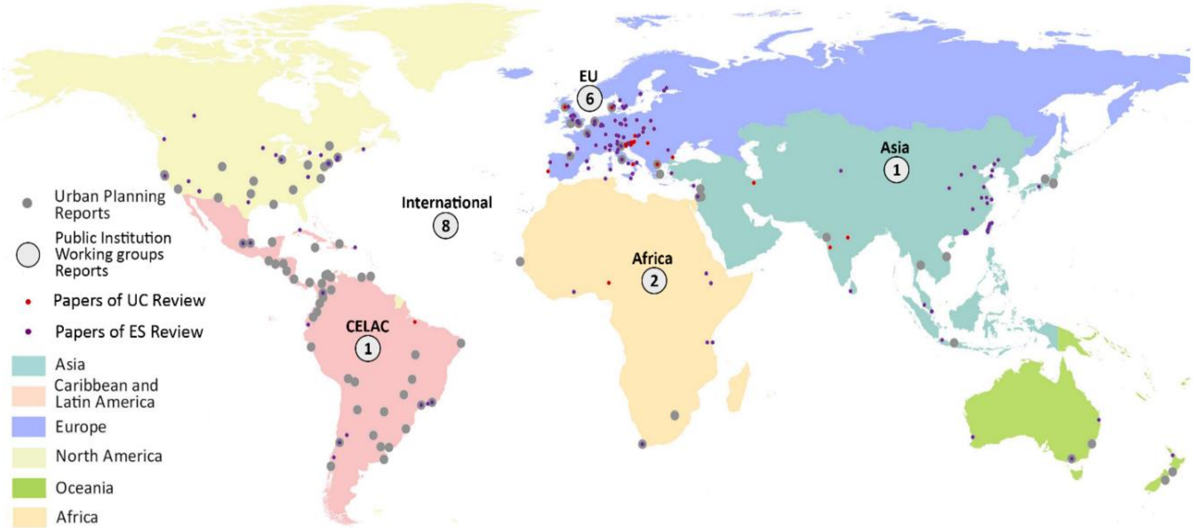

Figure 2.5. Location of the case studies in the reviewed documents

2.4.1. Classification of UC by type of document and contextual attributes

Based on the UC review, we identified 18 UC for urban sustainability and resilience and 58 associated sub-challenges. Table 2.1 shows the occurrence of UC and sub-challenges in the reviewed literature per type of document. At least 50% of all reviewed literature on UC focuses on *built-environment issues*, *physical health*, *green and circular economy,* and *material & solid waste management*. Public institution reports and urban planning documents mention *water management* and *mobility* in at least 50% of the documents. Around half of public institution reports consider *energy* and *governance*. Similarly, only urban planning documents consider *social vulnerability* and *climate change* in at least 50% of the cases. In fact, urban planning documents is the only type of literature where a sub-challenge (*vulnerability to human/natural disasters*) is present in at least half of the documents.

The review of the most frequently mentioned UC and sub-challenges (i.e. those present in at least 15% of the UC literature) across urban contextual classes (e.g. medium size urban areas) helps us to understand similarities and differences among specific urban contexts (Table 2.2). Regarding similarities, the *green and circular economy* is the only UC mentioned in at least 50% of the case studies in every contextual class. Other UC, such as *social vulnerability*, *built environment*, *mobility*, *water management*, *material & solid waste management* and *physical health*, show a frequency close to 50% in most of the contextual classes. When looking for differences, some UC and sub-challenges appear to be of higher interest in urban areas sharing certain conditions. For example regarding continents, *solid waste management*, *governance* and the sub-challenge of *vulnerability to human/natural disasters*, are only present in around 50% of the cases of urban areas belonging to the Community of Latin American and Caribbean States (CELAC), whilst *energy* and the sub-challenge of *employment (job) development* are mainly taken into consideration in European contexts. Also, more than 50% of all cases in the review (bold numbers in Table 2.2) identifying *wastewater management, expenditure and debt management,* and the sub-challenge *urban violence and insecurity* come from CELAC urban areas. In terms of urban size, *socio-spatial equity*, *climate change* and the sub-challenge *flooding risk*, are mostly studied in large metropolitan areas. Moreover, more than 50% of all cases in the review for the sub-challenges of *ageing and inadequate infrastructure* and *urban heat island effect, sea level rise,* and *vulnerability to disease outbreak* correspond to large metropolitan areas. Climate classes constitute a particular case, where the sub-challenge *energy efficiency* appears to be of interest only in urban contexts in the climatic class Cfb.

The different prioritisation of UC according to particular contextual conditions (e.g. urban size, continent) suggests that the presence of some UC might be more likely where specific social and biophysical contextual factors occur. We are not able to identify if causal relationships exist or not, and in this case only nexuses are identified. In the future, in-depth studies of UC in regard to contextual classes can help us understand whether these links are causal relationships or not. For example, the UC *socio-spatial equity* appears to be closely associated with large metropolitan areas, and therefore it might be worth investigating whether there is a causality associated with urban size. Future studies could also inform us whether lessons learnt from other urban contexts regarding the mitigation of UC through the ES supplied by NBS are transferable or not.

Part of our results on UC across urban contextual classes (Table 2.2) have also been stressed in the review of Dobbs et al. (2019). Their review emphasises *governance* and *vulnerability to human/natural disasters* as UC intrinsic to CELAC countries. As implied by these authors, these UC are triggered by a combination of social, political and biophysical factors specific to CELAC countries. Consequently, in their opinion, global urban ES lessons extracted predominantly from studies in northern developed countries and applied directly to advise on urban strategies and policies in CELAC countries could generate socially, environmentally and economically mismatched decisions (Dobbs *et al.*, 2019). Instead, win-win situations could arise if policymakers and urban planners of municipalities with similar contextual conditions and UC collaborate and exchange knowledge about their research and experiences on urban ES or policies, strategies and interventions on NBS. Consistently, scholars argued that NBS should not be copied from one place and applied exactly as they are to others (Haase, 2017; Dorst *et al.*, 2019). NBS should be sensitive both to the socio-spatial context in which they are applied, as well as to the specific UC that they aim to tackle, in order to be considered “solutions”(Haase, 2017; Dorst *et al.*, 2019).

Table 2.1. Urban Challenges and sub-challenges considered in the literature review

Urban Challenges and sub-challenges	**UP**	**SA-UC**	**PI/WG**	**SA-ES**
Total Number of Documents	87	29	18	178
Socio-spatial Equity	42	10	7	8
Socio-spatial Segregation	7	3	0	0
Lack of Gender Equity	2	0	1	0
Lack of Age Equity	1	0	1	0
Lack of Racial Equity	6	0	0	0
Lack of Income Class Equity	11	0	0	2
Social Cohesion	29	6	7	3
Lack of Inclusion of Immigrants and Refugees	3	1	3	0
Lack of Community-Building	8	3	0	0
Lack of Public Spaces for Social Interaction	6	1	0	0
Social Vulnerability	65	5	5	11
Vulnerability to Human/Natural Disasters	49	1	3	10
Vulnerability to Disease Outbreak	19	0	0	0
Vulnerability to Terrorism	7	0	0	0
Urban Violence & Insecurity	28	1	2	1
Demographic Dynamics	15	7	3	1
Increasing Population	9	4	0	1
Decreasing Population	1	3	1	0
Population Displacement	3	0	0	0
Ageing Population	5	1	2	0
Built Environment	48	15	11	5
Preservation of the Cultural Heritage and Identity	8	2	2	0
Affordable Housing	17	3	5	0
Lack of Liveable and Adaptable Public Spaces	16	4	5	4
Urban Sprawl	6	6	3	0
Informal Settlements	2	4	1	0
Mobility	55	10	11	0
Inadequate Public Transport	11	0	0	0
Ageing & Inadequate Infrastructure	30	5	3	0
Inadequate Non-Motorised Transport Systems	6	1	0	0
Physical Health	46	19	9	66
Lack of Air Quality	22	8	5	8
Lack of Soil Quality	2	1	0	3
Lack/Deficient Sanitation Systems & Water Quality	26	9	2	11
Mental Health	33	9	5	50
Urban Stress & Lack of Psychological Relaxation	4	0	1	3
Lack of Education and Training	24	8	3	0
Lack of Cultural & Leisure Opportunities	1	1	1	1

>≈50% <10%

Papers per type of document

Notes: UP = Urban Planning Reports; SA-UC = Scientific Articles from UC review; PI/WG = Public Institution and Working Groups Reports; SA-ES = Scientific Articles from ES review.

Table 2.1 (Continued). Urban Challenges and sub-challenges considered in the literature review

Urban Challenges and sub-challenges	**UP**	**SA**	**PI/WG**	**SA-ES**
Total Number of Documents	87	29	18	178
Green and Circular Economy	58	15	12	2
Economic Efficiency and Competitiveness	14	1	1	0
Lack of Economic Diversification	8	1	1	0
Employment (Job) Development	31	10	8	0
Innovation and Green Entrepreneurship	16	0	0	0
Climate Change	52	13	8	52
Greenhouse Gas Emissions	4	2	0	1
Urban Heat Island & Heatwaves	21	6	0	31
Sea Level Rise & Coastal Resilience	16	0	1	1
Water Management	59	11	9	45
Flooding Risk	27	3	0	32
Freshwater Shortage	33	7	2	11
Wastewater Management	26	4	3	11
Material & Solid Waste Management	43	16	9	6
Raw Material Shortage	8	2	0	0
Food Shortage	16	4	3	4
Solid Waste Management	31	10	5	2
Energy	31	9	9	10
Energy Efficiency	9	5	2	3
Increased Energy Demand	7	2	1	2
Lack of Diversification of Energy sources (Renewable Energy)	12	0	1	0
Biodiversity	16	9	4	18
Loss and Degradation of Habitats	10	3	2	7
Lack of Ecological Integrity and Connectivity	4	0	0	1
Digital Connectivity	21	3	5	0
Unreliable Digital Infrastructure	3	0	0	0
Insufficient Public Access to Open Data	2	0	0	0
Public Participation	31	4	3	0
Empowerment of Communities in Decision-making	14	0	0	0
Promotion of Stakeholder Involvement	16	1	1	0
Governance	39	5	10	0
Transparency	11	1	1	0
Relationship Private Stakeholders - Government	7	1	0	0
Collaborative Multi-level Governance	6	2	1	0
Empowerment of Local Representatives	3	0	1	0
Expenditure	23	0	4	0
Expenditure & Debt Management	17	0	1	0
Use of Taxes & Financial Autonomy	12	0	1	0

>≈50% <10%

Papers per type of document

Notes: UP = Urban Planning Reports; SA-UC = Scientific Articles from UC review; PI/WG = Public Institution and Working Groups Reports; SA-ES = Scientific Articles from ES review.

Table 2.2. Urban (sub)challenges most frequently assessed per contextual class in the literature review on urban challenges. Bold underlined numbers in a cell indicate that more than 50% of the case studies including a specific (sub)challenge come from that specific contextual class.

Urban Challenges & sub-challenges	No	Climate Classes*				Continents*				Size of Urban Areas					GNI Classes*		
		Aw	Cfa	Cfb	Csa	RA	AS	LA	EU	LMA	MA	MC	SC	VSC	HI	UMI	LMI
Cases per Contextual Class		16	21	39	9	20	18	45	47	49	30	20	17	10	77	42	11
Socio-spatial Equity	59	10	9	7	1	15	7	21	7	23	11	7	6	0	26	17	5
Social Cohesion	47	1	6	12	4	11	4	6	18	16	10	5	5	1	**29**	6	2
Lack of Inclusion of Immigrants and Refugees	7	0	2	1	0	3	0	0	**4**	3	1	0	0	0	**4**	0	0
Social Vulnerability	75	10	15	12	4	14	10	33	6	33	16	10	8	2	33	27	7
Vulnerability to Natural Disasters	53	9	11	7	4	7	7	**28**	3	24	14	6	6	0	20	22	6
Vulnerability to Disease Outbreak	19	1	7	5	2	2	4	5	4	**13**	3	2	1	0	**11**	6	1
Reduction of Urban violence & insecurity	31	7	4	2	2	6	2	**20**	1	14	5	5	4	1	9	15	4
Demographic Dynamics	26	1	5	7	2	2	5	1	8	12	1	4	2	0	**14**	3	2
Built Environment	85	11	11	16	6	12	13	25	25	31	16	12	7	2	36	27	9
Affordable Housing	25	1	5	4	1	9	4	2	6	11	2	3	2	0	**17**	3	1
Lack of Liveability & Adaptability of Public Spaces	35	1	6	13	2	6	2	5	**20**	14	6	7	3	0	**24**	6	1
Urban Sprawl	15	0	1	2	1	0	4	4	4	7	1	0	1	1	2	7	2
Mobility	83	10	11	18	5	14	12	29	18	**24**	18	8	8	10	**36**	28	6
Ageing & Inadequate Infrastructure	38	3	6	4	5	11	11	4	6	**20**	6	2	3	1	19	8	5
Physical Health	81	9	14	16	4	10	15	24	22	28	15	7	5	9	**39**	20	10
Lack of Air Quality	42	4	6	13	3	6	5	12	17	16	5	2	3	9	**23**	10	3
Lack/Deficient Sanitation Systems & Water Quality	44	5	6	12	2	5	9	14	13	16	7	4	4	9	**24**	11	6
Mental Health	62	6	9	23	2	9	6	18	25	13	15	11	7	8	**38**	14	5
Improved Education & Training	47	5	7	18	1	8	4	14	21	9	12	9	4	8	**30**	12	3
Green and Circular Economy	97	12	11	26	6	11	12	**27**	33	28	17	14	9	10	**50**	23	6
Enhancement of Economic Efficiency & Competitiveness	16	7	0	2	1	0	1	**13**	1	1	6	3	2	2	2	**10**	2
Employment (Job) Development	61	8	6	21	3	8	4	14	28	15	11	10	5	9	**37**	10	4
Innovation and Green Entrepreneurship	16	0	**9**	4	0	3	5	4	2	**10**	4	2	0	0	**12**	3	1
Climate Change	81	7	12	22	3	15	8	18	27	34	11	7	8	9	**49**	15	6
Urban Heat Island & Heatwaves	28	0	9	5	4	6	5	4	9	**19**	3	1	3	0	**19**	6	2
Sea Level Rise & Coastal Resilience	17	1	7	2	1	5	3	2	3	**11**	2	2	1	0	**11**	2	2

>≈50% <10%

Case studies per contextual

Notes: Aw = Tropical Savannah with dry winter; Cfa = Temperate climate without dry season and with hot summer; Cfb = Temperate climate without dry season and with warm summer; Csa = Mediterranean hot summer; LA = Community of Latin American and Caribbean States; RA = Rest of America; EU. = Europe; AS=Asia; LMA = Large Metropolitan Areas (>1.5 Million inhabitants); MA = Metropolitan Areas (500.000-1.5 Million inhabitants); MC = Medium Cities (200.000-500.000 inhabitants); SC = Small Cities (200.000-50.000 inhabitants); VSC= Very Small Cities (<50.000 inhabitants); HI = Higher income class country; UMI = Upper-medium income class country.

Table 2.2 (Continued). Urban (sub)challenges most frequently assessed per contextual class in the literature review on urban challenges. Bold underlined numbers in a cell indicate that more than 50% of the case studies including a specific (sub)challenge come from that specific contextual class.

Urban Challenges & sub-challenges	No	Climate classes*				Continents*				Size of Urban Areas					GNI Classes*		
		Aw	Cfa	Cfb	Csa	RA	AS	LA	EU	LMA	MA	MC	SC	VSC	HI	UMI	LMI
Cases per Contextual Class		16	21	39	9	20	18	45	47	49	30	20	17	10	77	42	11
Water Management	85	10	17	13	3	12	15	29	18	35	17	13	8	1	41	27	9
Flooding Risk	36	2	11	9	3	9	6	7	14	**25**	5	6	3	1	**28**	10	2
Freshwater Shortage	42	5	6	5	2	5	9	15	5	20	11	3	4	0	17	13	8
Wastewater Management	33	7	4	3	2	4	4	**19**	3	12	9	5	5	0	11	14	5
Material Solid Waste	81	7	11	21	5	9	9	23	24	26	13	10	9	9	**42**	20	6
Food Shortage	28	0	5	8	3	6	2	1	10	12	4	4	3	0	**19**	2	1
Solid Waste Management	53	7	7	13	3	5	8	22	13	16	9	6	7	9	24	17	5
Energy	56	1	7	18	4	8	8	7	22	20	7	3	5	10	**34**	8	1
Enhance Energy Efficiency	23	0	2	**12**	1	4	3	1	**12**	6	3	1	2	8	**16**	2	1
Biodiversity	42	2	4	18	5	5	7	5	**21**	11	5	5	5	10	**29**	5	3
Digital Connectivity	36	5	4	12	3	5	5	8	15	7	5	5	5	9	**20**	6	4
Public Participation	45	5	7	15	4	6	7	13	16	14	10	5	4	9	**27**	11	3
Empower Communities in Decision-making	14	4	3	2	2	2	4	6	1	4	5	1	3	1	5	**7**	1
Promote Stakeholder Involvement	18	0	3	3	4	4	4	4	4	**9**	4	2	1	1	**11**	4	1
Governance	59	10	5	8	3	8	4	25	13	13	14	9	11	0	23	17	6
Expenditure	27	7	5	2	0	3	1	**20**	1	3	11	4	4	1	7	**14**	2
Expenditure & Debt Management	18	4	4	1	0	2	0	**15**	0	3	**9**	1	3	1	5	**9**	2
Digital Connectivity	36	5	4	12	3	5	5	8	15	7	5	5	5	9	**20**	6	4

>≈50% <10%

Case studies per contextual

Notes: Aw = Tropical Savannah with dry winter; Cfa = Temperate climate without dry season and with hot summer; Cfb = Temperate climate without dry season and with warm summer; Csa = Mediterranean hot summer; LA = Community of Latin American and Caribbean States; RA = Rest of America; EU. = Europe; AS=Asia; LMA = Large Metropolitan Areas (>1.5 Million inhabitants); MA = Metropolitan Areas (500.000-1.5 Million inhabitants); MC = Medium Cities (200.000-500.000 inhabitants); SC = Small Cities (200.000-50.000 inhabitants); VSC= Very Small Cities (<50.000 inhabitants) HI = Higher income class country; UMI = Upper-medium income class country.

2.4.2. Identification of ecosystem services and nexuses with urban challenges

In the ES literature, the most frequently mentioned UC and sub-challenges (right-hand column of Table 2.1) are *physical health* (39% of the papers), *mental health* (29%), *climate change* (29%), *water management* (22%) and the sub-challenge *urban heat island and heatwaves* (15%). In contrast, several UC are not explicitly mentioned in the ES articles (e.g. *mobility, digital connectivity, governance*), which could be interpreted as a lack of direct causal relationship between ES and UC, and therefore NBS. In addition, for some UC (e.g. *social cohesion, green and circular economy*) either no sub-challenges are specified in the papers or it was difficult to distinguish them (e.g. *energy* sub-challenges) due to the use of vague terminology.

In terms of ES, regulation services are the most frequently assessed (131 papers), followed by cultural services (73 papers) and provisioning services (41 papers). This is consistent with previous literature reviews (Ziter 2016, Luederitz et al. 2015, Haase et al. 2014). The analysis of ES studies by urban contextual classes does not present major differences and therefore this aspect is not discussed. In addition, several ES classes of CICES v5.1 are not identified (e.g. *visual screening* and *smell reduction*) or appear very rarely (e.g. *noise attenuation*, *weathering processes*). A few of these ES classes, such as *smell reduction*, are not identified as independent classes either in CICES before version 5.1 or in other classifications (e.g. TEEB), which could explain the lack of related case studies. Furthermore, the assessment of some ES classes (e.g. *bioremediation*) is quite specific (i.e. pollutant by pollutant) and technically complex (e.g. analyses require detail characterisation of attributes and several years of experimentation in the field and lab testing). The lack of identification of these ES classes might also indicate that they have minor or no relevance to address UC or that urban NBS do not have the capacity to supply them. For the remaining ES, the review suggests multiple links occurring with UC for each ES, as visualised in Figure 2.6. The above results might help policymakers to frame their urban agendas, prioritising the supply of those ES classes for which at least a nexus with a UC has been identified reiteratively by academic papers.

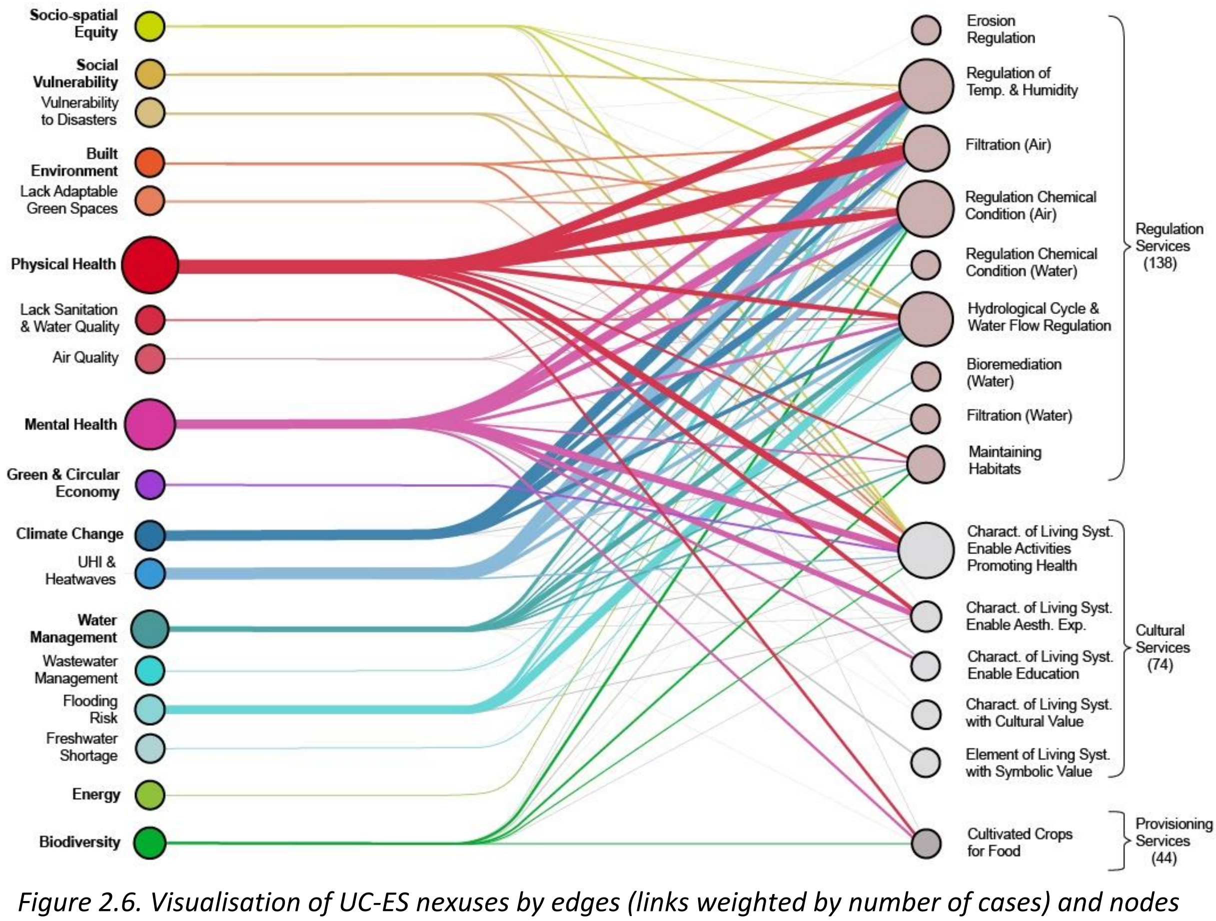

Figure 2.6. Visualisation of UC-ES nexuses by edges (links weighted by number of cases) and nodes (circles weighted by the number of ES and cases per ES). UC are written in bold. UC = Urban Challenges; ES = Ecosystem Services

The visualisation of the UC-ES nexuses shows that *physical health* and *mental health* are not only the most frequently mentioned UC, as shown in Table 2.1, but also are the ones with the highest number of links to different ES, including classes from all ES sections. The UC c*limate change* and the sub-challenges *urban heat island effect* and *heatwaves* are the only ones for which nexuses are mostly concentrated in few regulating ES. Overall, the most frequently mentioned UC show numerous links with the *regulation of chemical condition of the atmosphere*, *regulation of temperature and humidity*, *hydrological cycle and water flow regulation* and *filtration/sequestration/storage of pollutant* (air or water). In addition, *mental and physical health, biodiversity, built environment, lack of adaptable and green spaces, social vulnerability, vulnerability to disasters and socio-spatial equity* appear several times in relation to the *characteristics of living systems that enable activities promoting health*.

Cause-effect relationships underpinning the identified UC-ES nexuses are in many cases evident and can be easily understood. For example, this is the case for the relationship between *characteristics of living systems that enable activities promoting health* (ES class belonging to the ES section of “cultural services”) and the mitigation of *physical health* or *mental health* issues (see UC in Table 2.1). Similarly, the relationship between the above ES class and the UC *socio-spatial equity* can also be easily explained once it is known that several case studies describe a lack of adequate distribution of living systems (and therefore their associated *characteristics promoting health*) in urban areas. This is why the study of this ES class in regard to *socio-spatial equity* is recurrent in the ES papers reviewed. In fact, this outcome is consistent with several papers on environmental justice focused on urban inequity derived from the distribution of green areas (Lin, Meyers and Barnett, 2015; Shen, Sun and Che, 2017; Anguelovski *et al.*, 2018; Wu *et al.*, 2018). In contrast, for other UC-ES nexuses, such as *characteristics of living systems that enable activities promoting health* (ES) and *biodiversity* (UC), the direct causality is not evident. However, it might be that an indirect causal relationship exists due to synergies and trade-offs between the supply of this ES and the supply of other ES related directly to the UC *biodiversity*. For example, Lin et al. (2018) show that an increase in the *maintenance of habitats and gene pool reserve* might also increase the presence of the *characteristics of living systems that enable activities promoting health* and mitigate *biodiversity* decline.

2.4.3. Limiting factors and UC-ES-NBS causal relationship groups

The results from the above sections, together with the characterisation of the nature of limiting factors (e.g. biophysical, technological) driving specific UC provide the basis to differentiate groups of UC-ES-NBS relationships (Figure 2.7), identifying when direct plausible causal relationship between UC-ES-NBS exist.

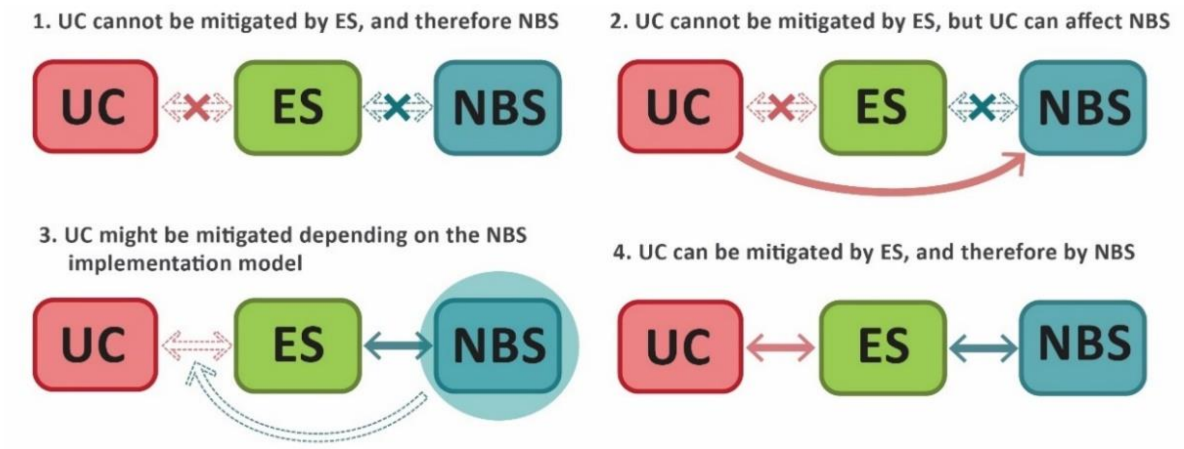

Figure 2.7. Scheme of the four types of causal relations among UC, ES and NBS; UC=Urban Challenge(s); ES=Ecosystem Service(s); NBS=nature-based solution(s).

In the first group of UC-ES-NBS (upper left, Figure 2.7) neither direct nor indirect plausible causal relationships occur. This is the case of *digital connectivity* and *mobility*, which are both limited by human and technology limiting factors, rather than biophysical ones. The enhancement of ES would neither be able to compensate for the limitations and nor mitigate or address these UC. For example, an NBS could make a road infrastructure more pleasant, or protect it against flooding, but it cannot overcome mobility issues due to infrastructure limitations. Nevertheless, these UC would not jeopardise either the implementation of NBS or their perception by people.

In the second group (upper right Figure 2.7), the limiting factors of UC are of human-social nature, such as decision-making issues (*governance, public participation and expenditure)* or changes in socio-economic trends (*demographic dynamics*), which cannot be compensated by ES. Some of these UC are relevant to several urban contexts (as shown in Table 2.2). In contrast to the first group, the planning and implementation of NBS in those urban contexts might need to tackle these UC or adapt to the consequences resulting from them to ensure the subsequent wide acceptance/use of the NBS implemented by society. For example, in the case of *demographic dynamics,* when related to an ageing population, NBS will not be able to mitigate this issue. However, if the ageing issue exists in an urban context, it might need to be considered when planning or designing NBS for mitigating other UC. This would be the case of a municipality with aging issues that implements urban forests to increase (in the long term) nature-based recreation with the aim of enhancing the physical health conditions of its citizens. In that case, to ensure that the accessibility and usability of the urban forests is adequate for an elderly population might be more relevant than in other contexts. As another example, to mitigate UC related to decision-making issues, representatives of different political parties, levels of governance, private partnerships and citizens need to be included early on in the planning and implementation phases of NBS and consensual planning should be followed to avoid failure or non-implementation due to citizens' opposition or changes in the government representatives. In the same sense, Dorst et al. (2019) state that the multifunctionality of NBS can be hampered by *governance* issues such as fragmentation (or lack of consensus) in the decision-making process.

In the third group (lower left Figure 2.7), the UC (*socio-spatial equity, social cohesion, social vulnerability, green and circular economy*) are partially driven by limiting biophysical factors and/or their enhancement might contribute to the mitigation of UC. However, the mitigation of UC depends on *how*, *where*, and *for whom* (i.e. the interests of which social groups are considered) NBS are implemented. For example, in terms of *how* to move towards *a green and circular economy*, business and financial mechanisms need to be considered in the implementation of NBS. This will ensure the economic exploitation of their products and services (Chen and Warren, 2011; Toxopeus and Polzin, 2017; TECNALIA *et al.*, 2018). Concerning *where* and *for whom*, in order to address *socio-spatial equity* factors such as location of NBS with respect to existing recreational areas (i.e. *where*), public accessibility, inclusion of different social groups in the plan, design, and implementation stages (i.e. *for whom*) and potentially governance aspects (e.g. to prevent green gentrification) need to be considered (Almohamad et al., 2018; Haase et al., 2017; Park & Kim, 2019). In this sense, as already illustrated by other scholars through empirical cases (Haase et al., 2017; Kotsila et al., 2020), NBS per se do not solve social cohesion issues. In order to tackle social and economic UC, the specific NBS implementation model chosen, including governance, finance and business mechanisms, is also relevant.

The fourth group (lower right Figure 2.7) occurs when the UC (or one of its sub-challenges) is driven by limiting biophysical factors or could be overcome by processes depending on biophysical attributes, which influence the ES supply. These are mainly the cases where the nexuses between UC and ES are evident or can be easily explained, as illustrated at the end of Section 2.4.2. For example, in cities suffering from the *urban heat island effect*, the solution requires the *regulation of temperature and humidity* through the enhancement of biophysical processes such as evapotranspiration and/or shading (see Zardo et al. 2017). Section 2.4.6 describes the identified social and biophysical attributes and processes, on which the supply of the most frequently mentioned ES (of this review) depend.

2.4.4. Classification and identification of NBS types

In the systematic and non-systematic literature reviews, solutions based on (or applied to) nature or living features were often framed under different ecological concepts: green infrastructure, urban green (and blue) spaces, services providing units, services providing elements, sustainable urban drainage systems and ecosystem-based adaptation. As already stated in Section 3.2, these ecological concepts and their specific solutions could be assimilated under the umbrella of NBS. For example, ecosystem-based adaptation refers to the protection, management and restoration of the spatial structures (Munang et al. 2013, Geneletti and Zardo 2016) corresponding to NBS Type 1 and 2. As another example, green infrastructure types, urban green (and blue) spaces, services providing units and sustainable urban drainage systems correspond to biophysical structures and in many cases are equivalent to specific MAES Ecosystem types (and NBS Type 3). Hence, following the NBS conceptualisation (Figure 2.4), specific solutions framed under these ecological concepts were included in the three main types of urban NBS, leading to the detailed classification of NBS types shown in Figures 2.8 and 2.9.

In the systematic literature, only urban case studies referring to (semi)natural ecosystems and NBS Type 3 are found. This illustrates a clear prevalence of physical solutions in urban ES studies, although the underlying reason is not entirely clear. It could be because there is still a strong need to bring back natural structures into cities before solutions focused on managing and restoring them (NBS Type 1 and 2) become relevant. It could also mean that the extensive and intensive modification of existing urban ecosystems and the creation of new ones are perceived as more effective solutions for addressing UC. Alternatively, the lack of Type 1 and 2 urban NBS in our review may be due to the difficulty of assessing them independently from the physical structures on which they are applied. For example, for the assessment of the individual contribution of NBS Type 1 to ES supply it might be necessary to compare changes in the supply of ES by the same physical structure before and after an NBS Type 1 or 2 was applied. Furthermore, urban environmental management and urban ecological restoration might not usually be framed in the research area of ES and therefore these related studies were not captured in our review.

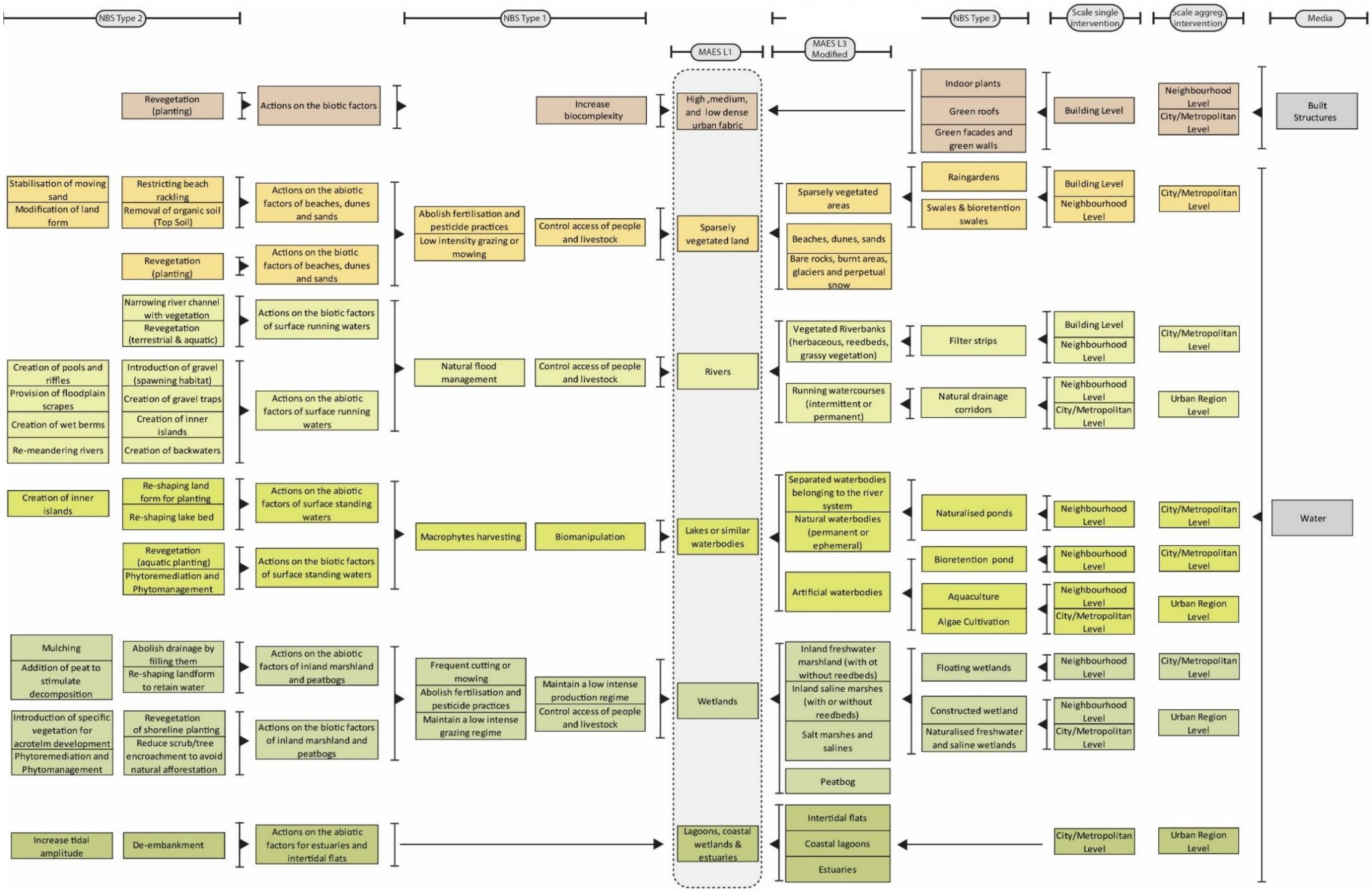

Figure 2.8. Built structure and water NBS Types 1, 2 and 3 (as described in Figure 2.4) corresponding to MAES Level 1 ecosystem types. NBS Type 2 are differentiated in actions that apply to the abiotic and the biotic factors of NBS. The spatial scales refer only to NBS Type 3. NBS = Nature-based solutions

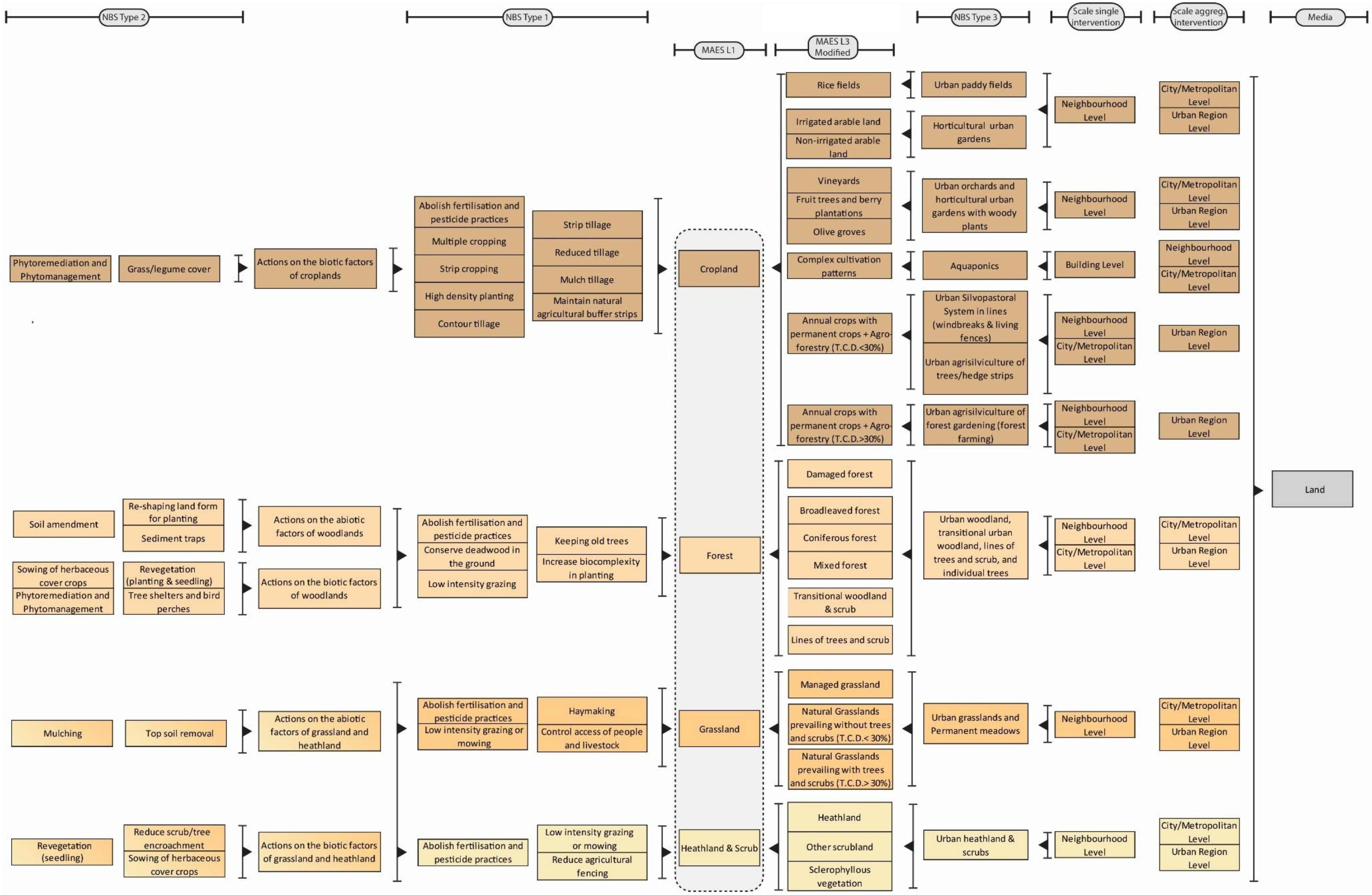

Figure 2.9. Built structure and water NBS Types 1, 2 and 3 (as described in Figure 2.4) corresponding to MAES Level 1 ecosystem types. NBS Type 2 are differentiated in actions that apply to the abiotic and the biotic factors of NBS. The spatial scales refer only to NBS Type 3. NBS = Nature-based solutions

The most frequently studied NBS Type 3 per type of media are green roofs, green walls, woodland-like structures, urban grasslands and meadows, urban scrubland and heathland, horticultural gardens, vegetated filter strips, swales, constructed wetlands, natural(ised) wetlands, natural(ised) ponds and bioretention basins (Figure 2.10). Similar to previous reviews, woodland-like structures appear as the most frequently studied supplier of ES (Haase et al., 2014; Keeler et al., 2019; Luederitz et al., 2015). In contrast to previous reviews, studies of ES supplied by green roofs have increased. For many other solutions, it is difficult to draw parallels with previous reviews due to a lack of common NBS classification and because not all of the previous reviews analysed solutions related to nature (e.g. urban fabric, land use mixture and infrastructure appear as ES suppliers in Haase et al. (2014)). This outcome illustrates the relevance of an agreed-upon NBS classification, to allow comparisons and facilitate transfer of information to support urban policies, strategies and interventions.

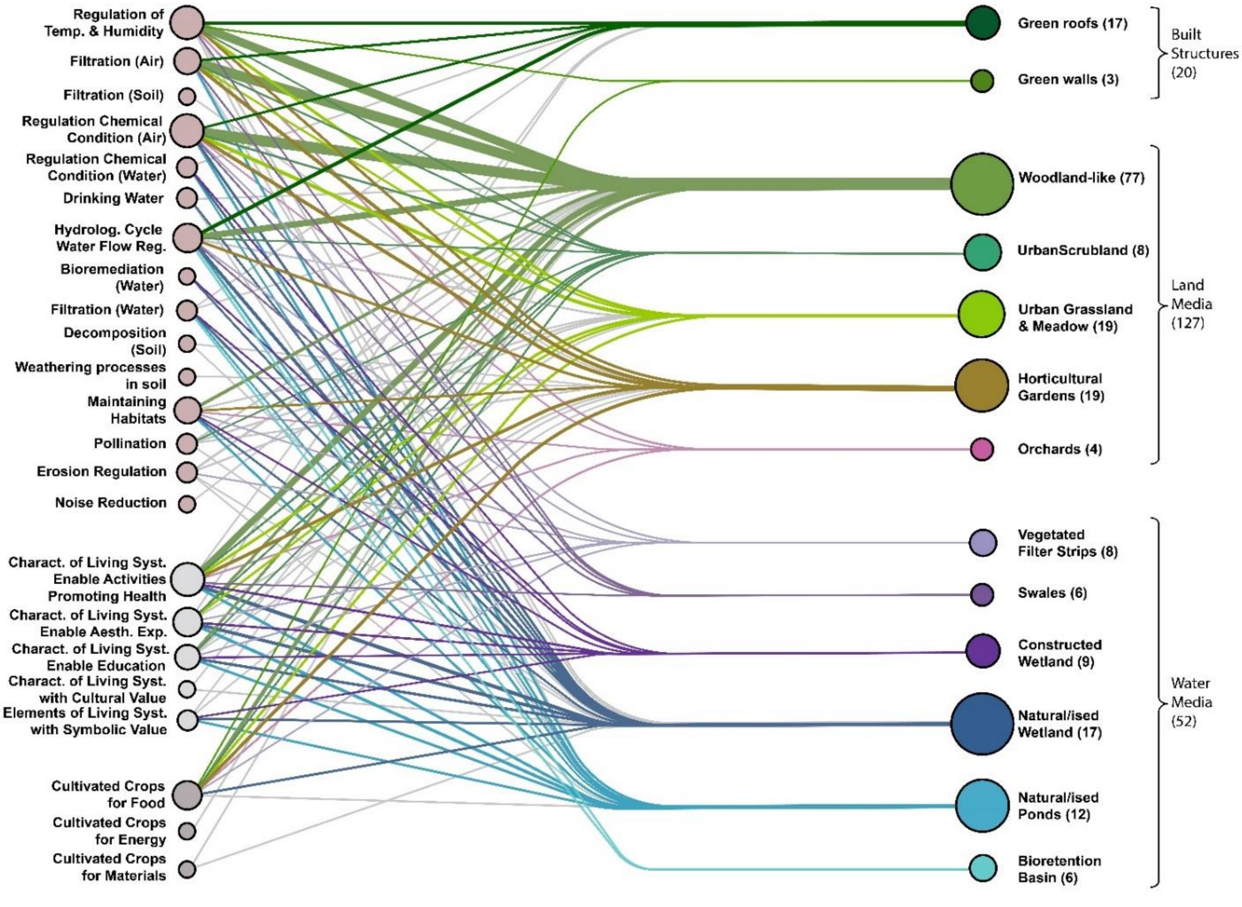

Figure 2.10. NBS (Type 3)-ES links identified in the ES review. Edges (links weighted by number of case studies) and Nodes (circles weighted by number of ES (or NBS) and number of case studies per ES). NBS = Nature-based solutions; ES = Ecosystem Services.

2.4.5. Identification of links between ecosystem services and nature-based solutions

Green roofs, woodland-like, urban grasslands and meadows, horticultural gardens, (natural(ised) wetlands and natural(ised) ponds have links to a higher number of ES classes than other NBS (Figure 2.10).

Regarding the specific ES-NBS nexuses, all the highlighted NBS except vegetated filter strips, green roofs and green walls appear to be strongly associated with *characteristics of living systems that enable activities promoting health*. In addition, green roofs (built structures media) are mostly linked to regulation services (i.e. *regulation of temperature and humidity, filtration, sequestration, storage of air pollutants and regulation of the hydrological cycle and water*). Urban woodland-like environments, urban grasslands and horticultural gardens (land media) are also often associated with the *regulation of temperature and humidity*, *regulation of chemical condition of the atmosphere, filtration, sequestration and storage of air pollutants, regulation of hydrological cycle and water flow* and *cultivated terrestrial plants for nutritional purposes*. Natural(ised) wetland, constructed and natural(ised) ponds (water media) prove to be relevant not only for regulation services but also for cultural ones, being mostly linked to the *regulation of chemical conditions* (water), *regulation of hydrological cycle and water, filtration (water), sequestration and storage of water pollutants, characteristics of living systems enabling aesthetics experiences,* and *education and symbolic value*.

As with the visualisation of the UC-ES nexuses (Figure 2.6), Figure 2.10 shows that the ES classes *smell reduction, erosion regulation* and *bioremediation* are not mentioned in connection with NBS or seem to have low momentum in urban studies. Figure 2.9 also presents ES classes rarely assessed in the case studies, which do not appear in Figure 2.6 (i.e. *drinking water, pollination, filtration (soil), decomposition (soil), weathering processes, noise reduction, cultivated crops for manufacturing purposes and cultivated crops for energy*). The differences between both figures occur because 55 of the 170 papers in the ES literature review do not clarify the specific UC that could be addressed by the specific ES studied therein. Consequently, links with UC are not established for all the ES classes identified in the literature review. The review of Haase et al. (2014) also shows the scarce consideration of many of the abovementioned ES classes in urban studies, making it less likely that this outcome is the result of an unnoticed bias in the current review. It might be that the roles of those ES classes in urban areas are not clear enough. A reason that could also explain why in this review clear connections with specific UC are not found for many of those ES classes.

Similar to the results outlined in Figure 2.6 for the UC-ES nexuses, multiple links between specific ES classes and NBS do also occur. This is unsurprising given that NBS are described as solutions capable of addressing multiple challenges and providing multiple benefits (see Section 1). To investigate the causality underpinning these nexuses, the processes and factors responsible are discussed below. However, in some cases, especially for cultural services, these processes and factors were difficult to identify. For example, in *characteristic of living systems that enable education* or *with symbolic values* or *resonant in cultural values*, the attributes are very subjective and it was not possible to identify the ones that were reiterated several times in the case studies and that did not rely on human agreements, such as the application of a legal protection status.

2.4.6. Interpretation of the links between ecosystem services and nature-based solutions

The main social and biophysical attributes and processes identified in the reviews, which influence the most frequently studied ES classes, are summarised in Table 2.3.

In most cases, biophysical processes are partially dependent on attributes of NBS. For example, the amount of tree canopy coverage, which influences shading, and therefore *regulation of temperature and humidity,* relates to a biotic component (trees) of woodland-like NBS type. In other cases, such as for the sub-challenge *lack of water quality,* it is important to know which pollutants are generating the issue. This helps understanding whether regulation services that could mitigate the issue (e.g. *filtration, sequestration, storage and accumulation by plants, microorganisms and algae*) can be provided by the NBS or not. In some cases it is difficult to state the suitability of an NBS for supplying an ES class without a detailed analysis of the UC and related limiting factors. In the case of water-related ES, the reviewed literature focused mainly on processes specific to nitrogen and phosphorus compounds (e.g. Adhikari et al., 2011; Adyel et al., 2016; Liquete et al., 2016; Nocco et al., 2016; Olguin et al., 2017; Sun et al., 2017) and very few papers on processes specific to other pollutants (e.g. Krzeminski et al., 2019). Several reviewed papers also emphasise that physical attributes of the human-made contexts and/or their social characteristics (e.g. people's perception) are relevant to the supply of ES by NBS, especially in the case of cultural services (e.g. Andersson-Sköld et al., 2018; Brill et al., 2017; Fry et al., 2009; Ode et al., 2008; Szücs et al., 2015).

The abovementioned findings confirm the importance of making explicit which attributes of specific NBS, and the urban context, influence the processes underpinning the supply of a particular ES. It is necessary to state whether certain ES-NBS nexuses are plausible causal relationships or not. These reflections are also stressed by Keeler et al. (2019), who state i) that decision-makers need more applied information describing the conditions (of NBS and contexts) in which specific approaches and NBS are effective for some UC; and ii) that many NBS studies overlook the influence of social, ecological and technological factors when assessing their performance. For example, the *regulation of hydrological cycle and water flow* depends on infiltration, with several papers in this review indicating that it is influenced by root distribution and depth, among other attributes (Kim et al. 2016; Zölch et al. 2017). However, root depth varies depending on whether we implement a horticultural garden or a woodland (two of the NBS indicated as suppliers of this ES class). It also varies inside the same NBS depending on attributes such as the plant species. Therefore, the contribution of NBS attributes should be carefully considered by policymakers and urban planners when designing, implementing or assessing NBS with the aim of enhancing the supply of specific ES and, as a consequence, addressing a particular UC.

Table 2.3. Processes and factors per ES class identified in the systematic and non-systematic literature review. Blue cells identify processes already described in an upper ES or an ES that influences the ES class analysed (references per process included in Annex 2.4). ES = Ecosystem Services

ES Classes	Related Processes (conditions and other ES)	Related Attributes and Associated Processes
Regulation of temperature and humidity	Shading	**Vegetation growth**, Tree canopy coverage, Dimension of trees, Leaf area
	Alteration of wind movement	**Vegetation growth**, Shape of the open space, Shape of the buildings
	Evapotranspiration	**Vegetation growth**, Land cover type, Temperature, Precipitation, Tree canopy coverage, Leaf area index, Root depth and distribution, Soil (vegetation) cover, Soil permeability, Soil texture, Soil moisture, *Shape of vegetated area*
	Insulation (buildings)	Soil depth, Thermal capacity of soil substrate (and building materials)
	Vegetation growth	Temperature, Precipitation, Growing season, Land cover type, Vegetation composition and density, Plant health condition, Light conditions, Landscape (vegetation) management, Soil texture, Soil nutrients, Soil depth, Dimension of tree & Wood density (only for trees)
Regulation of hydrological cycle and water flow	Interception	**Vegetation growth**, Precipitation, Tree canopy coverage, Leaf area index, Plant vegetation composition and density
	Detention	Precipitation, Land surface slope, Roughness of land surface
	Infiltration	**Vegetation growth, Detention, Soil storage**, Precipitation, Soil permeability, Soil moisture, Root distribution & depth
	Soil storage	**Infiltration**, **Evapotranspiration**, **Percolation**, Depth of depression storage, Soil texture, Soil depth
	Evapotranspiration	-
	Water run-off	**Infiltration**, **Soil Storage**, Precipitation, Land slope, Roughness of land surface
	Groundwater lateral flow	**Not identified**
	Percolation (deep infiltration)	**Infiltration, Storage**, *Soil permeability, Soil moisture*
	Vegetation growth	-
Regulation of chemical condition of the atmosphere	Carbon uptake (by vegetation and soil)	**Vegetation growth, Soil respiration**, Dimension of trees, Species wood density, *Carbon fraction of dry biomass (species specific),* Landscape (vegetation and soil) management.
	Organic matter decomposition	***Soil respiration, Vegetation growth***, *Degradation rates of organic matter types, Land management, Temperature*
	Soil respiration	*Soil texture, Soil moisture, Landscape management, Organic matter inflow, Percentage of humus, Microbiological activity, Temperature*
	Vegetation growth	-
Filtration, sequestration, storage, accumulation by microorganisms, algae, plants (air)	Dry deposition (on vegetation)	**Vegetation growth**, Tree canopy coverage, Vegetation composition & density, Canopy height, Leaf area index, Shape of leaves, Roughness of leaves (due to hair, Exudates), Phenology of Plant, Plant Diversity, Concentration of air pollutants, Deposition velocity, Distance from emission source
	Re-suspension	*Wind speed, PM amount in the leaf surface*
	Washing-off	*PM amount in the leaf surface,* **dry deposition of PM**, *Precipitation*
	Pollutants' plant uptake	**Vegetation growth**, Leaf area, Soil moisture, Growing season, Light condition, Health of trees, Stomatal resistance, *Aerodynamic resistance, Quasi-laminar boundary layer resistance, Canopy resistance (stomatal resistance, mesophyll resistance, cuticular resistance and soil resistance), Concentration of CO2*
	Biological emission of particulates (including pollen)	**Vegetation growth**, Total leaf biomass, Plant diversity, Plant composition, Type of pollination, Duration of pollen season, Height of plant
	Vegetation growth	-
Filtration, sequestration, storage, accumulation by microorg., algae, plants (water) / Regulation of the chemical condition of freshwaters by living processes	*Dry deposition (on water)*	*Deposition velocity*
	Settling (sedimentation)	Roughness of land surface
	Adsorption by sediments	**Not identified**
	Re-suspension (from water)	**Not identified**
	Nitrification	*Temperature, Soil moisture*
	Denitrification	*Temperature,* Dissolved oxygen, *Water saturation, Dissolved and particulate organic carbon, Soil depth, Soil permeability, pH, Soil cover*
	Ammonification	*Temperature, pH, C/N ratio, Available nutrients, Soil texture, Soil permeability*
	Mineralisation	*Temperature, Soil moisture*

Note: Words in italics are for processes, factors and references identified through the non-systematic review

Table 2.3 (Continued) Processes and factors per ES class identified in the systematic and non-systematic literature review. Blue cells identify processes already described in an upper ES or an ES that influences the ES class analysed (references per process included in Annex 2.4). ES = Ecosystem Services.

ES Classes	Related Processes (Conditions and other ES)	Related Attributes
Filtration, sequestration, storage, accumulation by microorganisms, algae, plants (water) / Regulation of the chemical condition of freshwaters by living processes	Plant & microbial uptake (and immobilisation)	**Vegetation growth**, Plant capacity to absorb inorganic nitrogen (N), Rate of conversion of N into biomass, Bioavailability of N, *Temperature, Soil moisture*
	Organic matter decomposition	-
	Leaching	**Plant & microbial uptake**, **Adsorption, Soil sorption, Infiltration**, Soil depth, Soil nutrients
	Volatilisation	**Not identified**
	Soil sorption	*Amount of organic matter, Degradability of the pollutant, Temperature, Precipitation*
	Vegetation growth	-
	Erosion regulation (ES)	-
	Water run-off	-
	Groundwater lateral flow	-
	Infiltration	-
	Percolation (deep infiltration)	-
Bioremediation by microorganisms, algae, plants and animals	*Phytoextraction (plant & microbial uptake)*	***Vegetation growth**, **Soil storage**, Hyperaccumulation capacity of plant, Plant biomass, Planting density, Cropping period, Compartmentalisation of pollutants in biomass, Rooting depth, Plant's resistance trait to the specific pollutant, Landscape management, Soil aeration, Presence of macronutrients, Soil microbiological activity, Soil pH*
	Phytodegradation	**Not identified**
	Rhizofiltration	***Vegetation growth**, Microbiological activity, Rooting depth, Exhudation*
	Phytostabilisation	*Microbiological activity*
	Phytovolatilisation	***Vegetation growth, Plant uptake, Evapotranspiration***
	Evapotranspiration	-
	Leaching	***Phytodegradation, Rhizofiltration, Phytostabilisation, Phytovolatilisation***
	Adsorption by soil	-
	Soil sorption	-
	Infiltration	-
	Vegetation growth	-
Maintaining nursery populations and habitats	Conservation of habitat	*Vegetation patches: abundance, richness, distribution& area (single patches)*
	Movement of species	*Dimension of vegetation patches, Number of vegetation patches, Distances between vegetation patches, Presence of barriers, Suitability of surrounding patches for movement*
Characteristics of living systems enabling activities promoting health or enjoyment	Accessibility (visual, physical, legal)	Presence of barriers (e.g. fences, dense vegetation), Public access allowance, *Proximity*
	Provision of recreational infrastructure	Footpaths and cycling routes, *Plant diversity adjacent to paths*, *Lighting along paths*, Sport facilities, Benches, Pleasant views, *Existence of forests, Water features, Parking lots*
	Perception of safety	*Lack of traffic, Availability of footpaths*
	Social characteristics	People's age, Health, Level of education
	Aesthetics (ES)	-
Characteristics of living systems enabling aesthetic experiences	Sensorial perception	Number of view axes, Presence & view of landmarks, Ratio between open spaces and forests, Existence of (mature) forests, Existence of fruit trees, Amount of natural vegetation, Amount of naturalised waterbodies, Diversity of natural features, Landscape management, Number of disturbing elements*, Shape diversity, Patch and edge attributes and Seasonal changes in vegetation & water* features
Plants cultivated for food purposes	Vegetation growth	-
	Pollination (ES)	*Abundance and distribution of pollinators, Availability of forage and nesting habitat, Landscape management*

Note: Words in italics are for processes, factors and references identified through the non-systematic review

As another outcome, some ES share several processes and attributes, which implies that interdependencies occur in the supply of ES (Table 2.3), as shown by other scholars (Bennett, Peterson and Gordon, 2009; Lorilla *et al.*, 2018). For example, as stated in several papers of this review, evapotranspiration is one of the main processes for *regulation of temperature and humidity*, as well as for *regulation of hydrological cycle* (Lundy and Wade, 2011; Skelton *et al.*, 2011; Nocco, Rouse and Balster, 2016; C. C. Reynolds *et al.*, 2017; Pappalardo *et al.*, 2017a; Zardo *et al.*, 2017). Evapotranspiration, more specifically transpiration, is also identified as an influence of several processes of *bioremediation by plants* (e.g. phytovolatilisation), since it affects the rate at which pollutants are taken up and expelled from plants (Pulford and Watson, 2003; Singh and Santal, 2015). Moreover, Table 2.3 also illustrates that the supply of some ES classes might be indirectly influenced by other ES, or might prevent their demand in the first place, such as in the case of *erosion regulation* (De Troyer, S. Mereta, *et al.*, 2016). Erosion dynamics influence the pollutants transported from one point to another, and therefore the input of pollutants to be filtrated, sequestrated or stored. Consequently, it seems necessary to study the supply of ES by NBS in bundles, defined as a set of ES that usually appear together (Yang *et al.*, 2015), instead of individually. The importance of studying ES in bundles is also stressed in the review of Smith et al. (2017). Such an approach may further allow accounting for ES synergies and trade-offs, and should be recommended when planning/designing, assessing and monitoring the performance of NBS. The consideration of ES in bundles should also be recommended by policies and guidelines supporting the mainstreaming of NBS or establishing rules for their adequate implementation

2.4.7. UC-ES-NBS Framework

Our results were used to create a network diagram outlining all the UC-ES-NBS relationships observed. An illustrative example of this diagram, built in Gephi, is shown in Figure 2.11 for the relationships of the UC *Built Environment*. Further information for the remaining UC-ES-NBS relationships is included in Annex 2.5 and 1.6.

Based on the evidence of this review, the UC-ES-NBS network diagram obtained as the final outcome makes explicit the nexuses that are plausible causal relationships between NBS, ES and ultimately UC. The diagram confirms that NBS provide multiple benefits and adequately address multiple challenges, even if not all. In addition, nexuses between specific classes of contextual attributes and UC that were highly reiterated (i.e. in more than 50% of the case studies including a specific class of contextual attribute) are also included. However, in the latter case plausible causal relationships were not studied.

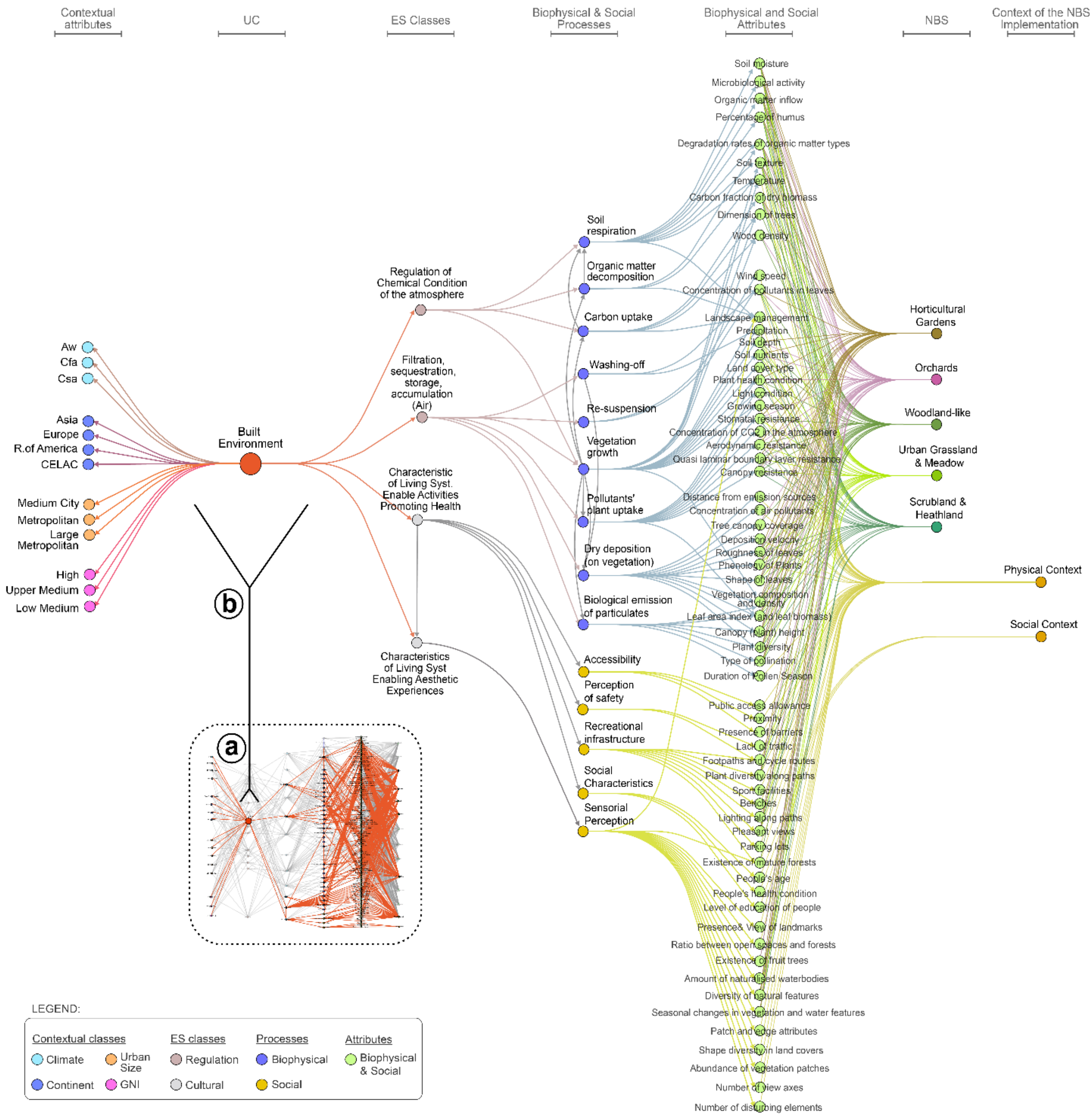

Figure 2.11. Illustrative diagram of Contextual attributes-UC-ES-Process-Attributes-NBS Network for the UC Built Environment. a) Diagram of the complete network, only relationships of the Built Environment are in colour; b) Zoom to the Built Environment Network Diagram. Arrows are non-weighted to facilitate visualisation. UC = Urban Challenges; ES = Ecosystem Services; NBS = Nature-based solutions

The network diagram, in the two formats (html and Gexf) provided in the Annex 2.6, can also be used by professionals of different backgrounds as a visual user-friendly output. The file in html format (accessible at https://mimes.list.lu/articles/network) allows an easy visualisation of the whole network and the detailed first order relationships without requiring any software. A more detailed visualisation (and manipulation) can be performed using the Gexf file, which shows relations of all the orders for each UC following the instructions included in the Annex 2.6. Therefore, readers can use one of the three alternatives included (Figures in Annex 2.5, or html and Gexf file in Annex 2.6) to see how specific UC can be addressed by means of ES, the processes on which the latter depend, and which attributes of NBS and their surrounding urban context mediate in those processes.

In combination with the NBS classification of Figure 2.8, the diagram could be used to identify the specific NBS of interest for a particular situation, also taking into account the suitability of spatial levels and ecosystem types. It also highlights the NBS that need to be further studied in order to understand their capability for addressing specific UC through the supply of ES. In summary, the framework presented here has a double utility. First, it depicts a preliminary framework, in progress, that disentangles a wide set of UC-ES-NBS relationships in a qualitative form, which could inform urban policies, strategies and interventions. Second, in the longer term, this framework could be used as a basis to move towards broad quantitative “correspondence” tables that consider causal pathways and where each NBS is scored against every UC on the basis of the ES classes supplied and their amount. The relevance of the second point is also anticipated in Elliot et al. (2019), who review the integration of urban ES modelling within urban metabolism frameworks.

The usefulness of this framework is illustrated by the municipalities of São Paulo (Brazil) and Xalapa (Mexico), two cases that appear in the UC review. These urban areas share all the contextual attributes studied in the UC review (i.e. large metropolitan urban areas, Cfa climate, CELAC countries, upper-medium income countries). Both cities identify *built environment* (illustrative UC depicted in Figure 2.11) as one of their main UC of concern in the reviewed urban planning documents. In a first instance, the framework presents policymakers and urban planning professionals of both cities with the bundle of ES (third column Figure 2.11) that they should prioritise to mitigate this UC. It also informs them about the underpinning biophysical and social processes to enhance those ES (fourth column Figure 2.11) and how they relate to attributes of NBS and the urban context (fifth column Figure 2.11). These details provide policymakers and urban planners with supporting information to better frame policies, strategies and interventions, as well as monitoring them. Thus, policymakers and urban planners could use the UC-ES-NBS network diagram and the NBS classification (Figure 2.8) in combination to better understand which NBS Type 3 could be more relevant in their specific contexts. For example, they could prioritise a list of suitable NBS based on the dominant urban ecosystems of their municipalities, and spatial levels of proposed interventions. Moreover, in Figure 2.8 they would be able to see non-physical solutions (NBS Type 1 and 2) applicable to selected NBS Type 3 or dominant natural ecosystems of their cities. Policy makers and urban planners might further investigate these non-physical solutions and if they find strong evidence of their value, include them as a complement in policies, strategies and interventions. This alternative might be especially relevant when the feasibility of extensive and intensive physical modifications is limited by space or budget.

2.5. Conclusion

This paper identifies and visualises the nexuses between urban challenges (UC), ecosystem services (ES) and nature-based solutions (NBS), in order to support the mainstreaming of NBS in urban policies and sustainable and resilient urban planning interventions and strategies.

The review of UC literature from science, policy and local urban planning perspectives allowed us to identify and classify 18 UC and 58 sub-challenges for sustainability and resilience. This classification, and its equivalent for NBS, could be further improved and used as a basis for future studies aimed at identifying quantitative causal relationships among UC, ES and NBS. The results show that all the above-mentioned perspectives focus similarly on some UC in particular, namely *built environment issues, physical health, green and circular economy, climate change, water management, and material and solid waste*. The local urban planning literature is the one reiterating more social-led challenges (e.g. *social vulnerability*), showing more concern about this kind of UC. Additionally, several UC (e.g. *socio-spatial equi*ty) are stressed differently depending on the specific urban contextual classes (e.g. large metropolitan areas). In the future, further characterisation of UC with respect to their specific urban conditions can be useful for identifying which challenges arise mainly in urban areas sharing specific contextual attributes. By doing this, these cities can join forces when researching potential solutions and better understand whether certain contextual attributes drive the emergence of specific UC.

The reviews conducted in this paper have permitted us to advance in the classification of NBS types starting from the three broad types defined by Eggermont et al. (2015). More specifically, the overlap between specific NBS Types 1, 2 and 3 has been shown explicitly with the interconnections in Figure 2.8. Furthermore, frequently mentioned ecological concepts used in the urban ES literature (e.g. sustainable urban drainage systems, ecosystem-based approach) have first been identified and then positioned under the NBS umbrella as part of its three broad types.

From the review of urban ES literature, specific nexuses between ES classes, UC and NBS have been identified. The type of relationship between the most frequently mentioned UC, groups of ES and NBS have been discussed, depicting one group with direct plausible causal relationships (albeit qualitative). In addition, social and biophysical attributes, and processes influencing 10 ES have been made explicit, including some of their feedback relationships. These results inform which attributes (of NBS and their contexts) need to be assessed when implementing NBS for the supply of specific ES. Socio-ecological processes are shared among different ES reinforcing the importance to account for ES in bundles to do not miss trade-offs and synergies. As a main output, a network of relationships among UC, ES, processes, attributes and NBS has been generated and exemplified for one UC in Figure 2.11. The complete network of relationships can be visualised in the files included in the Annex 2.6. This

UC-ES-NBS network can be used to provide qualitative advice on urban policies, strategies and interventions that intend to make use of NBS.

Further research is needed to move from this qualitative network towards a quantification of the impact of NBS in the supply of specific ES, and the contribution of these ES to mitigate or overcome specific UC. For example, starting from this preliminary work, NBS models (e.g. mechanistic models, system dynamic models) could be developed to quantify the supply of ES classes in bundles. Some examples of these type of models already exist for urbanised contexts such as ENVI-met, which will be described in Chapter 3. Such a type of model helps to acknowledge shared processes and interactions in the supply of ES and could depict quantitative cause-effect relationships between ES and NBS. Thus, they would be useful to advise on the effectiveness of NBS to address UC and further refine urban policies supporting the mainstreaming of NBS. Chapter 5 presents a modelling framework developed in this PhD thesis, which goes in that direction.

2.6. Additional Information

The Annex 2.7 provides all the data collected in the systematic UC literature review (Spreadsheet 01) and ES literature review (Spreadsheet 02).

Chapter 3

Identifying suitable methods and indicators for the assessment of nature-based solutions

3.1. Introduction

Chapter 2 has identified the potential value of several ecosystem services (ES), related socio-ecological processes and biophysical attributes of nature-based solutions (NBS) for the mitigation of various urban challenges (UC). By identifying these socio-ecological processes and biophysical attributes, it informs on which key elements ES assessment methods should focus to quantify the environmental value of NBS. However, as introduced in Chapter 1, a complete assessment of the contribution of NBS to sustainability and resilience should also consider the economic and social values. Therefore, more complete assessments of NBS need to integrate at least one of the other two dimensions.

In the current literature, an extensive number of ES assessment methods have already been developed to assess positive environmental, economic, and social values derived from ecosystems (Grace *et al.*, 2016; Cheng *et al.*, 2019). In fact, with the intention of creating an operational database of ES assessment methods a previous H2020 project, ESMERALDA, elaborated a detailed classification of them (Santos-Martin *et al.*, 2018).

Environmental values are assessed making use of biophysical assessment methods, which are usually focused on measuring ecosystems' capacity to supply ES in biophysical units (Potschin-Young *et al.*, 2018). In other words, they measure biophysical attributes (ecosystem structure) and ecological processes as parameter-proxies of ES. Economic and social values derived from ecosystems (e.g. harvested wood of a forest) can be (partially) measured making use of biophysical methods (Potschin-Young *et al.*, 2018).

Nevertheless, biophysical methods cannot provide a complete view of the economic welfare derived from ES, since they cannot explicitly acknowledge the demand. As a result, economic values are commonly measured in monetary units to quantify the human welfare generated by goods and benefits derived from ES (Pascual *et al.*, 2010). In practice, the generated human welfare is quantified in the form of total economic value (Figure 3.1.), which considers goods and benefits derived from ES in the form of use (i.e. direct, indirect, optional) and non-use (i.e. altruism, bequest, existence) values (Turner *et al.*, 2003).

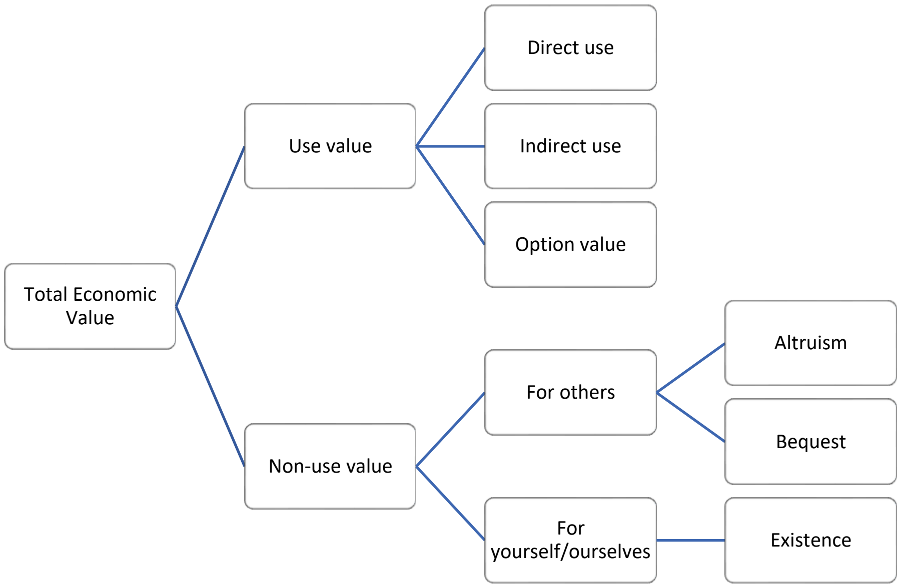

Figure 3.1. Total Economic Value framework

As illustrated in Table 3.1, different primary economic methods (i.e. based on direct data collection), mainly using monetary units (other units such as time can also be used), have been developed to compute use and non-use values (Pascual *et al.*, 2010; Brander, 2013). These methods have different limitations in terms of the accuracy of the estimated values due to their respective assumptions (see limitations in Table 3.1). For example, methods from stated preferences approaches such as contingent valuation are able to elicit both use and non-use values based on individual surveys and questionnaires. However, as a drawback are very time consuming and there is always the risk of overestimation due to the so-called hypothetical bias[5]. Additionally, secondary economic methods, value transfer methods, are also quite used (Costanza *et al.*, 1997, 2014; Troy and Wilson, 2006; European Environmental Agency, 2010; de Groot *et al.*, 2012). These methods take profit of the information obtained in existing primary valuations and transfer it to a new context.

In most cases, the goods and benefits derived from ES are public (e.g. cooling derived from *regulation of temperature and humidity*) instead of private, meaning that the welfare measured represents a societal benefit, and can be understood (in part) as a social value.

Although economic methods partially acknowledge the social dimension, they might not be always the most appropriate for measuring social values derived from ES. For example, this is the case of ES classes with a non-tangible nature such as those related to spiritual values

[5] Since people do not have to pay the amount of money that they state for a service/project/good in questionnaires, they might respond with monetary values significantly greater than what they would have accepted as actual payments.

(e.g. *elements of living systems with a symbolic value)* or those that partially depend on individual perceptions and socio-cultural constructs (e.g. *characteristics of living systems that enable aesthetic experiences)*. In the above examples, a monetary valuation might be limited and the use of social assessment methods, which elicit social values derived from ES in non-monetary units, could be more appropriate (Santos-Martín *et al.*, 2018).

Table 3.1. Classes of ES monetary valuation methods (based on: Pascual et al., 2010; Brander, 2013).

Approach		Method	TEV values	Limitations
Market valuation	Priced based	Market prices	Direct and indirect use	Only marginal value or ES traded. Prices may be distorted (non-competitive markets, taxes, subsidies)
		Public pricing	Direct and indirect use	Only indirect link to consumers' preferences
	Cost based	Avoided damage costs	Direct and indirect use	Difficult to estimate side effects in absence of ES
		Restoration costs	Direct and indirect use	Capturing only direct cost component (not ES benefits)
		Replacement costs	Direct and indirect use	Capturing only direct cost component (not ES benefits)
		Opportunity cost	Direct and indirect use	Capturing only one indirect cost component (not ES benefits)
	Production based	Production function	Indirect use	Complex methodology. Lack of data
		Net factor income	Indirect use	Tendency of overestimation (net profit attributed only to ES)
Revealed preferences		Travel cost method	Direct (indirect) use	Difficult to collect non distorted data (trips with multiple purposes/locations, perceived vs real costs)
		Hedonic pricing	Direct and indirect use	Risk of biased estimation (many factors involved). Capturing only preference of individuals close to natural capital.
Stated preference		Contingent valuation	Use and non-use	Complex and expensive methodology. Risk of biases in design and analysis
		Choice modelling	Use and non-use	Complex and expensive methodology. Risk of biases in design and analysis

An integrated ES assessment considering environmental, economic and social positive values (impacts) could provide a more complete estimation of the contribution of NBS to urban sustainability and resilience. However, as already introduced in Chapter 1, few ES scholars have started to investigate disservices derived from NBS, and the need to include them as part of NBS assessments (Lyytimäki and Sipilä, 2009; von Döhren and Haase, 2015; Schaubroeck, 2017). Other scholar have started to use life cycle costing (LCC) to integrate the monetary valuation of financial costs and ES to better understand the net contribution of NBS over their entire life cycle (Bianchini and Hewage, 2012; Perini and Rosasco, 2013). There are also emerging studies focusing on how management actions might reduce the positive impacts derived from NBS as a result of the resources used by them (Ingram and Fernandez, 2012; Ingram, 2013; Luck *et al.*, 2014; Mcpherson and Kendall, 2014; Mcpherson, Kendall and Albers, 2015; Petri *et al.*, 2016). Thus, to better quantify the net overall contribution of NBS to urban sustainability and resilience other flows different than ES might need to be considered.

Chapter 3 aims to identify potential biophysical and monetary assessment methods and related indicators that could support an assessment of the overall contribution of NBS to the sustainability and resilience of urbanised contexts. Following the logic of Chapter 2, Section 3.2.

describes a methodological procedure composed of a systematic review complemented with two non-systematic reviews. Section 3.3 describes and discusses the results. As outputs a set of assessment methods are present together with a summary table of suitable indicators discerned from them. Conclusions are drawn in Section 3.4.

3.2. Methods

First, a systematic review on urban ES studies is developed to identify the ES assessment methods used for the most frequently studied ES classes already identified in Chapter 2 (see the list in Table 2.3). As anticipated in Section 2.3, the systematic literature review on ES of Chapter 2 was also performed for the collection of data on ES methods needed for Chapter 3. Therefore, in Section 3.2.1 the description of the ES systematic review is focused only on new aspects specific of this chapter. Second, the systematic study is complemented with two non-systematic reviews. Figure 3.2 summarises the methodological procedure.

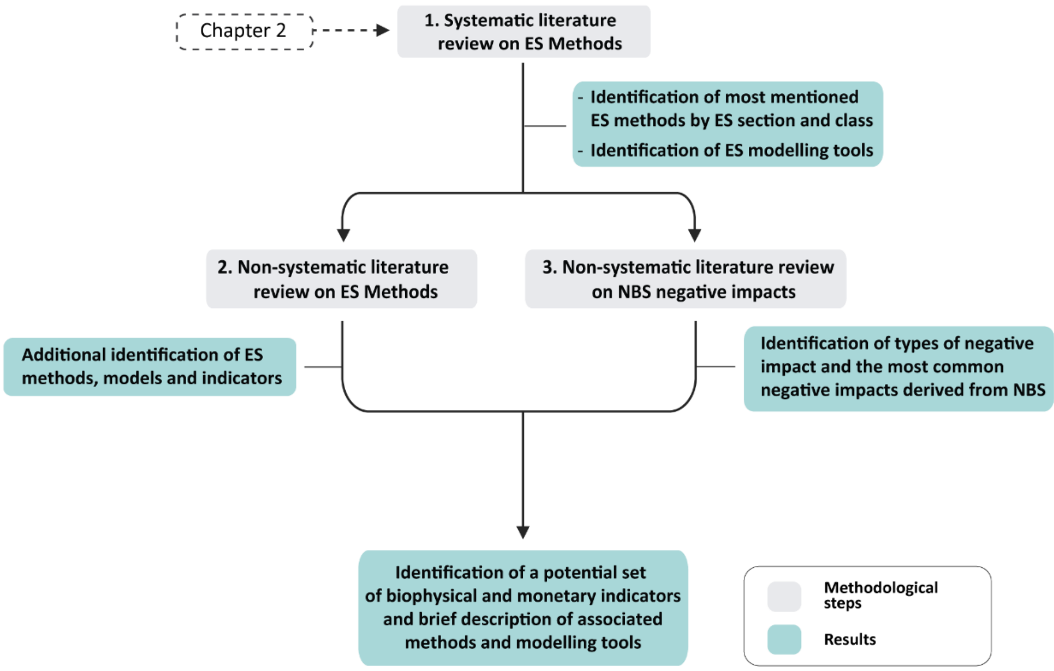

Figure 3.2. Methodological steps of the literature review

3.2.1. Systematic and non-systematic literature review on ES

The ES papers were analysed based on nine categories. Three categories from Chapter 2 were retained: ES sections, ES classes and specific types of NBS. The additional categories analysed included type of valuation (environmental, economic, social), type of data (quantitative, semi-qualitative, qualitative), type of units (biophysical, monetary, adimensional), spatial level of the assessments (global, urban region, metropolis, city, neighbourhood, block/site/object), specific methods and specific modelling tools/software.

For the category spatial extent of the assessment, the global level corresponds to studies focused on NBS in urbanised contexts, but where the scope is a global understanding independent of specific urban areas. Urban region, metropolis and city levels were differentiated according to their conceptualisation by Forman (2014). Neighbourhood level corresponds to studies focused on spatial extensions beyond specific sites or buildings, but which do not cover an entire urban area or a metropolitan district. Block/site/object level corresponds to studies focusing on NBS implemented on individual buildings, sites or on individual urban NBS where the context is disregarded (e.g. studies about experimental constructed wetlands in research facilities).

The use of specific ES methods were identified and organised according to the classification elaborated by Santos-Martin *et al.* (2018). They were visualised using an ordinal ranking approach supported on conditional tables. Specific modelling tools in the form of software or decision support systems were identified and analysed independently. They usually correspond to highly defined modelling frameworks that integrates more than one disciplinary approach, being of special interest for the overall aim of this PhD thesis.

A complementary non-systematic review on ES helped to fill gaps in the identification of modelling tools and indicators for specific ES classes of interest. Like in Chapter 2, it included papers on ES assessment and handbooks of ES process-based models.

3.2.2. Non-systematic literature review on negative impacts of NBS

A second non-systematic literature review was used to identify emerging studies that include the assessment of negative environmental, economic or social impacts derived from NBS or management actions. In this chapter, and for the rest of the manuscript, management actions are understood as human actions applied on NBS for their implementation, their maintenance and the collection and treatment of residues generated by them.

The review on negative impacts derived from NBS analysed nine categories equivalent to the ones considered in the review of ES papers. These emerging studies are coming from different disciplines and are framed under different but related accounting methodological approaches (see Hoogmartens *et al.* (2014) for a comparison of variants of Cost Benefit Analysis (CBA), life cycle assessment (LCA), and LCC). In some cases, these studies do not refer to the concepts of ES, NBS or related ones, neither use similar vocabulary. Consequently, a non-systematic approach focused on the most frequently NBS Type 3 identified in Chapter 3 was considered the most appropriate form for doing the literature review.

For the purpose of the review on negative impacts derived from NBS, a lower efficiency on the inherent capacity to supply an ES by an NBS was not assumed as a derived negative impact. For example, different constructed wetlands have different capacities to reduce suspended solid particles in the waterflow, which is part of the ES *water filtration by plants and soil.* However, the lower efficiency of some of them should not be interpreted as a negative impact on water quality derived from the constructed wetland. In addition, the review did not include studies focused only on grey components (e.g. a new type of rubber, perlite, cladding system) of hybrid NBS (e.g. green roof, constructed wetland) or on solutions placed on them (e.g. solar cells placed on an NBS) that do not form part of the NBS. Three main reasons support the last decision:

i) these components can be integrated into many products different than NBS;
ii) the assessments do not focus on the NBS itself; and
iii) the assessments do not focus on fundamental components that makes a solution an NBS (i.e. the ecosystem generated or its living components) or management actions applied on these components.

The outputs from this review were simply described and visualised as tables. As a result of their lower quantity, due to their emerging character, the use of conditional tables and quantification of higher or lower presence of specific groups was not considered necessary.

3.3. Results and Discussion

From the ES systematic review, 178 peer reviewed papers were examined. Biophysical methods (136) were the most used, followed by economic (46) and social methods (41). A combined use of biophysical and economic methods (26) was also more frequent than a combination of biophysical and social methods (17) or economic and social methods (5). Most of the papers provided its data on quantitative form (128). Almost all the papers using economic methods presented their results in monetary units (45). In fact, just in one case results were presented only aggregated in the form of a semi-qualitative scale (Hong and Guo, 2017). These results illustrate that most urban ES studies focus on assessing the capacity of ecosystems or NBS to supply ES, and that combination of biophysical and economic methods are preferred when both ES supply and ES demand are assessed.

From the non-systematic review on negative impacts, only 30 papers were reviewed, which included six review papers. Most of the studies were focusing on negative environmental and economic values, although few identified also social negative impacts. In all the papers making use of their own case studies, results were presented as quantitative data and LCA, financial LCC and financial CBA were the preferred methodological approaches.

3.3.1. Identification of spatial levels and ES methods by ES section and ES class

In terms of the spatial extent used to assess each ES class, studies at city level were the most common (Table 3.2). A global spatial extent is only used by Hellies, Deidda and Viola

(2018), when evaluating potential differences in water retention performance of green roofs in relation to climatic variations.

In the cases of *bioremediation (water)*, *filtration of pollutants (water)*, and *regulation of chemical conditions (water)*, studies at site level are used almost as much as those at city level. As indicated in Chapter 2, some ES such as *bioremediation* are technically complex and require a detailed characterisation of spatial attributes. This complexity and detailed characterisation explain the prevalence of site level studies for those ES classes where the focus is on understanding what produces changes in the ES supply capacity of very specific NBS.

Table 3.2. Spatial extent of the case studies considered in the literature review by paper including a specific ecosystem service class.

ES Section	ES class	No. Paper	Spatial Extent					
			Global	Urban Region	Metropolis	City	Neighbourhood	Block/site/object
Provisioning	Cultivated Crops for Nutrition	33	0	7	2	18	3	4
Regulation	Reg. Hydrological cycle & Water Flow	69	1	8	2	34	13	13
	Reg. Chemical Condition (water)	21	0	4	2	7	2	6
	Filtration Pollutants (air)	48	0	7	3	27	6	11
	Filtration Pollutants (water)	25	0	3	2	9	3	8
	Reg. Chem. Condition Atmosphere	61	0	8	3	38	7	8
	Regulation of Temp. & Humidity	63	0	6	5	36	9	13
	Bioremed. (water)	16	0	1	0	6	3	7
	Maintaining Habitats	38	0	6	2	23	4	8
Cultural	Charact. of Living Syst. Enabling Act. Promoting Health	68	0	11	5	35	12	10
	Charact. of Living Syst. Enabling Aesth. Exp.	38	0	5	2	18	6	10

Spatial level per No. papers on a ES class >≈50% <10%

For other ES classes, *regulation of the hydrological cycle, regulation of temperature & humidity, maintaining nursery population and habitats* and the two cultural ES, there is a more equilibrated distribution of studies at city, neighbourhood and site levels. This result might be explained by the importance of a multi-level spatial understanding to assess those ES classes. For example, in the case of *regulation of hydrological cycle and water flow* it might be necessary to assess the processes occurring in service providing areas, service connecting areas and service benefiting areas (Syrbe and Walz, 2012; Cortinovis and Geneletti, 2019). Studies focusing on service providing areas usually assess the capacity of specific sites or NBS to intercept, detent and infiltrate precipitation and run-off from other areas (from site to neighbourhood level). Studies focusing on connecting areas put more emphasis on the routing (movement) of the water run-off (and groundwater flows) in a catchment (from neighbourhood to urban region levels). Studies focused on service benefiting areas need to understand where the *regulation of hydrological cycle and water flow* is required to avoid human or property damages and losses (from urban region to neighbourhood levels). As another example, for *maintaining nursery populations and habitats*, the qualities of specific sites to host different species are relevant (site and neighbourhood levels), but the movement of species between sites in a broad landscape (urban region level) is also key to ensure the survival of populations. Consequently, the need of multiple spatial extensions depending on the specific focus of the assessment justifies a more equilibrated distribution of ES studies.

In terms of ES methods, spatial proxy methods are the most applied for many of the ES classes (Table 3.3). In fact, most case studies making use of spatial proxy methods are focused on city, metropolitan and urban region spatial levels. A great amount of these studies make use of land cover classes as spatial proxy of ES supply (e.g. Cabral *et al.*, 2016; Brill, Anderson and O'Farrell, 2017; Roussel *et al.*, 2017). As a result, they cannot differentiate ES supply between sites of the same land use class, neither the influence of specific biophysical nor management attributes occurring in specific sites. Hence, studies using land use/land cover classes as a spatial-proxy offer a limited understanding when applied at neighbourhood level or for informing specific urban planning actions (Cortinovis and Geneletti, 2019).

Despite the importance of spatial proxy methods, for some ES classes the use of process-based methods, field observations, use of statistical data and modelling tools is also quite common (Table 3.3). These are the cases of *regulation of hydrological cycle and water flow, filtration of pollutants by plants, regulation of the chemical condition of the atmosphere,* and *regulation of temperature and humidity*. As already stated above for some of them a complete understanding requires the use of a multi-level spatial analysis. In addition, all of these ES classes are the result of the interaction of multiple socio-ecological processes (La Notte et al, 2017), requiring to consider detail temporal resolutions that account for their changes over time. Hence, it explains the use of process-based methods as well as sophisticated modelling tools, that in many cases (e.g. i-Tree) require the use of local statistical data and/or field observations monitored at detailed temporal resolutions (e.g. hours, days) for the adjustment of the parametrisation of models to local conditions.

In the case of the two cultural ES and *maintaining nursery populations and habitats*, the application of preference assessments and narrative assessments (social assessment methods) appear also reiteratively used in the urban ES literature. In fact, for those ES classes social methods are more applied than economic methods. As already stated by Santos-Martin *et al.* (2018), this result illustrates that for the assessment of some ES classes, the use of social methods might be preferred to elicit differences in social values.

Table 3.3. Ecosystem service methods considered in the literature review by paper including a specific ecosystem service class.

Ecosystem Service Sections	Provisioning	Regulation								Cultural	
Ecosystem Service Classes	Cultivated Crops for Nutrition	Reg. Hydrological cycle & Water Flow	Reg. Chemical Condition (water)	Filtration Pollutants (air)	Filtration Pollutants (water)	Reg. Chem. Condition Atmosphere	Regulation of Temp. & Humidity	Biorremed. (water)	Maintaining Habitats	Charact. of Living Syst. Enabling Act. Promoting Health	Charact. of Living Syst. Enabling Aesth. Exp.
No. Papers	33	69	22	48	25	61	63	16	38	68	38
Spatial proxy methods	20	25	11	16	12	22	25	10	19	29	13
Phenomenological models	1	0	0	0	0	0	0	0	0	0	0
Process-based models	1	25	3	19	3	16	16	2	1	2	1
Statistical models	3	9	5	7	4	15	10	4	5	6	4
Ecological connectivity models	0	0	0	0	0	0	0	0	1	0	0
State and transition model	1	0	1	0	1	1	0	1	1	1	1
Integrated modelling framework	1	3	1	0	1	3	0	0	0	1	0
Field observations	7	21	8	15	9	26	19	9	12	11	4
Pseudo-observations (scenarios)	0	4	0	2	2	2	3	1	0	0	0
Surveys and questionnaires	6	5	3	4	3	5	7	2	10	11	9
Remote sensing & Earth obs.	6	10	2	9	3	12	12	2	6	9	4
Remote sensing & Earth obs. derivatives	1	3	1	4	4	3	5	0	0	1	1
Use of statistical, biophysical & socio-economic data	8	16	2	11	2	18	13	2	7	10	6
Market price	3	5	4	2	4	4	1	2	3	4	4
Replacement cost	2	4	2	2	2	2	1	0	1	1	2
Restoration cost	0	0	0	0	0	1	0	0	0	0	0
Avoided damage costs	0	3	0	5	0	3	3	0	1	1	2
Net factor income	1	0	2	0	1	1	0	1	1	1	1
Hedonic pricing	0	3	1	2	1	3	3	1	1	2	2
Travel cost	1	0	1	0	1	1	0	1	1	3	2
Contingent valuation	1	3	1	0	1	2	2	1	3	7	3
Choice modelling	0	2	0	1	0	1	1	0	0	2	2
Value transfer	3	3	3	3	3	6	4	2	6	5	5
Cost-Benefit Analysis	1	2	0	2	0	2	3	0	3	3	4
Preference assessment	3	5	2	5	2	5	7	1	8	11	9
Time-use assessment	0	0	0	0	0	0	0	0	0	1	1
Photo-elicitation surveys	0	0	0	0	0	0	0	0	0	2	1
Geo-tagged photo/Social Media	0	0	0	0	0	0	0	0	0	3	1
Narrative assessment	5	2	0	3	2	4	4	1	5	9	5
Participatory Geographic Information System & Field survey	1	1	0	3	0	0	1	0	0	6	3
Psychological tests	0	0	0	0	0	0	0	0	0	1	0
Modelling tools & Software	5	19	2	20	3	19	17	1	3	2	0

>≈50% <10%

ES Method per No. papers on a ES Class

3.3.2. Identification of ES modelling tools for the assessment of NBS

From the ES systematic and non-systematic review 21 modelling tools and software were identified (Table 3.4). Half of the tools (e.g. InVEST, TESSA, LUCI, RothC) were not initially developed for their application to urban areas. From this group, there are several tools, such as InVEST or LUCI, that rely on land use/land cover classes as a spatial proxy to assess the supply of multiple ES classes. Despite the abovementioned limitations, as an advantage these tools have low data requirements, allowing a rapid application. However, their assessment of multiple ES is made by independent models that does not acknowledge interrelations between processes and ES, neither their change over time. There are other modelling tools originally defined for rural areas, such as RothC, SWAT or BIOME that require more inputs (e.g. leaf area index, tree species, soil texture), but can identify better spatio-temporal changes in ES, being more appropriate for ES assessments at neighbourhood or site levels.

From the group of modelling tools specifically defined for urban studies, i-Tree, ENVI-met and Stormwater Management Model (SWMM) are already well-known and widely used by professionals and researchers (e.g. Lehmann *et al.*, 2014; Kim, Miller and Nowak, 2016; Zölch *et al.*, 2016; Bottalico *et al.*, 2017; Pappalardo *et al.*, 2017). SWMM was not born as a tool focused on ES, but it permits the calculation of stormwater dynamics over time making use of different infiltration models (i.e. Horton's, Green-Ampt and Curve Number methods), depending on the availability of data. In addition, it has been parametrised for modelling the performance of NBS (e.g. swales, green roofs) integrated as part of grey water collection networks (from site to metropolitan level). i-Tree is completely focused on the assessment of urban trees, requiring detailed input data about attributes such as species, tree size and health conditions. It permits a detailed assessment of the ES supplied by trees and their derived benefits in biophysical units, which can be converted to monetary units making use of default values (i-Tree, 2020). ENVI-met is more demanding in terms of data, since it requires a 3D characterisation of the space. However, it permits an integrated assessment of urban trees and built structures informing how to optimise both in terms of planning/design (Zölch *et al.*, 2016). The above urban ES modelling tools permit an accurate ES assessment of NBS, but they are very data demanding what could make them inappropriate to inform regular urban planning works (Cortinovis and Geneletti, 2018b).

From all the tools identified, only two focus on green walls and ponds and wetlands. The wetlands and water quality model (WWQM) of Chavan and Dennett (2008), models the retention of phosphorus and nitrogen in still waterbodies. In the case of green walls, the vertical greenery system model of Marchi *et al.* (2015), makes use of system dynamics modelling to assess the net carbon storage of green walls. It includes the impact of trimming of dead plants, treatment of biological waste as compost and its application onto agricultural soils. Therefore, different from the abovementioned models, the model of Marchi et al (2015) is the only identified that goes beyond the consideration of the operational phase of the NBS for the calculation of its net contribution..

Table 3.4. Modelling tools and software identified in the literature review

Name	Spatial Level	Processes & ES modelled/considered	Units	Source
TESSA	Neighbourhood to Site (Rural)	Storage and emission of carbon, emission of methane, N_2O emissions, water supply for irrigation and domestic consumption, water flow, yield of wild and cultivated goods, No of visitors (recreation)	Biophysical & Monetary	(Peh *et al.*, 2013)
LUCI	National to City (Rural)	Carbon storage, crop production, erosion risk, sediment transport, flood mitigation, habitat provision, nitrogen and phosphorus removal	Biophysical	(Trodahl *et al.*, 2017)
InVEST	Urban Region to City (Rural being adapted to Urban)	Carbon storage, water supply, nutrient & sediment transport, visibility (visual impact), visitation rates (recreation), shading, albedo, transpiration, water run-off reduction, crop & fish supply	Biophysical	(Cabral *et al.*, 2016)*
Drainmod Type	City/Neighbourhood (Rural)	Interception, evaporation, water run-off, infiltration, groundwater flow	Biophysical	(Koivusalo and Kokkonen, 2003)
Yasso v 7, v.15	National to Site (Rural)	Storage and emission of carbon in soil	Biophysical	(Liski, Tuomi and Rasinmäki, 2009)
RothC & RothPC-1				(Jenkinson and Coleman, 2008; Coleman and D. S. Jenkinson, 2014)
ECOSSE				(Smith *et al.*, 2010)
BIOME	Urban Region (Rural)	State and fluxes of carbon, nitrogen, and water in ecosystems	Biophysical	(Golinkoff, 2010)
CO2fix	Site (Rural)	Sequestration of carbon by trees and soil and economic value of tree biomass	Biophysical & Monetary	(Schelhaas *et al.*, 2004)
Soil & Water Assessment Tool	Urban Region/ Metropolitan (Rural)	Soil evaporation, tree transpiration, free evaporation, infiltration, percolation, nitrogen & phosphorus cycle, soil erosion, nutrient uptake, pollutants transport	Biophysical	(Shekhar and Xiong, 2008)
Wetlands water quality model (WWQM)	Site (Rural)	Nitrogen, phosphorus and sediment retention in wetlands	Biophysical	(Chavan and Dennett, 2008)
Temperature-Vegetation Index Model	Urban Region/ Metropolitan	Land surface temperature	Biophysical	(Yang *et al.*, 2017)
PANDORA	Urban Region to Neighbourhood	Bioenergy fluxes	Biophysical	(Pelorosso *et al.*, 2017a)
MODCEL	City/Neighbourhood	Interception, Infiltration, depression retention, water run-off	Biophysical	(Gomes Miguez *et al.*, 2017)
PERSiST	City/Neighbourhood	Infiltration, evapotranspiration, water run-off,	Biophysical	(Futter *et al.*, 2014)
ENVI-met	City to Neighbourhood	Solar radiation, transport & deposition of air pollutants, water & energy balance of green walls, wind flow, outdoor thermal comfort	Biophysical	(Zölch *et al.*, 2016)*
MIKE SHE	City to Site	Interception, infiltration, groundwater flow, water run-off, evapotranspiration, water movement over the landscape.	Biophysical	(Zölch *et al.*, 2017b)*
i-Tree	Metropolitan to Site	Tree growth and carbon storage, Dry deposition of pollutants in trees, interception, free evaporation, infiltration, water run-off, BVOC emission	Biophysical & Monetary	(Nowak, 2000) (Hirabayashi, Kroll and Nowak, 2015) (Hirabayashi, 2013)
Stormwater Management Model	Metropolitan to Site	Water run-off, infiltration, groundwater flow,	Biophysical	(Rossman and Huber, 2016)
Green Infrastructure Valuation Toolkit	Neighbourhood to Site	Carbon storage, transpiration, shading, interception, storage, infiltration, recreation opportunities, air pollution removal, stress alleviation, protection of habitats	Monetary, biophysical, adimensional	(The Mersey Forest *et al.*, 2018)
Vertical Greenery System Model	Block/Site/ Object	Carbon storage by green walls including end-of life (trimming of plants, composting and its use on soil)	Biophysical	(Marchi *et al.*, 2015)

Note: During the development of this thesis the modelling tool ARIES was in re-development and not accessible for review. It was then not included as part of the modelling tools reviewed.

* Due to the lack of a scientific peer-review paper introducing some tools, these references represent papers where the modelling tools were used and not a peer-review paper about the model itself.

3.3.3. Identification of negative impacts of NBS and methods to measure them

From the reviewed literature, four types of negative impacts were discerned (Table 3.5):

i) environmental, i.e. resulting from direct pollution or damage to ecosystems, which might also damage indirectly human health;
ii) financial costs associated to consumed materials and energy;
iv) financial costs associated to management actions, i.e. labour costs; and
v) social, i.e. direct damage of people's health or their enjoyment of the urban space.

Environmental and economic (financial) impacts were identified for all the NBS types. However, social impacts were only identified for the NBS Woodland-like (Table 3.5). Reviews on ecosystem disservices, which do not point to specific NBS or ecosystems, also helped to identify a broad list of social negative impacts that could be derived from multiple NBS (von Döhren and Haase, 2015; Vaz *et al.*, 2017).

Different to the review of Chapter 2, most of the studies identifying negative impacts were focused on green roofs, being mainly studied from an LCA or an LCC approach (e.g. Carter and Keeler, 2008; Peri *et al.*, 2012b; Ulubeyli and Arslan, 2017; Vacek, Struhala and Mat, 2017). This result is logic when considering that life cycle thinking approaches are already widely applied to assess the negative impacts of products from the construction sectors.

Most of the studies focusing on environmental impacts were LCA studies, except for woodland-like (Escobedo, Kroeger and Wagner, 2011; Russo *et al.*, 2017). They evaluated negative impacts only making use of midpoint and endpoint impact categories of well-known methods (e.g. ReCiPe 2016, IMPACT 2002+). In most of the cases, results were provided for all the available categories in an impact assessment method. Most case studies pointed to specific processes, such as manufacturing of specific materials, as major contributors of negative impacts. For example, technosphere processes related to implementation phases were identified as major contributors to negative impacts in the case of constructed wetlands (Lopsik, 2013). On the contrary, woodland-like studies focusing on global warming potential show that processes from implementation phases have a minor negative impact compared to arboricultural works during the operational phase and biological waste management at the end of life (Mcpherson, Kendall and Albers, 2015).

For the definition of the functional units in LCA studies, 50 years are commonly used as the average life time, including woodland-like (Kosareo and Ries, 2007; Sproul *et al.*, 2014; Mcpherson, Kendall and Albers, 2015; Rosasco and Perini, 2018). Functional units of NBS tend to be defined based on their physical properties (e.g. area, number of plants) or the amount of area that they benefit. For example, green roofs and woodland-like studies make use of impacts per square meter or tree as functional unit (Mcpherson, Kendall and Albers, 2015; Vacek, Struhala and Mat, 2017). Instead, studies on bioretention basins, swales and constructed wetlands focused on the amount of impermeabilized area that they serve or the potential amount of water volume that they would receive from serving areas over their lifetime (Flynn and Traver, 2013; Vineyard *et al.*, 2015; Xu *et al.*, 2017).

Table 3.5. Identification of the negative impacts of NBS and the main approaches to identify and assess them

NBS	Type of Negative Impact	Specific Negative Impact	Approach	Source
Green Roof	Environmental	Water consumption (irrigation), water pollution from fertilisers and pesticides,	Ecosystem Disservices	(Russo *et al.*, 2017)
	Environmental (ReCiPe 2008 & 2016)	Ozone layer depletion, photochemical oxidant formation, human toxicity, freshwater aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity	LCA (Impact Assessment)	(Saiz *et al.*, 2006; Kosareo and Ries, 2007; Vacek, Struhala and Mat, 2017)
	Financial Costs (Materials & Energy)	Vegetation, substrate (pumice, zeolite, compost, peat, organic fertiliser), water storage layer, drainage layer, fibre board (insulation), root barrier, waterproofing layer, geotextile, structural roof desk, rental of machinery (crane), fuel (transport & construction), electricity (construction & maintenance)	LCA (Life Cycle Inventory Phase), Financial Life Cycle Costing	(Wong *et al.*, 2003; Kosareo and Ries, 2007; Peri *et al.*, 2012a; Sproul *et al.*, 2014; Ulubeyli and Arslan, 2017; Vacek, Struhala and Mat, 2017)
	Financial Costs (Actions)	Raw material extraction, transport (raw material, product, construction site, disposal), material production, construction, maintenance, repair, replacement of components, refurbishment of green roof, de-construction, demolition, disposal, waste treatment		
Green Wall	Financial Costs (Materials & Energy)	Plants, auxiliary supporting systems, panels/plant boxes	Cost-Benefit Analysis	(Perini and Rosasco, 2013; Rosasco and Perini, 2018)
	Financial Costs (Actions)	Construction works, planting, pruning, cladding restoration, irrigation, plant replacement, pipes replacement, green layer disposal		
Bioretention Basin	Environmental (ReCiPe 2016)	Climate change, Ozone depletion, Human toxicity, Photochemical Oxidant Formation, Particle Matter Formation, Terrestrial Acidification, Freshwater Eutrophication, Marine eutrophication, Freshwater ecotoxicity, Marine ecotoxicity, Metal depletion, Fossil depletion	LCA (Inventory & Impact Phase) + financial LCC	(Xu *et al.*, 2017)
	Financial Costs (Materials & Energy)	Fuel, Electricity, Granite, Graded gravel, Cobblestone, PVC pipes, Turf, Non-woven fabrics, Concrete, Planting Soil, Flower Bed		
	Financial Costs (Actions)	Raw Material Production, transportation of material, excavation, Planting		
Vegetated Filter Strip & Swale	Environmental	<u>(Same midpoint impact categories than bioretention basin)</u>	LCA (Inventory & Impact Phase) + financial LCC	(Xu *et al.*, 2017)
	Financial Costs (Materials & Energy)	Graded gravel, PVC pipes, concrete, flower bed, turf, Non-woven fabric		
	Financial Costs (Actions)	Raw material production, transportation of material, excavation, planting		
Raingarden	Environmental (ReCiPe2008)	Global warming potential, acidification potential, human health - cancer, human health non-cancer, Respiratory effects, eutrophication potential, Ozone depletion potential, Eco-toxicity, Photochemical Oxidant formation, fossil fuel depletion	LCA (Impact Phase)	(Flynn and Traver, 2013; Vineyard *et al.*, 2015)
	Financial Costs (Materials & Energy)	compost, sand, stones, cement, asphalt, Bark for mulching, seedlings, tap water, machinery rental	LCC+LCA	(Vineyard *et al.*, 2015)
	Financial Costs (Actions)	Excavation, transportation of material and plants, management, end of life works		
Constructed Wetland	Environmental (ReCiPe2008)	Climate change, Ozone depletion, Human toxicity, Photochemical Oxidant Formation, Particle Matter Formation, Ionising radiation, Terrestrial Acidification, Freshwater Eutrophication, Marine eutrophication, Freshwater ecotoxicity, Marine ecotoxicity, Water depletion, Metal depletion, Fossil depletion, Land Occupation and transformation	LCA + financial LCC & Extended Exergy Accounting	(Lopsik, 2013; Casas *et al.*, 2017; Xu *et al.*, 2017; Wang *et al.*, 2018)
	Financial Costs (Materials & Energy)	Land, containers, pump, gravel, high density polyethylene, polymer coagulant, wastewater collection network, permutite, graded gravel, PVC pipes, concrete pipes, non-woven fabrics, excavation, planting soil, flower beds, fuel, electricity		
	Financial Costs (Actions)	Raw material production, transportation of materials to site, excavation, construction works, planting, operation, transportation of solids and sludge removed, waste management of solids removed (primary treatment -septic tank), waste management of sludge removed (secondary treatment),		

Table 3.5 (Continued). Identification of the negative impacts of NBS and the main approaches to identify and assess them

NBS	Type of Impact	Negative Impact	Approach	Source
Naturalised Wetland	Environmental	Global warming potential, eutrophication, non-renewable resource depletion, energy consumption, Acidification, Photochemical oxidant formation, Particulate matter, Waste	Emergy Accounting + LCA	(Duan *et al.*, 2011)
	Financial Costs (Materials & Energy)	Water, vegetation, gravel, steel, concrete, fuel, electricity		
	Financial Costs (Actions)	Construction labour, Maintenance & Management, Water quality maintenance		
Woodland-like	Environmental	Introduction of invasive species, displacement of native species, water consumption, water pollution from fertilisers, VOC emission, water quality (fertiliser run-off), air pollution from maintenance works, carbon and methane from litter decomposition, pests	Ecosystem Disservices	(Escobedo, Kroeger and Wagner, 2011; Russo *et al.*, 2017)
	Social	Allergies, refugia for vector-disease species, fruit fall problems, obscured views, fear of crime and safety hazards from tree fall	Ecosystem Disservices	(Escobedo, Kroeger and Wagner, 2011)
	Financial Costs (Materials & Energy)	Nursery materials (seeds, fiberglass stake, wire basket, trunk protector, low density polyethylene, polypropylene, polyethylene high density, polystyrene extruded, pumice, peat, fertiliser, pesticides, pesticide,	LCA (Inventory)	(Cambria and Pierangeli, 2011; Ingram, 2012, 2013)
	Financial Costs (Actions)	Seed collection, nursery management*, transport of plants stakes & other materials, excavation, planting, mulching, pruning, tree removal, stump grinding, chipping deadwood, replanting, transplants, pest-disease control, irrigation, pavement and sewer repair, unblockage of signs or light streets, damage to public infrastructure, debris clean-up, opportunity cost (space for other land uses), disposal of biological waste, biological waste management	LCA (Inventory) + Ecosystem Disservices	(Escobedo, Kroeger and Wagner, 2011; Kendall and Mcpherson, 2012; Vogt, Hauer and Fischer, 2015)
NBS & Ecosystems in general	Environmental	introduction of invasive species, displacement of endemic species, decrease in water quality and quantity, pests, VOC and particulate emission	Ecosystem disservices –Ecosystem	(von Döhren and Haase, 2015; Vaz *et al.*, 2017)
	Financial Costs (Materials, Energy & Actions)	Excrement from animals damaging buildings, Roots of plants damaging streets or paving, waste in the form of leaf and debris litter, maintenance costs (financial, opportunity, energy),	Ecosystem disservices - Material	
		Tree or branch falling on infrastructure or properties, Obstruction of views	Ecosystem disservices - Security and safety	
	Social	Pollen allergenicity, animal bites, zoonotic diseases transmitted to humans, direct attack by wild animals, tree or branch falling	Ecosystem disservices – Health	
		Species perceived as disgusting or irritating, Species or places considered unpleasant	Ecosystem disservices - Cultural	
		Anxiety from sounds/smell produced by animals disrupting physical connection with nature, presence of weed or mosquitoes, Obstruction of sunlight, Algal blooms impeding the use of waterbodies	Ecosystem disservices - Recreation	

3.3.4. Definition of a set of suitable indicators

From the review on negative impacts and ES derived from NBS a double set of biophysical and monetary indicators were identified (Table 3.6 and Table 3.7).

3.3.4.1. Negative impact indicators

For the assessment of negative environmental impacts, established LCA midpoint impact indicators were considered suitable (Table 3.6). First, estimation of midpoint impact categories involves a lower level of uncertainty that estimation of impact at endpoint, being therefore preferred. Second, the use of midpoint impact indicators would represent negative environmental values in the form of environmental effects, as it is the case for ES, and not as final damages to specific areas of protection (e.g. human health), which would be the case of endpoints. Consequently, representing both, negative and positive impacts, at a similar level of abstraction facilitates their comparison, especially in the case of alike categories. Third, LCA midpoint impact indicators are already widely used and standardised inside the LCA community of researchers and practitioners, already working on the assessment of NBS as it is illustrated in Section 3.3.3.

In terms of the specific impact assessment method (and associated biophysical indicators) to calculate midpoint impact categories, ReCiPe 2016 (Huijbregts *et al.*, 2017) was selected as the preferred method. ReCiPe is a well-known method since its previous version of 2008, and ReCiPe 2016 continuous to be widely used by LCA practitioners and researchers. Moreover, indicators for several midpoint impact categories of ReCiPe 2016 are analogue to indicators typically used for quantifying some ES classes. This would permit the quantification of net impacts for several categories since the same (or equivalent) unit of measurement are used.

Table 3.6. Potential set of indicators to be used as parameter-proxy for negative impacts

Midpoint impact category	Biophysical Indicator	Monetary Indicator	Source
Global warming	kg CO2 eq	Euro/kg CO2 eq	(Huijbregts *et al.*, 2017; S De Bruyn *et al.*, 2018)
Stratospheric ozone depletion	kg CFC11 eq	Euro/kg CFC11 eq	
Ionizing radiation	kBq Co-60 eq	Euro/kBq Co-60 eq	
Ozone formation, Human health	kg NOx eq	Euro/kg NOx eq	
Fine particulate matter formation	kg PM2.5 eq	Euro/kg PM2.5 eq	
Terrestrial acidification	kg SO2 eq	Euro/kg SO2 eq	
Freshwater eutrophication	kg P eq	Euro/kg P eq	
Marine eutrophication	kg N eq	Euro/kg N eq	
Human carcinogenic toxicity	kg 1,4-DCB	Euro/kg 1,4-DCB	
Human non-carcinogenic toxicity	kg 1,4-DCB	Euro/kg 1,4-DCB	
Water consumption	m3	Euro/m3	
Land Occupation & Transformation	ha Crop eq	Euro/ha	

In terms of monetary indicators for LCA midpoint impact category, the non-systematic review identified a recent publication called the Environmental Prices Handbook (De Bruyn *et al.*, 2018) as a state-of-the art reference. It permits to internalise the monetary value of negative environmental impacts or avoided negative environmental impacts (i.e. those derived from recycling or re-utilisation of waste) calculated as midpoint or endpoint impact categories. The handbook uses value transfer methods taking profit of primary valuation studies available in the scientific literature supported on enough evidence and whose values are adapted for its application to LCA studies developed in the EU context. The authors prioritise the use of primary studies based on market valuation methods to avoid uncertainty associated with the stated preferences of local social actors, since the scope is its wide application in the EU. For the monetisation of impacts, De Bruyn *et al.* (2018) make use of ReCiPe 2016 impact assessment method to track the cause-effect chains between emissions of substances or noise, the impact of them at midpoint and their derived consequences in terms of damage at end point As a result, they do not necessarily need to find monetary values for all substances, which would be quite time consuming. Instead, they can focus directly on select the most accurate valuations to document key substances and midpoint and endpoint indicators and by taking profit of the cause-effect pathways transfer values to a wider dataset of substances.

For ES classes equivalent to midpoint impact categories (e.*g. global warming* and *regulation of chemical composition of the atmosphere*), to be consistent, the monetisation is also based on the values proposed by De Bruyn *et al.* (2018). The following section mentions specific monetary valuation methods. A detailed description of the monetary valuation methods was not considered necessary since they are already well documented in key references already mentioned (Pascual *et al.*, 2010; Brander, 2013).

3.3.4.2. Positive impact indicators

The definition of the potential set of ES indicators for the assessment of NBS is synthesised in Table 3.7. The selection of the indicator/s per each ES class is briefly justified in the following lines based on the results of this review.

For the *regulation of chemical condition in the atmosphere*, carbon dioxide (CO_2) sequestration and storage is usually employed as biophysical indicator (Vaccari *et al.*, 2013; Kim, Miller and Nowak, 2016; C. Reynolds *et al.*, 2017). Several methods and modelling tools (e.g. i-Tree) make use of allometric equations (derived from statistical models) to calculate biomass growth rates of woody plants from where CO_2 storage can be inferred (e.g. Nowak *et al.*, 2008; Vaccari *et al.*, 2013; Andersson, Dickin and Rosemarin, 2016). For non-woody plants (e.g. grasses, sedges and rushes), few modelling tools used for urban forests (e.g. CO2fix, BIOME) are also adapted to model carbon sequestration of herbaceous plants. These models usually assign insignificant biomass to the stem and incorporate high turnovers of foliage and roots to acknowledge the different growth behaviour of non-woody plants (Schelhaas *et al.*, 2004). For soil carbon storage, modelling tools such as the RothC model assess carbon storage based on the rate of litter decomposition (Coleman and D. S. Jenkinson, 2014). From biophysical values of CO_2 storage in trees and soil, monetary values can be easily calculated based social cost of carbon or voluntary markets. The Environmental Prices Handbook (De Bruyn *et al.*, 2018), defines the value of monetisation of CO_2 making use of a combination of abatement cost and avoided damage cost methods. The latter is selected as the preferred method, since it is easier to measure and relate to specific financial costs.

Table 3.7. Potential set of ES indicators to be used as parameter-proxy per ecosystem service class

ES Class	Biophysical Indicator	Monetary Indicator	Source
Regulation of chemical composition of atmosphere	Carbon sequestration by vegetation ([CO_2])	Euro/ kg CO_2	(Schelhaas *et al.*, 2004; Coleman and Jenkinson, 2014)
	Carbon sequestration by soil ([CO_2])		
Regulation of temperature and humidity	Reduction in energy consumption for temperature regulation (kWh/ m^2year)	Euro/kWh	(Moss *et al.*, 2019)
Regulation of hydrological cycle and water flow	Total run-off volume (m^3)	Euro/m^3	(Rossman and Huber, 2016)
Filtration by plants (air)	Removal of pollutants of common air quality index([CO], [SO_2], [NO_2], [O_3], [$PM_{2.5}$], [PM_{10}])	Euro/ kg ([CO], [SO_2], [NO_2], [O_3], [$PM_{2.5}$], [PM_{10}])	(Hirabayashi, Kroll and Nowak, 2015)
Regulation of the chemical condition of freshwaters by living processes & Filtration by plants (water)	Load reduction of nitrogen (t/ha year)	Euro/ kg (N)	(Liquete *et al.*, 2016b)
	Load reduction of phosphorus (t/ha year)	Euro/kg (P)	(Cabral *et al.*, 2016)
Bioremediation by plants	Removal of common heavy metals ([Pb], [Cu], [Ni], [As], [Hg], [Cd], [Cr], [Zn])	Euro/kg (([Pb], [Cu], [Ni], [As], [Hg], [Cd], [Cr], [Zn])	(Adhikari *et al.*, 2011; Zanin and Bortolini, 2018; Krzeminski *et al.*, 2019)
Characteristic of living systems enabling activities promoting health or enjoyment	Number of visitors	Euro/visitor	(Liquete *et al.*, 2016b; Lupp *et al.*, 2016)
	Willingness to travel (Km) based on tree height (m)	-	(Filyushkina *et al.*, 2017)
	Willingness to walk towards an urban green spaces (Minutes)	-	(Ta, Tardieu and Levrel, 2020)*
Characteristic of living systems enabling aesthetic experiences	Diversity of landscape features (can be measured adapting Shannon diversity index)	-	(Szücs, Anders and Bürger-Arndt, 2015)
	Presence of water	-	Visulands Framework QLRT-2001-01017 (Ode, Tveit and Fry, 2010)
	Size of open land patches	-	
	Shape of open land patches	-	
	Number of patches of open land	-	
	Seasonal variability of vegetation (% of deciduous trees, annual herbaceous, and crops)	-	(Ode, Tveit and Fry, 2008; Fry *et al.*, 2009)
Maintaining nursery populations and habitats	Landscape metrics (e.g. mean patch size, number of patches. Patch density, Class area, Largest Patch index)	-	(Mörtberg and Wallentinus, 2000; Tian *et al.*, 2011; Li, Chen and He, 2015)
	Graph-based metrics	-	(Pelorosso *et al.*, 2017a)
	Shannon diversity index (included as part of landscape metrics)	-	(Russo *et al.*, 2017)
Cultivated plants grown for nutritional or material purposes	Average crop yield or biomass harvested (kg/ yr)	Euro/ kg (specific crop or type of biomass harvested)	(Maes *et al.*, 2016)

*Original source substituted by the most recent version of the document in form of a pre-print

For the *regulation of temperature and humidity*, in the case of water related NBS and those relying on herbaceous plants (e.g. swales, green roofs, green walls), evapotranspiration is usually considered the main process contributing to this ES (Francis and Jensen, 2017). The assessment of urban forests in modelling tools such as ENVI-met or methods such as the one proposed by Moss *et al* (2019) also identify evapotranspiration as a key process. Following Moss *et al* (2019), evapotranspirated water is converted into saved energy (kWh) of air conditioning (A/C). As a biophysical indicator, saved energy can be easily understood by a broad audience, and it can be easily monetised based on local market prices for electricity.

In the case of *regulation of hydrological cycle and water* flow at a service providing unit, total avoided run-off volume appears as a simple biophysical indicator that can be easily understood for valuing the contribution of NBS at site level. Despite its simplicity, its accurate assessment require a detailed modelling interrelating interception, evapotranspiration, infiltration, percolation (deep infiltration) and duration and intensity of rainfall events (Hirabayashi, 2013; Rossman and Huber, 2016; Pappalardo *et al.*, 2017b; Zölch *et al.*, 2017a). From avoided run-off volumes, monetisation could be calculated as avoided depuration of grey waters collected from public spaces. Local prices on depuration of grey water and its posterior provisioning as tap water for consumption can be used as specific monetary values.

To assess the *regulation of the chemical conditions of freshwaters*, the literature stressed the use of several variables such as: concentration of nitrogenous and phosphorous compounds (especially nitrates (NO_3^-), ammonium (NH_4^+), phosphates (PO_4^{3-}), total phosphorus, and total nitrogen), total dissolved solids, total suspended solids, chlorophyll-a, dissolved oxygen, biological oxygen demand, and electrical conductivity (Rooney *et al.*, 2015; De Troyer, S. T. Mereta, *et al.*, 2016; Jujnovsky *et al.*, 2017; Olguín *et al.*, 2017). In the case of LCA studies, nitrogen equivalent and phosphorus equivalent are already used as midpoint indicators (ReCiPe 2016) to quantify negative impacts on freshwater and marine eutrophication respectively. Nitrates and phosphates are included in the list of main pollutants of the EU Water Framework Directive 2000/60/EC. In addition, EU urban wastewater Directive 91/271/EEC establishes a maximum level of total nitrogen and total phosphorus that can be discharged by sewage treatment works into freshwater to avoid eutrophication issues. In this sense, a mandatory continuous monitoring of nitrogen and phosphorus for urban wastewater should exist in EU municipalities, making available input data about baseline conditions. In terms of reviewed modelling tools, the WWQM model is focused on calculating the load reduction of nitrogen and phosphorus in still waterbodies (Chavan and Dennett, 2008). Therefore, load reduction of nitrogen and phosphorus appears as suitable potential indicators for assessing the *regulation of the chemical conditions of freshwaters* and their values would permit a direct comparison with marine and freshwater eutrophication values. In terms of monetisation, to be consistent with negative environmental impacts, the abatement-cost for reducing nitrogen and phosphorus in surface waters used by De Bruyn *et al.* (2018) appears as the most suitable indicator.

For *filtration of air pollutants by plants*, the literature reviewed focus on the assessment of particulate matter ($PM_{2.5}$ and PM_{10}), SO_2, CO, NO_2, and O_3. (Jim and Chen, 2008; Selmi *et* *al.*, 2016; Bottalico, Travaglini, Chirici, Garfi, *et al.*, 2017). This list of pollutants is well-aligned with the common pollutants considered by the National Ambient Air Quality Standards of the Clean Air Act of United States (except for lead, not included) and the EU Air Quality Directive 2008/50/EC (except for lead, benzene, arsenic, cadmium, nickel and polycyclic aromatic hydrocarbons). These pollutants are also modelled as part of the i-Tree dry deposition model. From biophysical values, monetisation can be obtained by calculating the avoided damage cost to human health of each specific substance, as it is calculated by De Bruyn *et al.* (2018).

For the assessment of *bioremediation,* the reduction of heavy metals (Lead (Pb), Cadmium (Cd), Zinc (Zn), Nickel (Ni), Copper (Cu), Chromium (Cr(IV)), Mercurium (Hg), and Arsenic (As)) appear as the most suitable indicators, based on the few studies focusing on this ES class (Adhikari *et al.*, 2011; Zanin and Bortolini, 2018; Krzeminski *et al.*, 2019). These pollutant have a widespread presence in most urban brownfields, and their impact on human and ecosystem health is already well-enough understood (Thornton *et al.*, 2008; Sacristan, Peñarroya and Recatala, 2015; Qian *et al.*, 2017). The designation of a contaminated site, in the EU is usually done after assessing the levels of these pollutants. Thus, it was assumed that baseline data on the concentration of heavy metals in soil and water would be available for EU brownfield sites. A suitable monetisation could be based on the avoided damage cost to human health of these pollutants, but illustrative examples were not identified.

In terms of *maintenance of nursery populations and habitats*, in the literature reviewed, abundance, richness and diversity of species and habitats were identified as common applied indicators (Lowenstein, Matteson and Minor, 2015; Graça *et al.*, 2017; Russo *et al.*, 2017; Graca *et al.*, 2018; Müller *et al.*, 2018). In fact, the review of Russo *et al.* (2017), identified Shannon diversity index as a widely used indicator applied to assess the contribution of specific NBS to the maintenance of populations and habitats. The use landscape metrics such as mean patch size and mean patch density of natural or naturalised land covers also appeared few times in the literature review (Mörtberg and Wallentinus, 2000; Tian *et al.*, 2011; Li, Chen and He, 2015). In a broad spatial extent, such as urban regions, landscape metrics could be useful to inform how a specific NBS intervention or a combination of them might influence ecological connectivity. Similarly, graph based metrics such as those used in PANDORA (Pelorosso *et al.*, 2016) could also be used to inform changes in ecological connectivity, such as it is done in Chapter 4. However, a more specific study is required to identify a specific set of landscape metrics and graph-based metrics to inform how NBS interventions might contribute to the *maintenance of nursery populations and habitats* at urban region level. In terms of monetisation, as mentioned in Section 3.3.4, studies valuing this ES in monetary units do not represent this ES class. Instead, they represent the total value of entire ecosystems (including other ES classes) or represent the value of another ES influenced by this one. In fact, our review shows that a social assessment of this ES is usually preferred to economic assessments (see Section 3.3.1), which could be explained by the difficulty of differentiating clearly its value from other ES. Therefore, monetisation does not seem appropriate for this ES to avoid double-counting issues or the use of monetary values that represent a different ES.

Regarding *characteristics of living systems enabling aesthetics experiences*, the non-systematic review identified as suitable indicators size, shape, and number of open land patches together with diversity of landscape features and seasonal variation of vegetation (Tveit, Ode and Fry, 2006; Ode, Tveit and Fry, 2008, 2010). The first three indicators were defined in the European project Visulands (QLRT-2001- 01017) and have been already tested in rural areas. They appear also suitable for the assessment of NBS in urbanised contexts. In addition, diversity of landscape features was also identified in the consulted literature (Szücs, Anders and Bürger-Arndt, 2015; Brill, Anderson and O'Farrell, 2017; Andersson-Sköld *et al.*, 2018) and could be evaluated making use of Shannon diversity index. As an indicator measuring the seasonal variability of vegetation (Ode, Tveit and Fry, 2008) permits the assessment of short term changes in the aesthetic of the landscape, which could be easily measured by the amount of deciduous trees, annual herbaceous and seasonal crops present in an NBS intervention. All these indicators can be calculated making use of available landscaping documentation of NBS projects, which in principle make them suitable for use at site or neighbourhood spatial levels. Like for *maintenance of nursery populations and habitats*, a straightforward monetisation for this ES class, that does not implicitly value other ES classes, was not identified. Therefore, monetisation is not recommended.

In terms of *characteristics of living systems enabling activities promoting health or enjoyment*, the number of people visiting NBS interventions and the average amount of time spent on them were straightforward indicators identified during the literature review (Dennis and James, 2016; Liquete *et al.*, 2016b; Lupp *et al.*, 2016; Moseley *et al.*, 2017). However, to predict number of visitors for future NBS interventions is not straightforward. In the non-systematic review, one study assessed the willingness to move (in time) to visit an urban green space based on its biophysical attributes (Ta, Tardieu and Levrel, 2020). This method was found suitable to infer the potential number of visitors in a future NBS. The study of Filyushkina *et al.* (2017) inferred the willingness to travel a longer or shorter distance to visit a woodland based on its current dominant tree height. As the authors explained tree height is perceived by people as a representation of the maturity of the ecosystem (Filyushkina *et al.*, 2017). The later study was also found suitable to understand how potential visitors of an NBS intervention might variate over time because of the maturity of the ecosystem. In terms of monetisation, the potential number of visitors could be converted into monetary units making use of the willingness to pay for accessing urban green spaces in an urban area, following the approach of Bernath and Roschewitz (2008).

Lastly, for the ES *cultivated plants,* the quantification of crop yield or biomass harvested could be used as a suitable indicator. In fact, the indicator is already proposed in the urban pilot case studies of MAES (Maes *et al.*, 2016). Both indicators can be easy converted into monetary values based on local market prices for the specific crops or plants harvested.

3.4. Conclusion

Chapter 3 has identified the most used biophysical and monetary assessment methods considered to evaluate different ES classes and negative impacts derived from NBS. It has also identified the most common spatial extents at which different ES classes are studied in urbanised contexts and the need of multi-level spatial approaches for some of them. Moreover, it has been able to present a list of social, environmental and financial negative impacts derived from urban NBS by making use of ecosystem disservices, LCA, LCC and CBA literature. As a final output, Chapter 3 has presented a potential set of biophysical and monetary indicators to assess positive and negative impacts derived from NBS.

In terms of potential set of monetary valuation methods per each category/class, following the approach of De Bruyn *et al.* (2018) when possible market valuations were preferred over revealed and stated preferences approaches. In principle, market valuations can estimate use values (or loss of them), derived from positive and negative environmental impacts, with less risk of bias. Additionally, they are do not dependent on preferences of local social actors related to a specific context, being more easily transferable to other contexts. However, different than for LCA midpoint categories, for ES classes this was no always possible, due to the intangible nature of cultural services. Specifically, for *characteristics of living systems enabling activities promoting health or enjoyment* the study of Bernath and Roschewitz (2008) based on contingent valuation was found the most suitable primary valuation study to use.

As limitations, suitable monetisation methods were not identified for the ES classes *maintenance of nursery populations and habitats* and *characteristics of living systems enabling aesthetics experiences*. In fact, for both ES classes very specific indicators were not able to be defined either. For the case of *maintenance of nursery populations and habitats,* Chapter 4 expands the work of Chapter 3 and identifies suitable landscape metrics and graph-based metrics as well as additional modelling tools that can be used to assess this ES at urban region spatial extents.

In future works, the outputs identified in this chapter together with the ones identified in Chapter 2 could be used as a starting point for the definition of NBS modelling frameworks that go beyond the assessment of environmental impacts. Such a modelling framework is developed and presented in Chapter 5.

3.5. Additional Information

The Annex 3.1 contains the data collected in the systematic ES literature review (Spreadsheet 01) and the non-systematic Negative Impact review (Spreadsheet 02).

Chapter 4

Combining landscape ecology methods to assess the role of nature-based solutions in biodiversity conservation††

4.1. Introduction

Chapter 2 has highlighted the decline of biodiversity as one of the reiterated challenges in urbanised contexts, present in at least 15% of the urban challenge (UC) literature (see Section 2.4.1). It has also identified as sub-challenges (or major specific threating causes) of biodiversity, the loss and degradation of habitats and the lack of ecological integrity and ecological connectivity. In fact, habitat loss and fragmentation (i.e. increase in the subdivision of habitats) as a result of land cover conversion are widely recognised as a major threat to biodiversity conservation (Jaeger, 2000; Adriaensen *et al.*, 2003; Scolozzi and Geneletti, 2012a; Madadi *et al.*, 2017). Both of them together with the creation of new barriers (e.g. increase of unsuitable land covers or linear infrastructures impeding species movement) could end reducing ecological connectivity for different species (Januchowski-Hartley *et al.*, 2013; Mimet, Clauzel and Foltête, 2016).

Chapter 2 also showed that for *maintaining nursery populations and habitats* to avoid the decline of biodiversity, vegetation patches are of key importance (see Table 2.3 in Section 2.4.6). It is not only relevant its presence, but also the size of individual patches, their abundance in the space, their specific spatial distribution (being distance between patches important), and the suitability of the surrounding matrix (i.e. portion of the landscape with non-habitat patches for the species of interest) for movement. For example, if the distance between suitable patches surpasses a species-specific threshold (e.g. maximum distance the species could move over a certain land cover), a reduction of the ecological connectivity for that species will occur (Edelsparre, Shahid and Fitzpatrick, 2018). In this sense, to minimise

†† Chapter 4 is based on:
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Roles of other authors:
Benedetto Rugani, Davide Geneletti and Guido Sonnemann were academic supervisors of the paper.
Alya Bolowich contributed to the GIS tasks, looking for data about the surrogate species and the review of the paper.
Thomas Elliot contributed to the review of the paper

habitat loss, fragmentation, and reduction of ecological connectivity, the changes in land cover and their spatial configuration, including the removal or implementation of NBS, are relevant. Hence, design and planning professionals need accessible, easy to use, and robust methods to assess projects and plans proposing land cover changes at early stages.

Chapter 3 has also identified two suitable methods to assess habitat quality, habitat loss, fragmentation and ecological connectivity:

i) The use of indices based on graphs to assess ecological connectivity and their impact on ES (Pelorosso *et al.*, 2017b);
ii) the use of landscape metrics to inform quality of vegetation patches as habitat, their loss and fragmentation (Mörtberg and Wallentinus, 2000).

For two decades landscape metrics have been used as indicators to analyse changes in landscape patterns to provide information on the potential impacts on abiotic and biotic functions (Lausch and Herzog, 2002). By characterising changes in the composition and configuration of landscape patterns (e.g. inter-patch distance, patch density) at different levels (patch, class, and landscape), landscape metrics can be used to measure habitat loss, fragmentation, and changes in structural connectivity, i.e. the degree to which a landscape mosaic does or does not facilitate the movement of a species among patches (Taylor *et al.*, 1993). The use of graph theory is also considered useful in representing landscapes as a set of nodes (patches) and links (connection between two nodes, at first based on distance), from which graph-based indices known as connectivity indices have been developed to measure changes in structural connectivity (Saura and Pascual-Hortal, 2007; Zemanova *et al.*, 2017). However, despite not being spot in Chapter 3, there are other landscape ecology methods to assess habitat loss, fragmentation, and ecological connectivity, and therefore impacts on the supply of the ecosystem services (ES) *maintaining nursery populations and habitats*.

Recently, connectivity models have been developed for measuring functional connectivity, i.e. responses of individuals to landscape elements and their spatial configuration in the landscape mosaic (Tischendorf and Fahrig, 2000; Kindlmann and Burel, 2008). Examples of these model types include least-cost path analysis, circuit theory (both based on graph theory), matrix theory, and agent or individual-based modelling (see Kool, Moilanen and Treml (2013) for a detailed explanation on connectivity models). Moreover, methodologies combining connectivity models with connectivity indices or landscape metrics are also emerging. For example, connectivity indices can also be used in combination with least-cost path modelling approaches when aspects such as the landscape resistance to the movement of specific species in a study area needs to be taken into account for the definition of links (Scolozzi and Geneletti, 2012b).

Landscape metrics, connectivity indices, and connectivity models could be of great value to assess at early planning/design stages the impact of biodiversity of projects and plans proposing land cover changes. However, the plethora of tools makes it difficult for design and planning professionals to identify the most suitable metrics, indices or models to use, especially when in some cases there is still no agreement among experts (Calabrese and Fagan, 2004). In addition, scientific studies often only take into account one representative species when assessing the potential effects of land use changes or the adequacy of ecological corridors (e.g. Benedek et al. 2011; Gray et al. 2016), although recent multi-species studies are emerging (e.g. Pereira, Saura, and Jordán 2017; Pereira 2018). The use of single-species studies to guide spatial plans could generate a bias towards a better conservation of certain groups, making plans for broader biological conservation ineffective. It then becomes relevant for design and planning professionals to draft planning alternatives making use of a larger set of representative species, while balancing the amount of data required to ensure the feasibility (in time and cost) of their assessments.

The aim of Chapter 4 is two-fold:

i) to expand the review of Chapter 3 and select appropriate landscape metrics, connectivity indices and functional connectivity tools accessible to designers/planners for assessing landscape fragmentation and ecological connectivity in urbanised contexts.
ii) to apply in combination the selected metrics, indices, and models to a specific case study facing biodiversity decline as a societal challenge and simulating a scenario similar to the ones design and planning professionals might have to respond in practice.

The country of Luxembourg was selected as a case study because it is already one of the most (habitat) fragmented countries in the EU (European Environmental Agency, 2011). Additionally, its population is expected to almost double between 2015 and 2060, reaching one million people by 2062 (Eurostat, 2015). As a consequence of its population growth, it is expected an increase of urban development and productive land uses such as cropland at the expense of ecosystems such as forests. Moreover, Luxembourg conditions correspond to more than one of the contextual classes in which biodiversity decline have been most stressed as a major UC (see Table 2.2, Section 2.4.1). Luxembourg it is a European high-income country mainly formed by very small single urban areas, but which in fact due to its special character acts as a large metropolitan area. According to the EU-OECD definition of functional urban areas (Dijlkstra and Poelman, 2012), the entire territory of Luxembourg is defined as a unique functional urban area. Thus, in practice it is an urbanised context where the rural zones act as the hinterland of small and very small cities. This overlapping condition, like the one of other territories such as Singapore, makes it a quite special case for spatial planning purposes.

4.2. Materials & Methods

Figure 4.1 summarises the methodological steps from the selection of metrics, connectivity indices and models to the mapping and interpretation of results. The following sub-sections describe each step in detail.

4.2.1. Selection of landscape metrics, connectivity indices and ecological connectivity models

An initial literature review of case studies, comparative studies, and critical reviews was performed on Scopus including the terms fragmentation, landscape metrics, and connectivity (see specific search string in Table S0, Annex 4.1). The search was limited to the last 10 years to ensure the selection of up-to-date applications of landscape metrics, connectivity indices and models. This search returned 158 papers. From those, only 57 (Table S1, Annex 4.1) ultimately assessed fragmentation (25 papers), connectivity (25 papers), or both (seven papers) using landscape metrics or ecological connectivity indices or connectivity models. The latter were classified by modelling approaches based on the classification of Kool, Moilanen and Treml (2013).

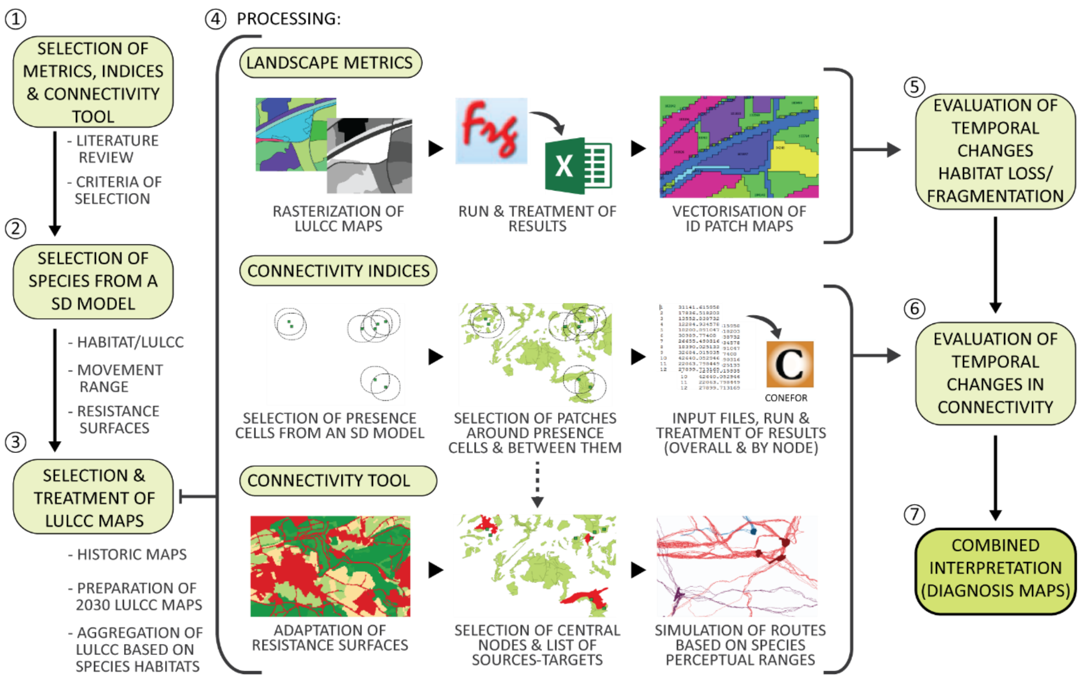

Figure 4.1. Flowchart of the methodological steps; SD=Species Distribution; LULCC=land use/cover classes

Supported on the discussions in the papers and key references (e.g. Kindlmann & Burel, 2008; Uuemaa, Mander, & Marja, 2013) about adequacy and limitations, 20 landscape metrics (landscape, class, and patch level), 9 connectivity indices, and 19 ecological connectivity tools/software were pre-selected for the assessment of habitat loss and fragmentation, and ecological connectivity. As a condition, the pre-selected metrics and indices should be able to be calculated making use of Fragstats v 4.4 (McGarigal, Cushman and Ene, 2012) or Conefor v 2.4 (Saura and Torné, 2012). Both software are open-source and widely used by scientists for an automatised calculation of metrics and connectivity indices, compatible with Geographical Information Systems (GIS) outputs, facilitating the integration of results into spatial outputs. The preselected metrics were also mentioned at least three times in the literature review in habitat fragmentation studies (Table S2a, Annex 4.1).

The metrics were narrowed down to a smaller set of 12 landscape metrics (Table 4.1) according to three criteria: simplicity, lack of redundancy, and history of application. If landscape metrics presented no difference in terms of adequacy, the most simple ones (i.e. fewer geometrical attributes and less mathematical operations) were prioritised. The history of application shows that the adequacy of the metrics have already been demonstrated, reinforcing their robustness.

From the nine connectivity indices, four were selected (Table 4.1), the only ones mentioned at least three times in the papers (Table S2b, Annex 4.1). Also, a review by Baranyi, Saura, Podani, et al. (2011) showed that three out of these four connectivity indices (i.e. the ones that can be run at node level) stood out for their capacity to capture most of the variability in the connectivity changes of patches. In other words, these indices are non-redundant and complementary indicators that will ensure time-effectiveness in terms of analysis and posterior interpretation of the assessments.

With respect to the 17 ecological connectivity tools/software, 12 were identified in the literature, and the other seven were already known by the authors (see the list of tools and short description in Table S3, Annex 4.1). Only eight were freely available and not dependent on commercial software (i.e. UNICOR, Guidos, Connectivity Analysis Toolkit, Maxent, Circuitscape, Condatis, Graphab, and LSCorridors), which was an essential criterion to ensure accessibility to tools for planners. From these, UNICOR and Guidos were excluded because of their lack of accessibility in terms of technical knowledge required. UNICOR was excluded because it requires additional software (ZonationX), python packages and needs to be run through a Python editor, making it less user-friendly. Guidos toolbox includes many tools, but it is a software more tailored to experts with a strong technical background in image analysis for ecological purposes. Later, taking into account the redundancy of modelling approaches, the Connectivity Analysis toolkit was excluded because it is based on the assessment of centrality connectivity indices, which would include the connectivity indices calculated through Conefor. Outputs from Maxent in the form of a species distribution grid at 1 km resolution, already developed by Titeux et al. (2013) in a previous work, are used as inputs in the case study to give information on the presence/absence of species (see Section 2.5 for further details). Hence, four potential tools/software were preselected (i.e. Circuitscape, Condatis, Graphab, and LSCorridors) and the differences in the rationales and underlying assumptions of their modelling methods were further described in Table 4.2.

Circuitscape (McRae, Shah and Mohapatra, 2013) and Condatis (Hodgson *et al.*, 2012) both apply circuit theory, while Graphab is based on graph theory, allowing calculations to be made based on Euclidean distance or cost distances (Foltête, Clauzel and Vuidel, 2012). Conefor also allows the calculation of connectivity indices based on Euclidean and cost distance. Instead, LSCorridors uses the least-cost path analyses (Ribeiro *et al.*, 2017).

Table 4.1. Type and description of the selected landscape metrics and connectivity indices (see McGarigal, Cushman and Ene, 2012; Saura and Torné, 2012 for more detailed information about metrics and indices).

Type	Metrics / Indices	Name	Level	Units	Range	Function and rationale
Landscape Metric (Area)	PLAND	Percent land area	C	% of total land	0-100	Explains the amount of habitat (using land use/cover class as proxy) over time to understand habitat loss; 0=all patches of habitat disappeared; 100=One habitat occupies all the landscape.
	AREA (AW, CV)	Area	P/C	Ha	0-total class area	Information about the average-weighted (AW) amount of patch area in each land use/cover class and its coefficient of variation (CV) describes the viable area for habitat specialist species. A reduction in AREA at class level usually indicates increasing fragmentation of a land use/cover class. The AREA at patch level provides the dimension of each patch making use of the same raster used for class level analysis, which could be used in combination with class level results to ascertain which patches are below AW AREA.
	CORE (AW*, CV)	Core area	C	Ha	0-total class area	Average-weighted (AW) amount of inner patch area in each land use/cover class and its coefficient of variation (CV) of the patch areas; describe the suitable area for habitat specialist species taking into account the outer areas of the patch that they will not use. Reduction in CORE_AREA indicates fragmentation and loss of suitable habitats and shows whether the patches are below a minimum threshold for specific species.
	LPI	Largest patch index	C	% of total land	0-100	Percentage of total land occupied by the largest patch in a specific land use/cover class. Reductions in LPI indicate fragmentation of an LCC (class level) or increased patchiness (landscape level).
	ED	Edge density	C	Edge Metres/Ha	0-Limitless	Increase usually indicates fragmentation of patches. If (CORE) AREA increases and ED decreases, growth occurred as an expansion of existing patches. Together with AREA and LPI, ED reduction shows increased patchiness of each land use/cover class.
Landscape Metric (Shape)	SHAPE (AW*, CV)	Shape	P/C	None	1-Limitless	Average-weighted (AW) shape of patches of each land use/cover class and its coefficient of variation (CV) explain the naturalness of the patches. Values close to one indicate that patches are quite regular (man-made). When a value increases beyond one, patches are irregular in form and more naturalised. However, for urban land use/cover class, irregular patches usually indicate increased sprawl.
	PD	Patch density	C	No./100 Ha	0-Limitless	Amount of fragmentation (increase of PD) or aggregation (decrease of PD) of patches of a land use/cover class or all the land cover patches in the study area.
Landscape Metric (Aggregation)	NLSI	Normalised landscape shape index	C	None	0-1	Values close to one mean that patches are isolated or uniformly distributed, whilst values close to zero indicate increased clumpiness**.
	MESH	Effective mesh size	C	Ha	Ratio of cell size to Total landscape area	Probability that two random patches of a land cover class (or different land cover classes at the landscape level) are connected in the study area. A decrease indicates higher fragmentation, indicating loss of structural connectivity.
	ENN (AW*, CV)*	Euclidean nearest-neighbour distance	C	Metres	0-Limitless	Metric that informs about distance to the nearest patch of the same land use/cover class. It measures loss of structural connectivity without considering the characteristics of the other land use/cover class or the presence of barriers and how these increase or decrease the cost of movement for different species.
	CONTAG	Contagion index	L	None	0-100	Overall clumpiness of all the land use/cover class in the landscape; but also the interspersion of patches of different land covers in the study area. Values close to zero indicate patches maximally disaggregated or equally interspersed; a value approaching 100 shows that all the LCC are maximally aggregated.
Landscape Metric (Diversity)	SHDI	Shannon's diversity index	L	None	0-Limitless	Evaluate the increase in the diversity and heterogeneity of land covers in the landscape. For the same area of study, a decrease in values indicates increased homogenisation of the land use/cover class, which can be interpreted as loss of habitats.

* AW offers a landscape-centric perspective that considers the abundance of patches, giving more weight to bigger patches showing the average patch area/core area/shape/distance to other patches that an individual from a species in a random point of the study area may encounter.

** Clumpiness refers to the level of aggregation of each of the different land use/cover class patches, indicating whether or not they are distributed as clumps or as close adjacent groups in the landscape.

Table 4.1 (Continued). Type and description of the selected landscape metrics and connectivity indices (see McGarigal, Cushman and Ene, 2012; Saura and Torné, 2012 for more detailed information about metrics and indices)

Type	Metrics / Indices	Name	Level	Units	Range	Function and rationale
Connectivity Indices***	BC, BC(IIC), BC(PC)	Betweenness centrality	N	None	0-1	These correspond to the degree to which the movement (Euclidean distance or cost-distance***) between other nodes (patches) passes through a particular patch. *BC* only considers the number of movements between patches that go through a particular patch. *BC(IIC)* and *BC(PC)*, however, take into account an additional attribute (in this paper, the attribute is area) of the patches that are being connected through a particular node from a binary approach (IIC) and a probabilistic one (PC).
	IIC (dIntra, dFlux, dConnect)	Integral index of connectivity	O/N	None	0-1	A binary index (graphs with unweighted links) that measures whether a patch (node) is connected (value 1) or not (value 0), taking into account a maximum distance of dispersal. The value increases with increased connectivity. For an overall connectivity assessment ("class level"), a value of one indicates that the entire area of study is occupied by the habitat studied. Their components provide the following information at the patch (node)-level: internal connectivity in each patch (dIntra); importance of the patch for the current dispersal of individuals (dFlux); and loss of connectivity in the network if a specific patch is lost (dConnect).
	PC (PCintra, PCflux, PCconnect)	Probability of connectivity	O/N	None	0-1	A probabilistic index (graphs with weighted links) that indicates the probability that two animals randomly located in the study area are in habitats connected to each other, instead of binary values such as IIC. Similar to an IIC increase from 0 to 1, PC can be run to obtain an overall value or value by patch accounting for the same components.
	ECC, (ECC(IIC), ECC(PC))	Equivalent connected area	O	Area (m^2, ha, km^2,...)	0-Total Landscape Area	EC is defined as the area of a single habitat patch that would provide the same connectivity as the existing habitat patterns of the area of study. EC(IIC) corresponds to the area equivalent to the connectivity value of IIC and EC(PC) to the area equivalent to the connectivity value of PC. This index can be run only for overall connectivity assessments.

*** If the connectivity indices are calculated based on Euclidean distances, the analysis is structural. However, with modifications, these can be calculated based on cost distances by using resistance surfaces, which makes them adequate for functional connectivity analysis. **P** = patch, **C** = class, **L** = landscape; **O** = Overall connectivity analysis (equivalent to class level); **N** = node level analysis (equivalent to patch level)

Out of the above four tools, LSCorridors was selected for the application in Luxembourg. The model mitigates a few of the common limitations of least-cost path and circuit theory models (i.e. assumption of omniscience and no influence of surrounding patches (Palmer, Coulon and Travis, 2011; Coulon *et al.*, 2015; Delattre, Baudry and Burel, 2018)) by including stochastic variation, species perception, and landscape influence. Furthermore, LSCorridors permits the assessment of connectivity using a modelling approach different to graph theory only (i.e. least-cost path), which is already considered via the connectivity indices. Like other least-cost path models, LSCorridors requires the definition of a resistance surface with the cost of movement for each patch (cell) of the study area. Environmental stochasticity, as defined by Fujiwara and Takada (2017), is integrated in the four different route simulation methods: measures by pixel (MP), measures by landscape-minimum, average, and maximum (MLmin, MLavg, and MLmax). The first method (MP) adds a random variability in the resistance surface, while the ML methods also integrate the landscape influence by considering how the value of the cells inside a moving window (equivalent to the species perception) influence the value of the central cell (Ribeiro *et al.*, 2017). In other words, in MLmin, MLavg, and MLmax, the value of each resistance surface cell is substituted by the minimum, average or maximum value of the surrounding pixels inside the moving window (Ribeiro *et al.*, 2017). It is suggested using MP, MLmin, and MLavg to model the movement of generalist species, while MLmax is recommended for specialist species because it generates more restrictive corridor routes (Ribeiro, Silveira dos Santos, Dodonov, et al. 2017). Since we have selected species with different degrees of specialism (see Section 2.3), all four-simulation methods were used in our case study

Table 4.2. Connectivity modelling approaches selected (rationale and assumptions). The organisation follows the classification of modelling approaches of Kool, Moilanen and Treml (2013).

Modelling approaches	Rationale	Assumptions
Circuit theory	A mathematical approach to calculate the path of least resistance through which an electrical current can travel in a circuit of multiple parallel paths (Svoboda and Dorf, 2003). In ecological connectivity it provides a map of possible pathways (Mcrae *et al.*, 2008).	Models usually assume patch homogeneity, employ land use/cover class as proxy and do not consider the direction of movement through a cell or the characteristic of surrounding patches. It assumes the individual has perfect knowledge of their surroundings (omniscience).
Graph theory	It explains the landscape as a set of nodes and edges. Movement (called a "walk") can occur between nodes (usually geometrically represented as the centroid of patches) only if an edge connection exists between those nodes (Bunn, Urban and Keitt, 2000; Kent, 2009).	Distance between unconnected nodes (nodes that are not connected by a walk of edges) is infinite. Least-cost path and circuit theory can be integrated with graph theory by adjusting the Euclidian length of edges according to their weighted length (Bunn et al, 2000; McRae et al, 2008).
Least-cost path	It assumes a cost per type of patch based on their attributes. Euclidian distances are weighted by their costs and the minimum sum of cost-weighted distances is the least-cost path (Bunn, Urban and Keitt, 2000; Zetterberg, Mörtberg and Balfors, 2010)	Models usually assume patch homogeneity, do not consider direction of movement, characteristics of surrounding patches, and assume omniscience.

4.2.2. Selection of species

The animal species were selected from an existing dataset that included the potential distribution of species (i.e. presence/absence matrix) calculated using the species distribution model of Titeux et al. (2013). The following non-excluding criteria were used for the selection of species (Table 4.3):

- Conservation status according to the European Habitats Directive. Priority was given to species with a bad or inadequate conservation status, but those with a favourable status were included if relevant for the other criteria. The use of the European Habitats Directive conservation status list was preferred to others because EU spatial planners are obliged to take this into account, since non-favourable conservation status and the priority species and habitats indicated in the Directive's Annexes are a means of establishing priority settings.

- Balance distribution of taxonomic classes and types of consumers. At least one species, if possible two, per taxonomic class were selected to ensure the presence of different taxonomic classes. In the case of mammals, differentiation between primary consumers (e.g. rodents) and secondary to quaternary consumers (e.g. foxes, wildcats) was also taken into account. Birds were preselected (*Accipiter gentilis, Anthus pratensis,* and *Terastres bonasia*), but due to the lack of data, they were eventually excluded. Fish were not included in the study, since they would require a very specific habitat fragmentation and connectivity analysis that cannot be developed by making use of national land use/cover class maps.

- The use of representative or surrogate species. Surrogate species are those that can provide a good representation of a larger group of species and types of habitats they are associated with, such as keystone, umbrella and flagship species among others (Caro and O'Doherty, 1999; Favreau *et al.*, 2006). We prioritise species that were recognised as habitat specialists, instead of generalists, as well as those already recognised in the literature as adequate surrogates for other species in forest, grassland and wetland habitats. In connectivity analysis, surrogate species can be also used (e.g. Mortelliti et al. 2009) to represent species with a different capacity of movement. As a result, we ensured some variety in the mobility range of the selected species.

The use of the criteria stated above prevented skewing the analysis towards a specific taxonomic group and specific habitat specialist. This ensured that locally vulnerable species were considered, whilst reducing the economic burden of addressing individual species' requirements. This methodological choice was selected to simulate a scenario similar to the ones environmental planners might face in practice during the definition of urban development strategies or ecological corridors.

Table 4.3. Description of the conservation status, land cover preferences, range of movement and representativeness as surrogate species of the species selected for the case study.

Species	Taxonomic group	Conservation status*	Land Cover preferences**	Range of movement***	Representativeness
Maculinea arion (Large blue)[a]	Butterfly	Bad	Grassland and Pasture (Spitzer *et al.*, 2009)	390 m as the lowest mean distance to suitable patches (Schneider and Fry, 2001). 500 m assumed as maximum distance	A surrogate for the conservation of grassland invertebrates including other butterflies (Spitzer *et al.*, 2009; Sielezniew, Włostowski and Dziekańska, 2010).
Triturus cristatus (Great crested newt)[a]	Amphibian	Inadequate	Woodlands and scrubland surrounding ponds(Edgar and Bird, 2006; Vuorio, Reunanen and Tikkanen, 2016)	Dispersal up to 860 (Edgar and Bird 2006). 1 km assumed as maximum distance	A surrogate for wetland conservation (Denoël *et al.*, 2013; Unglaub *et al.*, 2015).
Alytes obstetricans (Common midwife toad)[a]	Amphibian	Inadequate	Woodlands, scrublands and scarce vegetated areas surrounding ponds (Bosch et al. 2016)	1 km assumed as maximum distance	Extensively studied, but it is not a surrogate. Included to ensure at least a second amphibian with similar land cover preferences.
Lacerta agilis (Sand Lizard)[a]	Reptile	Bad	Grassland, pasture and rocky areas (Ceirans, 2007; Russell, 2012)	Evidence of short dispersal less than 150 m (Olsson 1997). 500 m assumed as maximum distance	A surrogate used in connectivity analysis since changes in structural connectivity of their habitats strongly match changes in functional connectivity (Rödder *et al.*, 2016).
Myotis bechsteinii (Bechstein's bat)	Mammal	Inadequate	Deciduous woodlands (Dietz and Pir, 2009; Watts *et al.*, 2010)	Less than 1 km to long distance foraging sites Dietz and Pir, 2009); 1 km assumed as maximum distance	Suggested as target species whose habitat protection could benefit other forest dwelling bat species in Luxembourg (Dietz and Pir, 2009).
Felis silvestris silvestris (European wildcat)	Mammal	Inadequate	Woodlands and scrublands (Klar *et al.*, 2008, 2012; Lozano, 2010)	2 km assumed as maximum distance	The species is usually selected in connectivity studies as a surrogate of woodland medium size carnivores (e.g. Gurrutxaga, Lozano, and del Gabriel 2010; Lozano 2010).
Martes martes (Pine marten)	Mammal	Inadequate	Woodlands (Pereboom *et al.*, 2008; Ruiz-González *et al.*, 2014)	Recorded daily distance is 2.1 km, even if maximum linear distance 860 m (Pereboom *et al.*, 2008). 2 km assumed as maximum distance.	The species is usually selected in connectivity studies as a surrogate of woodland medium size carnivores (Pereboom *et al.*, 2008; Gurrutxaga, Lozano and del Gabriel, 2010).
Muscardinus avellanarius (Hazel dormouse)	Mammal	Favourable	Deciduous Woodlands (Bani *et al.*, 2017)	Short dispersal of 500 m in forests and 300 m in open land (Bani *et al.*, 2017). 500 m assumed as maximum distance	A surrogate used in connectivity models (Dietz *et al.*, 2018) representative of of red squirrels (Mortelliti, Santulli Sanzo and Boitani, 2009), an endangered species.

* Extracted from the Habitat Directive Report of 2007-2012 of Luxembourg (Titeux, Mestdagh and Cantú-Salazar, 2013)
** Habitat preferences simplified based on land cover preferences without consideration of other features or microhabitat preferences.
*** Maximum distance used by the authors for LSCorridor and Conefor supported on data from literature review. The range of movement for *Alytes obstetricans*, and *Felis silvestris silvestris* was assumed similar to *Triturus cristatus* and *Martes martes*, respectively.
a. Species for which connectivity indices were also studied at patch (node) level and preferred routes of movement were calculated with LScorridors.

4.2.3. Selection and treatment of land use/cover class maps

The landscape metrics and connectivity indices were calculated for 1999, 2007 and 2030 by using the Luxembourgish land use/cover class maps of 1999 and 2007 (scale 1:20.000, a minimum mapping width of 4 m), the Urban Atlas of 2012 (minimum mapping unit of 0.25 Ha and minimum mapping width of 10 m), and a set of potential urban and infrastructure developments for 2030, which were taken from the Luxembourgish online geographic portal (geoportail.lu). The land cover maps of 1999 and 2007 are the only existing Luxembourgish national land cover maps. The development plans found on the geographic portal came from the Luxembourgish sectoral plans (*Plans sectoriels*, *Administration du Cadastre et de la Topographie*) and were digitalised by the authors. In order to create a land use/cover class map for 2030, the urban areas of the Urban Atlas of 2012 (European Union, 2018) were extracted and overlapped on the land use/cover class map of 2007, substituting non-urban land covers with new urban ones. Next, the digitalised 2030 future urban and infrastructure developments taken from the geographic portal were overlapped. This was used as a plausible scenario of urban growth for 2030, assuming that the changes in the land use/cover class will mainly be attributed to urban development. It was not possible to see changes between non-urban land use/cover class (e.g. grasslands converting to croplands or vice versa). The aggregation of land use/cover class for the raster (Table 4.4) was applied taking into account the habitat preferences of the species studied (Table 4.3).

4.2.4. Landscape metrics, connectivity indices and models.

For the calculation of landscape metrics, all land use/cover class maps were rasterised at a resolution of 10 metres (maximum resolution possible due to the minimum mapping width of the Urban Atlas 2012). The use of a high resolution minimises the loss of accuracy when transforming land cover maps into raster, especially for transport infrastructure land use/cover class, and therefore minimising the impacts on metrics results calculated in Fragstats v4.4. In addition, none of the selected species required an edge depth (i.e. width of the habitat edge used to identify core habitat) below 10 metres since their adequate minimum habitat was higher (Olsson, 1997; Edgar and Bird, 2006; Pereboom *et al.*, 2008; Dietz and Pir, 2009; Lozano, 2010; Bosch et al., 2016; Bani *et al.*, 2017). In the maps, to ensure the continuity of the roads and rail infrastructure in rural areas and their disappearance in favour of urban LULLCs inside settlements, urban land use/cover classes and later transport infrastructure were prioritised in the rasterisation. Otherwise, the fragmentation effect created by these barriers (i.e. transport infrastructure and urban settlements) is underestimated. The results obtained in Fragstats v4.4 provided the increase/reduction of metric values between 1999 and 2007 and 2007 and 2030. The patch maps were vectorised in QGIS v2.14, and the patch level metric values were joined to their attribute tables. This permits the spatial association of patch level values to specific patches, which is necessary for comparisons between class level and patch level values. Since the rasterisation slightly affects values of area and shape, this step was necessary to ensure coherence between class level and patch level values.

For the calculation of connectivity indices, the species distribution model from Titeux et al., (2013) was used to narrow down the nodes analysed to patches that show presence. This step was necessary since the use of all possible patches in Luxembourg at 10x10m resolution required an excessive computational demand. This step reduced the computational power demand and made the analysis feasible in terms of time-consumption, keeping in mind constraints also relevant in real planning processes. Only preferred land use/cover class patches for each species in a 1 km buffer around the presence cells and those preferred land use/cover class patches in between buffers were selected as nodes for the calculation of connectivity indices. Overall, structural connectivity analysis was run for all species. Node-level connectivity analysis (i.e. taking in consideration individual patches) was only done for the four non-mammalian species (*Maculinea arion, Triturus cristatus, Alytes obstetricans, Lacerta agilis*) since the computational power demand required was still excessive to model connectivity at node level for mammals. Once obtained, the differences in all connectivity indices between periods were calculated. At the node-level, the results of Betweeness Centrality (BC, BC(IIC) and BC(PC) variants), inter-patch components of Integral Index of Connectivity, and Probability of Connectivity (dConnector and dFlux (IIC and PC)) were associated with their specific patches to identify highly valuable patches (i.e. ones above the 95th percentile value for all the indices) for each year analysed (1999, 2007, 2030).

Preferred routes of movement were calculated with LSCorridors only for the four species for which node-level analysis was done to avoid excessive computational power demand. These species were the only ones for which node-level (structural connectivity) and functional connectivity results were combined. The resistance surfaces of these species were obtained from studies in similar European contexts that developed them based on empirical studies for the same or similar species. For example, in the case of *Maculinea arion*, a grassland butterfly, only the resistance values of land covers existing in our study area were kept. The values of the resistance surfaces were harmonised for all species to a shared value range from 1 to 1000 (Table 4.4). The presence cells per species obtained from the species distribution models of Titeux et al., (2013) were used to select the pairs of sources (starting patches) and targets (end patches) for the routes that were calculated, as this was the most up-to-date species distribution data available for Luxembourg. For all species, 100 m was assumed as the perceptual range, since a low perceptual range was indicated in the literature for all of these species or similar ones.

*Table 4.4. Aggregation of land use/cover classes of Luxembourg maps and resistance values**

Aggregated land use/cover class**	Maculinea arion	Triturus cristatus	Alytes obstetricans	Lacerta agilis
High-medium density urban areas	1000	1000	1000	1000
Low-density urban areas	1000	1000	1000	1000
Roads and railways	750	1000	1000	1000
Cropland	500	750	750	975
Pasture	100	250	250	1
Grasslands	100	500	500	1
Scrubland	1000	500	500	800
Deciduous woodland	1000	500	500	975
Coniferous woodland	1000	500	500	800
Mixed woodland	1000	500	500	900
Water	1000	1	1	1000
Wetlands	1000	250	250	1000
Rockland	100	500	500	1000

*The resistance surfaces were obtained from the following references: *Maculinea arion* (Schneider and Fry, 2005); *Triturus cristatus* & *Alytes obstetricans* (Arntzen *et al.*, 2017); *Lacerta agilis* (Russell, 2012). Later, their range of values was harmonised to share the same scale. ** Land use/cover classes corresponding to the ATT Codes can be found in the SI (Table S4)

4.3. Results

4.3.1. Landscape Metrics

A first diagnosis of the landscape metrics shows that there are no dramatic changes over time (between 1999 and 2030) for all the metrics. Additionally, the values of Shape Average-weighted (SHAPE_AW), Largest Patch Index (LPI), and Normalised Landscape Shape Index (NLSI) are almost constant, which makes these metrics not sensitive enough for interpretation of changes in Luxembourg. Also, the changes in Core Area Average-weighted (CORE_AW) are equivalent to Area Average-weighted (AREA_AW). Because of this, AREA_AW, simpler than CORE_AW, seems to be sufficient for the interpretation of changes in Luxembourg (see further details in Table S5, Annex 4.1). An initial diagnosis as this one can inform spatial planners, in this case Luxembourgish, about metrics from the initial set selected that might not be sensitive enough or worth to be retained for the assessment of alternative planning options.

At the landscape level, the Shannon diversity index (SHDI) shows a slight increase in the diversity of land use/cover class (from 1.90 to 1.95), which could be result of the increase in urban land covers. Contagion (CONTAG) slightly decreases (from 53.09 to 49.37), which might indicate an initial tendency towards an increased uniformity in the spatial distribution of land use/cover classes. This kind of analysis can provide spatial planners with relevant information on changes in the relation among land cover/use classes, and therefore on the general landscape character.

At the class level, cropland, mixed woodland, coniferous woodland, deciduous woodland, and pasture are the types of non-urban land use/cover classes that occupy most of the landscape, and they are the most affected by changes from 1999 to 2007 and the expected changes from 2007 to 2030 (see Figure 4.2). Cropland and mixed woodland in particular show a net increase of their area (increasing related habitats) during the whole period, while other land covers decreased. These land use/cover classes are the most relevant in terms of area and changes along time and are also the preferred land use/cover classes of our selected species. Therefore, the analysis of the remaining metrics (Table 4.5) only focuses on these thematic classes, since these would be the most informative for future land use planning in Luxembourg. Possible applications of the same approach to other contexts should be done according to a similar type of exercise. This means focusing the landscape metric interpretation on the most representative land covers (in area, changes along time, and relevance to surrogate species selected) to ensure time-effectiveness and relevance of the assessment to develop spatial planning recommendations.

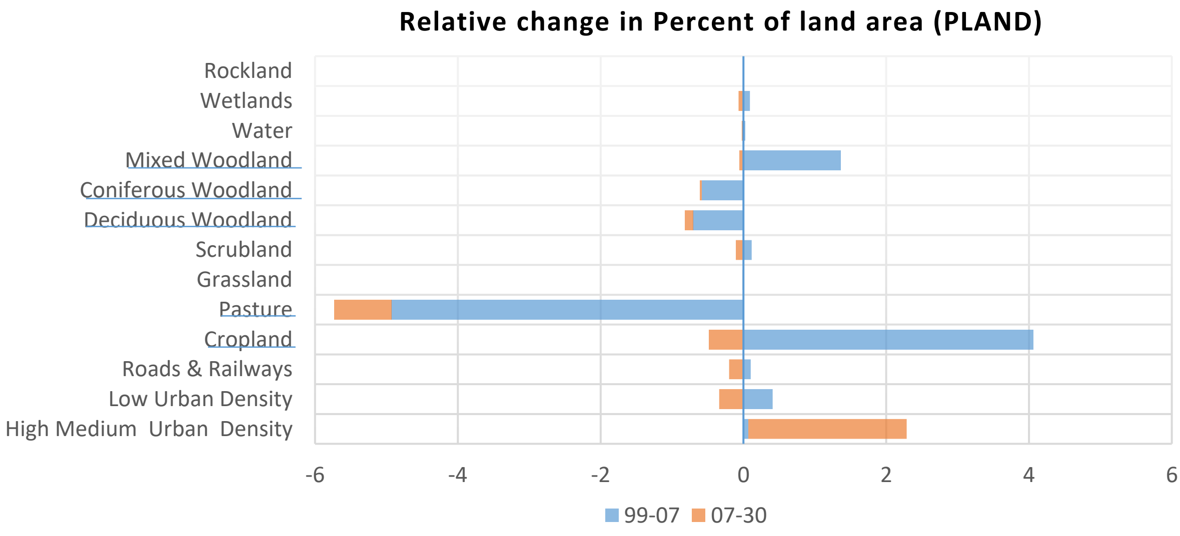

Figure 4.2. Relative change in Percent of Land Area (PLAND) of each land use/cover class since 1999. The blue bars show the changes from 1999 to 2007, and the orange bars show the changes from 2007 to 2030. Pasture, cropland, and woodland land use/cover classes are

Table 4.5. Relative change in percentage of class level metric values from 1999 (99) to 2007(07) and 2007(07) to 2030 (30)

Land Cover Classes and timeframes	Edge Density (ED)		Patch Density (PD)		Average-weighted area (AREA_AW)		Average Euclidean Nearest Neighbour Distance (ENN_AW)		Effective mesh size (MESH)	
	99-07	07-30	99-07	07-30	99-07	07-30	99-07	07-30	99-07	07-30
Cropland	12.65	-2.62	4.29	-2.16	22.33	-0.2	-12.55	0.47	47.7	6.52
Pasture	-12.22	-3.05	-2.44	-4.2	-10.58	-0.66	10.48	0.85	-25.33	-5.24
Deciduous Woodland	0.14	-0.9	0.9	-2.01	-6.77	-0.02	1.18	-0.22	-10	-3.92
Coniferous Woodland	-6.02	-0.5	-5.26	-1.01	-6.23	0.16	5.76	-0.09	-14.06	-2.72
Mixed Woodland	27.99	-2.02	23.92	-2.56	19.67	0.28	-17.99	0.37	58.38	2.64

In the case of cropland and mixed woodland, the increase in AREA_AW and effective mesh (MESH) from 1999 to 2007 indicate a reduction of the fragmentation and net gain of the two land use/cover classes. The increase in patch density (PD) also shows that new cropland and mixed woodland patches are generated. Additionally, the decrease of Euclidean neighbour (ENN) shows an increase of structural connectivity between cropland patches. From 2007 to 2030, for both cropland and woodland, the slight increase in ENN and the reduction of edge density (ED) and PD identifies a minor loss of entire existing patches.

For pasture and coniferous woodland, the decrease in AREA_AW and MESH explains the reduction in the size of patches. The decrease in ED and PD shows that this decrease was more related to the loss of entire patches than to their fragmentation. The overall increase in ENN, except for pasture, shows reduction of structural connectivity. A slight decrease of ENN in pasture from 2007 to 2030 seems to be the result of losing some of the most isolated patches. However, for deciduous woodland an increase in ED and PD and a decrease in AREA_AW and MESH up to 2007 shows the loss of area due to fragmentation. Instead, in

2030 the reduction of ED and PD identifies the loss of entire patches. ENN shows changes equivalent to those of coniferous woodland for the isolation of patches.

Following landscape and class level analysis, a visualisation of areas at patch level shows that for pasture, cropland, and coniferous woodland, the larger patches (i.e. those above AREA_AW at class level) were mainly lost and gained from 1999 to 2007 (Figure 4.3). However, for deciduous and mixed woodland, the changes in larger patches are more homogeneously distributed. The latter is also the case for the changes in all land use/cover classes from 2007 to 2030. The visualisation of landscape metric results at the patch level, supported on class level thresholds, can inform spatial planners, in this case Luxembourgish, about zones were the loss and fragmentation of habitats are more intense and for which mitigation measures might be more urgent. This information is useful when considering future land use/cover changes in specific areas, but also when developing landscape management interventions for them.

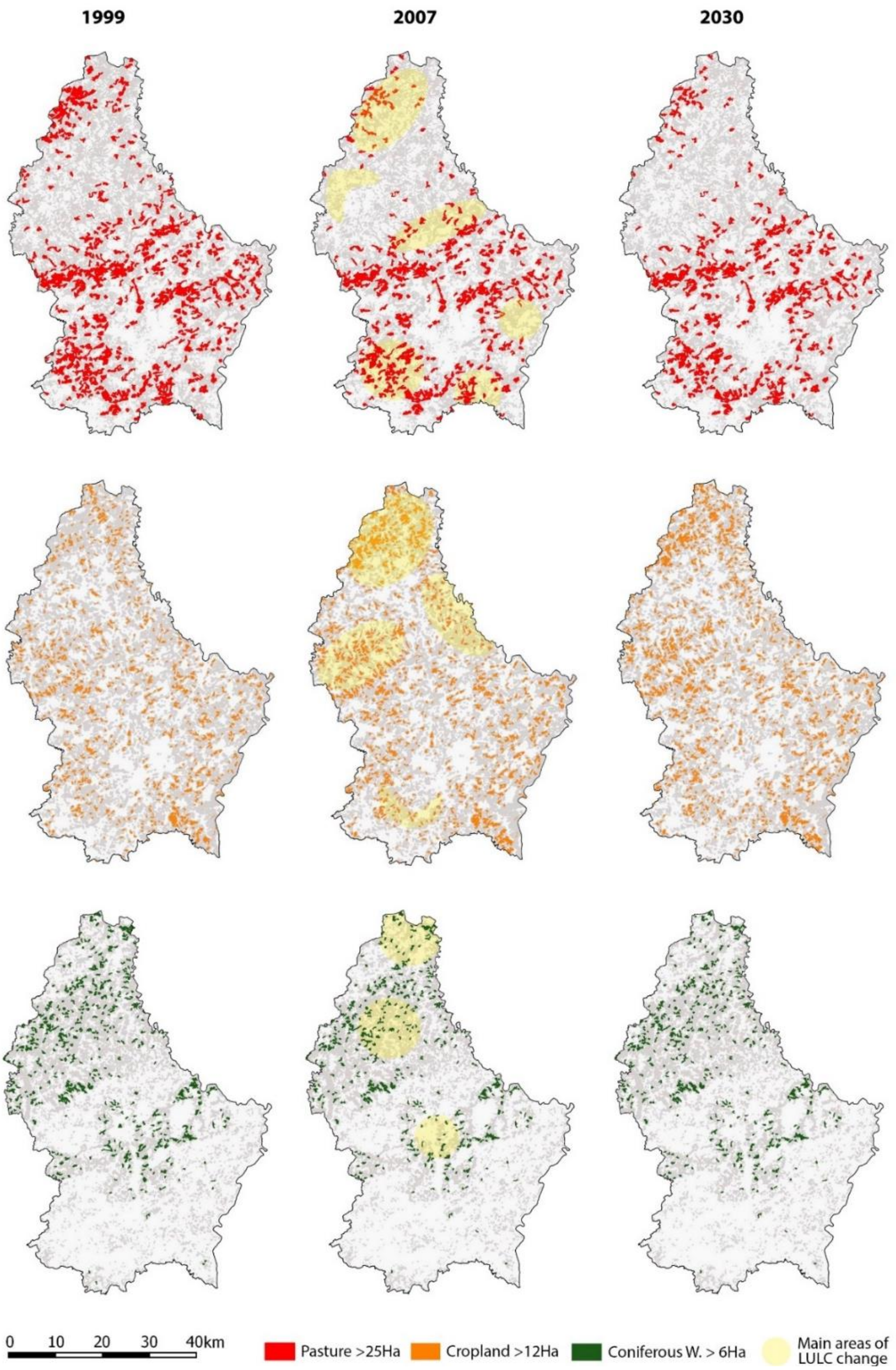

Figure 4.3. Distribution of large patches (i.e. those above AREA_AW) in pasture, cropland and coniferous woodland in 1999, 2007 and 2030 (years organised by columns). The areas highlighted in yellow indicate zones where intense loss of large patches occurred for p

4.3.2. Connectivity indices

An analysis of the results of connectivity indices showed changes in structural connectivity in Luxembourg using a graph-theory method as well as the observation of the performance of binary and probabilistic indices. The main results are described below, for the rest of the connectivity results please refer to Table S6-S10 in the Annex 4.1.

The relative change in the overall value for IIC and PC highlights a reduction of the ecological connectivity for all the selected species from 1999 to 2030 (Table 4.6). ECC makes the implication of this loss clearer, by translating it into an equivalent area of habitat lost if all the connected patches were one single patch. This is quite relevant for species such as *Maculinea arion* or *Felis silvestris,* which lost an equivalent of almost 25% and 33% of habitat, respectively (Figure 4.6). For *Maculinea arion, Alytes obstetricans, Martes martes,* and *Muscardinus avellanarius*, the values of IIC and ECC (IIC) in one of the periods are contradictory with the values of PC and ECC (PC), showing an enhancement of ecological connectivity. This is a consequence of the limitations of binary indices compared to probabilistic ones, which should be considered by spatial planners when using binary indices to analyse planning alternatives, since they consider patches connected or not connected in a more simple form (see Section 4.2 for further explanation).

In the case of *Maculinea arion*, the major decrease in structural connectivity will occur from 2007 to 2030 due to the urban development anticipated in one of the locations where this species is present. In addition, due to the reduced amount of patches for this species, any habitat loss will have a relevant effect on the decrease of ecological connectivity and therefore this sensitivity needs to be considered when making further changes to land use/land covers. For the rest of the species, the major decrease in connectivity occurred from 1999 to 2007. In other geographical contexts, similar exercises might be useful to inform spatial planners about relevant impacts for some species (especially those with a reduced local habitat distribution like *Maculinea arion* in Luxembourg), which could be overlooked if only landscape metrics analysis are performed.

Table 4.6. Relative change (%) of the overall values of the connectivity indices since 1999. IIC=Integral Index of Connectivity; ECC(IIC) = Connected area equivalent to IIC value; PC = Probability of Connectivity; ECC(PC) = Connected area equivalent to PC value

Species	Timeframes	IIC	ECC(IIC)	PC	ECC(PC)
Maculinea arion	99-07	3.37	1.67	-9.59	-4.91
	07-30	-35.64	-19.37	-39.81	-23.95
Triturus cristatus	99-07	-12.87	-6.66	-17.23	-9.02
	07-30	-9.88	-5.45	-4.72	-2.63
Alytes obstetricans	99-07	5.54	2.73	-21.58	-11.45
	07-30	-8.14	-4.04	-11.98	-7.04
Lacerta agilis	99-07	-26.69	-14.38	-30.46	-16.61
	07-30	-8.72	-5.25	-7.83	-4.83
Myotis bechsteinii	99-07	-23.91	-12.77	-32.41	-17.79
	07-30	-0.81	-0.46	-0.54	-0.32
Muscardinus avellanarius	99-07	-17.81	-9.34	-26.46	-14.25
	07-30	4.62	2.51	-0.31	-0.18
Martes martes	99-07	0.46	0.23	-2.22	-1.12
	07-30	-1.16	-0.58	-2.1	-1.07
Felis silvestris	99-07	-34.24	-18.91	-55.27	-33.12
	07-30	-1.01	-0.63	-2.674	-2.03

The analysis of the connectivity indices by node for *Maculinea arion, Triturus cristatus, Alytes obstetricans,* and *Lacerta agilis* indicates that there is a low centrality in all patches (i.e. low values for BC, Tables S7-S10, Annex 4.1), and therefore, there are no key patches influencing the dispersal of individuals. However, if centrality is weighted by area (i.e. BC(IIC), BC(PC)), a few of the key patches for dispersal can be identified. This can be observed, for example, in the individual BC(IIC) results mapped in Figure 4.4a. The combined analysis of dflux (IIC, PC), dConnector (IIC, PC), and BC(IIC) and BC(PC) identifies "key patches" that are simultaneously relevant as dispersal sources and sinks to maintain the connectivity between other patches and as current stepping stones from a probabilistic and a binary perspective (Figure 4.4b). In the case of *Maculinea arion*, part of the "key patches" identified for 1999 and 2007 could be lost due to the urban development predicted for 2030. This result could be useful to inform modifications in the Sectorial Plan of Luxembourg to avoid decline of this species (see Figure 4.6 in Section 4). In other countries, similar applications might be useful to identify minor specific patch changes affecting species which show loss of overall connectivity along time (like *Maculinea arion* in Luxembourg). This can help spatial planners to draft local urban plans or to make more specific mandatory mitigation actions associated with these plans.

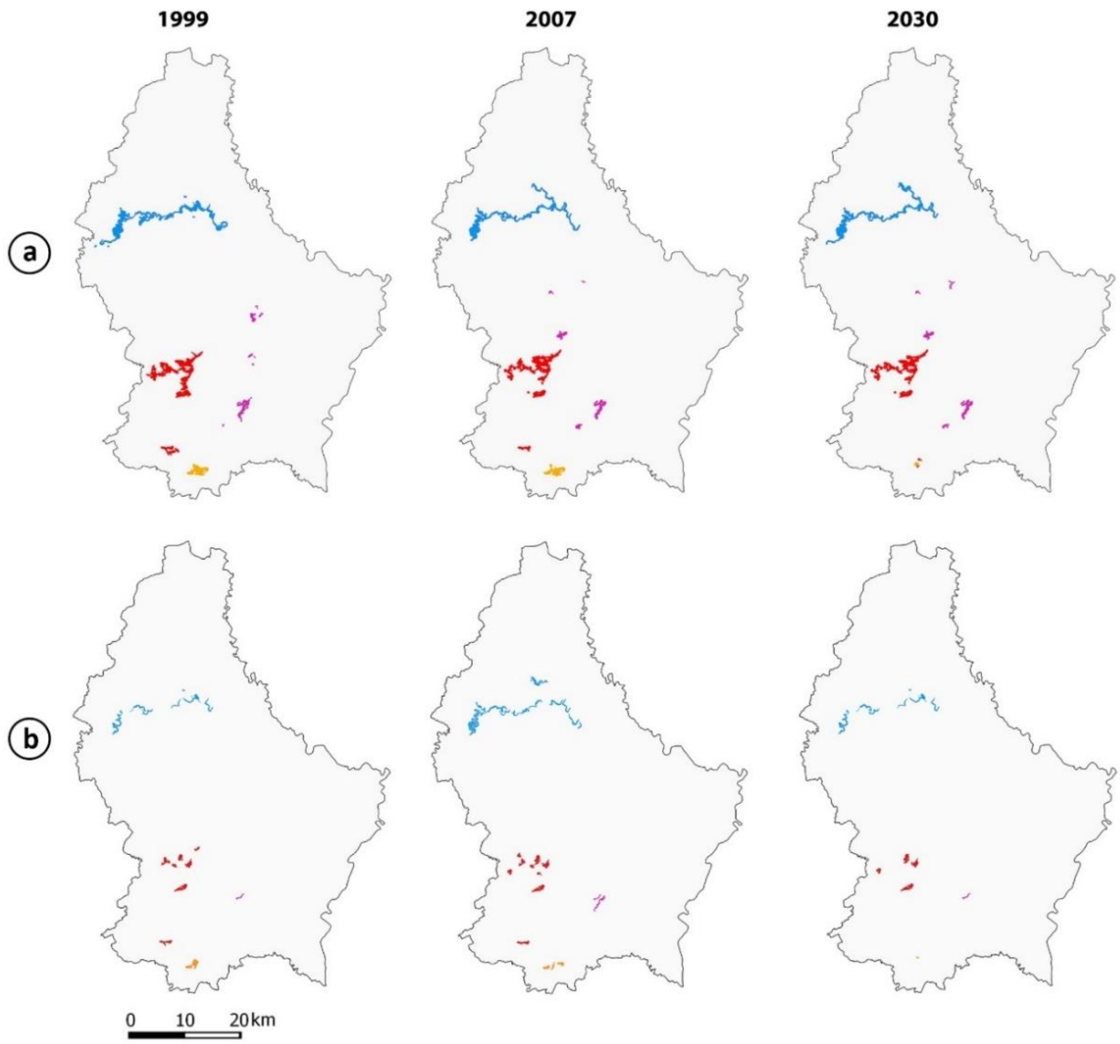

Figure 4.4. a) Patches with values above the 95th percentile for BC(PC); b) Patches with values above the 95th percentile for dflux (IIC, PC), dConnector (IIC, PC), BC(IIC), and BC(PC). See reduction in key patches when the 95th percentile values for several indices need to be fulfilled. *Maculinea arion* (orange), *Lacerta agilis* (red), *Alytes obstetricans* (blue), and *Triturus cristatus* (purple) from 1999 to 2030.

4.3.3. Functional Connectivity tool (LSCorridors)

The functional connectivity analysis based on the least-cost path approach helped us to identify preferred routes of movement for the selected species in the expected land use/cover class mosaic of 2030. In some cases, preferred routes traverse urban areas, which could be explained by the adjacency of most of the sources and targets to urban areas (Figure 4.5e). This is also explained by the much longer distances (and higher cost per route) that would be required to avoid them and by the introduction of stochastic variation in original resistance surfaces by LSCorridors. In Luxembourg, this situation is more common in the southern areas due to the increased urbanisation, highly limiting the options of movement of different species. The results also show that there are few overlaps between the routes of the different species (Figure 4.5). This evidence a common spatial planning problematic, the difficulty of prioritising ecological corridors when preferences and distributions of several groups of species do not match and need to be taken into account.

Some of the routes for some species (e.g. *Lacerta agilis*) also show a good match with Natura 2000 areas (Figure 4.5e). Overlaps like this one could be used by spatial planners to reinforce the relevance of protected areas for potential animal movement at a national level, and not only for the conservation of the fauna and flora within these areas. In other cases, such as for *Alytes obstetricans*, there are certain areas, such as in the north of Luxembourg (Figure 4.5e), where movement between sources does not match Natura 2000 since sources and targets are not yet associated with them. Spatial planners could use this kind of result when discussing new conservation that should be included in an existing network as well as to establish landscape management plans around protected areas that could also influence future local urban plans. Both outputs show an example of how spatial planners, in this case Luxembourgish, could use functional connectivity results together with existing protected areas, such as Nature2000, to prioritise conservation patches along preferred routes of movement to build ecological corridors.

Additionally, the results of the preferred routes of movement complement the connectivity analysis made by the indices, as these can help to identify whether the key connectivity patches identified in Figure 4.4 in 2007 are still maintained in 2030. These results also identify whether the preferred routes of movement overlap with those patches. In terms of spatial planning, in this case for Luxembourg, the latter reinforces the value of protecting specific patches of habitats, since structural and functional analysis highlight their relevance, and act as another example of how to prioritise new protected areas for ecological corridors. Further explanations about the limitations and the relevance of the combination of outputs for spatial planning are discussed in Section 4.3.

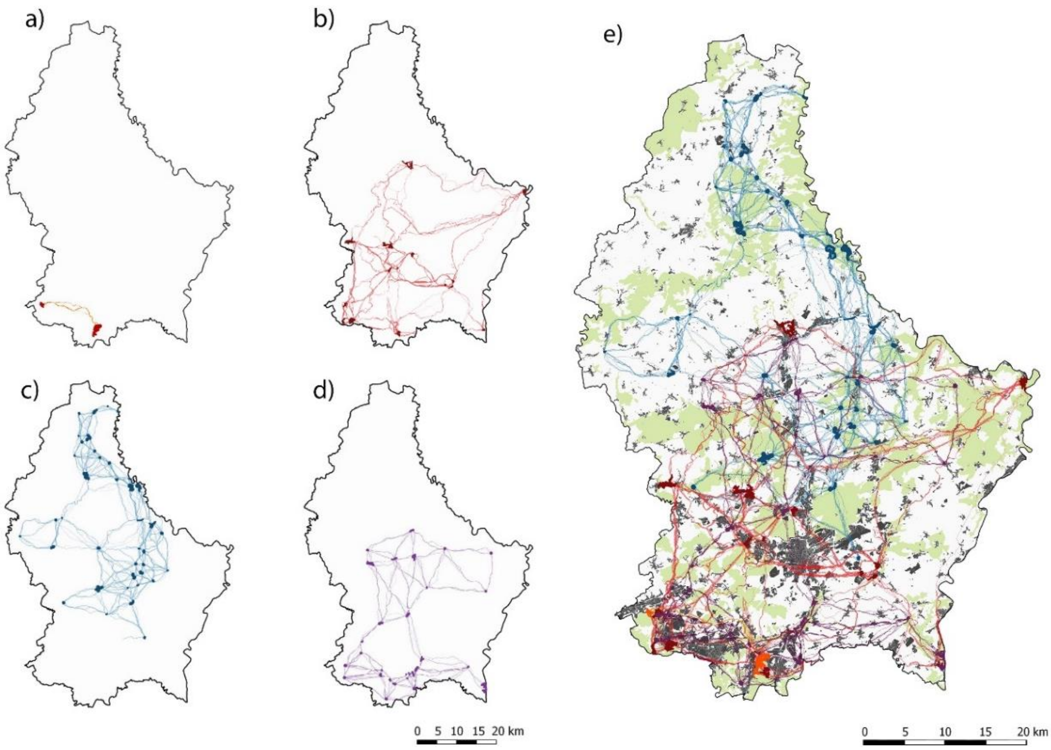

Figure 4.5. Preferred routes of movement modelled in LSCorridors for 2030. a) Maculinea arion; b) Lacerta agilis; c) Alytes obstetricans; d) Triturus cristatus; e) Overlapping of the preferred routes of movements for the different species over urban areas (grey) and Natura 2000 sites (green).

4.4. Discussion

4.4.1. Habitat loss, fragmentation, and ecological connectivity in Luxembourg

The analysis of landscape metrics and connectivity indices shows that a reduction of structural connectivity for all selected species is associated with the loss and fragmentation of pastures, deciduous and coniferous woodlands, grasslands, and rocky areas. It should be noted that a decrease occurs in low density urban areas and transport infrastructure in PLAND (as shown in Figure 4.2), which is due to an unavoidable aggregation in the base maps used as inputs. However, from the results, it is clear that the overarching trend is represented by an increase in total urban land area by 2030, consequently reducing non-urban land covers and related habitats.

The period from 1999 to 2007 has not only the greatest observed habitat loss, but also the highest reduction in ecological connectivity for all species, except *Maculinea arion*. However, a reduction in habitat loss from 2007 to 2030, albeit smaller than the previous period, is not followed by a similar decrease in connectivity in all cases, as shown by connectivity indices. For some cases (e.g. *Martes martes, Alytes obstetricans*), the decrease in connectivity is equally as relevant as in the period from 1999 to 2007. This might be related to the fact that the analysis by connectivity indices only considered part of the preferred land use/cover class patches of each species (i.e. patches showing species presence and those in between them in the species distribution model of Titeux et al., (2013)). It is worth remarking, however, that the reduction in ecological connectivity is not always linearly related to habitat loss and fragmentation (Zeigler and Fagan, 2014; Thompson, Rayfield and Gonzalez, 2017; Edelsparre, Shahid and Fitzpatrick, 2018), something which could further explain this result. For example, in the case of *Martes martes*, almost all the woodland patches in Luxembourg were taken into account in the analysis of connectivity indices due to the well-spread distribution of this species (presence cells) in the landscape. Despite a smaller reduction of preferred land use/cover classes in 2007, the loss of structural connectivity, as shown by connectivity indices, in 2007 and 2030 is almost the same. Moreover, the overall abrupt loss of connectivity for *Maculinea arion* in 2030, supported by the identification of highly valuable patches when applying connectivity metrics, may be caused by the potential loss of key patches in 2030, a result of the new developments proposed in the sectoral plans of Luxembourg (Figure 4.6); these impacts cannot be ascertained only from applying landscape metrics.

Regarding the spatial configuration of land use/cover classes, a potential loss of the most isolated patches of pastures and coniferous and deciduous woodland for 2030, shown by a decrease in ENN, might imply a reduction in the spatial distribution of species such as *Muscardinus avellanarius* or *Maculinea arion*, which are specifically dependent on those land use/cover classes. Concurrently, the mapping of patches above the AREA_AM (Figure 4.3) identifies a concentrated loss of pastures and coniferous woodland from 1999 to 2007 in zones where preferred movement routes for *Lacerta agilis* and *Triturus obstetricans* exist. As a consequence, a loss of redundancy in potential habitats might be occurring in Luxembourg for several species, which could affect the connectivity of their habitats by jeopardising the movement of individuals between local populations or their future migration to alternative habitats if changes in the local conditions occur.

The results are coherent with a few previous studies in Luxembourg or surrounding areas. Regarding landscape fragmentation, studies from the European Environmental Agency (2011, 2017), using MESH as landscape metric, show that Luxembourg and its surrounding territories are highly fragmented, being one of the most fragmented in Europe. Regarding connectivity, a study by Filz *et al.* (2013) shows low butterfly connectivity for calcareous grasslands in an area of south-western Germany very close to Luxembourg, being similar to our results. No other research for Luxembourg or adjacent territories looking at temporal changes in habitat fragmentation and ecological connectivity were found that could inform or be compared against these results.

4.4.2. Limitations of metrics, indices and the connectivity model

The application of different landscape assessment techniques in this paper points toward a number of potential limitations that need to be further discussed to support the interpretation of the results obtained. The most important limitations are summarised and discussed below:

- *the sensitivity of landscape metrics to spatial and thematic resolution and the difficulty of their interpretation when subtle changes occur*.

  It is well known that the values of landscape metrics strongly depend on the resolution of the rasterisation process and also are affected by the aggregation of land use/cover class used (Huang, Geiger and Kupfer, 2006; Buyantuyev, Wu and Gries, 2010). For example, in the case study proposed here, a resolution of 15 metres was also tested (Table S10, Annex 4.1) for which the PLAND value of roads and railways was more overestimated due to rasterisation than at 10 m (since keeping transport infrastructure land covers was prioritised in the rasterisation), further impacting other metrics. The problem of maintaining adequate proportions of transport infrastructure classes during rasterisations has already been shown in previous studies (e.g. Wickham and Riitters, 1995). Additionally, in cases such as that of Luxembourg (i.e. no dramatic changes in the land use/cover class), the values of some metrics might not help to explain fragmentation trends (e.g. LPI, LSI). For other metrics, transformation might be required to facilitate their interpretation and explanation to non-experts (e.g. showing their value changes relatively), which could in some cases misinterpret the meaning.

- *The potential contradictions between binary and probabilistic connectivity indices*.

  The connectivity indices we have considered are either binary or probabilistic. Binary indices are deterministic and only tell us that the patches are connected or not connected. On the other hand, probabilistic indices incorporate randomness based on probability distributions. The probability distributions work as weighting values for the likelihood of a given decision. A probabilistic model provides sets of connectivity indices according to their probability (Saura and Pascual-Hortal, 2007). For example, agent-based models depend on probabilistic indices for the likelihood of the agent's choices of movement from one patch to another. Sometimes we found that these two types of indices gave different results. For example, IICconnector (binary) gives a higher value to short and intermediate distance patches than PCconnector (probabilistic), which affects the overall connectivity value (Bodin and Saura, 2010). In this sense, if land use/cover class conversion generates an increase in adjacent patches of the preferred land use/cover class, and only a few patches are lost, the overall IIC value could increase. This seems to be the reason for results with a positive relative change in overall IIC values when PC values were negative (see Table 4.5). Hence, as recommended by Saura and Pascual-Hortal (2007), when data is available, spatial planers should prioritise PC above IIC analysis, since it seems to be more adequate and will avoid oversimplification of patch connections.

- *an under-estimation of urban land use/cover classes resistance by the modelled preferred routes and connectivity indices*.

  Regarding the ecological connectivity model, in some cases, the preferred routes of species cross urban areas, even if this is not very likely to occur since the selected species would tend to avoid them. However, the extensive presence of settlements (i.e. southwestern Luxembourg, Luxembourg City area) in between source and target patches limits the creation of alternative paths, since the accumulated cost due to an increasing length would be much higher than for paths crossing cells classified as urban. Also, the MP, MLmin, and MLavg paths (most of those crossing urban settlements) created with LSCorridors add stochastic variation to the resistance values. This affects the cost of movement, and makes the paths less restrictive than in the case of MLmax. In order to better adapt the resistance surfaces to Luxembourg, and to adjust the stochastic variation applied in LScorridors, their refinement based on empirical animal movement studies applied to Luxembourg would be relevant. This refinement is a common step in the creation of resistance values, but something that has not yet been done in Luxembourg. With respect to the identification of key patches based on connectivity indices, these are not sensitive to the presence of settlements as the case of *Triturus cristatus* demonstrates (Figure 4.6), where key patches are identified in zones surrounded by urban areas, and these are challenging for animals to traverse. This is a limitation of a simple application of connectivity indices based on a structural connectivity perspective, which uses Euclidean distance instead of a functional perspective that makes use of the least-cost path analysis (Saura and Pascual-Hortal, 2007). But also, it is affected by the fact that many patches with the presence of the study species were adjacent to existing settlements. To cover the limitations of least-cost path (LSCorridors) and the simple application of graph-theory indices (Conefor), models using more advanced heuristic mathematical algorithms could be applied to reproduce ecological corridors. For example, particle swarm (e.g. Liu *et al.*, 2012) and ant colony optimisation (e.g. (Yang, Zheng and Lv, 2012) are two special variants of genetic algorithms based on the movement of animals (bees and ants, respectively) which make use of machine learning. They incorporate a random mutation of weighting values with a clustering algorithm to generate the most probable paths of movement (Dorigo, Birattari and Stutzle, 2006). However, these methods could be too complex to be technically accessible to planning professionals and no application of them was found during our initial literature review.

- *the high computational capacity required to run the models*.

  The analysis of different species at a national level required the authors to reduce the number of patches to be considered for Conefor 2.6 and LSCorridor due to the excessive computational power demand. However, for the mammalian species (*Muscardinus avellanarius, Martes martes, Myotis bechsteinii,* and *Felis silvestris*), this was still too heavy to process with the available equipment, and it was not possible to run the analysis at node level (Conefor 2.6). In many cases, running the tools required more than 24 hours of processing per studied year and species (e.g. input files for Conefor 2.6, and node level analysis). Moreover, for LSCorridors, although the number of simulations was limited to 40 per pair of patches, the outputs for some species occupied more than 20Gb. The computational power demand may make the use of these tools difficult for extensive areas by spatial planning practitioners who may not have access to advanced IT infrastructure. Therefore, spatial planners in Luxembourg as well as from other contexts should consider computational power demand from the very beginning before applying this methodological approach to new assessments.

4.4.3. Implications and opportunities for spatial planning

Despite the abovementioned limitations, outputs likes the ones obtained from using the selected techniques may be useful for spatial planners during assessment phases or the drafting of strategies and plans. Furthermore, the specific combination of techniques linking structural (landscape metrics and connectivity indices) and functional analysis (LSCorridors) might be useful to advance the practical utility of landscape ecology techniques for spatial planning and similar purposes. By modelling the preferred routes of movement, the value of some key patches identified by connectivity indices were reinforced, and others were marked as less relevant as a result of the lack of consideration of barriers (i.e. urban areas) by connectivity indices.

As part of spatial planning works, the combined outputs could be used to improve the diagnosis of current (or potential) habitat status by linking connectivity loss to habitat loss and fragmentation and showing relevant routes of movement and areas to protect in comprehensive maps. In this sense, exercises like this one could be useful for spatial planners in the development (or modification) of strategies and plans. The most relevant routes of movement could be selected by manually removing paths crossing urban areas and giving priority to routes where more simulated paths are adjacent or overlap. These routes could be combined with the key connectivity patches, and the visualisation of patch level results. Then, these outputs can be overlapped onto key features of existing plans (e.g. ecological corridors, protected areas, new areas for housing allocation), and used to support future planning decisions (e.g. selection among alternative spatial development strategies).

To illustrate the above suggestions, Figure 4.6a shows the results of our case study overlapped onto the current Landscape Plan (*Plan Sectoriel du Paysage*) and the Housing Plan (*Plan Sectoriel du Logement*). Map windows (boxes b-e) present examples of the potential use of the results to inform planning. Figure 4.6b, which shows a match between a preferred route of movement and a specific ecological corridor , can be used to confirm the relevance of the latter, and to identify which specific group or surrogate species (in this case *Alytes obstetricans*) moves through this corridor (see other illustrative examples in Pereira, Saura, and Jordán 2017; Pereira 2018). As another example, as indicated in section 3.3, an overlap of key patches from the connectivity indices and preferred routes of movement could reinforce the conservation value of specific patches. If these overlaps are adjacent to existing protected areas (as shown in Figure 4.6c for *Lacerta agilis*), the outputs could support the designation of new protected areas, the extension of existing ones or could inform specific landscape management interventions to enhance routes of movement. Similarly, adjacency of preferred routes of movement of different surrogate species, could serve to prioritise new ecological corridors or strengthen the importance of areas already protected. An example is shown in Figure 4.6d where the routes of *Lacerta agilis* and *Triturus cristatus* are adjacent and greatly overlap with a Nature2000 area. Moreover, the outputs could be used to identify areas not to be urbanised. This is illustrated in Figure 4.6e, which shows a new housing area proposed in the Housing Plan that will cause the loss of the key connectivity patch of *Maculinea arion*, already described in Section 3.2.

Regarding the innovation of this combination of techniques for spatial planning, very few studies were found that simultaneously used landscape metrics, connectivity indices and models based on circuit theory, least-cost path, or techniques such as agent-based modelling (Loro *et al.*, 2015; C. Chen *et al.*, 2017; Simpkins *et al.*, 2018). In the literature review, there were studies combining landscape metrics and connectivity indices (e.g. Elliot *et al.*, 2014; Zemanova *et al.*, 2017) or connectivity indices with least-cost path or circuit theory (e.g. Lechner *et al.*, 2015; Poodat *et al.*, 2015). However, these studies did not go from landscape metrics to least-cost path or circuit theory. To the authors' best knowledge, a combination of the three types of tools has never been applied to study an entire country before. This is most likely due to the difficulty of assessing habitat loss, fragmentation, and connectivity at a high resolution for vast areas. Additionally, the use of surrogate species is considered useful when optimising the conservation of a small set of species with similar ecological requirements and for limited environmental gradients (Mortelliti, Santulli Sanzo and Boitani, 2009). The case of Luxembourg is a suitable exception to facilitate and demonstrate such a methodological approach.

In the case of Luxembourg, its small size combined with its intense population growth requires that planners balance smartly the growth and associated urban development with the protection of habitats. Due to this urgent need and its relatively small size, Luxembourg offers an ideal context to advance the combination of structural and functional landscape ecology analysis for the optimisation of national spatial plans. The coincidence of governmental and study area boundaries could foster the integration of these types of analysis into broader socio-ecological evaluations of national policies and strategies, such as urban development evaluations. Moreover, the scope of the present work offers an appropriated context for socio-ecological transboundary spatial planning studies by making use of the Greater Region (a transnational cooperation structure between the territories of Luxembourg, Belgium, France, and Germany) or Benelux (Belgium, the Netherlands, and

Luxembourg) as case study areas. Such studies might foster international collaboration around Luxembourg for the protection of species, particularly those that share political borders. As urban plans are developed in Luxembourg, and the Greater Region, we advise detailed, up-to-date studies such as this one before the urban plan is put into place. In such instances, the future urban development is an added factor in how species will distribute themselves, becoming a main part of the future of urban planning.

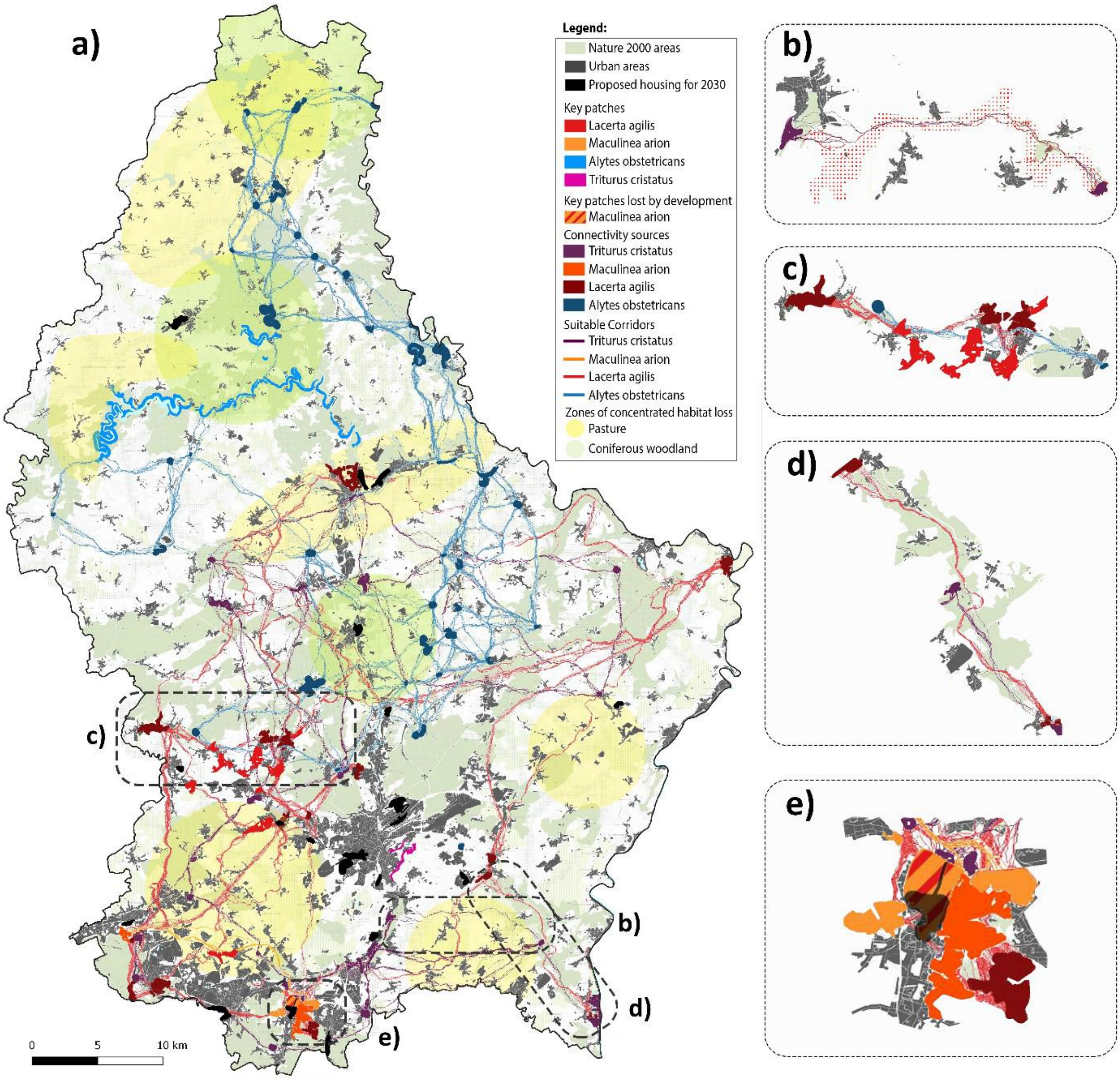

Figure 4.6. a) Illustrative diagnosis based on the combination of outputs overlapped onto the "réseaux écologiques" map extracted from the Plan Sectoriel du Paysage (2014) and the current and future development areas (the new housing proposed by the Plan Sectoriel du Logement (2014) is included in black); b) Zoom of proposed ecological corridor (highlighted in red) matching the simulated route of movement for Triturus cristatus*; c) Zoom to a simulated route matching key patches; d) Zoom to simulated routes for* Lacerta agilis *and* Triturus cristatus *matching Natura 2000 areas; e) Zoom to a settlement in the south of Luxembourg, overlapping the specific housing development (semi-opaque polygon in black) proposed in the Plan Sectoriel du Logement that contributes to the loss of a key patch for* Maculinea arion*.*

Finally, replication of studies like the present one can add strength to existing international conservation networks, such as Natura 2000 area (overlappings with Nature 2000 illustrated for Luxembourg in Figure 4.5 and 4.6), to ensure that the most valuable areas are protected (e.g. Pereira et al., 2017; Santiago Saura & Pascual-Hortal, 2007) and to support spatial planning processes.

4.5. Conclusion

The combined analysis of metrics and connectivity indices shows an increased fragmentation and loss of habitats as well as a reduction of ecological connectivity in Luxembourg from 1999 to 2007 with regard to the selected species (*Maculinea arion, Lacerta agilis, Triturus cristatus, Alytes obstetricans, Martes martes, Felis silvestris silvestris, Muscardinus avellanarius*). The analysis of the proposed urban development up to 2030 shows that this trend will continue, potentially causing a decline in the species population. The selected species are representative of different groups (mammals, reptiles, amphibians, butterflies), habitat specialists (e.g. grasslands, woodlands), and ranges of animal movements. In other words, the conversion of land use/cover classes from non-urban to urban up to 2030 might also affect other species with similar characteristics. Thus, from 2007 up to 2030 it is expected a decrease in the potential capacity to supply the ES class *maintenance of nursery population and habitats* by the ecosystems of Luxembourg.

The combined use of landscape metrics and connectivity indices selected in Chapter 3 can be easily replicated by planners/designers. It would provide them a better understanding as to how land use/cover class conversion or changes in the landscape structure affect ecological connectivity. The combination of metrics and indices would avoid not noticing significant impacts derived from proposed land use/cover changes, which could be missed that when using only one group. Additionally, as it was done in the case study the relevance of key patches identified by connectivity indices can be supported by outputs of LSCorridors when preferred routes of movements overlap with those patches. Therefore, as shown in this exercise, the combined use of different tools proved to be effective in providing useful spatial information during the definition of urban plans and projects, including those integrating NBS, to avoid biodiversity impacts.

When applying functional connectivity models, such as LS Corridor, the use of national empirical studies monitoring animal movement, if available, could allow better adaptation of resistance surfaces improving the quality of the simulated routes. Finally, functional connectivity outputs may be compared against results of other connectivity models/software (e.g. Circuitscape, Condatis, Graphab) or advanced models such as particle swarm (Liu *et al.*, 2012), which are based upon other methods (i.e. circuit theory, graph theory, genetic algorithms) to support testing and validation (from different angles) of the robustness of simulated routes.

When measuring habitat loss and fragmentation, coupling landscape metrics, connectivity indices and least-cost path models for the concurrent analysis of several surrogate species reduces the limitation of applying individual techniques focused on single species. Hence, in spatial planning exercises similar uses of a multitude of techniques at different levels of detail should be encouraged to support nature conservation policies, strategies or plans to better anticipate the negative ecological effects of future urban planning actions or landscape design interventions, and their impact on biodiversity conservation.

In future studies, analyses of landscape metrics and connectivity indices applied on a regular grid at the municipal or canton level could improve the comparison of the different levels of fragmentation and habitat loss among zones (e.g. cities, neighbourhoods) in an urban region. Moreover, the specific combination of landscape ecology methods presented in this chapter could be incorporated into complex modelling frameworks studying other ES classes as well as negative impacts. Chapter 5 introduces in its conceptual framework, how this integration could be developed.

Chapter 5

Combining life cycle assessment, system dynamics and ecosystem services to assess nature-based solutions: an application to urban forest[‡‡]

5.1. Introduction

As Chapter 2 illustrates not all the nature-based solutions (NBS) are able to address all types of urban challenges (UC). The capability of an NBS to address specific UC strongly depends on the ES classes (and amount of them) that it can supply and the capability of these ecosystem services (ES) for mitigating the UC of interest. Consequently, the capacity of urban NBS to mitigate environmental impacts, sudden shocks (e.g. heatwaves) and chronic stresses (e.g. recurrent droughts) needs to be assessed in advance before implementing them.

Besides the positive impacts of NBS, as described in Chapter 3, few authors have started investigating their negative impacts, such as ecosystem disservices (von Döhren and Haase, 2015; Schaubroeck, 2017) or from the management practices (e.g. arboricultural actions) applied on them (e.g. Ingram and Fernandez, 2012; Ingram, 2013; Mcpherson and Kendall, 2014; Mcpherson, Kendall and Albers, 2015; Petri *et al.*, 2016). Similarly, NBS generate biological waste from urban trees as raw material inputs that causes relevant financial costs, even if few emergent studies advocate the re-utilisation of this waste to reduce environmental impacts and convert waste management from a financial costs into a benefit (Nowak, Greenfield and Ash, 2019). Hence, it is necessary to account for positive and negative impacts over the entire life cycle of NBS to better understand the overall net positive contribution of NBS and whether they can address UC.

‡‡ Chapter 5 is based on:
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Roles of other authors:
Benedetto Rugani, Davide Geneletti and Guido Sonnemann were the academic supervisors of the paper.
Claudio Petucco reviewed the conceptual methodological framework from an economic disciplinary approach and contributed to the reviewed of the paper
Thomas Elliot contributed to the review of the paper

When accounting for positive environmental impacts in the form of ES, it is necessary to use assessment methods that consider the changes in ES supply over time and when they are required by the population due to variations in ecological pressures (Sutherland *et al.*, 2018). As shown in Chapter 3, simple methods might oversee these temporal variations. As already stated by several scholars, ES assessments also need to take into account temporal and spatial dimensions (Bagstad *et al.*, 2013; Elliot *et al.*, 2019). In addition, Grêt-Regamey et al. (2017) advert that ES studies in many cases offer oversimplified approaches, assess a reduced number of ES, lack inclusion of monetary valuation and do not acknowledge uncertainty.

Regarding the use of oversimplified approaches, a review from Haase et al. (2014) shows that many urban ES studies only account for changes in ES supply as a consequence of urbanisation. These studies assume that specific land use/cover classes will provide the same ES supply everywhere independently of the ecosystem's maturation or land management of specific plots. However, the use of land use/cover classes might not permit to assess adequately urban ES such as regulating services (Cortinovis and Geneletti, 2019). They might not permit either to capture changes derived from planning actions developed at the very local scale, such as the implementation of NBS (Cortinovis and Geneletti, 2018a, 2019).

In many cases, current ES assessments are unable to consider multiple ES simultaneously due to a lack of scientific and empirical evidence on several ES and the difficulty in measuring them as tangible local benefits (Cortinovis and Geneletti, 2019). As a consequence, synergies and trade-offs among bundles of ES cannot be fully considered (Cord *et al.*, 2017).

As illustrated in Chapter 3, monetary valuation is not yet broadly applied in urban ES assessments, it is already included in few tools suited for urban areas such as i-Tree and the Green Infrastructure Valuation Toolkit. However, these tools rarely include negative externalities or associated costs that occur over the life cycle of NBS (i.e. from the supply of natural and human-made elements to implement NBS up to their end of life). For example, i-Tree monetise several ES (e.g. carbon sequestration, air pollution removal) and allows the consideration of changes in the value scale. However, it does not simulate financial costs, which the user should provide separately (i-Tree, 2020). The Green Infrastructure Valuation Toolkit calculates ES monetary values, but it does not include capital expenditures (i.e. investment costs) and operational costs (The Mersey Forest *et al.*, 2018). However, none of the NBS modelling tools identified in Chapter 3 quantify financial costs, ecosystem services and negative externalities (i.e. non-marketable disservices, such as the expenditure for air pollution mitigation derived from biological waste management) concurrently.

The aim of this Chapter is the development of a modelling framework that combines methods from life cycle thinking, urban ecology, and ecosystem services to assess in comprehensive form the contribution of NBS to urban sustainability and resilience. The work presented here builds on the findings of Chapter 2, 3, and 4.

The use of system dynamics modelling is a core element of the modelling framework interrelating methods from the different disciplinary approaches. The use of system dynamics modelling have been shown adequate to represent the ecological components and their complex interactions (e.g. Ouyang et al. 2007; Jerez et al. 2015; Marchi et al. 2015; Mohammed and Babatunde 2017), demonstrating potential to avoid oversimplified assessments.

In the next section, the conceptualisation of the combined modelling framework is illustrated. In Section 3, the modelling framework is used for the development of a proof of concept model for urban forests. In Section 4, the proof of concept model is tested in a case study in Madrid (Spain) to show its potential value for NBS sustainability and resilience assessments. In Section 5, the limitations and advantages of the modelling framework, and the proof of concept model are discussed. The last section summarises the most relevant findings and introduce future works.

5.2. Conceptualisation of the combined modelling framework

The following section describes: i) the spatial, temporal and thematic dimensions of the conceptual modelling framework; ii) how the environmental impacts, externalities and financial benefits and costs are accounted over the entire life cycle of NBS; and iii) the interaction and description of the main components of the conceptual modelling framework.

5.2.1. Spatial, temporal, and thematic dimensions of the modelling framework

In order to capture changes at the very local scale, the modelling framework should consider a spatial extent that can cover interventions developed at neighbourhood level. High spatial resolution is also necessary to discern the variation in ES flows of different parts of the same intervention and between similar alternatives (Cortinovis and Geneletti, 2018b). However, to represent adequately some ES (e.g. *maintaining nursery populations and habitats*) the spatial extent should go beyond the neighbourhood level. Similarly, to understand whether the supply of an ES significantly contributes to the mitigation of an UC or not, a spatial extent broader than the neighbourhood level might be required. In fact, the spatial extent depends partially on the spatial variability of the ecological pressure(s) underlying the UC of interest, and the spatial variability of the population exposure and vulnerability, which inform about the real need for ES demand (Baro *et al.*, 2016). Moreover, negative environmental impacts derived from management actions applied to NBS, such as extraction of raw material and management of its waste, might not occur in the same place where the NBS is implemented and the ES are generated. Thus, the modelling framework must consider multiple spatial extents and resolutions.

Concerning the temporal dimension, environmental impacts, benefits and costs arise at different points in time. Consequently, the temporal extent of the modelling framework should cover all life cycle phases of an NBS (Ottelé *et al.*, 2011). For NBS, these phases can be synthesised in implementation phase (i.e. from sowing or raw material extraction until the NBS is put in place), operational phase (i.e. from NBS implementation up to the death of the entire NBS or part of their components), and end of life phase (i.e. from collection of dead components up to the end of waste management treatments). Additionally, the temporal resolution of the modelling framework should be aligned to the temporal variability of the socio-ecological processes influencing ES flows. For example, to adequately quantify some ES (e.g. *regulation of hydrological cycle and water flow*), a detailed temporal resolution is necessary. Otherwise, the modelling framework could miss changes in socio-ecological dynamics (e.g. variations in the soil water balance due to the interaction between precipitation, evaporation, infiltration and percolation) that occur at very short time scales (Almeida and Sands, 2016). Similarly, the temporal resolution should also capture the temporal variability of the ecological pressures influencing ES flows and ES demand. The latter is relevant to quantify when ES supply become an actual societal benefit in the form of a positive externality (i.e. a good or service for which a market does not exist, such as nature-based recreation). Hence, as in the case of the spatial dimension several temporal resolutions are required together with the consideration of long temporal extents to include all life cycle phases of NBS.

To simulate ES supply and when ES are required, the thematic extent should include not only components of the ecological system but also the human system. In addition, as already anticipated by Haase *et al.*, (2014) and Cortinovis and Geneletti (2019), for urban ES assessments the thematic resolution used to represent urban areas should move beyond land-use/cover. The thematic resolution should be higher in order to represent and categorise individual attributes contributing to the processes influencing ES supply and when they are required by the population. This means that the thematic resolution of the framework should be able to distinguish between NBS types and between variations in the same NBS type that would lead to different ES flows.

To balance the multiple needs in terms of spatial, temporal and thematic dimensions, the modelling framework is developed at two levels: foreground and background. The foreground corresponds with the system dynamics core of the modelling framework. It focuses on quantifying locally generated ES and intermediate outputs (e.g. amount of waste generated) in biophysical units. The background level uses the foreground inputs (e.g. number of trees planted) and its intermediate outputs (e.g. amount of biological waste generated) as inputs to calculate final outputs in the form of environmental impacts and economic values. It makes use of existing inventories and databases (e.g. environmental inventories, price databases) and relies on methods such as Life Cycle Assessment (LCA), ecological connectivity modelling and economic valuation methods to calculate final outputs.

The foreground is framed for assessments at the neighbourhood level and makes use of a default spatial resolution of few meters. Three temporal resolutions are considered (daily, monthly and yearly). A daily time-step is used for socio-ecological processes with fast variations over time such as in the case of tree transpiration. A monthly time-step is used for processes where changes are seasonal or for which assuming a monthly linear behaviour (e.g. tree growth) do not lead to significant inaccuracies. The yearly time-step is used for aggregating intermediate outputs and when changes in net ES supply over several years need to be quantified. A long temporal extent is used to cover the entire operational life of an NBS, until major modifications on them are required or until the social actors bearing their costs and benefits change. Previous urban NBS studies have considered 50 years as the default operational life for NBS (Ottelé *et al.*, 2011; Perini and Rosasco, 2013; Broun *et al.*, 2014). This assumed lifetime is also aligned with the one of buildings and related urban infrastructure. From the perspective of social actors, around 35-40 years correspond to the average working life of people in the EU-27, and around 63 years to their life expectancy once they become adults (Eurostat, 2020a, 2020b). In this sense, the use of a default operational life of 50 years ensures that the overall contribution of NBS to sustainability and resilience is accounted inside a temporal extent equivalent to one adult generation. The thematic resolution differentiates between main biotic and abiotic components of NBS (i.e. water, soil, vegetation) and specific variations (e.g. vegetation species, soil texture) on them. The thematic resolution also plays a role in how some key management actions are undertaken, such as pruning and replanting.

Different to the foreground level, in the background level the calculations assume a static condition and in many cases are not spatially explicit. Making use of a dynamic approach to calculate negative environmental impacts that occur beyond the local scale, at different moments, and involve a diverse set of natural and technological components would be too complicated. The use of a static condition facilitates the use of a broad set of databases and inventories which are not compatible with dynamic assessments.

5.2.2. Accounting for environmental impacts, externalities and financial benefits and costs over the entire life cycle of NBS

As described in Chapter 1 and Chapter 3, to provide a more complete assessment of the contribution of NBS to sustainability and resilience assessments should go beyond the environmental dimension. Consequently, the modelling framework accounts for the contribution of NBS to sustainability and resilience in the form of environmental and economic values (Figure 5.1). Environmental values are result of the quantification of environmental impacts (positive and negative), which are calculated in biophysical units. Their calculation is required for the quantification of externalities, which are a monetisation of environmental impacts. Economic values are the result of the quantification of financial benefits, financial costs and externalities (positive and negative), which are calculated in monetary units. Financial costs can occur in the form of capital expenditures (implementation phase), operational expenditures (operational phase) and end of life costs. Similarly, positive and negative externalities can be observed in all project phases. In this sense, quantification of costs and benefits covers the entire life cycle of NBS, include cash flows already internalised (financial values) and monetised environmental impacts (externalities). These aspects makes the monetary quantification in the modelling framework very similar to the concepts of full environmental life cycle costing and environmental cost benefit analysis (Hoogmartens *et al.*, 2014; Schaubroeck, Petucco and Benetto, 2019). The two main differences between both approaches remain vague for NBS (i.e. product vs project and lifetime vs life cycle). For simplicity, since the focus is on NBS interventions, closer to a project than to a product, hereafter there is only reference to cost-benefit analysis.

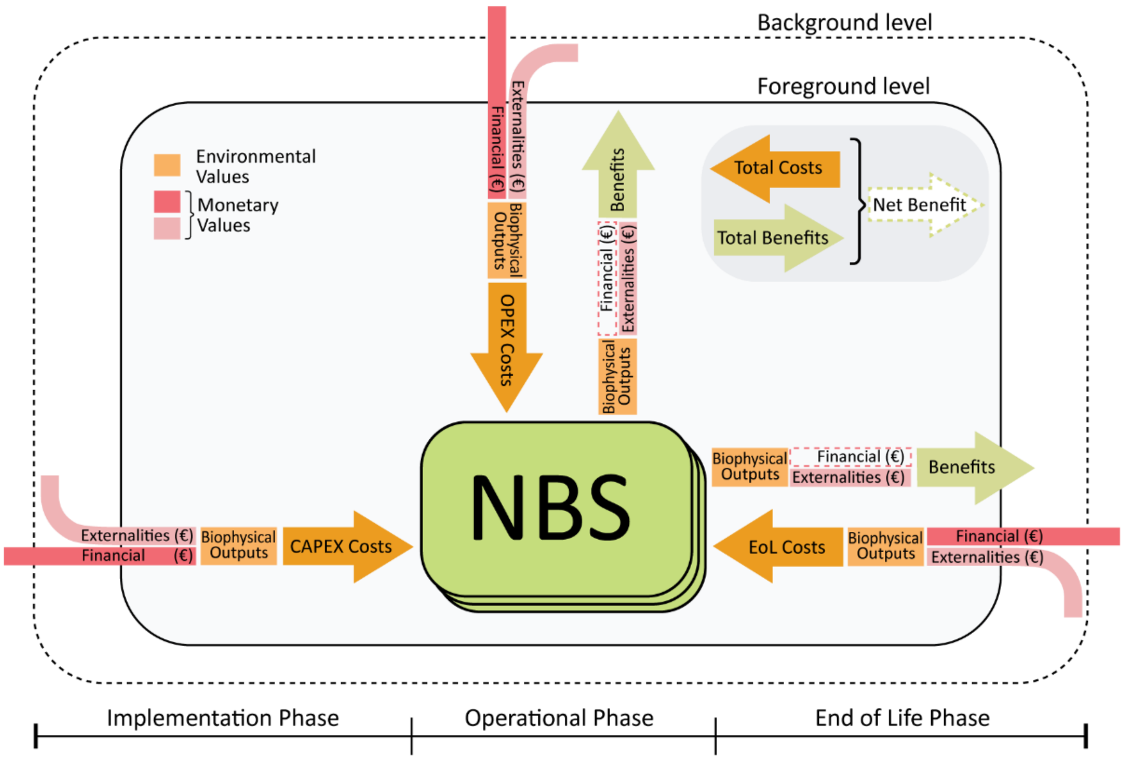

Figure 5.1. Types of costs and benefits associated with nature-based solutions (NBS) considered in the modelling framework by applying life cycle thinking principles; CAPEX: capital expenditures; OPEX: operational expenditures; EoL: end of life costs.

Positive environmental impacts are mainly accounted in the foreground level as ES supplied by NBS for which a demand exists. ES classes are considered only when presenting a causal relationship with UC, as identified in Chapter 2. The link with UC is assumed to reflect an actual under supply of those ES. For most of those ES, it is assumed that there is always a constant local demand over time. For example, as illustrated by Elliot, Babí Almenar and Rugani (2020) for the case of Lisbon, besides the global demand, local *regulation of chemical condition of the atmosphere* in the form of CO_2 storage is always needed because it cannot overcome the local emission of greenhouse gases. Certain ES, however, are demanded only when specific thresholds are overpassed. Consequently, the benefits generated by these ES are accounted for only when they are actually needed by citizens. For example, *regulation of temperature & humidity* in the form of cooling is relevant, and hence measured, only in the hottest periods of the year (Moss *et al.*, 2019). Negative environmental impacts and few positive environmental impacts are accounted for in the background level as LCA midpoint impact categories. Positive impacts are rare and only occur when outputs generated from waste (end of life) can serve as inputs for new technosphere processes, avoiding the consumption of additional resources. To further investigate whether *regenerated* materials would substitute *new* raw materials in real markets was outside the scope of this research.

As illustrated in Figure 5.2, besides ES, the foreground level also generates intermediate biophysical outputs quantifying management actions and the generated biological waste. These outputs are required for the calculation of operational expenditures and end of life expenditures in the economic valuation. Additionally, the inputs of the foreground level defining the implemented elements (e.g. amount of vegetation planted and their species) are also used to inform the calculation of the capital expenditures. Financial costs from capital expenditures and operational expenditures are computed based on the quantities of inputs and intermediate outputs, which are converted into monetary units making use of available national price databases or local studies (Figure 5.2). For some elements, the financial cost might be already provided in the bill of quantities of the project, in those cases it is not necessary the use of price databases, because the cost is already known (and provided) by the built environment professional. The characterisation of the technosphere processes associated with the inputs and intermediate outputs described above permit the calculation of the negative environmental impacts as outlined in the above paragraph. As described in Chapter 3, and illustrated on the bottom of Figure 5.2, negative environmental impacts are converted into negative externalities making use of the environmental prices defined by De Bruyn *et al.* (2018). Similarly, ES outputs are converted into positive externalities making use of values from existing local environmental economic studies. If local studies are not available a value transfer approach is used based on existing scientific literature.

Based on the point in time at which positive and negative environmental impacts are arising, discounting can be applied to the calculation of externalities. However, De Bruyn *et al.* (2018), following the indications of the Dutch Discount Rate Working Group, does not recommend modifying the price of externalities generated by future environmental impacts that end damaging human health. Hence, discounting is considered only as an option in the modelling framework, but by default it will not be applied. When, discounting is selected a three percent annual discount rate was assumed in line with previous studies on NBS (e.g. Foudi *et al.*, 2017; Silvennoinen *et al.*, 2017; Johnson and Geisendorf, 2019). For sake of consistency, when discounting is selected it applies to all impacts.

5.2.3. Description of the main components and interactions in the conceptual modelling framework.

The interactions between the foreground and background level and between the modules of the foreground level (the system dynamics core) are summarised in Figure 5.2. These interactions and the structure of the background and foreground levels will be explained in the following sections

5.2.3.1. Background level: Ecological Connectivity Assessment and Life Cycle Assessment

As described in Section 2.2, most positive environmental impacts are quantified in the foreground level. Few are quantified in the background level, through LCA, as avoided environmental impacts. However, there are positive impacts that due to the required spatial extension for their assessments need to be calculated in the background level. In those cases, the roles are inverted, and the background level becomes the main modelling component. This is the case of the ES *maintaining nursery populations and habitats*, which is related to biodiversity issues and for which the use of integrated ecological connectivity assessment seems the most appropriate methodological approach as already illustrated in Chapter 4. In the case for ecological connectivity assessments (see right side of Figure 5.2), the foreground could provide intermediate spatial outputs tracking changes on biophysical attributes (e.g. tree height, crown width) that inform about the maturation of the ecosystem over time. These results could be used as proxy-parameters to inform when mature habitat patches, as those identified in land cover maps, are generated or have disappeared for a specific neighbourhood level intervention. Ecological connectivity assessments could use the latter data to evaluate whether changes in one NBS intervention (or several of them) influence the ecological connectivity in the broader urbanised context.

For the calculations of negative environmental impacts through LCA, the foreground inputs and intermediate outputs (i.e. management actions and generated biowaste) provide the project specific information. As illustrated in Chapter 3, in many LCA and LCC studies on NBS, a unit of area (e.g. square meter) and the NBS lifetime are used to define the functional unit. For example, in the case of green roofs a square meter of green roof over 50 years is a common functional unit. In this modelling framework, area and lifetime are also used as the functional unit by default. This is justified on simplicity of this functional unit, that it suits a large variation of NBS Types (e.g. green roof, green walls, woodland-like), and the fact that all the environmental impacts over the entire life cycle of NBS could be related to a unit of area.

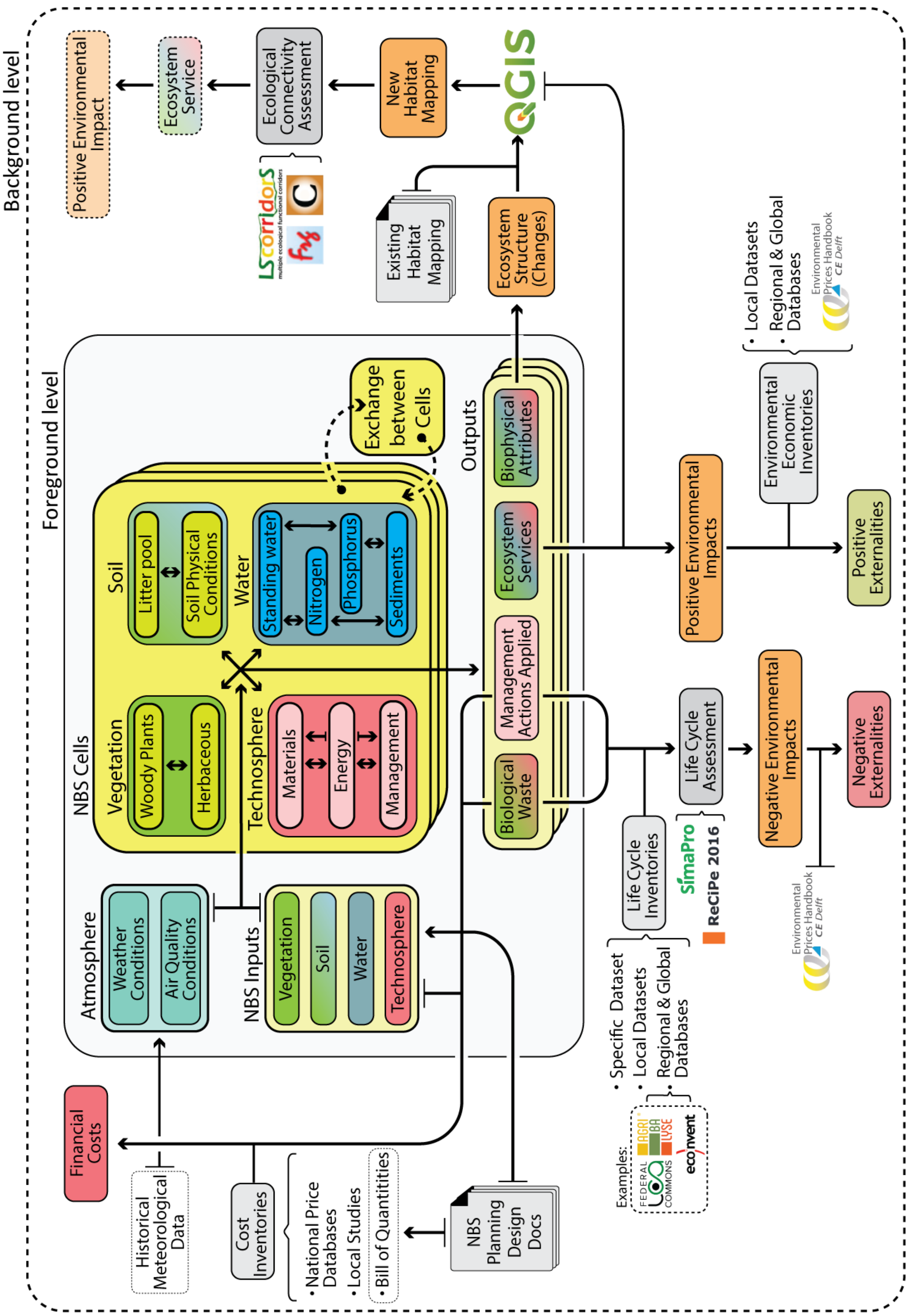

Figure 5.2. Schematic diagram of the main interactions among the components of the modelling framework. To facilitate the visualisation compartments in the sub-modules of NBS cells are represented as individual entities even if this might not be always the case.

In the background, the technosphere processes required for the management actions and waste treatment need to be documented and quantified (life cycle inventory analysis), including processes that occurred in the past. For example, as illustrated in Chapter 3, tree planting as a management action should also include all the nursery processes (from sowing up to selling the tree) and the transport of the tree to the site unless these are accounted as independent actions. The bill of quantities of projects could already describe partially processes related to inputs such as transport of plants. To finalise the documentation of processes, in most cases scientific literature and databases of life cycle inventories need to be used (Figure 5.2). For example, for the case study in this Chapter, the processes for tree planting (i.e. aggregating processes from sowing up to planting on site) pruning and replanting (i.e. including tree removal plus another planting) are documented and quantified making use of life cycle inventories (i.e. Agrybalyse, ecoinvent) and scientific literature (Ingram and Fernandez, 2012; Ingram, 2013; Luck *et al.*, 2014; Mcpherson and Kendall, 2014; Mcpherson, Kendall and Albers, 2015; Petri *et al.*, 2016).

Once the processes are documented and quantified, the negative impact are calculated as midpoint impact categories. These categories quantify impacts in the form of environmental effects, as it is the case for ES, and not as final damages to specific areas of protection (e.g. human health). Representing both, negative and positive impacts, at a similar level of abstraction facilitates their comparison, especially for alike categories. For example, *global warming potential* and *regulation of the chemical condition of the atmosphere*. The former measures greenhouse gas emissions and the latter greenhouse gas sequestration. In addition, both use the indicator mass units of CO_2 equivalent as a parameter-proxy.

In terms of the specific life cycle impact assessment method, ReCiPE 2016 (Huijbregts *et al.*, 2017) and the Environmental Footprint 3.0 developed by the Joint Research Centre were selected as the preferred methods. In fact, for the illustrative case study in this chapter, negative impacts of the different management actions were calculated with both methods and equivalent mid-point categories compared. It was seen that results were similar and there were not significant differences (see Annex 5.1). However, it was considered that the midpoint categories of ReCiPE 2016 could be easily understood by non-LCA experts. Additionally, the midpoint categories had more equivalences with the ES classes of interest. Moreover, the environmental prices proposed by De Bruyn *et al.* (2018), are calculated for ReCiPE 2016 and its midpoint and endpoint categories. Therefore, ReCiPe 2016 was proposed as the default life cycle impact assessment method to quantify negative impact assessments.

5.2.3.2. Foreground level: System dynamics core model

The system dynamics core model describes the evolution of the NBS over time and quantifies their positive and negative impacts. It is defined by four modules: Atmosphere, NBS Inputs, NBS cells, and Outputs. Each of these modules is described in the following lines.

The atmosphere module is defined by the sub-modules weather conditions and air quality conditions and acts as a daily weather generator. The sub-module weather conditions characterise the daily temperature (average, maximum, minimum and dew), vapour pressure deficit, precipitation, average wind speed, atmospheric pressure, cloud fraction and solar radiation. The sub-module air quality conditions characterise the daily average ambient levels (i.e. atmospheric concentration) of air pollutants (i.e. CO, SO_2, NO_2, O_3 and PM_{10}) commonly used in the definition of well-known air quality index (i.e. AQI, CAQI, EAQI). Both sub-modules work at a neighbourhood resolution, this means that the daily values defined by them apply equally to all the NBS cells.

The NBS inputs module is defined by the sub-modules vegetation, soil, water and technosphere. It is the module where the thematic resolution of each of the components of each NBS cells sub-module is characterised. In other words, this module contains the parametrisation of the variables defining categorical variations in the NBS attributes of influencing the socio-ecological processes in the model. For example, in the case of vegetation, the species is one of the most important biotic attributes to which the parametrisation for multiple processes (e.g. growth rate, leaf fall, maximum transpiration) is associated. The technical documentation of the specific NBS project indicates which categorical variations correspond to the attributes of each NBS cell. Only the categorical variations (e.g. clay soil texture) of attributes for which its parametrisation is already defined in the NBS input module can be assessed for specific NBS interventions. Thus, an extensive library of categorical variations for the key attributes is needed to adequately represent real interventions.

The NBS cells module is composed of the sub-modules vegetation, soil, water and technosphere. NBS interventions to be assessed are split in multiple NBS cells of few square metres. Each cell represent its piece of intervention as best as possible making use of the library of categorical variations of attributes defined in the module of NBS inputs. This means that the cell is considered the minimum divisible unit whose attributes and behaviours are spatially homogeneous. The NBS cells module is where changes in biophysical attributes, socio-ecological processes and the derived ES flows and biological waste are quantified and later stored in the Output sub-module. It is also where the applied management actions are quantified. These changes depend on interactions among each of the sub-modules. For example, as a management action it might be established in the NBS inputs that trees must be replanted once they die, but replanting also depends on other sub-modules (e.g. month of the year).

The vegetation sub-module is formed by the compartments woody plants and herbaceous plants. Two compartments are required because these groups of plants differ in their growth and the role of their inner structures in different processes. It is where vegetation growth occurs together with associated changes in biophysical attributes (e.g. root depth, leaf area) that influence socio-ecological processes such as rain interception, transpiration, evaporation, air pollution filtration, biological waste generation.

The soil sub-module is formed by the compartments litter pools and the soil physical conditions. The former contains the biotic conditions and it is where the litter, humus, and microbiota interact. The latter is where physical conditions such as percentage of clay or soil bulk density are defined. The interactions in the soil sub-module influence litter decomposition, soil carbon emission, soil evaporation, water storage, tree transpiration, tree morbidity, and management actions such as irrigation.

The water sub-module is used to represent NBS related to water ecosystem and includes the following compartments: free-standing water, nitrogen pool, phosphorus pool, and sediments (settling of suspended solid). The free-standing water defines a simple water balance model of the NBS. The nitrogen pool is the compartment where the mineralisation/nitrification, denitrification, and volatilisation processes occur. The phosphorus compartment and the sediment compartment are where the interactions influencing the removal of phosphorus through settling occur. Three main assumptions were made in the conceptualisation of the water sub-module. First, there is no carbon sequestration; second, resuspension is negligible and thus omitted; third, there is no seasonal or permanent water stratification independent of water depth. The last two assumptions, are usually applied in wetland models (e.g. Lee, Mostaghimi and Wynn, 2002; Chavan and Dennett, 2008; Neitsch et al., 2011) to consider that water flow and nutrients (phosphorus and nitrogen) are evenly mixed through the waterbody. As explained in Grafius *et al.* (2016), carbon sequestration occurring in water NBS, it is not usually considered, and therefore it is not taken into account in this sub-module. In fact, carbon dynamics have a high temporal variability in water NBS (i.e. it moves from net emission to sequestration due to changes in conditions (Schäfer *et al.*, 2014)), which are difficult to predict.

The technosphere sub-module is composed of the compartments stocks of materials, energy, and management actions used in NBS. Materials and energy could be very relevant in the interactions of hybrid NBS such as green roofs or green walls. Management compartment represent human activities applied on vegetation, soil, and water to maintain an NBS over time. They represent a non-physical attribute, and therefore could be very relevant to study the effects of NBS Type 1 (i.e. better management of an ecosystem) or NBS Type 2 (i.e. restoration/reclamation of an ecosystem) applied on existing ecosystems. More explanation about the management compartment is provided in Section 3 as part of the description of the proof-of-concept urban forest model.

More detailed explanations about the atmosphere, vegetation, soil and technosphere sub-models are provided in Section 5.3 as part of the description of the proof-of-concept urban forest model.

5.3. Proof-of-concept of NB$ for urban forest

Based on the above modelling framework, a system dynamics model (foreground level) specific for urban forest was built. It was developed making use of SIMILE (https://www.simulistics.com/), a proprietary software for visual declarative modelling useful to build spatially-explicit and time-dependent system dynamics models.

5.3.1. Outputs of the urban forest model

The system dynamics model was developed to predict the performance of urban trees with respect to six ES, three types of biological waste generation (dead stem biomass, dead branch biomass and leaf litter), and six applied management actions. The inputs number of planted trees of each species and size are also used as outputs calculate monetary values.

Regarding ES, four regulating ES, a provisioning ES and a cultural ES are quantified:

i) regulation of chemical composition of the atmosphere;
ii) regulation of temperature and humidity;
iii) regulation of water flow and hydrological cycle;
iv) filtration of air pollutants by plants;
v) fibres and other materials from cultivated plants; and
vi) characteristics of living systems that enable activities promoting health.

In terms of applied management actions, the following are quantified:

i) pruning;
ii) replanting;
iii) irrigation; and
iv) collection of biological waste.

The collection of biological waste is implicitly calculated as part of the modelling of biological waste. Depending on the management actions applied to each cell, dead wood and leaf litter can be either collected (and in what percentage) or left to decompose via the soil sub-module. Only the collected dead wood and leaf litter become biological waste. In addition, quantifying biological waste permits the calculation of waste disposal and waste treatment in the background level. As part of the decisions taken in the background level, it is possible to define via a dedicated parameter the proportion of the biological waste (i.e. dead wood) that could be re-used and hence accounted as a provisioning ES. Figure 5.3 summarises all the above outputs, the indicators used to represent them, and their link to positive and negative environmental impacts and externalities. Default monetary values for the conversion of impacts to externalities are provided in Annex 5.2, based on the set of indicators already presented in Chapter 3.

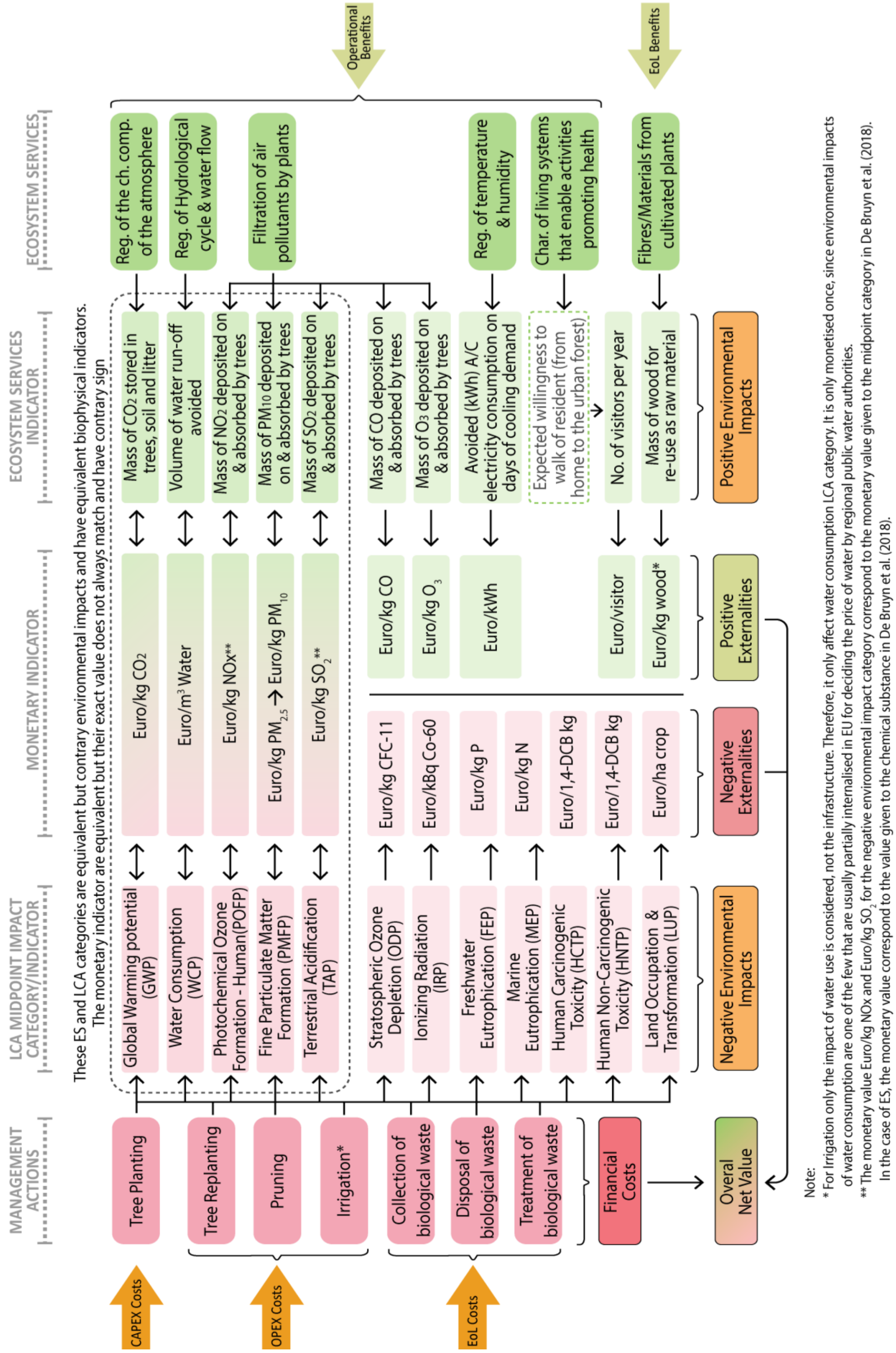

Figure 5.3. Ecosystem Services and Management Actions considered in the urban forest model, their interrelation and their conversion to negative and positive environmental impacts, financial costs and positive and negative externalities. All the costs (outputs), except irrigation (only contributes to Water Consumption), contribute to environmental impacts of each of the LCA midpoint impact categories included.

5.3.2. Spatial, temporal and thematic dimensions of the system dynamics model

As described in Section 5.2, the system dynamics model was defined with a default temporal extent of 50 years and it performs based on three temporal resolutions (daily, monthly, yearly). For the spatial resolution of each cell, it was considered that urban forests usually have a low tree density. As an example, from the 26 green open space sampled by Cariñanos et al. (2017) as representative of Spanish green open spaces, none of them surpass a tree density of 25 m^2/tree. In addition, the average crown width for many adult tree species is between 5 and 10 meters (see a repesentative list of common urban tree species in Chanes and Castano (1969)). Consequently, the cells are defined by default at a resolution of 10x10m and it was assumed that each cell could contain from one to four trees. Since the model needs to be spatially homogeneous in attributes, if more than one tree is defined inside a cell it needs to be of the same species and age. Temporal extent and spatial resolution can be adjusted on a case by case basis if needed.

Concerning thematic extent, categorical variations are included for the following attributes: climate, tree species, soil texture, soil cover, paving, irrigation, pruning, and intensity of biological waste removal.

Categorical variations for climate are defined making use of the Köppen-Geiger climate classes (Kottek *et al.*, 2006). Different climate classes influence the tree species parametrisation. Trees of the same species planted in a different climate differ in the value of parameters and in some cases in their growth equation. Thirty-one tree species were parametrised for all the socio-ecological processes. The twelve soil textures defined in the standard soil texture triangle were parametrised. For soil cover two options are possible, grass or bare soil, both of them influence litter decomposition. For paving, the model differentiates between non-paved ground and paved ground. Paved ground is defined as fully impermeable and does not differentiate types of pavement.

Concerning the management attributes, pruning and irrigation are optional, hence they can be activated or excluded from the model. When activated, irrigation only occurs when the soil water balance approaches the wilting point. The model assumes that water loss in the irrigation infrastructure and in the soil do not occur, being the plant able to take profit of all the water irrigated. If pruning is activated, it occurs by default at three or five year intervals, which is considered the optimal pruning frequency for most species (Mcpherson, Kendall and Albers, 2015). Specifically, only “safety pruning” is modelled, i.e. removal of broken branches and structural flaws (Mcpherson, Kendall and Albers, 2015), which should correspond to a removal of about 10% of the crown. However, a lower level of crown pruning is assumed, which is set at 5% removal, because the model already accounts for natural loss of death of branches (corresponding to an annual 1% of crown loss). The intensity of biological waste removal is considered in four intervals: no removal (0% litter), partial removal (25%), intense removal (50%), complete removal (100%). By default, complete removal of waste is applied to trees in paved cells because they usually represent zones with intense pedestrian movement. Consequently, risk of branch and tree falling should be completely avoided as well as nuisances and obstruction to movement derived from litter.

5.3.3. Description of the main structure of the system dynamics model and the modelled processes

As described in Section 2, the system dynamics model is composed of four modules (Figure 5.4) that interact with each other to generate the outputs. The key aspects of the Output module and the NBS Inputs module (the thematic extent and resolution) have been described in Section 3.1 and 3.2 respectively. In this section, the Atmosphere module and the NBS cells module are described. For a complete documentation of the model or its detailed visual declarative representation in SIMILE see Annex 5.3.

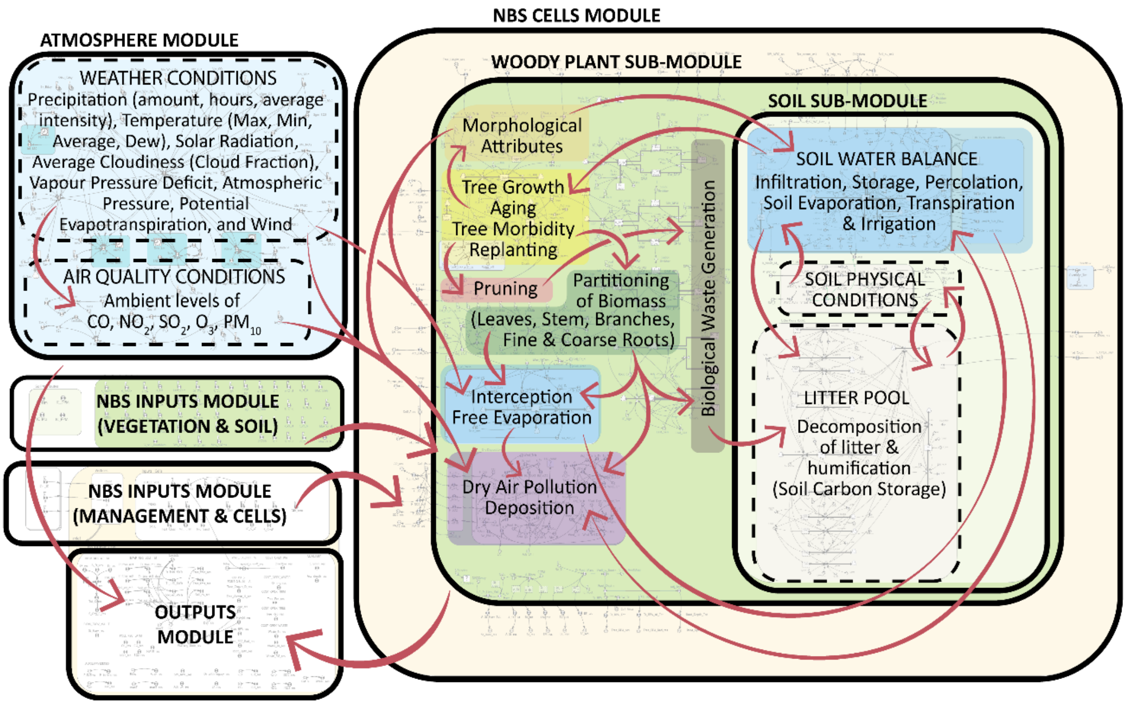

Figure 5.4. Current urban forest model representation in SIMILE showing the modules NBS Cells, NBS Input, Atmosphere, and Outputs and the interaction between their processes.

5.3.3.1. Atmosphere module

The atmosphere module is a stochastic weather generator that simulates daily weather and air quality variables that are required for the socio-ecological processes modelled in the NBS cell module. This module requires daily historical data as inputs to compute the parameters defining the statistical distribution of the atmospheric variables (list of variables included in Weather Conditions in Figure 5.4). These parameters are then used by the daily weather generator. They could be adjusted to represent variations result of climate change scenarios or other type of scenario (e.g. progressive changes in vehicles engines leading to reduction on ambient levels of specific air pollutants). In terms of the historical baseline, to ensure an adequate accuracy, the historical data should be obtained from close monitoring stations or as a substitute from remote sensing data with enough spatial and temporal resolution. The generation of daily values representative of the local conditions are preferred to a direct use of historical data because i) behaviour of NBS is not compared always against the same values; ii) temporal extents can be adjusted more freely; and iii) modifications in statistical parameters used as inputs could permit a representation of future transitions in local weather and/or air quality conditions.

The simulation of precipitation, wind, and atmospheric pressure considers variations in the statistical distribution of variables at monthly level to acknowledge the seasonal changes over the year. Previous studies showed that air temperature and solar radiation of wet and dry days in the same month require many times an independent parametrisation of the statistical distribution (Nicks and Harp, 1980; Richardson, 1981). Consequently, besides monthly variations, the simulation of air temperature and solar radiation also considers variations in the statistical distribution of variables between rainy and non-rainy days inside each month. The occurrence and amount of precipitation and average wind speed is modelled according to the CLIGEN model of the USDA developed by Nicks (1975). The duration of the daily storm event is modelled according to the CLIGEN model, the documentation of the EPIC model (Sharpley and Williams, 1990) and Lobo *et al.* (2015). Air temperature (max, min, average and dew) and average solar radiation are modelled following the equations defined by Nicks (1975) and Nicks and Harp (1980). Average cloud fraction is modelled based on the adaptation of the Angstrom-Prescott model defined by Luo, Hamilton and Han (2010). Potential evapotranspiration is modelled according to the Hargreaves method, (Hargreaves and Samani, 1985), since it only requires air temperature as input data. Modelling evapotranspiration according to the Penman-Monteith equation is usually preferred. However, it requires a parametrisation of an excessive number of variables, including woody plant attributes, that was not possible to characterise for a long temporal extension. Vapour pressure deficit is modelled using the method described in the FAO guidelines (Allen *et al.*, 1998).

The modelling of air pollutant ambient level is also generated with stochastic simulation following the same logic than for weather variables. Several authors have illustrated that independently of the specific urban context and the average temporal resolution used common air pollutants such as CO, SO, O_3, NO_2 and PM_{10} follow log normal frequency distributions (Larsen, 1971; Bencala and Seinfeld, 1976). Additionally, many scholars have shown that pollutant species variate over the year having periods of higher concentration in certain seasons (Fernández Jiménez, Climent-Font and Sánchez Antón, 2003; Salvador *et al.*, 2011; Kassomenos *et al.*, 2014). For example, SO_2, CO and NOx are higher in winter and O_3 is higher in summer. PM_{10} usually is higher in winter, but also during days of high atmospheric pressure. Instead, $PM_{2.5}$ tend to be higher in summer. In general, pollutants' concentration is influenced by mixing depth (influenced by air temperature) and wind speed. Moreover, some air pollutants, especially PM_{10} and $PM_{2.5}$ are lower during rainy days. Based on the above evidence, air pollutants are simulated stochastically according to a log-normal distribution differentiating variations in statistical distribution by month and by rainy or non-rainy day.

5.3.3.4. Woody plant module

The woody plant module simulates the following processes:

i) tree growth;
ii) the partition of the tree biomass in different compartments (stem, branches, foliage, coarse roots, and fine roots);
iii) changes in biophysical attributes (crown diameter, crown height, tree height, root area and root depth);
iv) tree evaporation;
v) the interception of rainfall by canopy;
vi) air pollutants removal (CO, SO_2, NO_2, O_3 and PM_{10});
vii) tree morbidity; and
viii) biological waste generation by each tree compartment naturally or as a result of management actions.

First, the module calculates the increase of the diameter at breast height (Dbh) per monthly time step making use of available allometric equations for trees species in urban areas. Most of the allometric equations Age-Dbh parametrised in the library of the NBS inputs module have been obtained from the Urban Tree Database (McPherson, van Doorn and Peper, 2016). Growth-reducing factors are applied to the standard radial growth rate to mimic the effect of drought (based on tree drought tolerance index of Ninements and Valladares (2006)) or paved conditions.

Once the Dbh by timestep is obtained, the model uses it as intermediate input to calculate the dry biomass stocks of each compartment (i.e. stem, branches, roots, foliage biomass and leaf area) based on the allometric equations of Forrester *et al.* (2017) and McPherson, van Doorn and Peper (2016). The modelling of biomass growth is needed to compute tree carbon storage and biological waste generation. Biophysical attributes such as tree height, crown radius and crown height are also modelled mainly through allometric equations from the Urban Tree Database. Crown radius will permit to know the canopy area per cell, needed to calculate leaf area index (LAI). The tree root area is modelled by assuming it equivalent to the root protection area as defined by the British Standards 5837:2012 (British Standards, 2012). Tree root area and root depth, based on (Soares and Almeida, 2001) defines how much of the water available in soil is directly available to trees.

As described in Chapter 3 - Section 3.3.4.2, to track the evolution of tree height over time is necessary since it is used as a parameter proxy of the perceived maturity of a tree (following the approach of Filyushkina *et al.* (2017)), and consequently the urban forest when extrapolated to all the wooded cells representing the NBS. In this sense, following the definition of forests by FAO (FAO, 2000), forest trees should have a height of 5 metres and the wooded area should cover a minimum of 0.5 ha and have a minimum width of 20 meters. In the urban forest model, for wooded areas with average tree height above 5 metres the average willingness to walk (in minutes) estimated by Ta, Tardieu and Levrel (2020) for Paris is assumed. When available, this value could be substituted by more accurate values obtained from local studies. The willingness to walk increase linearly with height up to 5 meters, beyond this height willingness to walk does not change. The minimum value given to the smallest tree sapling that can be planted (i.e. one year old) is equivalent to the baseline value for green open spaces with no trees that was also estimated by Ta, Tardieu and Levrel (2020). Once the willingness to walk (in minutes) for the entire NBS intervention is estimated, the number of visitors per year can be extrapolated in the background level applying least-cost path (in a similar way than applied to animals in Chapter 4) and obtaining the amount of street network, and therefore residents, serviced by the urban forest (see illustrative example in the Figure 5.10 of the case study of this Chapter)

Leaf area per time step is used to calculate LAI per canopy area, which together with the ratio of cell covered by the tree canopy is used to model daily canopy interception and free tree evaporation. To use LAI per canopy area instead of LAI per cell area allows modelling more accurately socio-ecological processes controlled by thresholds when trees are very small and when their crowns are very large and overpass the area of the cells. LAI and the soil water balance are used to estimate daily tree transpiration and soil evaporation based on SWAT model equations (Neitsch *et al.*, 2011). LAI together with tree height, tree transpiration, and most of the variables of the atmosphere module are used to calculate dry deposition of the different pollutants based on i-Tree ECO equations adjusted to daily time steps instead of hourly time steps (Hirabayashi, 2013, 2016).

All the ES flows are constrained by tree mortality, which is simulated through stochastic equations to account for the variation of this ecological process. It would be difficult to treat mortality as a deterministic process, especially since the causes of death, their importance, and interrelation are still not fully understood. The default death probability included in the model relies on the tree death statistics of Nowak, Kuroda and Crane (2004) for Baltimore[8], which are related to the different Dbh (age) of trees. Additionally, probability of death increases beyond the default probability depending on three stressors (i.e. drought, waterlogging, and paved conditions). These stressors are among the most relevant for growth and death for which roughly quantitative characterisations are discussed in the scientific

[8] The parametrisation of the stochastic equation could be adjusted according to local statistics of tree death if those are available.

literature (Nowak, Kuroda and Crane, 2004; Roman and Scatena, 2011; Koeser *et al.*, 2014; Roman, Battles and McBride, 2014; Ko *et al.*, 2015b, 2015a; Y. Chen *et al.*, 2017). Moreover, the stochastic equation always includes a residual probability of death even if none stressors occur, to account for unobserved and exogenous stressors (e.g. pests) influencing death. The use of stochastic equations for the assessment of NBS requires to replicate simulations several times to ensure that the mean and standard deviation of each output are representative of the range of potential values.

Plant litter generation is calculated making use of the biomass per compartment, which permits differentiation of the type of waste. Stem residues are only produced once a tree dies. Branch residues comprise dead branches, pruned branches, or all the branches if a tree dies. Leaf litter is calculated making use of leaf turnover rates characterised for evergreen broadleaves, deciduous broadleaves, and coniferous in BIOME-BGC model (White *et al.*, 2000). In the case of tree death, all the foliage becomes plant litter. Roots are split in fine roots and coarse roots following White *et al.* (2000). Fine roots turnover is modelled as in BIOME-BGC and coarse roots turnover only occurs if tree dies. Plant litter produced in the roots compartment serve as input in the organic matter decomposition process simulated in the soil module and cannot be accounted as biological waste removed from site by management actions. All residues are split into a decomposable fraction (more easily degradable) and a resistant fraction, according to the parametrisation described in Shirato and Yokozawa (2006). If residues are not collected as biological waste, they become inputs of the soil module to calculate soil carbon storage.

5.3.3.5. Soil module

The soil module simulates the following processes, which are needed to estimate the soil water balance:

i) litter decomposition and organic carbon retained in the soil;
ii) infiltration of throughfall (i.e. net rainfall after interception) and associated overflow;
iii) percolation;
iv) soil water movement due to water pressure gradients;
v) soil evaporation;
vi) tree transpiration; and
vii) vii) irrigation. Modelling all the above processes.

The soil module assumes a maximum root depth of two meters, and it is split by default in eight horizons of 25 cm depth. Litter decomposition and additional organic carbon retention is only modelled in the topsoil horizon. The soil is split in horizons to permit a more accurate modelling of the available water for plants over their growth. The depth of each horizon and its specific soil physical conditions can be adjusted on a case by case basis.

The maximum root depth is defined based on the work of Crow (2005). He indicates that usually tree roots, especially in artificialized environments, do not penetrate a depth greater than 2 meters. The ones that go more in-depth tend to correspond with species (e.g. *Quercus robur*) that maintain the growth of their tap root (i.e. primary vertical root) at adulthood. Crow (2005) also illustrates that in general 80-90% of the roots tend to be within the 60 cm of soil, and that a 90-99% stay in the first meter of soil. Day *et al* (2010) also describes that in most cases roots are mostly concentrated in the upper 30cm of soil, since they tend to exploit more the upper horizons of soil. Despite it is a generalisation, for the purpose of this proof of concept model two meters of maximum root depth is assumed adequate for all tree species.

The soil physical conditions are defined making use of the equation of Saxton and Rawls (2006), which model the relationship between soil texture (including organic matter), field capacity, wilting point, water content at saturation and saturated hydraulic conductivity. Default values for bulk density are defined per soil texture class making use of the values proposed by USDA for optimal growth of plants (USDA, 2020). Consequently, soil compaction is disregarded as a stressor in the model even if it widely occurs in urban green open spaces.

Organic carbon retention is modelled at monthly time steps. Four compartments are used (decomposable plant material, resistant plant material, humus, and microbial biomass), with equations adapted from the RothC model for agricultural soils (Coleman and D. S. Jenkinson, 2014). The decomposable plant material and resistant plant material obtained from the tree model, once converted into carbon, and the initial amount of humus and microbial biomass per cell are used as inputs for the soil module to calculate organic carbon retention. Each fraction of the plant residues decay becoming carbon microbiota, carbon humus, and emitting atmospheric carbon. In all the compartments litter decomposition decay is influenced by the amount of organic matter, the conversion proportions between compartments, the intrinsic decomposition rate constants of each compartment, and the rate modifying factors for temperature, soil moisture and soil cover as established in the original RothC model. The total organic matter present in the model in each time step is calculated as the sum of the organic matter in all the compartments. For changes in the soil physical conditions (soil texture) over time, only the proportion of humified organic carbon is considered and up to a maximum value of an 8 %, because higher values in agricultural or urban soils are highly unlikely.

The infiltration of throughfall is modelled at daily time steps considering the soil texture of each soil horizon, which influences water content at saturation (and consequently maximum water storage). Infiltration is based on a pre-calculated infiltration table at hourly level developed outside the model making use of the Green-Ampt method as described in SWMM (Rossman and Huber, 2016). This approach for estimating infiltration is used because it is simple enough, and already widely used in practice since it represents the real process well enough real infiltration rates (Kale and Sahoo, 2011).Its use in the model at daily time steps required to know the values of storm duration, and associated average intensity of daily precipitation, which are calculated in the atmosphere module. The precalculated table is obtained for all the possible changes in the soil of the percentage of sand, clay and silt on a 5% interval. Percolation only occurs when field capacity is overpassed, and it is modelled according to SWAT (Neitsch *et al.*, 2011). Additionally, soil water upward and downward movement to equilibrate water pressure gradients between adjacent horizons are modelled making use of the Darcy's equation as described in the work of Soares and Almeida (2001). The soil water balance is completed with the modelling of tree transpiration and soil evaporation described in the woody plant module. When irrigation is activated as a management action, it only occurs when the available soil water is below a 20% of its potential capacity and it is equivalent to the transpiration demanded by the tree.

As visually illustrated in Figure 5.4, the four modules and compartments described above interact with each other for the simulation of processes and generation of outputs. This proof of concept of the urban forest model is already operational and has been tested in the case study presented in the following section. Additionally, in Chapter 6, the urban forest model is also used to exemplify how the conceptual modelling framework could become a simple to use decision support system.

5.4. Application of the urban forest model to an illustrative case study

The model was applied to the urban forest of La Mancha in the west phase of Valdebebas Park (Figure 5.5a). This green open space is part of a new urban development close to Barajas airport (Madrid, Spain), and the west phase is the only currently developed. Valdebebas Park covers an area of around 140 ha, of which 17.33 ha correspond to La Mancha.

Six scenarios were considered, and their cost and benefits assessed over 50 years. The performance of five types of NBS cells (with specific combination of tree species, tree age at planting, and soil texture) that define the urban forest of La Mancha was also assessed at individual level. The five types were compared against four alternative (hypothetic) versions developed on paved ground. The six scenarios help to illustrate the use of the model for assessing the cumulative long-term cost and benefits of urban forests. The specific types of urban forest cells help to illustrate the use of the model for assessing the performance of urban forests against shocks (specific events) and stresses in specific periods over the simulacion horizon. The default spatial resolution of the urban forest model was used (10x10 metres). Ecological connectivity assessment is not included as part of the background assessments. Since there are stochastic equations in the model, more than one replication of the simulations is necessary. For this illustrative exercise, 50 replications of the simulations were considered sufficient to ensure the convergence of the stochastic variable. In Chapter 6, a similar exercise is repeated with 75 replications, where for each new replication successive difference in the mean and standard deviation of values is assessed to confirm if these are enough to be representative of the potential range of values.

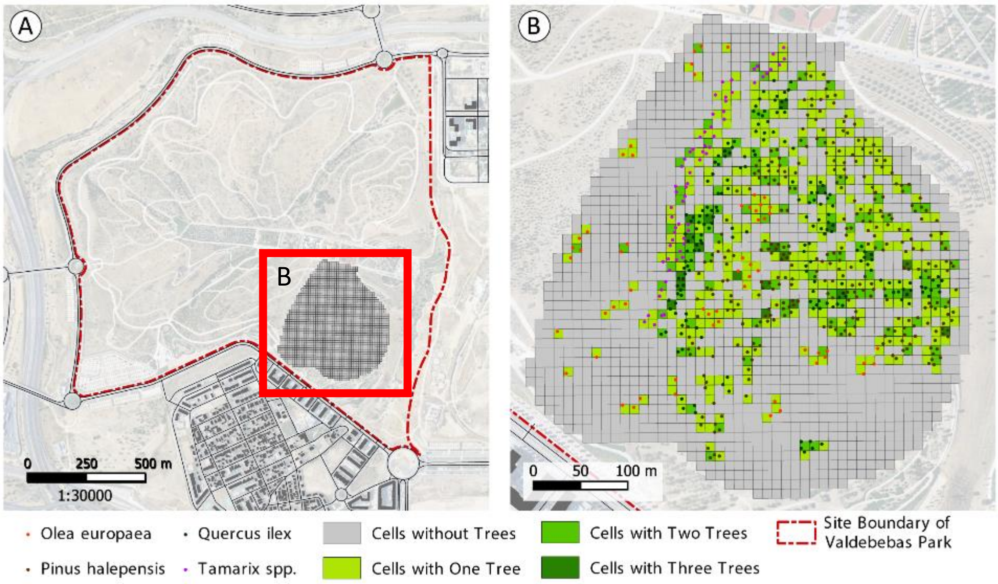

Figure 5.5. A) Site boundary of Valdebebas Park with the zone La Mancha mapped; B) Zoom of La Mancha showing cells including trees the distribution of tree species.

5.4.1. Preparation of the input data

The historical data on meteorological conditions were obtained from the monitoring station of Barajas airport. The historical data on air pollutants were measured by the two closest monitoring stations to Valdebebas Park (*Sanchinarro* and *Barajas Pueblo*), both classified as urban background monitoring station (i.e. located far away from high traffic areas). Both sets of data were used to calculate the statistical parameters required as input by the Atmosphere module. Despite the atmosphere module was developed as a weather generator to be able to simulate daily weather conditions beyond the historical ones, for the purpose of this exercise it was not parametrised according to climate change scenarios. Therefore, daily weather conditions generated emulate historical weather conditions.

The tree and soil input data for the model were extracted from standard urban plans and reports provided by the Council of Madrid and prepared before the construction phase of the park. This documentation included: i) the latest version of the landscape plan; ii) an associated georeferenced map showing the location of the tree species; iii) a soil survey; iv) and a disaggregated bill of quantities with the financial costs.

From the georeferenced map, only the 484 cells with tree occurrence were selected for the modelling (Figure 5.5b). The urban forest of La Mancha is composed of five tree species (*Olea europaea, Quercus ilex, Pinus halepensis, Tamarix bobeana* and *Tamarix canariensis)* always planted in non-paved ground and with the soil covered by grass. For all the allometric equations, equations developed for the same species and equivalent climate class were prioritised. When equations for the same climate class were not available, equations from the most similar classes were used as proxies. In some cases (e.g. Tamarix), equations were not at species level, and equations for a close species of the same genus were used as a substituted, modelling tree growth at a genus level. Except for Tamarix species, the allometric equations to model Age-Dbh relationship are extracted from the Urban Tree Database (McPherson, van Doorn and Peper, 2016). For Quercus ilex, the allometric equations to model biomass growth of different compartments were also extracted from the Urban Tree Database. For the rest of the species, biomass growth allometric equations were obtained from the scientific literature. For *Pinus halepensis,* equations were obtained from Spanish local studies (López-Serrano *et al.*, 2000; Montero, Ruiz-peinado and Muñoz, 2005). For *Olea europaea*, the equations were obtained from Italian local studies (Brunori *et al.*, 2017). *Tamarix bobeana* and *Tamarix canariensis* were modelled at genus level, *Tamarix spp*. Their allometric equation to model Age-Dbh relationship was obtained from a study of *Tamarix ramosissima* in Nevada (USA) (Haigh, 1998). Their equations for biomass growth were also based on studies of *Tamarix ramosissima* in arid and semi-arid conditions in the USA (Sala, Smith and Devitt, 1996; Wei *et al.*, 2012)

The results from the soil survey were used to calculate the specific soil texture (i.e. percentage of sand and percentage of clay in the soil) and initial organic matter content per cell. The results were corresponding only to the soil texture in the first meter of soil. As a simplification for this exercise, and due to the lack of data, the values for the soil texture reported were used for the entire soil profile. Inverse distance weighing was used to extrapolate values for the entire area from the point measurements of the soil survey. This is not usually recommended for large areas in regional studies, or where there are abrupt changes in key abiotic attributes of soil formation (e.g. lithology). However, it was considered adequate for the illustrative purpose of this exercise because of the short distance between measurements, their very similar values, and the lack of enough data points to apply kriging[9].

Regarding landscape management, the landscape plan anticipates irrigation over several years after tree planting (it was assumed up to 10 years after planting), after this period irrigation is not applied. It is expected that during the first 10 years dead trees are replanted each year, and after 10 years each three years. Trees are replanted with the same species and at the same age at which they were planted for the first time. In the documentation provided there was no reference to other management actions such as pruning, removal of trees or collection of biological waste (i.e. leaf litter dead stem wood and dead branches). Thus, different options for these management actions were considered in the simulated scenarios.

In terms of financial costs, the unit prices for the supply of trees (from nurseries), their transport and planting were obtained from the disaggregated bill of quantities provided by

[9] Kriging is a geostatistical interpolation method that acknowledges distance and degree of variation between measured data points to estimate values in the area between measurements (Paramasivam and Venkatramanan, 2019).

the Council of Madrid. The costs from the disposal of biological waste and its treatment were calculated making use of historical costs from 2017 to 2018 obtained from the last strategy of residues of Madrid and the Annual Reports (from 2009 to 2018) of the main waste treatment plant of Madrid (Madrid City Council, 2018; Parque Tecnológico de Valdemingomez, 2018). The rest of the financial costs are calculated making use of two national construction cost databases (Base Paisajismo, 2019; IVE, 2019). The specific unit prices used for financial costs are included in Annex 5.2.

To monetise positive and negative environmental impacts as externalities the default values of the urban forest model were used (see Annex 5.2), since local monetisation studies were not available. Therefore, value transfer data was used as illustrated in Petucco *et al.* (2018). Since the original sources of monetisation are variegated and values are given in Euros referred to different base years, all the monetary values were adjusted to a common base (i.e. Euro 2018 for EU-28). The values were corrected by inflation (using the GDP deflector data provided by the World Bank), purchasing power parity (using the PPP exchange rates computed by the World Bank) and, when necessary, by the average income (based on the GDP per capita data in 2010 USD from the World Bank database) to adjust the willingness to pay for an ES to the local economic conditions. A unitary income elasticity of Willingness-To-Pay (WTP) was used as suggested in Tyllianakis and Skuras (2016). For this exercise, it was assumed that prices and costs are constant over time, which may not be the case in the reality. However, the development of dynamic price models was outside the scope of testing the urban forest model. Discounting is not considered in this exercise.

5.4.2. Scenarios and types of urban forest cells

The six scenarios considered for testing the urban forest model are summarised in Figure 5.6a. These scenarios were defined based on differences in design/planning actions (implementation phase), management actions (operational phase) and the management of biological waste (end-of-life phase).

In terms of design/planning and management actions, two alternatives were considered:

- Current La Mancha. This alternative seemingly corresponds to the implementation of the case study described in the documentation provided by the Council of Madrid (Figure 5.6 a). Few modifications were required to adapt the current capacities of the model to the case study. For example, *Tamarix boveana* and *Tamarix canariensis* needed to be modelled at genus level (*Tamarix spp*). As another example, shrubs and individual herbaceous plants could not be considered. The case study is described in the documentation as a naturalistic urban forest. Thus, no pruning is expected, and removal of trees and biological waste should only occur when trees die.

· Paved La Mancha. In this hypothetic case the urban forest only includes a monoculture of Quercus ilex planted at 2 years old on paved ground (Figure 5.6a). The density of trees corresponds to only one tree per cell. Security pruning (i.e. pruning to avoid the risk of branch falling on people) is expected each five years and leaf litter and branch litter should be always collected. In fact, this hypothetic case study where trees are placed at very low density, on paved ground and with a more intensive management emulates street trees.

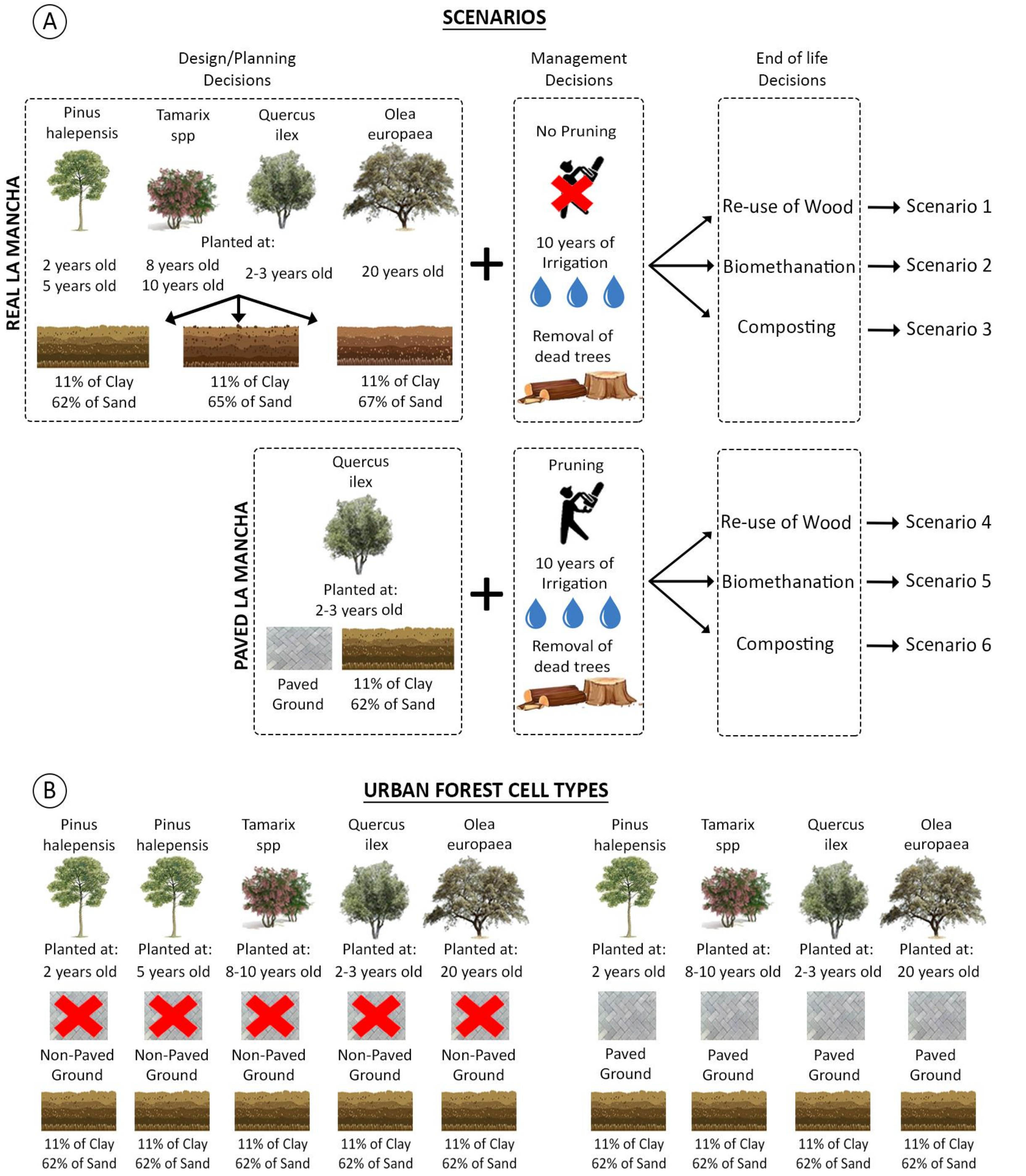

Figure 5.6. A) Graphical summary of the six different scenarios assessed; B) graphical summary of the nine different urban forest cell types assessed.

In terms of management of the biological waste, three options (i.e. composting, biomethanation, and re-utilisation of dead wood as raw material) were considered for each of the two design/planning alternatives, which makes six scenarios. For composting and biomethanation, it was assumed that all the biological waste is transported to Valdemingomez waste treatment complex, as it currently occurs with the biological waste generated in Madrid. For the re-utilisation of dead wood, it was assumed that dead stem wood could be used as raw material for lumber wood, chipped dead branch wood as raw material for woodchips, and that leaf litter is treated through biomethanation. This last alternative considered the transport from the case study to sawmills and panel board industries (based on average distances in Madrid) as the last process of the end of life.

The input characteristics of the nine types of urban forest cell assessed at individual level are summarised in Figure 5.6b.These urban forest cell types level represent the main variations in tree species and tree size (age) at planting present in the design/planning alternative of Current La Mancha and the versions of them planted on paved ground. Originally, existing variations in soil texture were also considered relevant to define the specific urban forest cell types assessed. However, the variations in soil texture present in Current La Mancha respond only to small variations in the percentage of sand that were only providing minimal changes in specific outputs. Thus, for simplicity all the urban forest cell types have the exact same soil texture.

5.5. Results

5.5.1. Performance of the urban forest cell types against shocks and stresses

As introduced in Section 1, the supply and demand of ES change over time. In some cases, e.g. regulating ES, these changes could be very relevant for the actual benefits derived from ES flows and therefore the performance of NBS against sudden shocks or chronic stresses. This section illustrates the use of the urban forest model to predict changes in the average daily supply of the ES *filtration by plants* per month over time (Figure 5.7). It also illustrates the differences over time in tree transpiration, a process contributing to the ES *regulation of temperature and humidity,* when there is no-demand and when there is demand for this ES (Figure 5.8). At the end, it is presented how tree mortality might influence the long-term supply of a cultural ES such as *characteristics of living systems that enable activities promoting health* (Figure 5.9) as well as the recreational benefits derived from it.

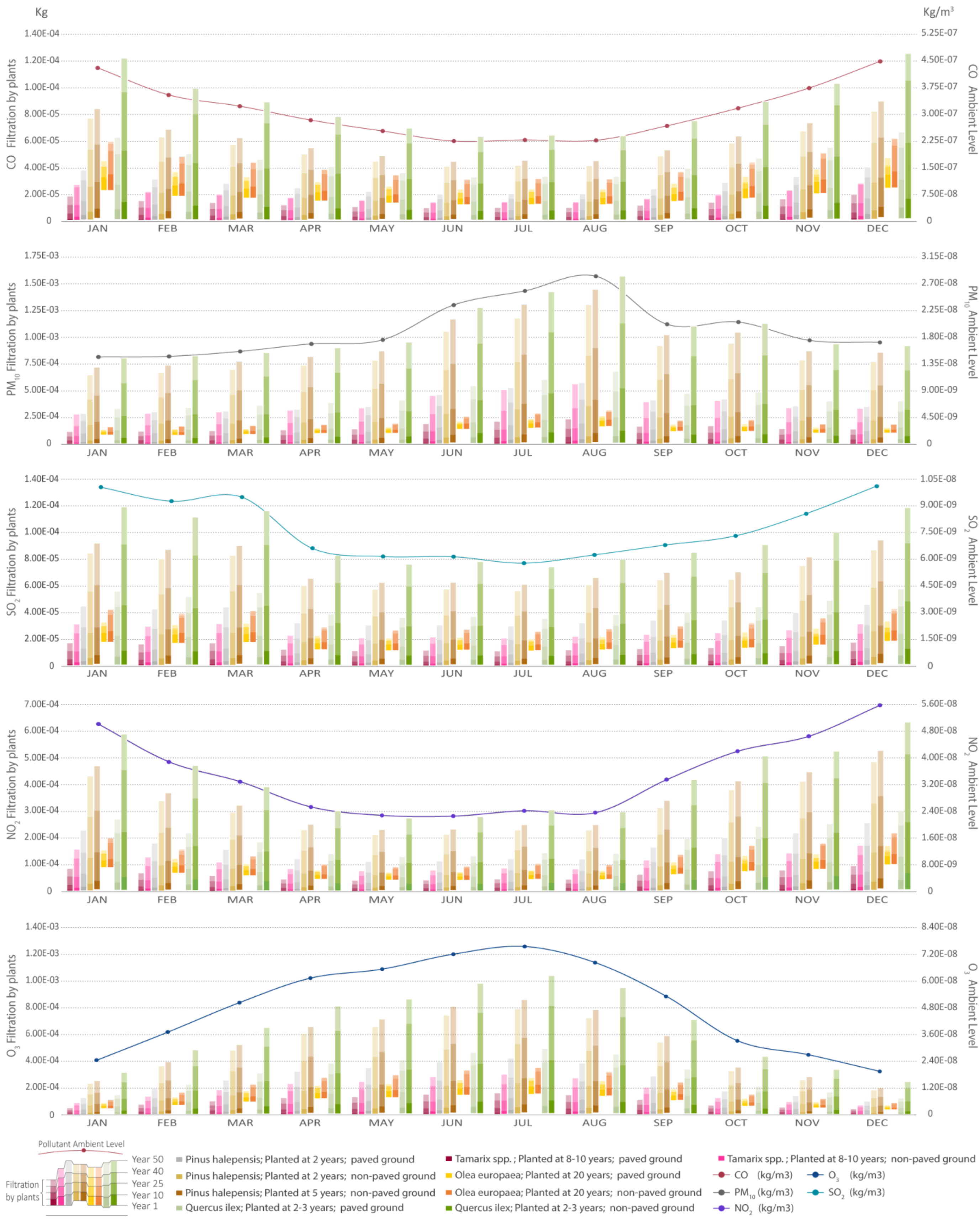

Figure 5.7. Evolution over 50 years of average daily filtration of air pollutants per month visualised against average monthly air pollutant ambient levels (Visual overlapping between lines of Ambient Level *and bars of* Filtration by plants *do not have a quantitative meaning).*

Regarding average daily *filtration by plants* of air pollutants (CO, PM_{10}, SO_2, NO_2 and O_3), Figure 5.7 clearly shows that after 25 years of the urban forest implementation, *Quercus ilex* planted around 2-3 years on non-paved ground performs better than the rest for all the pollutants in each of the months of the year. During the first 10 years, *Olea europaea* (non-paved and paved) are the best performers due to their already mature condition, and much higher leaf area. Between 10 and 25 years, also *Pinus halepensis* planted at 2 and 5 years old on non-paved ground overcomes both *Olea europaea* in performance. For some substances, the differences over time between *Quercus ilex* non-paved and *Pinus halepensis* non-paved are not relevant and might not imply significant differences in their capacity to mitigate air pollution. For example, for both urban forest cell types daily filtration of PM_{10} on summer months is quite similar. These are the months when atmospheric PM_{10} ambient level is higher in Madrid (see Figure 5.7) and in few occasions the recommended threshold of $5 \cdot 10^{-8}$ µg/m^3 (World Health Organisation, 2005) is overpassed. However, for the rest of the urban forest cell types the differences are substantial. Therefore, since Madrid overpasses the maximum ambient levels of NO_2 and O_3 permitted (Air Quality Directive 2080/50/EC) a few times each year favouring a greater planting of one of the lowest performers in the city might not be ideal to mitigate air pollution problematics

Concerning *regulation of temperature and humidity* for all the cell types, Figure 5.8 illustrates a clear difference between the average daily transpiration when air temperature is below the thresholds of comfort, and when they are overpassed, and the ES is demanded. In the latter case, water is also scarce for trees since the water soil balance is low due to the increased soil evaporation and tree transpiration demand. This is a consequence of continuous high temperatures and a low occurrence of rain in Madrid, on average around 2 days per month in July and August. Figure 5.8 also show differences between specific urban forest cell types. For example, after 25 years *Quercus ilex* on non-paved ground appears as the best performers for average and highest daily transpiration. However, when looking at demand it is not always the best performer. In fact, during demand periods *Olea europaea* on non-paved ground performs like *Quercus ilex* after 25 years and at 40 and 50 years clearly outperform it in average and highest transpiration. This can be explained by the low maximum transpiration of this species as indicated in Fernández and Moreno (2008). Regarding differences between paved and non-paved cells, transpiration in paved versions is much lower, being close to zero. Despite that lower transpiration could be expected in paved cells due to less access to water, their values are likely and will be discussed further in Section 5.6.

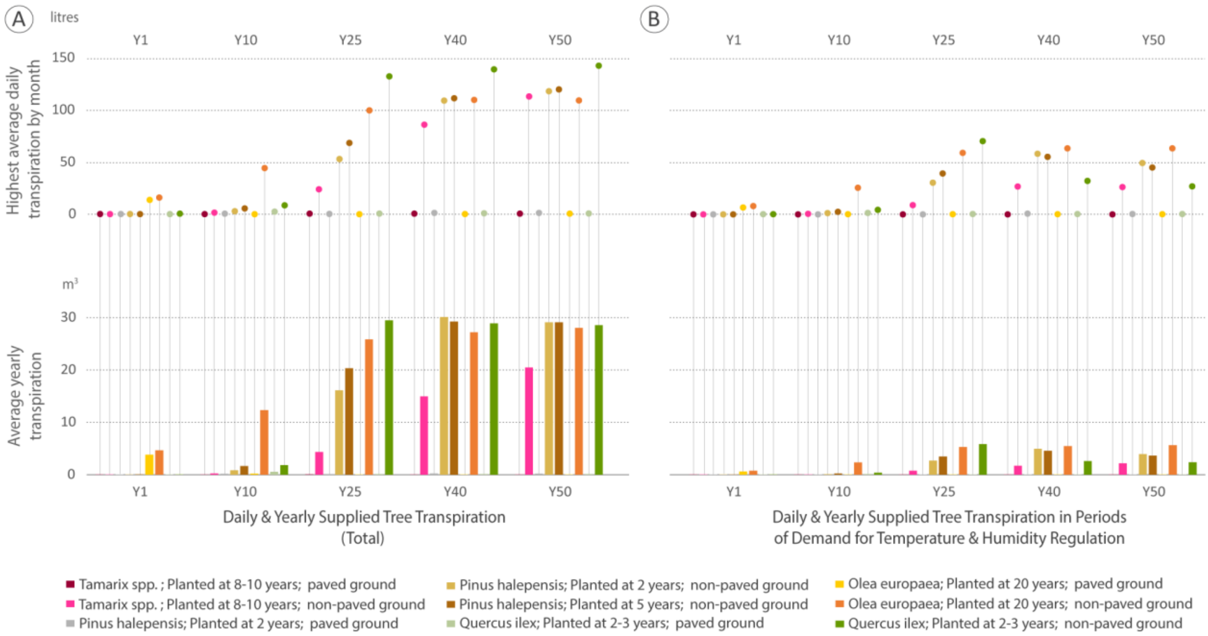

Figure 5.8. a) Evolution over 50 years of average daily and yearly total supplied tree transpiration b) Evolution over 50 years of daily and yearly supplied tree transpiration when society demands it (i.e. when maximum and average daily air temperature overpass the

With respect to the expected willingness to walk, Figure 5.9 shows that an increased cumulative mortality could end reducing it in the long term. This can be observed for *Quercus ilex* paved and *Olea europaea* paved for which expected willingness to walk is slightly reduced after 25 and 40 years. Their increased cumulative mortality implies that dead individuals are substituted by younger ones (same age as first time planted) or are not present for few years, which could end making the urban forest look younger (smaller) and less attractive to users that need to walk further distances.

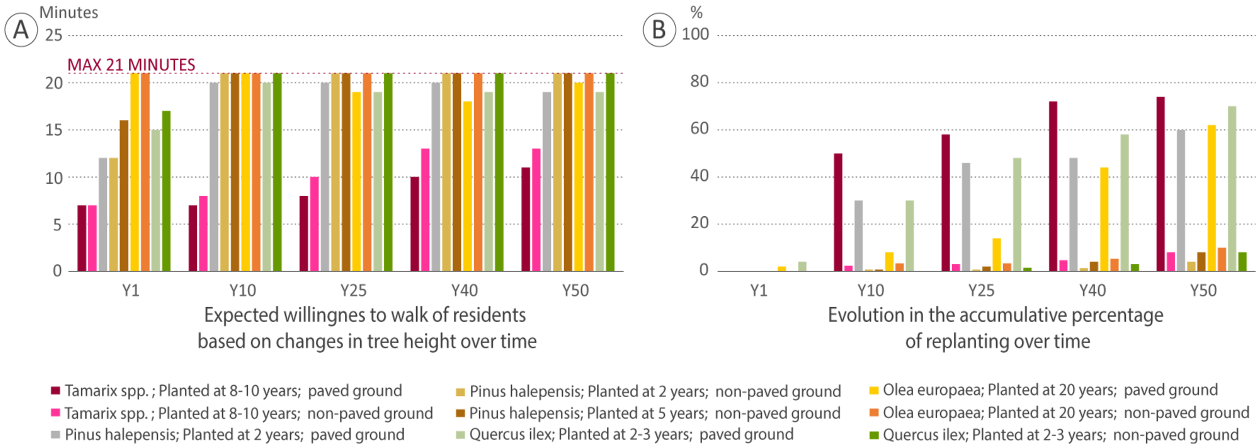

Figure 5.9. a) Evolution over 50 years in the expected willingness to walk of residents to visit urban forests only formed by each cell type as result of changes in tree height; b) Evolution over 50 years in the expected accumulative percentage of tree replanting

5.5.2. Cumulative long-term performance cost-efficiency of La Mancha scenarios

In terms of positive environmental impacts derived from ES flows, scenarios 1 to 3 (S1 to S3) perform much better than scenarios 4 to 6 (S4 to S6) (Table 5.1a). Due to their non-paved condition, S1 to S3 avoid much more water run-off than S4 to S6. As another example, S4 to S6 filtrate much less air pollutants due to the lower number of trees, canopy growth (and their associated leaf area) and transpiration rates. The latter process influences the canopy stomatal resistance for deposition of air pollutants (Baldocchi, Hicks and Camara, 1987). S4 to S6 perform slightly better on *characteristics of living systems that enable activities promoting health* than S1 to S3 because of their higher values for willingness to walk during years 1 to 10 and 29 to 34 (details on Annex 5.4). Consequently, during several years S1 to S3 have a lower potential number of visitors (Figure 5.10 show how potential visitors are obtained based on willingness to walk values). S4 also performs better in the use of *fibres from cultivated plants*, what is expected because only S1 and S4 consider the re-use of biological waste as raw material, and only in S4 all the litter and dead wood generated is collected.

The monetarisation of positive environmental impacts (Table 5.1b) informs on the most relevant ES in terms of societal benefits. It shows clearly that *characteristics of living systems that enable activities promoting health* are the most relevant ES by two orders of magnitude. It also emphasises the higher value of *regulation of hydrological cycle and water flow, regulation of chemical condition of the atmosphere*, *regulation of temperature and humidity* and *filtration by plants* of PM_{10} compared to the rest of the regulating ES.

Table 5.1. . A) Evolution over 50 years of cumulative positive environmental impacts in the form of ecosystem services provided by the urban forest; B) Evolution over 50 years of cumulative positive externalities provided by the urban forest (Values in Euro 2018)- S = Scenario.

A) CUMULATIVE POSITIVE ENVIRONMENTAL IMPACTS

Positive Environmental Impacts (Biophysical Units)	Units	Year 1				Year 10				Year 25				Year 40				Year 50			
		S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6
Regulation of hydrological cycle & water flow	1000 m³ avoided run-off	16.99		0.2		172.26		2.64		430.15		10.74		686.85		25.94		859.72		39.69	
Filtration of pollutants by plants	kg CO filtrated	0.57		0.18		12.11		6.33		71.71		43.06		198.84		123.47		322.66		196.4	
	kg NO_2 filtrated	2.14		0.7		45.76		24.29		289.61		167.89		865.86		498.48		1481.52		811.45	
	kg SO_2 filtrated	0.49		0.16		10.46		5.53		65.73		38.26		195.19		113.28		332.32		184.18	
	kg O_3 filtrated	3.24		1.05		70.07		37.02		444.42		257.17		1331.73		764.49		2280.74		1246.57	
	kg PM_{10} filtrated	3.83		1.36		84.47		45.76		586.27		319.51		1906.16		989.34		3422.06		1657.25	
Regulation of chemical condition of the atmosphere	t CO_2 stored in trees, litter & soil	52.3		5.92		239.13		83.66		1070.06		451.16		2662		1193.79		3959.49		1751.25	
Regulation of temperature & humidity	1000 kWh A/C Avoided	32.23		3.26		361.16		35.5		1210.76		143.03		2401.53		334.82		3248.59		505.58	
Characteristics of living systems that enable activities promoting health*	Willingness to walk (min.)*	15		16		20		20		21		19		21		19		21		20	
	1000 Potential Visitors/ year	8.28		8.43		143.02		218.66		498.86		505.5		854.71		892.72		1091.94		1110.12	
Fibres & other materials from cultivated plants**	t lumber wood from dead trees	0.0007	-	0.07	-	0.17	-	3.55	-	0.32	-	33.82	-	1.07	-	94.33	-	2.1	-	191.2	-
	t woodchips from dead trees	0.0004	-	0.04	-	0.15	-	2.75	-	0.24	-	30.79	-	0.75	-	103.18	-	1.5	-	196.49	-

B) CUMULATIVE POSITIVE EXTERNALITIES

Positive Externalities (Monetary Units)	Units	Year 1				Year 10				Year 25				Year 40				Year 50			
		S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6
Regulation of hydrological cycle & water flow	Euro	6,041		70		61,255		939		152,962		3,818		244,244		9,226		305,716		14,115	
Filtration of pollutants by plants	Euro (CO)	0		0.01		0.64		0.33		3.79		2.28		11		6.53		17		10	
	Euro (NO_2)	32		10		681		361		4,307		2,497		12,877		7,413		22,033		12,068	
	Euro (SO_2)	6		1.81		121		64		760		442		2,256		1,309		3,840		2,128	
	Euro (O_3)	30		10		642		339		4,072		2,357		12,203		7,006		20,900		11,423	
	Euro (PM_{10})	151		54		3,327		1,802		23,093		12,586		75,084		38,970		134,795		65,279	
Regulation of chemical condition of the atmosphere	Euro	2,974		337		13,601		4,758		60,859		25,659		151,399		67,896		225,193		99,601	
Regulation of temperature & humidity	Euro	1,321		133		14,807		1,456		49,641		5,864		98,463		13,727		133,192		20,729	
Characteristics of living systems that enable activities promoting health	Euro	259,484		264,184		4,481,953		6,852,342		15,633,639		15,841,727		26,785,324		27,976,692		34,219,781		34,789,642	
Fibres& other materials from cultivated plants*	Euro (lumber wood)	0.09	-	2.38	-	21	-	163	-	38	-	1,826	-	125	-	6,119	-	245	-	11,654	-
	Euro (wood chips)	0.02	-	7.98	-	15	-	424	-	22	-	4,037	-	66	-	11,259	-	133	-	22,821	-

* Willingness to walk is not provided as an accumulative value. Thus, the value indicated represents the willingness to walk in minutes at the specific year presented.

** Only scenario 1 and scenario 4 includes the re-utilisation of wood waste as input material. Thus, for scenarios 2, 3, 5, and 6 this service is not accounted.

*** Only scenario 1 and scenario 4 includes the re-utilisation of wood waste as input material for processing. Thus, for scenarios 2, 3, 5, and 6 this service is not accounted. In this table only the monetary value of the wood as a material is considered. Financial costs and negative externalities derived from the transport of the wood are included in Table 3b and 5.

Relative Colour Scale

<10%	<15%	<20%	<30%	<50%	<75%	<100%

For the monetarisation, the colour scale is generated excluding values of *Characteristics of living systems that enable activities promoting health* to avoid the scale being insensitive to changes

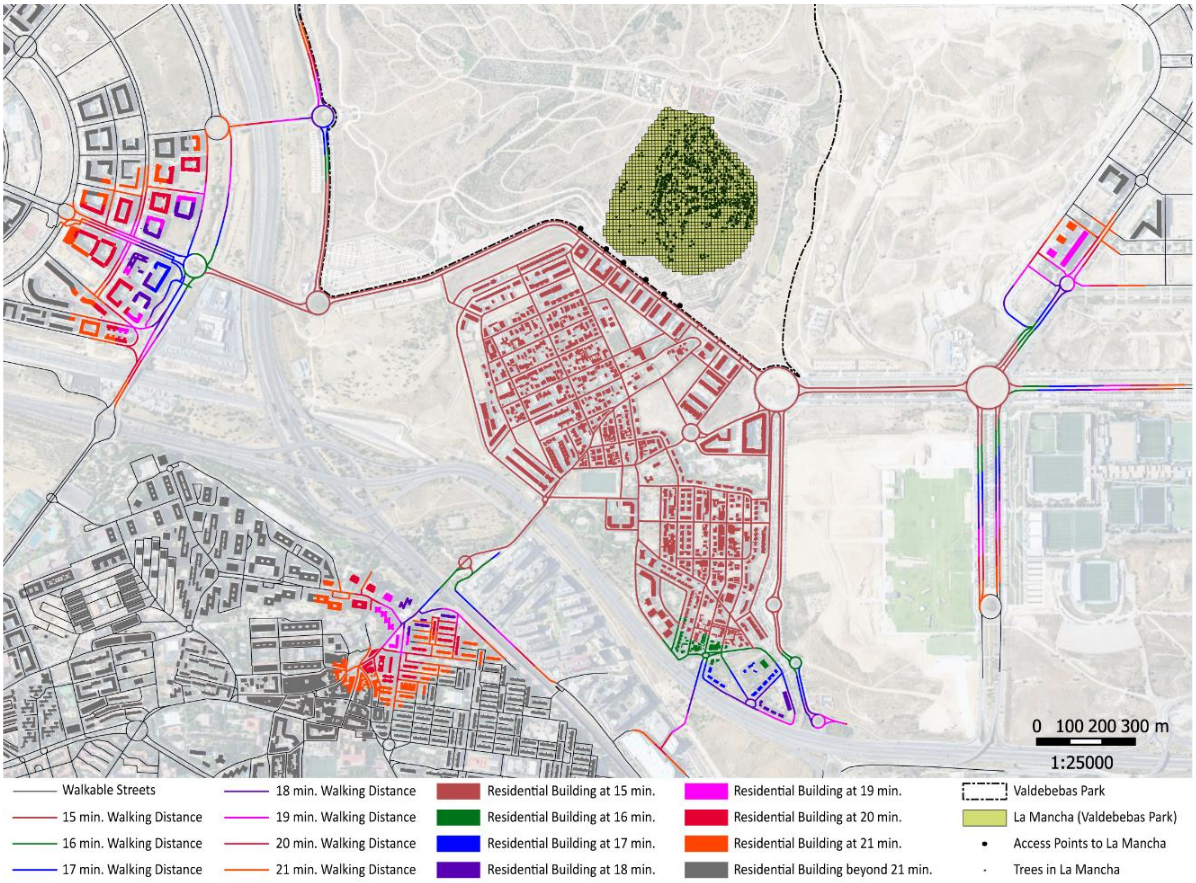

Figure 5.10. Evolution of the potential visitors (residents) of La Mancha for Scenarios 1 to 3, as a function of walking distance over time associated with the increasing maturity of the urban forest. The legend includes the years after implementation (Y) that correspond with each walking distance in minutes.

The input data of each scenario and the quantitative modelling of management works during the operational life and end of life are summarised in Table 5.2a. These are intermediate outputs that serve to calculate the financial costs (Table 5.2b), the negative environmental impacts (Table 5.3) and the negative externalities (Table 5.4). Results of Table 5.2a can help to anticipate, *grosso modo,* the scenarios with higher costs. For example, it is quite clear that S4 to S6 will have more negative externalities and financial costs than S1 to S3 due to the much greater amount of *replanting*, *pruning*, and *waste management of leaf litter*. A quick look to Table 5.2b and 5.4 confirms this anticipated result.

Table 5.3 permits an identification of two mid-point impact categories (stratospheric ozone depletion and terrestrial acidification) for which in S5 and S6 a positive environmental impact is generated. These values correspond to avoided environmental impacts due to the generation of biogas from biological waste. In the case of S2 and S3, the biological waste is so low that does not overcome the negative impacts from other management actions. In addition, the monetisation of negative environmental impacts (Table 5.4) helps to *identify global warming potential*, *particulate matter formation*, *human non-carcinogenic toxicity*, and *land occupation and transformation* as the most relevant impacts categories in terms of societal costs.

Table 5.2. A) Evolution over 50 years of cumulative management actions applied on the urban forest; B) Evolution over 50 years of the cumulative financial costs generated as a result of management actions (values in Euro 2018). S = Scenario.

A) CUMULATIVE MANAGEMENT ACTIONS APPLIED ON THE URBAN FOREST

Management Actions	Unit	Year 1				Year 10				Year 25				Year 40				Year 50			
		S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6
Planting	No. Trees	641		484		641		484		641		484		641		484		641		484	
Replanting	No. Trees	0.2		19.4		5.7		145.2		10.2		203.3		17.0		280.7		21.9		338.8	
Pruning	No. Trees	-		0		-		464.6		-		1703.7		-		2904		-		3697.8	
Disposal of dead branch wood for its re-use	t of woodchips	0.0004	-	0.04	-	0.2	-	2.8	-	0.2	-	30.8	-	0.7	-	103.2	-	1.5	-	196.5	-
Disposal of dead stem wood for its re-use	t of lumber wood	0.0007	-	0.1	- -	0.2	-	3.6	-	0.3	-	33.8	-	1.1	-	94.3	-	2.1	-	191.2	-
Waste treatment of dead branch wood	t of woodchips	-	0.0004	-	0.04	-	0.2	-	2.8	-	0.2	-	30.8	-	0.7	-	103.2	-	1.5	-	196.5
Waste treatment of dead stem wood	t of lumber wood	-	0.0007	-	0.1	-	0.2	-	3.6	-	0.3	-	33.8	-	1.1	-	94.3	-	2.1	-	191.2
Waste treatment of leaf litter	t of leaf litter	0		0.5		0.1		20.3		0.1		147.9		0.4		456.8		0.9		768.2	
Irrigation	m^3 of water	1.1		0.0		29.1		0.6		38.3		8.7		38.6		14.1		38.9		14.2	

B) CUMULATIVE FINANCIAL COSTS DERIVED FROM MANAGEMENT ACTIONS

Financial Costs	Unit	Year 1				Year 10				Year 25				Year 40				Year 50			
		S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6	S1	S2-3	S4	S5-6
Planting	Euro	103890		8867		103890		8867		103890		8867		103890		8867		103890		8867	
Re-Planting	Euro	47	52	2345	2979	1508	1694	17588	22342	2589	2923	28141	35748	4379	4937	34004	43195	5650	6367	41040	52132
Pruning	Euro	-		0		-		465		-		1704		-		2904		-		3698	
Disposal of dead branch wood for its re-use	Euro	0.04	-	4.4	-	17	-	302	-	26	-	3380	-	82	-	11326	-	165	-	21569	-
Disposal of dead stem wood for its re-use	Euro	0.08	-	7.3	-	19	-	390	-	36	-	3712	-	117	-	10354	-	231	-	20988	-
Waste treatment of dead branch wood	Euro	-	0.05	-	6	-	22	-	400	-	35	-	4476	-	109	-	14999	-	218	-	28563
Waste treatment of dead stem wood	Euro	-	0.11	-	10	-	25	-	516	-	47	-	4916	-	155	-	13713	-	306	-	27794
Waste treatment of leaf litter	Euro	0.05		78		11		2956		19		21495		64		66405		129		111677	
Irrigation	Euro	1.38		0.05		36		0.74		47		11		48		17		48		17	

Relative Colour Scale

<10%	<15%	<20%	<35%	<50%	<60%	<100%

Table 5.3. Evolution over 50 years of cumulative negative environmental impacts generated as a result of human actions on the urban forest. S = Scenario.

Negative Environmental Impacts	Unit	Year 1						Year 10						Year 25						Year 40						Year 50					
		S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6
Global warming Potential	t CO2 eq	14.8	14.8	14.8	9.8	10.1	10.1	15.3	15.4	15.4	37.6	40.2	40.9	15.7	15.9	15.9	94.2	99.7	105.8	16.3	16.7	16.7	154.9	164.3	183.2	16.8	17.3	17.4	203.2	218.2	251.5
Stratospheric ozone depletion	kg CFC11 eq	0.008	0.008	0.008	0.005	0.006	0.006	0.008	0.008	0.008	0.015	0.014	0.034	0.008	0.008	0.009	-0.009	-0.039	0.127	0.009	0.008	0.01	-0.124	-0.22	0.292	0.009	0.007	0.011	-0.25	-0.44	0.464
Ionizing radiation	MBq Co-60 eq	0.3	0.3	0.3	0.2	0.2	0.2	0.3	0.3	0.3	0.6	0.6	0.7	0.3	0.3	0.3	1.3	1.3	2	0.3	0.3	0.4	2	1.9	4.1	0.4	0.4	0.4	2.5	2.3	6.2
Ozone formation, Human health	kg NOx eq	108.3	108.3	108.3	55.1	56.2	56.2	110.9	111.2	111.2	197.2	205.6	206.2	113	113.6	113.6	481	496	500.8	116.2	117.2	117.2	753.1	774.4	789.3	118.5	119.8	119.9	951.5	981.4	1007.8
Particulate matter formation	kg PM2.5 eq	28.6	28.6	28.6	15.4	15.7	15.9	29.2	29.2	29.4	40.9	42.5	51	29.7	29.8	30	77.1	72.3	140.5	30.4	30.4	31.1	87.8	65.9	275.8	30.9	30.8	32.2	85.9	39.5	410.3
Terrestrial acidification	kg SO2 eq	55.5	55.5	55.5	30.6	31.2	32.8	56.7	56.5	57.5	73.8	71.9	140.3	57.7	57.4	59.2	55.5	-10.7	535	58.9	57.4	63.2	-175.5	-393.8	1286.6	59.6	56.2	67.8	-440.1	-876.3	2092.4
Freshwater eutrophication	kg P eq	1.4	1.4	1.4	1.3	1.3	1.3	1.5	1.5	1.5	10.6	10.7	10.8	1.6	1.6	1.6	29.9	30	30.6	1.8	1.8	1.8	48.2	48.5	50.3	1.9	1.9	1.9	61.4	61.7	64.9
Marine eutrophication	kg N eq	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	2.3	2.3	2.3	0.2	0.2	0.2	6.7	6.9	6.9	0.3	0.3	0.3	11.2	11.6	11.5	0.3	0.3	0.3	14.5	15.2	15.1
Human carcinogenic toxicity	t 1,4-DCB eq	0.3	0.3	0.3	0.2	0.3	0.3	0.3	0.3	0.3	0.9	0.9	1	0.3	0.3	0.3	2.2	2.3	2.6	0.3	0.3	0.3	3.6	3.9	4.8	0.3	0.4	0.4	4.8	5.2	6.8
Human non-carcinogenic toxicity	t 1,4-DCB eq	8.3	8.3	8.3	6.2	6.3	6.3	8.5	8.5	8.5	14.1	14.3	14.6	8.7	8.7	8.7	29.6	30	32	8.9	8.9	8.9	47.9	48.3	54.5	9.1	9.1	9.2	64.2	64.7	75.7
Water consumption	m3 water eq	62.6	62.6	62.6	37.1	37.3	37.4	92.7	92.8	92.8	169.7	171.4	172.8	103.7	103.8	103.8	438.1	440.5	451.7	106.7	106.9	107	693.9	696.3	730.6	108.9	109.2	109.4	879.3	881.7	942.2
Land occupation & transformation	ha crop eq	1.3	1.3	1.3	1	1	1	1.4	1.4	1.4	1.5	1.5	1.5	1.4	1.4	1.4	2	2	2	1.4	1.4	1.4	2.4	2.4	2.6	1.4	1.4	1.4	2.7	2.7	3.1

Relative Colour Scale

>50%	>40%	>30%	>20%	>10%	<10%	<15%	<20%	<35%	<50%	<75%	<100%

Table 5.4. Evolution over 50 years of cumulative negative externalities generated as a result of human actions on the urban forest (values in Euro 2018). S = Scenario.

Negative Externalities	Unit	Year 1				Year 10						Year 25						Year 40						Year 50					
		S1-3	S4	S5	S6	S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6
Global warming Potential	Euro	840	557	576	577	868	874	875	2,136	2,285	2,329	892	903	904	5,359	5,672	6,020	929	948	951	8,809	9,347	10,419	956	982	990	11,558	12,411	14,304
Stratospheric ozone depletion	Euro	0.20	0.20	0.20	0.20	0.30	0.20	0.30	0.50	0.40	1.10	0.30	0.30	0.30	-0.30	-1.20	3.90	0.30	0.20	0.30	-3.80	-6.70	8.90	0.30	0.20	0.30	-7.60	-13	14
Ionizing radiation	Euro	19	14	14	14	19	19	19	37	38	43	20	20	20	78	78	121	20	20	21	116	111	241	21	21	22	145	133	363
Ozone formation, Human health	Euro	120	61	62	62	123	123	123	218	227	228	125	126	126	532	548	554	128	130	130	832	856	872	131	132	133	1,052	1,085	1,114
Particulate matter formation	Euro	1,638	882	900	912	1,672	1,675	1,682	2,343	2,433	2,923	1,699	1,705	1,718	4,418	4,144	8,049	1,740	1,742	1,784	5,029	3,776	15,802	1,768	1,762	1,844	4,920	2,262	23,506
Terrestrial acidification	Euro	277	153	156	164	283	282	287	369	359	701	288	287	296	277	-54	2,672	294	287	316	-877	-1,967	6,426	298	281	338	-2,198	-4,376	10,450
Freshwater eutrophication	Euro	2.50	2.50	2.50	2.50	2.80	2.80	2.80	20	20	20	3.00	3.00	3.00	56	56	57	3.30	3.30	3.30	90	91	94	3.50	3.50	3.60	115	115	121
Marine eutrophication	Euro	0.60	0.60	0.60	0.60	0.70	0.70	0.70	7.10	7.20	7.20	0.70	0.70	0.70	21	21	21	0.80	0.80	0.80	35	36	36	0.90	0.90	0.90	45	47	47
Human carcinogenic toxicity	Euro	30	24	25	25	31	32	32	86	92	95	32	33	33	216	228	257	34	34	35	361	385	474	35	36	36	481	520	678
Human non-carcinogenic toxicity	Euro	830	619	623	623	847	848	849	1,403	1,429	1,454	862	864	864	2,946	2,987	3,188	885	888	891	4,768	4,809	5,430	904	908	912	6,391	6,439	7,535
Water consumption	Euro	49	29	29	29	73	73	73	134	135	136	82	82	82	346	348	356	84	84	84	548	549	576	86	86	86	694	696	744
Land occupation & transformation	Euro	1,142	882	882	882	1,153	1,153	1,153	1,238	1,238	1,246	1,163	1,163	1,163	1,674	1,670	1,739	1,177	1,177	1,178	2,022	2,011	2,222	1,187	1,187	1,189	2,309	2,286	2,659

Relative Colour Scale

-5%	>-10%	>-20%	<10%	<15%	<20%	<35%	<50%	<60%	<100%

As a final output, the monetisation of environmental impacts allows the integration of financial costs and externalities providing a simple overall value of the economic performance of urban forest over time (Figure 5.11a and b). This type of results helps to identify when break-even points are achieved, and net benefits are starting to be obtained. It also helps to identify when different scenarios have the same net value.

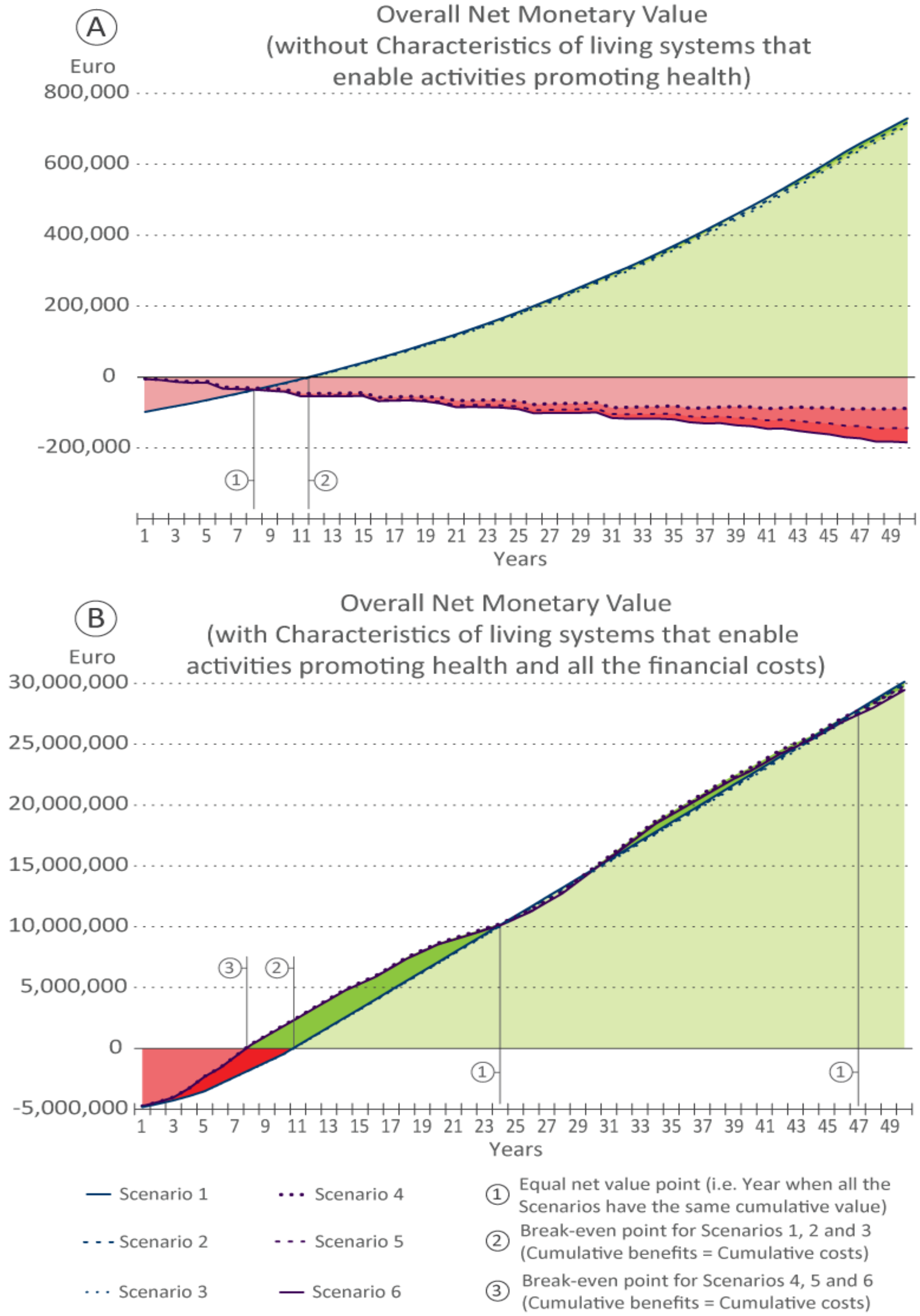

Figure 5.11.A) Evolution over 50 years of cumulative net monetary value of the urban forest without including the service characteristics of living systems that enable activities promoting health; B) Evolution over 50 years of cumulative net monetary value of the urban forest considering all costs and benefits, including financial costs not directly related to trees extracted from the bill of quantities.

The same approach could be applied to environmental impacts where ES and mid-point impact categories of LCA share units of measurement. This is the case of *global warming potential* and *regulation of chemical condition of the atmosphere* (Figure 5.12a) and *particulate matter formation* and *filtration by plant* of PM_{10} (Figure 5.12b).

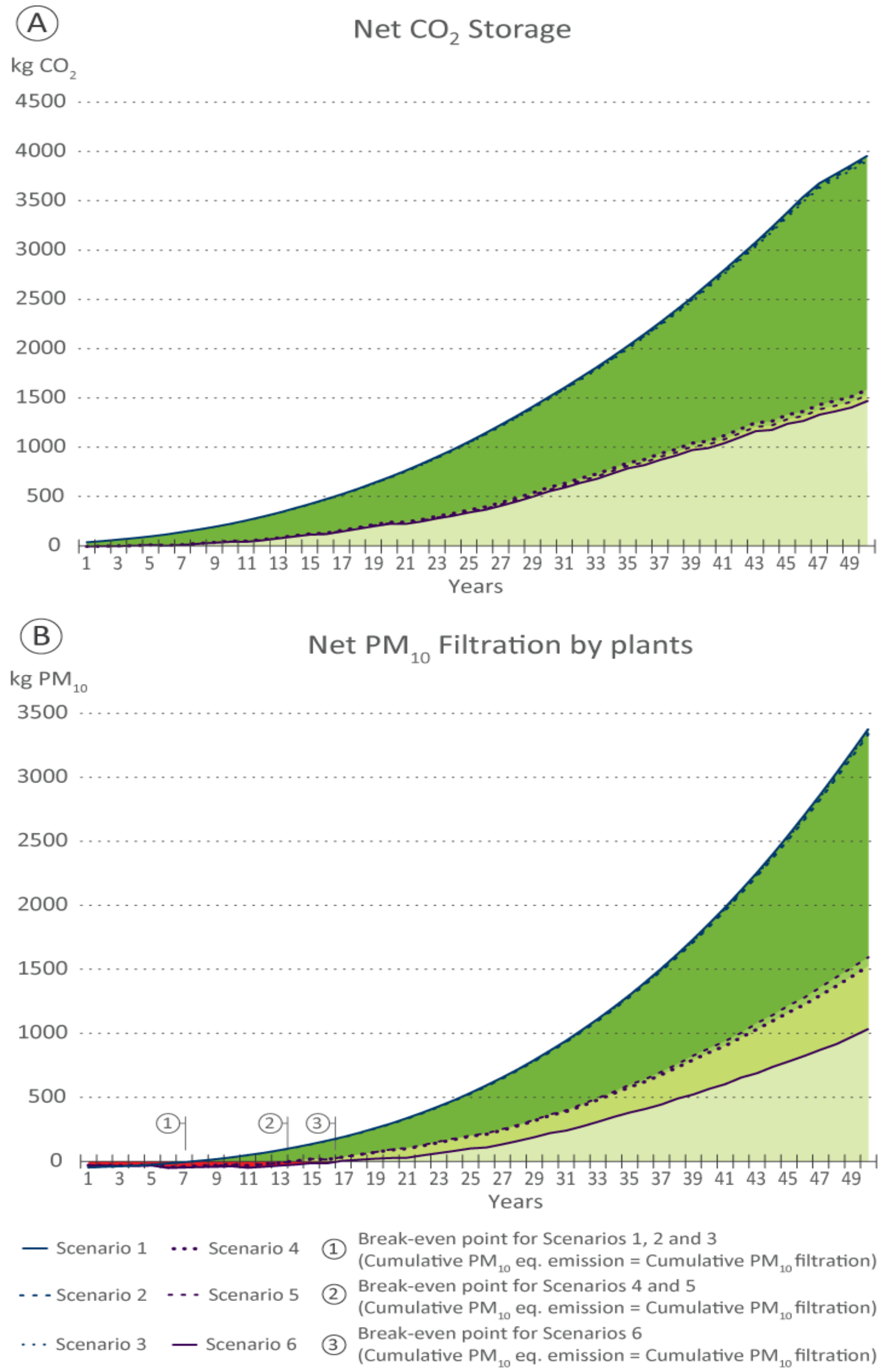

Figure 5.12. A) Evolution over 50 years of net CO2 storage of the urban forest considering the CO2 eq. emissions in all the life cycle phases; B) Evolution over 50 years of cumulative net PM10 filtration of the urban forest considering PM_{10} eq. emissions in all the life cycle phases.

For the case of particulate matter formation, $PM_{2.5}$ eq. is converted first to PM_{10} eq. based on the conversion factor proposed by De Bruyn *et al.* (2018). As a result, the overall performance for specific environmental impacts (negative and positive) over time can be assessed considering all the life cycle phases.

5.6. Discussion and Conclusion

5.6.1. Novelty underpinning the conceptual modelling framework

The conceptual modelling framework outlined in this Chapter represents a novel integrated methodology for the assessment of the contribution of NBS interventions to sustainability and resilience. To the best of the authors' knowledge, no other solutions exist capable to integrate the strengths of life cycle thinking, ES and system dynamics approaches on the specific case of NBS.

The representation of outputs in biophysical and monetary units enables to encompass both, the environmental and economic dimensions in the NBS assessment. For example, it permits quantification of the costs for society of specific decisions offering the outputs to non-expert users in a metric that they can easily understand. However, to improve the communication of biophysical results by non-experts it might be necessary to provide a reference against which NBS can be compared. In addition, if biophysical and monetary results outputs are used as two different set of impacts, there is a risk to double count the environmental impacts. In this sense, it is worth making clear to future users that the biophysical and monetary outputs of the conceptual modelling framework are complementary and should not be used as a representation of independent set of impacts.

The consideration of multiple positive and negative environmental impacts looks to overcome bias in current NBS assessments. It can help improving the quantification of their net contribution to urban sustainability, offering a useful tool for urban planning decisions. It also permits differentiation of choices (e.g. paved ground) with alternative consequences in the net contribution of NBS. For example, the comparison among the Current Park and the Paved Park illustrated the strong negative consequences that few design/planning decisions, such as extensive paving of an urban forest, could have on the ES supply. As another example, the comparison between "Paved Park dead wood re-use" scenario and "Paved Park composting" scenario helped to understand the impacts (e.g. in terms of particulate matter formation, Table 5.4) of alternative management decisions could have regarding the performance of common urban solutions, such as urban trees. However, the modelling framework risks being excessively time consuming for its use in regular urban planning decisions. It requires the application of multiple methods, steps, and needs multiple types of input data. Therefore, advancements should be made to move from the current modelling framework to a user-friendly tool for built environment professionals, where most of the time-consuming tasks remains on the side of the modeller.

In the modelling framework, the risk of an oversimplified assessment is avoided modelling at two-levels. On one hand, the foreground level works at daily, monthly and yearly resolutions, at a detail spatial resolution, and with thematic resolutions that goes beyond the use of land cover classes for local NBS assessments. On the other hand, the background level permits consideration of assessments at a larger spatial extent. In this sense, it permits the integration of environmental impacts that do not occur locally and therefore cannot be mapped as part of local assessments.

Following this exercise, the potential of the conceptual modelling framework for the development of tangible generalisable, quantitative and time-efficient and simple to use tools is showcased in Chapter 6, also making use of the urban forest model.

5.6.2. Urban forest model: advantages and limitations

The application of the urban forest model to Valdebebas, showed clearly that S1 to S3, the current plan/design implemented (Real La Mancha), performed in terms of environmental impacts much better than S4 to S6 (Paved La Mancha), where the urban forest was defined emulating street tree conditions. It also showcased, that for the cases of S1 to S3, which were treated more like a naturalised forest, changes in end-of-life processes were not significant. Instead, in S4 to S6 variations in the end-of-life processes provided significant changes.

In terms of monetary units, when all the benefits and costs, except those derived from *characteristics of living systems enabling activities promoting health* (recreation), were considered (Figure 5.11a) it is also clear that S1 to S3 performed much better than S4 to S6. Moreover, differences between S4 to S6 can also be observed due to the variations in their end-of-life processes, being S4 the best option among them. However, in terms of overall net benefit when accounting for recreational benefits and all the financial costs (not only those related to trees) provided in the original bill of quantities (Figure 5.11b), the differences between S1 to S6 are not evident in overall monetary units. This is because of the extreme high value of recreational benefits compared to others, which it is logic when thinking that recreation is traditionally the main positive externality provided by green open spaces. In fact, for the visualisation of Figure 5.11b, it was decided to include also financial costs not related to trees, since *characteristics of living systems enabling activities promoting health* and its derived recreational benefit do not only depend on trees. Then, it was considered unfair to include the benefits derived from this ES and not the associated costs on equipments (e.g. benches, paved paths, earth movements to make land more accessible) needed for the supply of this ES. Hence, this difference in the overall net value shows how important is to provide also disaggregated results and more than one type of value (economic and biophysical), otherwise differences in performance among scenarios would not have been evident.

The case study illustrates that the urban forest model provides a comprehensive spatial-temporal-thematic representation of urban forests, adequate for a detailed assessment of interventions such as single urban woodlands or landscape plans of street trees. It shows that the model is sensitive to changes in ES flow result of changes in tree species, tree age (dimensions), ground conditions (soil texture, initial soil organic matter, and impermeabilization), and basic management actions (irrigation, pruning, harvesting, and removal of plant residues). The model also acknowledges the effect of meteorological conditions on the behaviour of the NBS, allowing the use and generalisation of the model for different urban contexts.

Nevertheless, due to the high complexity and lack of complete knowledge about the interactions occurring among the elements of an urban NBS, the urban forest model necessarily simplifies the representation of some aspects – translating it in a reduced number of equations – of this coupled human-nature system. For example, all trees in the same cell had to be represented as the same species and age, even if was not always the case, because cells are the minimum unit of differentiation. As another example, a full soil water balance cannot be considered since the influence of groundwater through water table movements is difficult to be characterised. Neither alternative sources of water than precipitation or irrigation for paved trees are acknowledged, ending in an excessive underestimation of tree transpiration under paved conditions. The consequences can be observed clearly in Figure 5.8, where after the initial period with irrigation, paved trees had a very low transpiration, almost close to zero. All these simplifications provide variable and structural uncertainty in the model, which can only be mitigated by improving the collection of data for local urban forest inventories.

The temporal resolution of the model accounts for seasonality influencing ES supply when and when it is not demanded. Figure 5.8 illustrates clearly this aspect, by showing how different is tree transpiration when demand is present than when it is not. This kind of results may help more informed decisions to ensure that urban forests are able to supply ES when shocks such as heat waves occur. For example, in terms of management it may inform when irrigation should be applied. Additionally, it may aid in the selection of components (e.g. tree species, size of trees) during the design and planning works to ensure high supply capacity of specific ES classes in the long term. Accounting for this temporal dynamism is quite relevant for regulating ES, helping to avoid overestimation in their supply capacity (Sutherland et al., 2018).

Concerning multifunctionality, the model shows a good potential to study ES trade-offs and synergies, since ecological processes are represented interrelated, and design and management decisions modifying them influence all the ES depending on them. For example, changes in tree growth rate not only influence tree and soil carbon sequestration in the model, but also pollutants removal, hydrological cycle and water flow regulation and temperature and humidity regulation. As another example, Figure 5.9 illustrates how death rate may end influencing potential willingness to walk of residents, due to a lower average height of the trees, associated to the perception that the urban forest is still very young and not that attractive. The link between death rate and the perception of the forest might not be obvious at first and consequently might be missed in models that assess ES one by one. However, in the model interactions among cells, which could be relevant for several regulating ES, are still not considered. Consequently, effects on ES supply derived from the spatial configuration are not acknowledged.

For sake of simplicity, this chapter showed only average values across multiple simulations. However, the model includes stochastic components, and each simulation is therefore different in values. It is possible to illustrate the known variation in results in the form of standard deviation, as it will be showcased in Chapter 6. Visualisation of the standard deviations of values can provide more transparency about the outputs of the model and their reliability to support decision making.

Chapter 6

An online decision support system to aid in the planning and design of cost-effective nature-based solutions†††

6.1. Introduction

The capability to address multiple societal challenges and to provide multiple co-benefits in a cost-effective way needs to be easily quantified before fully mainstreaming urban nature-based solutions (NBS). However, in many cases ecosystem services (ES) assessment methods are non-generalizable, or provide qualitative outputs (Haase *et al.*, 2014; Cortinovis and Geneletti, 2019). Consequently, generalizable ES assessment methods that provide quantitative results and are applicable to a broad range of urban settings are necessary.

As, already introduced in Chapter 3, several ES generalizable methods are already developed in the form of ES modelling tools (e.g. InVEST, ARIES, LUCI, SolVES), which could act as decision support systems (DSS). Most of these modelling tools assess ES based on land use/cover classes and are focused on large scales. However, land use/cover classes do not offer adequate baseline data to assess some types of urban ES such as regulating services (Cortinovis and Geneletti, 2019). Then, ES tools relying on land use/cover data might not be suitable for urban studies, especially for the assessment of specific urban NBS interventions. Only few modelling tools (e.g. i-Tree, ENVI-met) are focused on assessing specific urban NBS interventions. However, these few tools still show some of the typical limitations of ES methods (Bagstad *et al.*, 2013; Grêt-Regamey *et al.*, 2017; Cortinovis and Geneletti, 2019): i) highly time-consuming data collection; ii) difficult practical integration into urban planning

††† Chapter 6 is based on:
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Roles of other authors:
Benedetto Rugani, was the academic supervisor of the paper.
Claudio Petucco contributed to the review of the paper.
Tomas Navarrete Gutiérrez and Laurent Chion developed the coding for the calculation service and the graphical user interface of the decision support system. They also contributed to the review of the paper

and design processes; iii) lack of monetisation of ES values; iv) lack of consideration of ES demand and costs; v) and lack of simultaneous modelling of multiple ES over time.

As discussed in Chapter 5, collection of data is very time consuming because it is necessary to take into account the multiple attributes of NBS (e.g. vegetation species, size of trees, soil characteristics). For existing NBS interventions, detailed data is in many cases not available (Petucco *et al.*, 2018). Moreover, data on future NBS interventions is unknown at early stages of its planning/design because site surveys have not been done yet (e.g. soil characteristics) or some aspects of the NBS (e.g. vegetation species) are still not defined.

To ensure a broad integration of evaluation procedures into urban planning and design processes, modelling tools also need to be user-friendly and to require a low computational effort. Environmental assessments are generally absent or done only at advance project stages (Badach *et al.*, 2018; Oliver and Pearl, 2018). However, their integration at early design stages is more desirable. As suggested in the MacLeamy curve (Figure 6.1), changes at early stages have less cost and are more capable to influence functionality of interventions (Landscape Institute, 2016; Hollberg *et al.*, 2018). In this sense, ES modelling tools need to provide visually attractive and understandable outputs so that built environment professionals can easily use them to communicate about NBS value with non-technical stakeholders and the broad public.

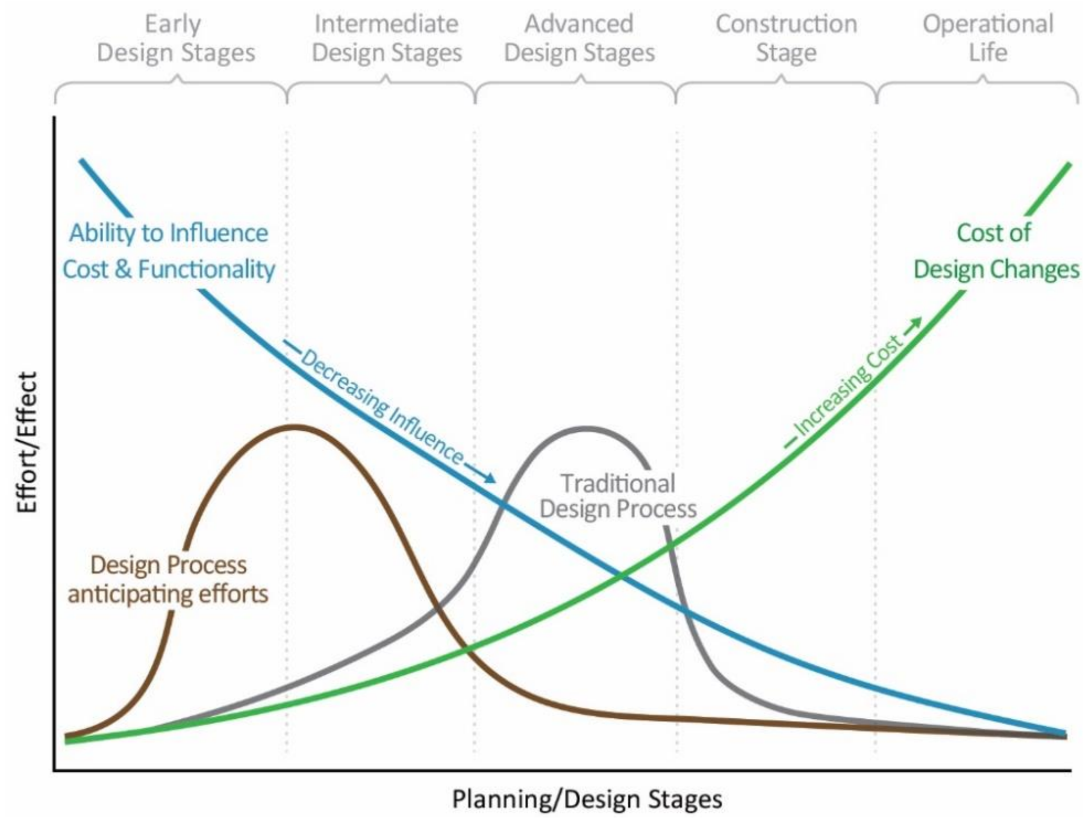

Figure 6.1. The MacLeamy curve. Adapted from Landscape Institute (2016) and Davis (2013)

Most ES modelling tools provide outputs only in biophysical or social values because they are designed for experts and may be hard to understand by other stakeholders. For example, ENVI-met only uses biophysical values to inform experts about changes in microclimate conditions (e.g. temperature and humidity regulation, air pollution removal) due to new urban interventions. However, complementing biophysical and/or social values with monetary estimations can facilitate the understanding or the model outcomes by decision makers and other stakeholders. In addition, existing ES modelling tools usually do not consider negative externalities (i.e. damages to goods and services for which a market does not exist), nor ES demand, and calculate ES supply for a specific point in time or for a short period not taking into account a life cycle thinking perspective. This gives a partial picture about NBS that impedes to see the evolution of benefits and costs over their entire life cycle.

The aim of Chapter 6 is to integrate the modelling framework set up in Chapter 5 into a prototype DSS, named NBenefit$®, that overcomes the limitations of current modelling tools to assess NBS. By overcoming current limitations, NBenefit$ looks to facilitate the use of NBS assessment tools in urban planning and landscape design processes, especially at early design stages. The prototype DSS makes use of NBS archetypes. NBS archetypes refer to the specific combination of key abiotic (e.g. soil texture), biotic (e.g. plant species) and management (e.g. irrigation) attributes defining variations in a specific NBS Type (e.g. urban forest, green roof). The compilation and use of NBS archetypes is a simple form of providing ready to use variations of specific NBS Types for built environment professionals. They can test how modifications in the attributes of an NBS makes a specific NBS project more or less cost-effective.

The following section describes the design, building and operation of NBenefit$. Then, to illustrate its functionality, the calculation and results of 48 urban forest archetypes for the environmental conditions of Madrid (Spain) are used as support. At the end, we discuss current limitations and advantages of NBenefit$ and we introduce future works.

6.2. NBenefit$ Decision Support System

The design, building and operation stages of NBenefit$ are summarised in Figure 6.2. The design stage needs to be repeated each time a new NBS Type is included in the DSS. For the calculation of new NBS archetypes of NBS Types already integrated in the DSS, only the building stage needs to be repeated.

To pre-calculate the archetypes, the system dynamics model presented in Chapter 5 assesses their performance (in biophysical units) during the operational stage. The input characteristics of the archetype and part of the outputs from the system dynamics model are used to evaluate the negative impacts (in biophysical units) generated from the implementation stage up to the end of life. These costs (negative impacts) and benefits (positive impacts) in biophysical units are later monetarised making use of value transfer. These results represent the net externality value of the archetype, onto which the financial costs are added based on national cost databases and public available data on expenses. The final monetary results offer a better understanding of the net value of an NBS archetype by year and overall. Both biophysical and monetary values per archetype are stored in NBenefit$ database.

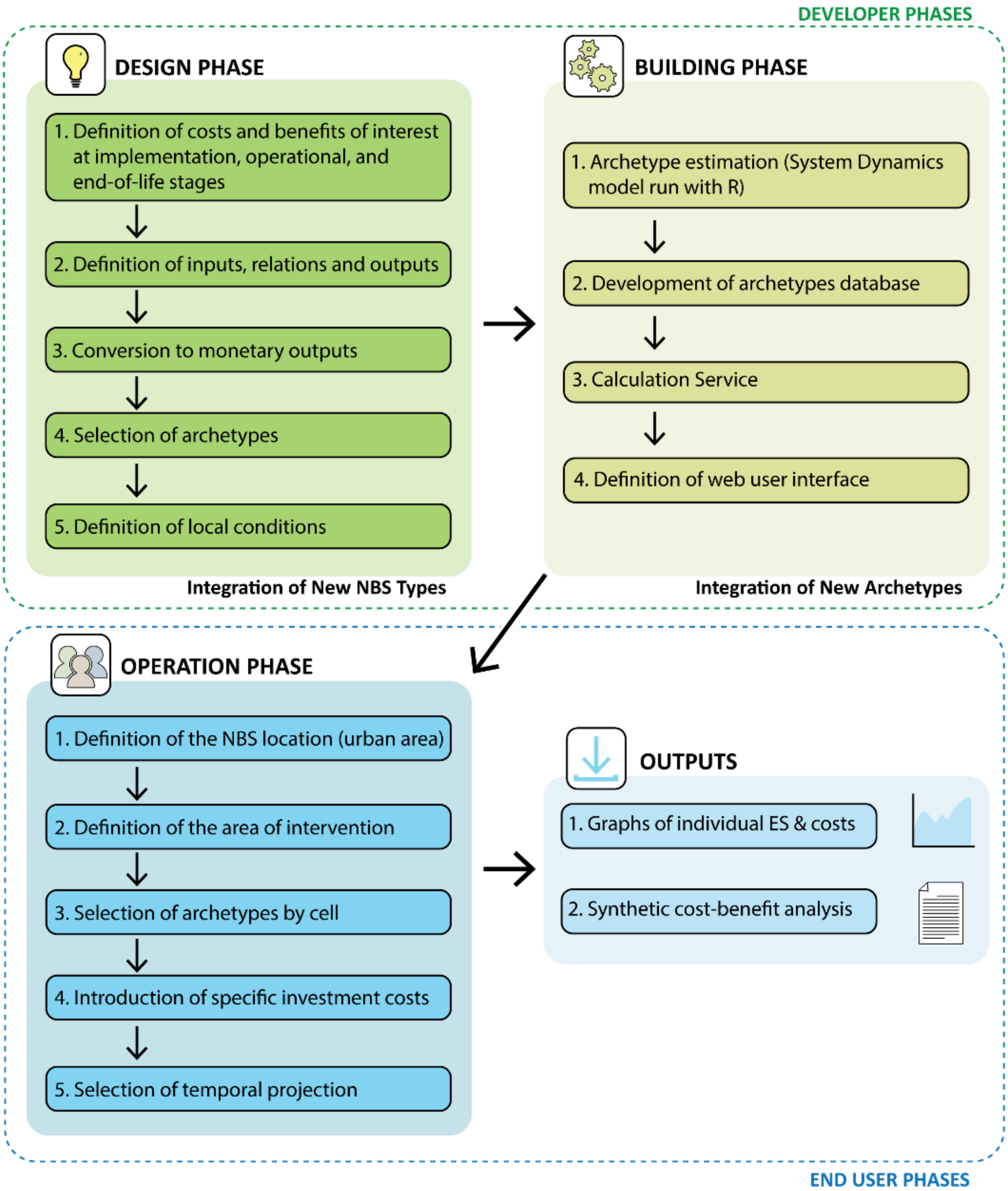

Figure 6.2. Schematic diagram of the Design, Building and Operation phases to develop the overall NBenefit$ architecture.

NBenefit$ provides two types of outputs: i) a graph of the evolution of each cost and benefit flow (financial and externality) over time in biophysical and monetary units; ii) and a simplified cost-benefit analysis of the entire life cycle of the NBS.

6.2.1. Design Phase

For each new NBS Type a system dynamics model, as the one representing the foreground level in Chapter 5, is developed in Simile (https://www.simulistics.com/).

The first step is to identify for each NBS Type the main benefits and costs to be quantified over the entire life cycle (operational, implementation and end-of-life). This step has been already explained in detailed in the description of the conceptual modelling framework in Chapter 5. The benefits and cost selected, like in Chapter 5 are based on the outputs of Chapter 2 and 3.

As a second step, the relevant socio-ecological processes and human actions that generate the above outputs are identified together with the attributes of the NBS and the urban context influencing these processes and actions. As described in Chapter 5, for the implementation and end-of-life stages, the processes of interest are defined and pre-calculated supported on the use of life cycle inventories and are not included in the system dynamics model. As also described in Chapter 5, for the operational life, processes and attributes of interest are also defined based on from the outputs of Chapter 2 and 3. For urban forest, Table 6.1 summarises the cost and benefits, the life cycle stages in which they occur, the processes modelled, the main input characteristics influencing those processes, and the method used for calculation. Different to Chapter 5, several end-of-life alternatives are not tested and only composting is considered. In addition, *characteristics of living systems to enable activities promoting health* is not included in the list of ES assessed. Figure 6.3, as an adaptation of Figure 5.3 for the ES and costs considered in this chapter, summarises the biophysical indicators used to represent cost and benefits in biophysical units, and for which cost and benefits indicators are equivalents.

In the third step, positive and negative externalities in biophysical units are converted to monetary units following the steps of Chapter 5. For the externality costs of the implementation and the end-of-life, the monetary estimation is defined starting from the handbook of environmental prices for the European Union of De Bruyn et al (2018). For the benefits of the operational life, the values from De Bruyn et al (2018) are complemented with the default values used in Chapter 5 (see Annex 5.2). In the case that the monetary estimation of externalities is not covered in the previous two sources, scientific literature should be used to cover the gaps. Different to Chapter 5, in order to be more transparent about the known uncertainty of the monetisation methodologies, monetary estimations are defined by mean, upper and lower values. Later in the operation phase, NBenefit$ users can indicate a discount factor for future benefits and costs (i.e. operational life and end-of-life).

Table 6.1. Costs and benefits of urban forests, main processes influencing outputs, main input characteristics influencing processes, and methodological approaches used for calculation.

	Costs & Benefits	Life-cycle stage	Main Processes	Main Input Characteristics	Method for Calculation
COST	**Tree planting****	Implementation	Tree management in nursery, transport to site, planting	Species, initial tree size/year at planting, average transport distance, planting techniques & machinery	Inputs of System Dynamics Model + LCA
	Amount of tree replanting* (due to premature death)**	Operational	Plants morbidity and plant mortality	Species, initial tree size, stress factors (paving, drought, waterlogging), mortality statistics****	System dynamics Model + LCA
	Pruning	Operational	Vegetation growth (branches)	Species, initial tree size, percent of branch to be pruned, pruning techniques & machinery	System dynamics Model + LCA
	Irrigation	Operational	Storage of available soil water, soil evaporation, vegetation transpiration, infiltration, percolation	Soil texture, initial available soil water	System dynamics Model
	Management of waste from litter and dead wood	End-of-life	Leaf, branch decay, and plant mortality	Species, initial tree size, mortality records, branch and leaf decay rate, plant management	System Dynamics Model + LCA
BENEFIT (Ecosystem Services)	**Regulation of the chemical composition of the atmosphere**	Operational	Vegetation growth, drought and waterlogging, death	Species, initial tree size, growth rate, decay rate, soil texture, threshold to stress factors	System Dynamics Model
	Regulation of temperature and humidity	Operational	Free evaporation during rainy days, soil evaporation, vegetation transpiration	Species, initial tree size, growth rate, ratio tree size to leaf area, daily precipitation, daily temperature, dormant periods (deciduous trees), soil texture, available soil water	System Dynamics Model
	Hydrological cycle and water flow regulation	Operational	Vegetation interception, infiltration, percolation, soil evaporation, vegetation transpiration	Species, initial tree size, growth rate, ratio tree size to leaf area, daily precipitation, daily temperature, dormant periods (deciduous trees), soil texture, available soil water	System Dynamics Model
	Filtration of pollutants by plants	Operational	Dry deposition, vegetation transpiration	Species, initial tree size, tree height, growth rate, ratio tree size to leaf area, precipitation, temperature, air pollutant levels (CO, NO_2, SO_2, O_3, PM10), wind, solar radiation	System Dynamics Model

Notes: LCA = Life Cycle Assessment

* A specific tree-planting scheme is decided by the landscape designer, the client and in some cases guided by external consultants as part of the design process. The model only accounts for the differences in cost due to species, sizes (age = amount of time in the nursery), and planting system.

** The model assumes that the same species and initial planting size is used for replanting if the tree dies. It assumes mandatory replanting after one year if the tree is planted less than 10 years ago, and after three years if the tree was planted for the first time more than 10 years ago. Therefore, in the model plants mortality is the only processes influencing tree replanting.

**** Currently mortality statistics are taken from existing literature, but could be obtained from local data (including the influence of stress factors) if data is available.

In the fourth step, the archetypes for specific NBS Types are defined at different levels of detail to meet the needs of different NBS design stages. Data available at different planning/design stages, and the detail of the proposed interventions are taken into account to generate the outputs expected by professionals at each stage.

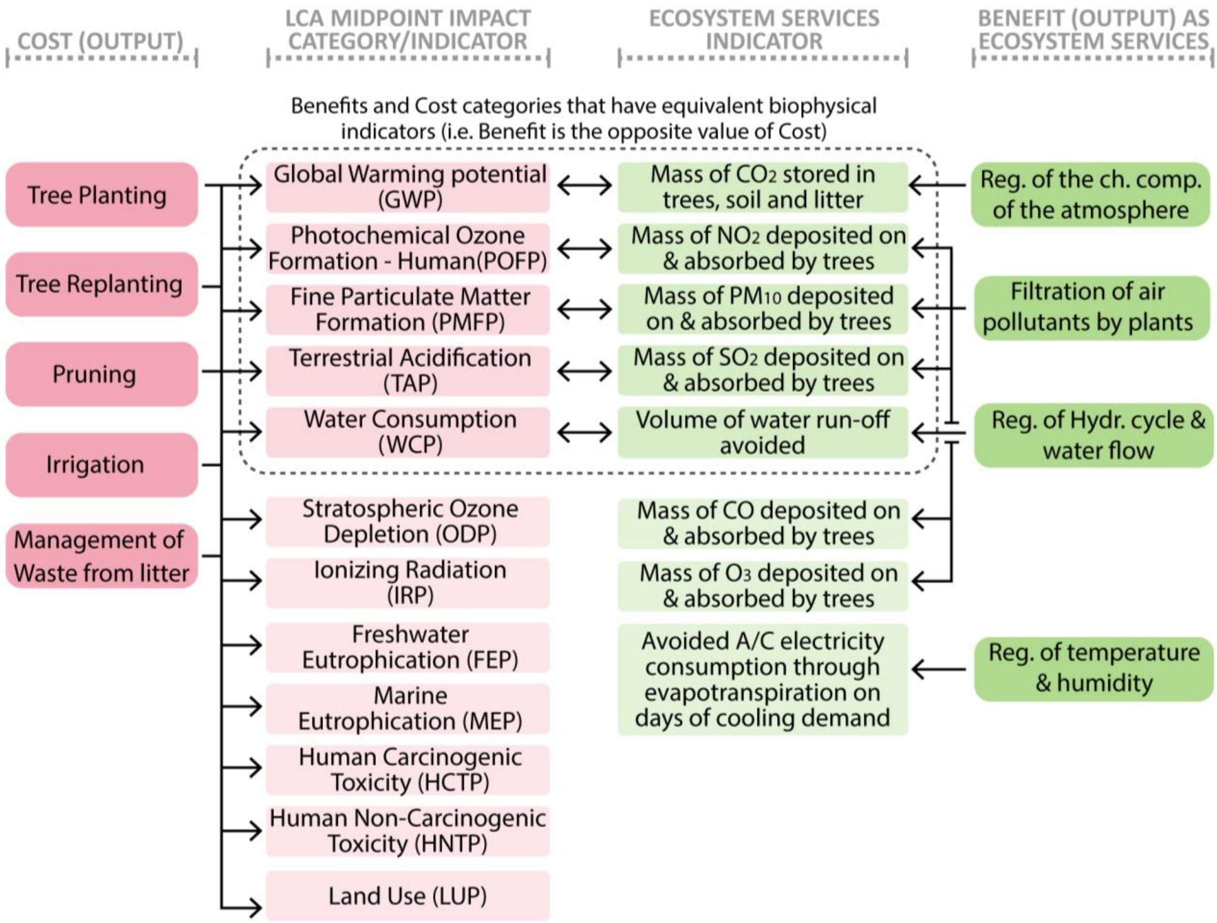

Figure 6.3. Biophysical indicators that represent cost and benefits in the form of negative and positive environmental impacts. The cost and benefits that have equivalent indicators are highlighted. All the costs (outputs), contribute to environmental impacts represented by each of the LCA midpoint impact categories included.

At early planning/design stages such as strategic definition of projects, briefing or concept design (see RIBA (2013) for an example of design stages), detailed data on site conditions (e.g. soil characteristics, existing vegetation) and a good definition of the proposal is generally not available. In fact, the data produced by built environment professionals tends to be similar to zoning diagrams equivalent to land-use/cover maps. This is why, only indicative information about the benefits of different basic NBS alternatives might be required. Then, NBenefit$ archetypes used at this stage are similar to land use/cover classes and correspond to an aggregation of detailed archetypes.

At intermediate and advanced stages, when the project is being developed and technical aspects are being defined, there is usually more detailed data about the site conditions and the proposed intervention (e.g. type of vegetation, species, size at planting, their location on site) and potential future NBS management. At these stages, outputs have to provide detailed information to compare alternatives in terms of benefits and costs with a good level of accuracy. Outputs of archetypes at these stages help to see what happens if abiotic or biotic conditions are modified (e.g. by soil amendments) or landscape management actions are changed. Then, NBenefit$ archetypes used at these stages differentiate clearly vegetation species, size at planting, soil texture, and management actions.

For the case of urban forests, Figure 6.4 showcases the current available values per each core attribute and their potential combination, which define the maximum number of archetypes possible. Additionally, Table 6.2 illustrates the levels of detail in values for the attribute tree species considering the planning/design stage in which they would be used most often. The current list of tree species selected correspond to species typical of European urban landscapes for which data was available. The species were selected with the support of French experts on plants and landscape design, making use of a 2007 French street tree census (Gutleben and Goumot, 2015) and a compilation of species from 81 nurseries catalogues and collections (including botanical gardens).

The last step in the Design Phase is the definition of the local context, regarding meteorological conditions (i.e. temperature, wind and rain), air quality (i.e. concentration of CO, SO_2, NO_2, O_3, and PM10), and country. As explained in Chapter 5, the modelling of the meteorological and air quality conditions is generated in the atmosphere module of the system dynamics model underpinning NBenefit$. The notion of country is needed to transfer and adjust the ES monetary values when country-specific estimates are not available. In this case a benefit transfer method is applied to correct for the purchasing power parity and the income effect (Plummer, 2009; Brouwer and Navrud, 2015; Petucco *et al.*, 2018).

Figure 6.4. Core biotic, abiotic and management attributes that define urban forest archetypes in the system dynamics model underneath NBenefit$

Table 6.2. Urban forest archetypes. 1st and 2nd level aggregated archetypes correspond to detailed land use/cover classes as the ones used in early planning/design stages.

Early planning/design stages		Intermediate/Advanced planning/design stages
2nd Level Aggregated Archetypes* **	**1st level Aggregated Archetypes** **	**Disaggregated archetypes***
Mixed Urban Forest (75% Evergreen/25% Deciduous)	Evergreen Urban Forest	*Quercus ilex*
		Eucaliptus globulus
		Brachychiton populneum
		Acacia melanoxylon
		Magnolia grandiflora
		Ceratonia siliqua
		Cedrus deodara
		Juniperus virginiana
		Ilex Opaca
		Prunus caroliniana
Mixed Urban Forest (50% Evergreen/50% Deciduous)		*Pinus sylvestris*
		Pinus nigra
		Pinus strobus
		Pinus radiata
	Deciduous Urban Forest	*Platanus acerifolia*
		Tilia cordata
		Acer palmatum
		Prunus serrulata
		Aesculus hippocastanum
Mixed Urban Forest (25% Evergreen/75% Deciduous)		*Quercus palustris*
		Fraxinus americana
		Celtis occidentalis
		Populus balsamifera subsp. Trichocarpa
		Pyrus calleryana 'Bradford'
		Robinia pseudoacacia
		Carpinus betulus 'fastigiata'
		Betula pendula
		Liquidambar styraciflua

Notes: * Disaggregated archetypes correspond to perennial and deciduous species extensively used in European urban areas for which allometric equations exist.
** 1st level aggregated archetypes are made by equal combination of disaggregated archetypes split by evergreen and deciduous.
***2nd level aggregated archetypes are made by combination of evergreen and deciduous 1st level aggregated archetypes.

6.2.2. Building Phase

In this phase, the modelling of archetypes per NBS Type are related to the web user interface by making use of four core components: offline archetype model, archetype database, calculation service, and web user interface. These components and their relationships are illustrated in Figure 6.5.

As a first step, the biophysical outputs of the archetypes defined in the Design Phase are estimated and later monetised without applying discounting. For the operational life, ES and the costs of each archetype are pre-calculated with the system dynamics model built in Simile, which is compiled and run making use of the open statistical software R (R Core Team, 2020). The outputs are calculated at daily and monthly temporal resolutions, and then aggregated into yearly time steps. This is done because the scope of NBenefit$ is to inform on the performance of NBS over its life cycle. Modelling at detailed temporal resolution is necessary because most of the socio-ecological processes modelled are scale sensitive due to their non-linearity (e.g. water infiltration, transpiration). Otherwise, the model would fail to account for dynamics (e.g. soil water balance) that occur at a much shorter time scale (Almeida and Sands, 2016). Then, without the modelling at a detail temporal resolution a temporal scale mismatch might occur and led to very inaccurate results.

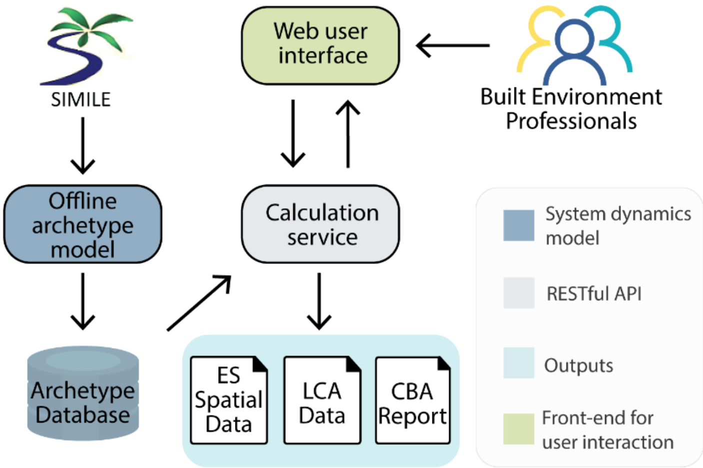

Figure 6.5. Scheme of the main components of the decision support system (DSS) relationship among them and type of outputs; CBA = costs-benefits analysis; ES = ecosystem services; API = application programming interface

As described in Chapter 5, some of the socio-ecological processes (e.g. rainfall, tree morbidity) are modelled as stochastic equations. Therefore, for each archetype, simulations are replicated enough times to ensure that the mean and standard deviation of each output are representative of the range of potential values. To achieve a satisfactory number of replications, successive differences in the yearly mean and standard deviation for all the outputs are analysed with each additional replication (Grandison, 2020). Successively, the final externality costs of implementation, operational and end-of-life stages, are obtained with the support of life cycle assessment software SimaPro (PRéConsultants, 2020). The implementation and end of life stages are represented with no temporal duration as the beginning and end points of the archetype's life.

Once the biophysical outputs, negative and positive environmental impacts, of an archetype are estimated for specific local environmental conditions, they are converted into monetary units (in this Chapter, Euro 2018 for EU-27) and stored in the archetype database. This offline database provides the disaggregated values of the costs and benefits, from which the final values for a specific NBS implementation will be obtained. Then, the archetype database is connected to the web user interface through one RESTful Application Programming Interface (API), i.e. the calculation service. The RESTful API contacts multiple micro-services to execute the calculations. The deployment of these micro-services has been done using container technology. Each of the micro-services is a virtual application running in a container. Docker technology was used to create the container images and docker-compose is used to coordinate the network of containers so that they communicate with each other.

The calculation service is the RESTful API where the benefits and costs for a specific NBS project are re-calculated. This step is necessary because in most cases projects are composed of more than one NBS archetype. Hence, the final costs and benefits are the result of the combination of archetypes, and so results need to be re-calculated. The calculation service assigns archetypes to specific grid cells based on the user's selection. Then, per each specific output (e.g. temperature and humidity regulation), the calculation service provides results per cell. It further aggregates the values of all the cells that compose the NBS project in order to account for the overall costs and benefits of the entire intervention in biophysical and monetary units.

The web user interface is split in two sections: the mapping component and the sidebar (Figure 6.6). The mapping component is presented as a window that embeds an Openstreet map, limited to the geographical area of Europe. It also holds the cell grid. The sidebar helps the user to introduce the different NBS Types and archetypes applied to each cell in the grid or to the global computation. This data, constituted by a combination of archetypes associated with the cells in the grid, is sent to the calculation service to obtain the CBA results. The following section describes in more detail the operation of the web user interface.

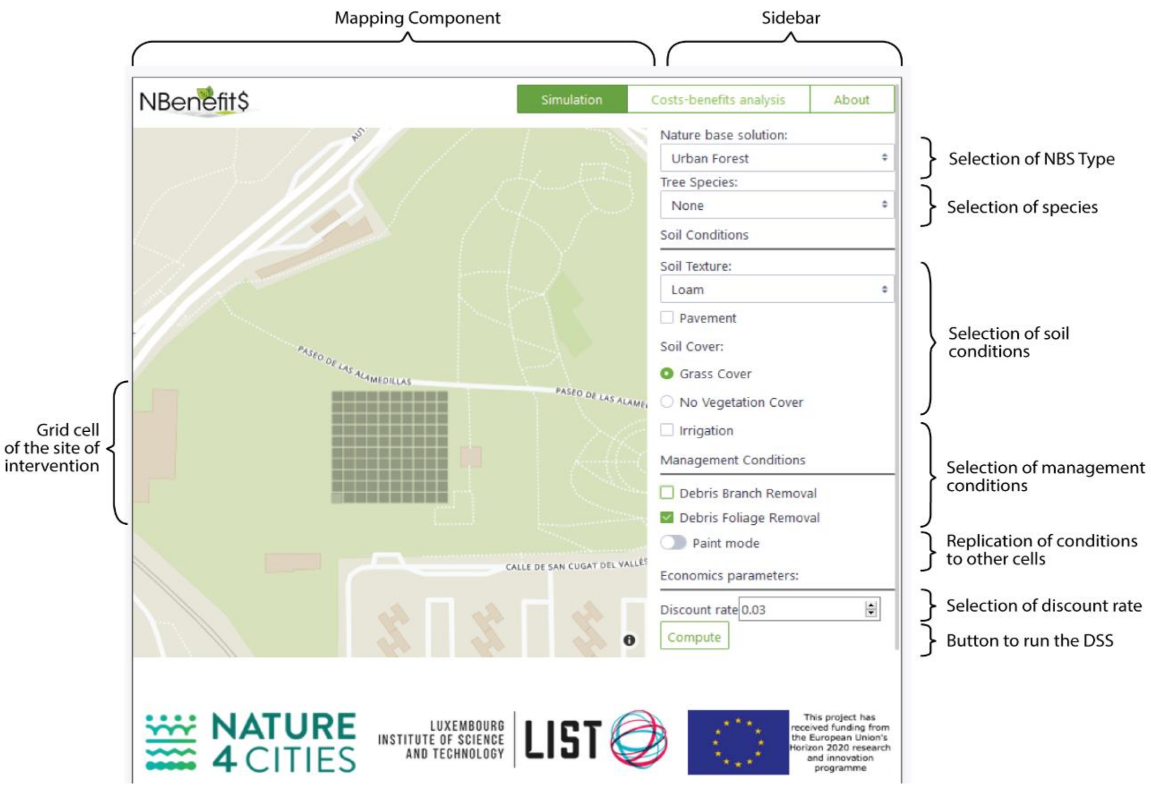

Figure 6.6. NBenefit$ web user interface, components and description

6.2.3. Operation Phase and Outputs

In the operation phase, the end user identifies the environmental conditions experienced by the NBS over its operational life and the core variables of each NBS applied.

First, the mapping component allows the user to look for the specific urban area and site where the NBS project will be implemented. Locations are associated with the parametrisation of specific environmental conditions. After the site of intervention is established in the mapping component, the user can define cell by cell the specific values (e.g. *Celtis australis*, sand, irrigated) of each core variable (e.g. vegetation species, soil texture, irrigation) of the NBS. If the same exact set of input values are shared by more than one cell, the user can activate a “paint mode” function and select the rest of the cells deemed to be equivalent. A specific set of values corresponds to a specific archetype in the database. Some cells can also be left empty if no NBS will be implemented there. Then, by providing the set of values for all the core variables the user indicates the archetype present in each cell (and the overall combination of archetypes for the entire NBS project) for which costs and benefits will be computed. Once the grid cells are characterized, the user can press “Compute”. This last function sends the data to be processed.

The core variables that define an archetype refer to the main abiotic, biotic, and management attributes that would influence significantly the socio-ecological processes and human actions responsible for generating ES and operational costs. In this sense, NBenefit$ allows multiple combinations of archetypes to ensure its applicability to complex projects. For example, a single urban forest might be composed of several tree species, planted at different sizes, and only some of them might be irrigated and/or under intensive management conditions. In addition, the values are presented in qualitative form to ensure they are easily understood by built environment professionals. For example, the DSS only requires the selection of the soil texture class instead of demanding the specific percentage of clay and sand. The tool offers the possibility to tailor the desired level of detail in the outputs in order to match the specific needs at different NBS design and planning phases. Correspondingly, the granularity of input data demanded increases with the level of detail in the output. This facilitates the use of the DSS from the early design/planning stages while providing accurate information at later stages. Both aspects are relevant to ensure the applicability of the DSS to real projects at different design/planning stages and by professionals with different computer-literacy.

As outputs for the overall project, NBenefit$ shows individual graphs for each impact/externality in biophysical and monetary units and a summary of cost and benefits (Figure 6.7). It also provides a downloadable report in PDF format that includes a balance of positive and negative environmental impacts in biophysical units and a cost-benefit analysis. The user can also download a spreadsheet file with the disaggregated benefits and costs per cell and time step, ready to be imported into Geographic Information System (GIS) software. These outputs can facilitate the communication with a broad range of stakeholders and decision makers.

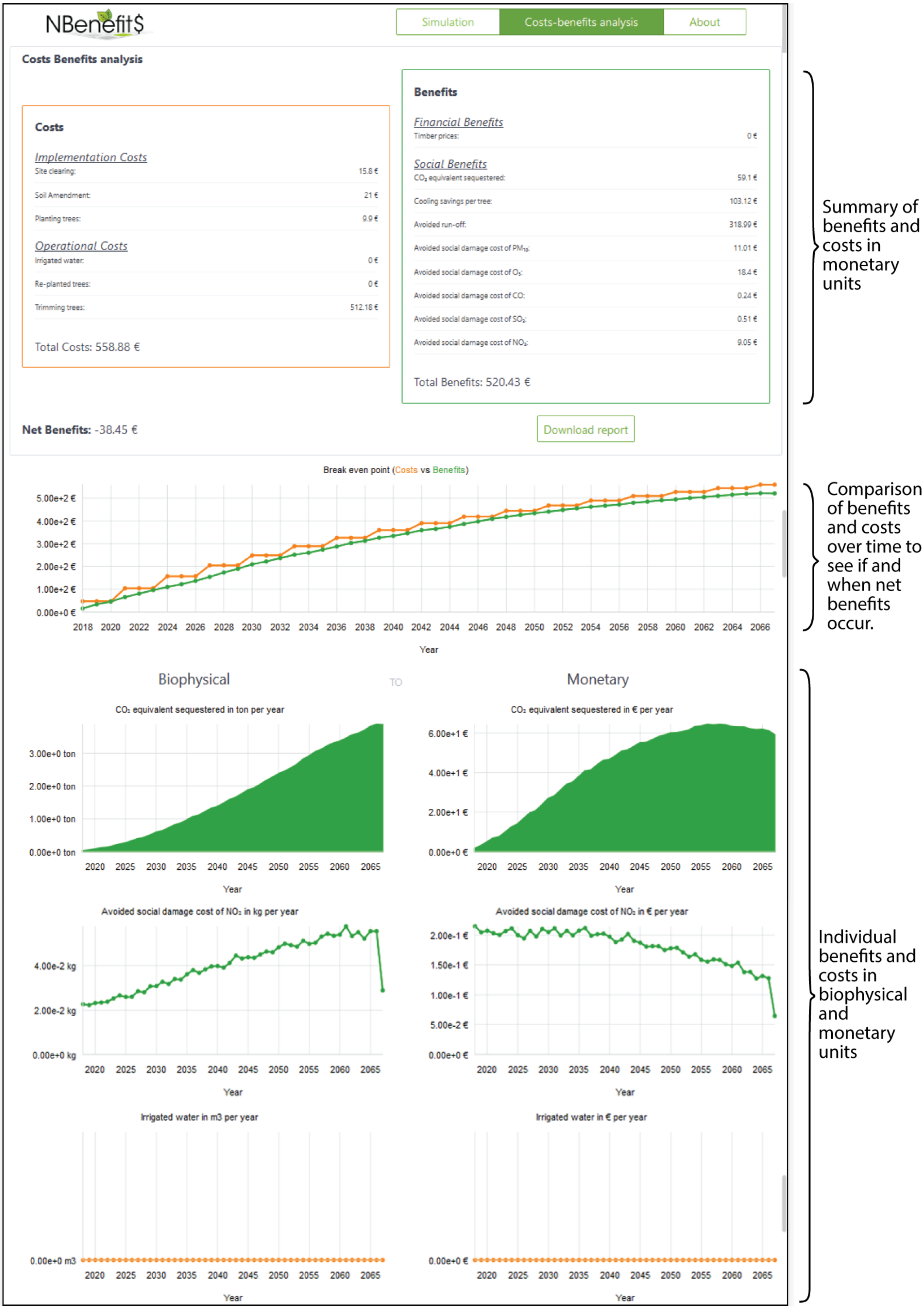

Figure 6.7. Online summary of cost benefit analysis and graph visualisation generated by the web user interface

For the balance of positive and negative environmental impacts (illustrative example in Section 6.3, Figure 6.14), the outputs are provided by process/activity per life cycle stage (e.g. pruning) and as an aggregated total for the entire life cycle. Negative environmental impacts are presented as LCA midpoint impact categories values. Positive environmental impacts as ES classes. For some LCA midpoint impact categories equivalent ES classes exist and therefore a balance over the entire life cycle can be done (see Section 2.1, Table 6.2). This is the case of *Global Warming Potential* that represents global warming impacts in the form of CO_2-equivalent emissions and *Regulation of the chemical condition of the atmosphere*, which represents the mitigation of global warming impacts in the form of biogenic CO_2 storage. However, some ES classes and midpoint impact categories do not have a corresponding equivalent and their total result are just presented distinctly. Furthermore, results by ES class and LCA midpoint impact category are represented in numerical and graphical format.

The graphical format intends to aid the interpretation of the DSS outputs. In the form of bars, the value per impact category for an average cell of the evaluated alternative (i.e. sum of all cell values divided by number of cell) is compared against the value generated by a relevant reference. This reference is represented by the impact of an average person in the world for the year 2010 (see Annex 6.1). This comparison is intended to inform on the magnitude of the impacts associated with the NBS, thus to allow an immediate understanding by non-experts.

In the case of cost-benefit analysis (illustrative example in Section 3), the outputs are provided by process/activity per type of cost and benefit (externality or financial) as well as a net value for the entire life cycle. Results are also presented in numerical and graphical format. In this case, the closest rounded value of the maximum cost or maximum benefit of the alternatives evaluated act as a reference against which the rest of the costs and benefits are compared. This comparison intends to communicate the contribution of each cost and benefit with respect to the net value of the best alternative. As previously anticipated, the spreadsheet file is an output to be used by the built environment professionals in the development of advanced spatial analysis. It includes the geometry data of each of the cells in Well-Known Text (WKT) format and a unique ID per time step (aggregated by year) and cell. The ID and the WKT permits to import the spreadsheet into commercial or open GIS software, which will retrieve the geometry of each cell, its geographical position, and keep the database of benefits and costs as an attribute table. In this sense, the user will be able to visualise the data by cell, filter the grid by specific time-steps, and overlap it onto the outputs of other assessments if these are also georeferenced.

6.3. Application of NBenefit$ to evaluate urban forests

To illustrate the functionality of NBenefit$, 48 urban forest archetypes were prepared for the environmental conditions of Barajas, in the northeast of Madrid (Spain). Section 3.1

provides an overview of these archetypes. Detailed output data for the 48 archetypes is also provided in Annex 6.2. In Section 3.2, three alternatives are tested for a hypothetical project of a small urban forest (0.1 Ha, 10 cells).

6.3.1. Biophysical and monetary outputs of NBenefit$ for 48 urban forest archetypes

The archetypes are presented in a graphical format (Figure 6.8) using grid cells such as the ones used in NBenefit$. The overall performance of each archetype in biophysical units as well as in monetary units are illustrated in Figure 6.9 and 6.10. Discounting is not applied for this illustrative exercise.

Tree Species & Size: Small Quercus ilex	Small Pinus sylvestris	Small Platanus acerifolia	Soil: Soil Texture, Soil Cover, Paving		Management: Litter Removal, Pruning, Irrigation
1	2	3	Clay, Covered by Grass, Non Paved	+	Low Maintenance, No Pruning, No Irrigation
4	5	6	Clay, Covered by Grass, Non Paved	+	Medium Maintenance, Pruning, No Irrigation
7	8	9	Clay, Covered by Grass, Non Paved	+	Low Maintenance, No Pruning, Irrigation
10	11	12	Clay, Covered by Grass, Non Paved	+	Medium Maintenance, Pruning, Irrigation
13	14	15	Clay, Covered by Grass, Paved	+	Low Maintenance, No Pruning, No Irrigation
16	17	18	Clay, Covered by Grass, Paved	+	Medium Maintenance, Pruning, No Irrigation
19	20	21	Clay, Covered by Grass, Paved	+	Low Maintenance, No Pruning, Irrigation
22	23	24	Clay, Covered by Grass, Paved	+	Medium Maintenance, Pruning, Irrigation
25	26	27	Sand, Covered by Grass, Non Paved	+	Low Maintenance, No Pruning, No Irrigation
28	29	30	Sand, Covered by Grass, Non Paved	+	Medium Maintenance, Pruning, No Irrigation
31	32	33	Sand, Covered by Grass, Non Paved	+	Low Maintenance, No Pruning, Irrigation
34	35	36	Sand, Covered by Grass, Non Paved	+	Medium Maintenance, Pruning, Irrigation
37	38	39	Sand, Covered by Grass, Paved	+	Low Maintenance, No Pruning, No Irrigation
40	41	42	Sand, Covered by Grass, Paved	+	Medium Maintenance, Pruning, No Irrigation
43	44	45	Sand, Covered by Grass, Paved	+	Low Maintenance, No Pruning, Irrigation
46	47	48	Sand, Covered by Grass, Paved	+	Medium Maintenance, Pruning, Irrigation

How to read the inputs of each archetype?
E.g. Archetype 11 = Pinus sylvestris planted small in a clay soil covered by grass and non paved, which will be under a low maintenance, with safety and/or formation pruning and with irrigation

Figure 6.8. Input characteristics of 48 illustrative urban forest archetypes modelled for the local conditions of Barajas in the northeast of Madrid (Spain).

The simulation of archetypes was replicated 75 times after an analysis of the successive difference in the mean and standard deviation of yearly outputs per archetype. This analysis showed that the 75 iterations were enough for most of the variables to be representative of the potential range of values (Illustrative examples visualised in Annex 6.3; numerical values for all the variables and archetypes in Annex 6.4). The values of standard deviation are provided as part of the biophysical outputs to inform the user about the known uncertainty of the results (Figures 6.11 to 6.13). The standard deviation of the biophysical output is also used to compute the lower and upper monetary benefits and costs (Figures 6.10 to 6.13).

As Figure 6.9 and 6.10 illustrate, the 48 archetypes perform rather differently in terms of benefits (positive externalities) and costs (externalities and financial). For example, in the case of archetype 9, the ES *Regulation of temperature and humidity* is the dominant benefit and *Irrigation* and *Management of waste from litter* the dominant financial and externality costs. Instead, for archetype 19, the ES *Regulation of chemical condition of the atmosphere* is the dominant ES in terms of monetary value, and *Management of waste from* litter the dominant financial and externality cost. However, when looking at the performance of archetype 19 in biophysical terms, it can be seen that it performs quite well for all the ES *Filtration of air pollutants by plants* compared to other archetypes, even if the absolute monetary values in terms of avoided damage are not significant.

The overall results also help to see how much the change in one or two input characteristics could affect the performance of an NBS. For example, archetypes 1 and 13 only differ in one input characteristic, the former has a non-paved soil and the latter a paved one, but that difference has a strong impact in the performance of the NBS regarding benefits and, partially, regarding costs. This is explained by the fact that paved soils impede infiltration of water, hence reducing water availability for the tree. As a consequence, the probability of water stress increases and the tree transpiration gets limited due to the lack of available water. Additionally, reduced transpiration also limits dry deposition of pollutants and the growth rate of the tree, and therefore diminishes carbon storage.

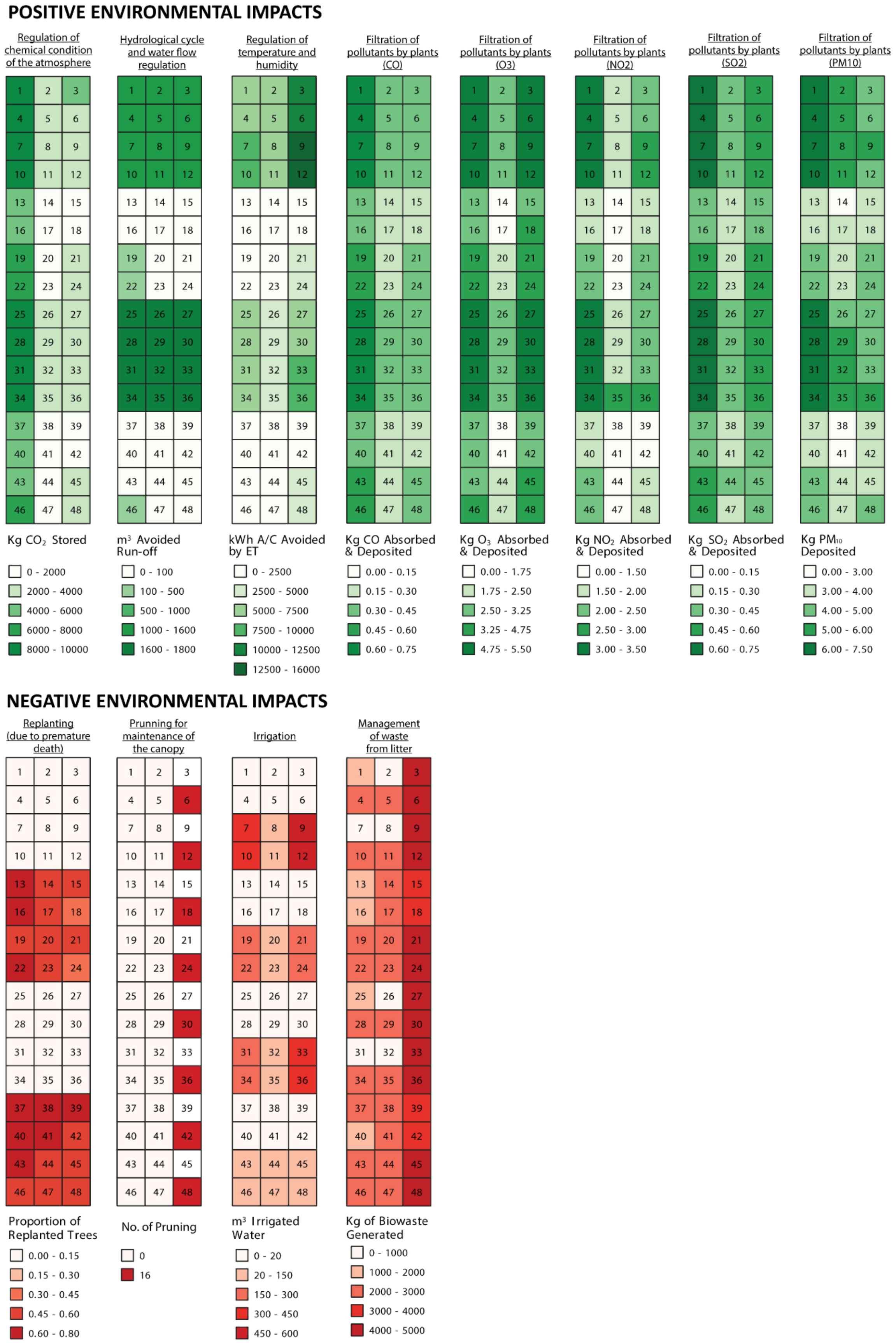

Figure 6.9. Mean cumulative benefits and costs over the operational stage of the 48 urban forest archetypes in biophysical units.

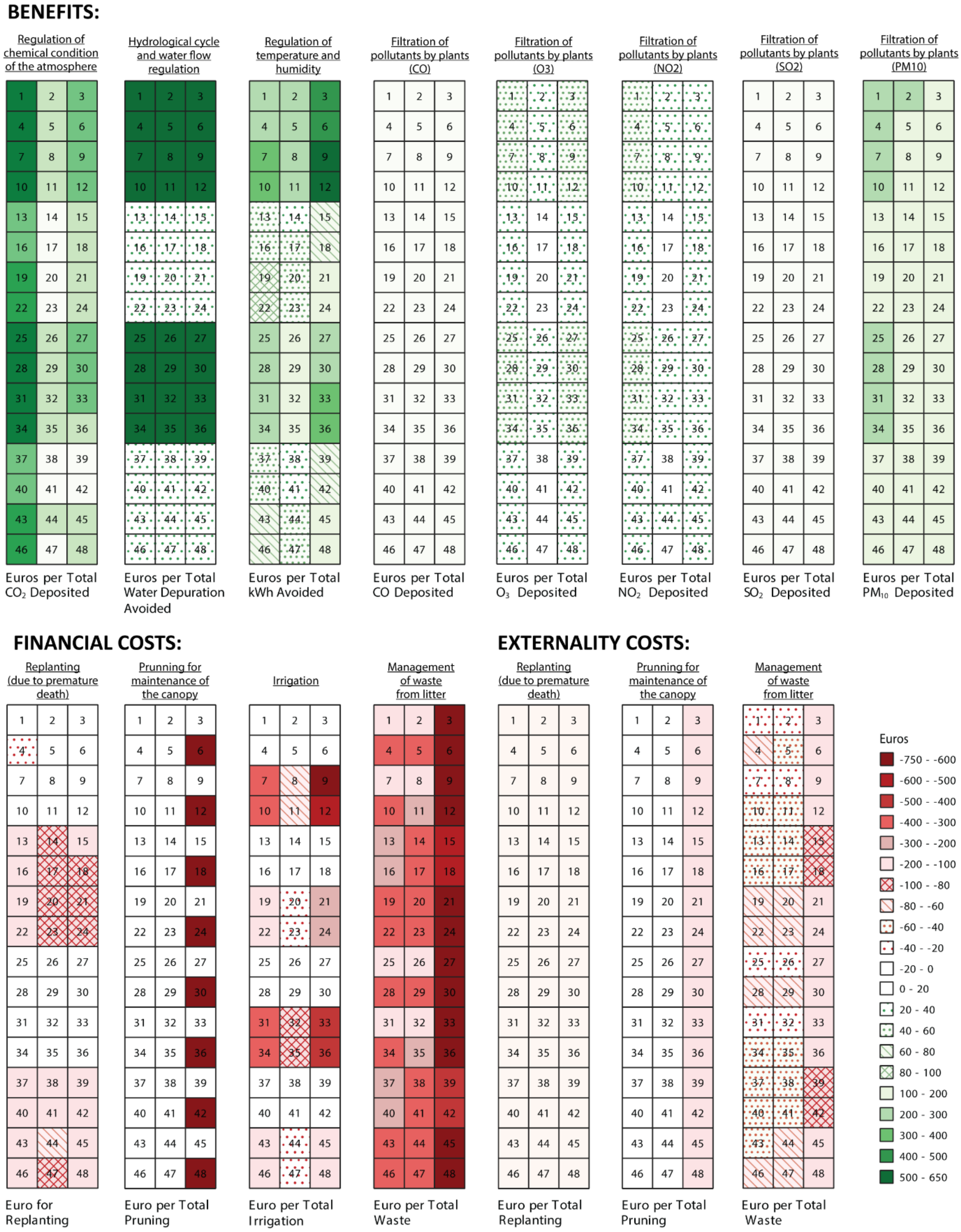

Figure 6.10. Mean cumulative benefits and costs (externalities and financial) over the operational stage of the 48 urban forest archetypes in monetary units.

Displaying results disaggregated by year and by ES and cost category (see Figures 6.11 to 6.13) helps to show the performance of an archetype over time and how long it takes to generate a significant benefit or cost. For example, in the case of *Regulation of temperature and humidity* (Figure 6.13), it can be seen that the yearly benefit is not always increasing and the change occurs at different rates. Figure 6.13 informs on how variable this benefit can be compared to others since it does not only depend on input characteristics of the archetype, but also on the variability of the meteorological conditions. Concurrently, including the standard deviation in the graphs as well as lower, central and upper values for the monetisation increases the transparency of the results. It allows the end user to better evaluate whether changes in performance between archetypes are relevant or not in each specific case. For example, archetype 2 provides a higher *Regulation of the chemical condition of the atmosphere* than archetype 48, but as Figure 6.12 illustrates this difference might not be significant in terms of monetary benefits. Instead, when both archetypes are compared against archetype 1 it appears clearly that the latter outperforms them.

The different visualisations offered by NBenefit$ show the practical utility of the DSS for built environment professionals and decision makers during the definition of a project. For example, a professional who has to fulfil cost-efficiency or sustainability criteria or who needs to demonstrate a project return of investment might be more interested in the cumulative results of a solution. Instead, the disaggregated results in biophysical and monetary units would be more appealing for professionals requiring a detailed evaluation of specific aspects of NBS performance.

Disaggregated results could also be more interesting in situations when some ES or costs are the key points of interest for local circumstances. For instance, in a city like Madrid that shows recurrent problems of air pollution, the biophysical performance of a tree in terms of *Filtration of air pollutants by plants* might become more relevant than other ES and cost categories. In those cases, the preference of archetype 19 for decision makers might be driven more by its biophysical performance than its economic one.

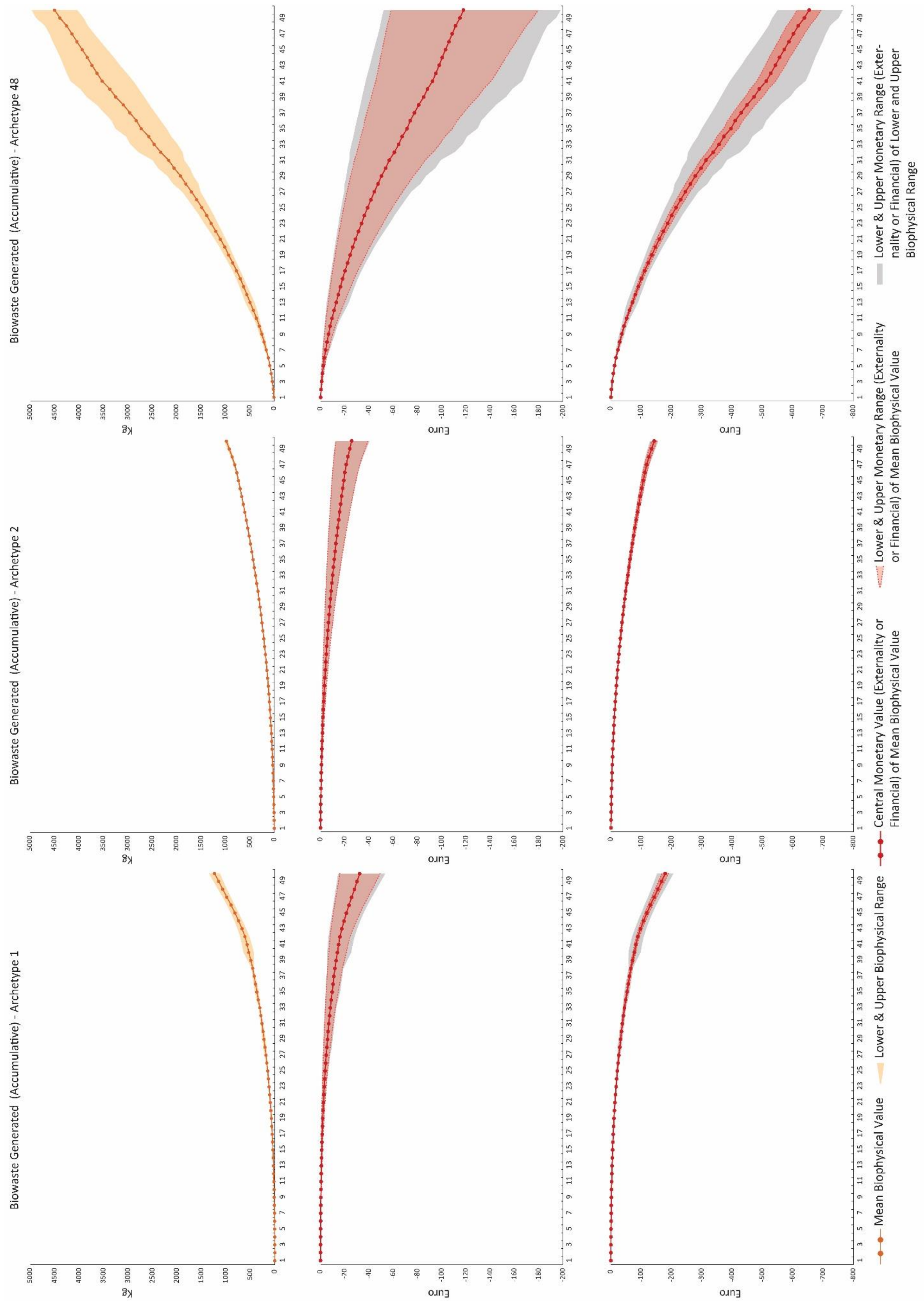

Figure 6.11. Comparison of biophysical and monetary performance over time for "Management of waste from litter" for archetypes 1, 2 and 48. Results are shown as accumulative values over time (i.e. value at year 50 is the accumulated value from Year 1 to 50).

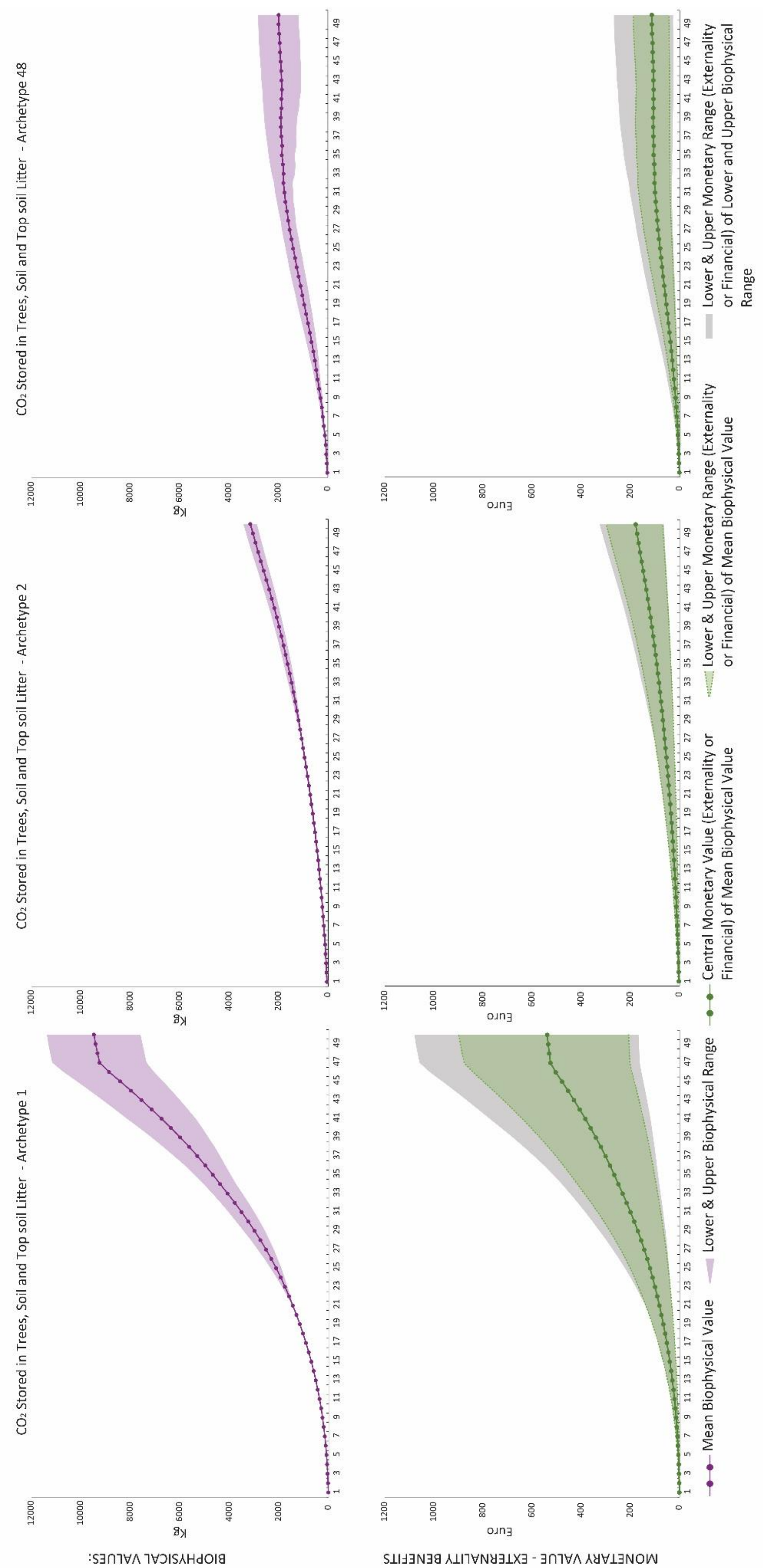

Figure 6.12. Comparison of biophysical and monetary performance over time for "Regulation of chemical condition of the atmosphere" for archetypes 1, 2 and 48. Results are shown as accumulative values over time (i.e. value at year 50 is the accumulated value from Ye

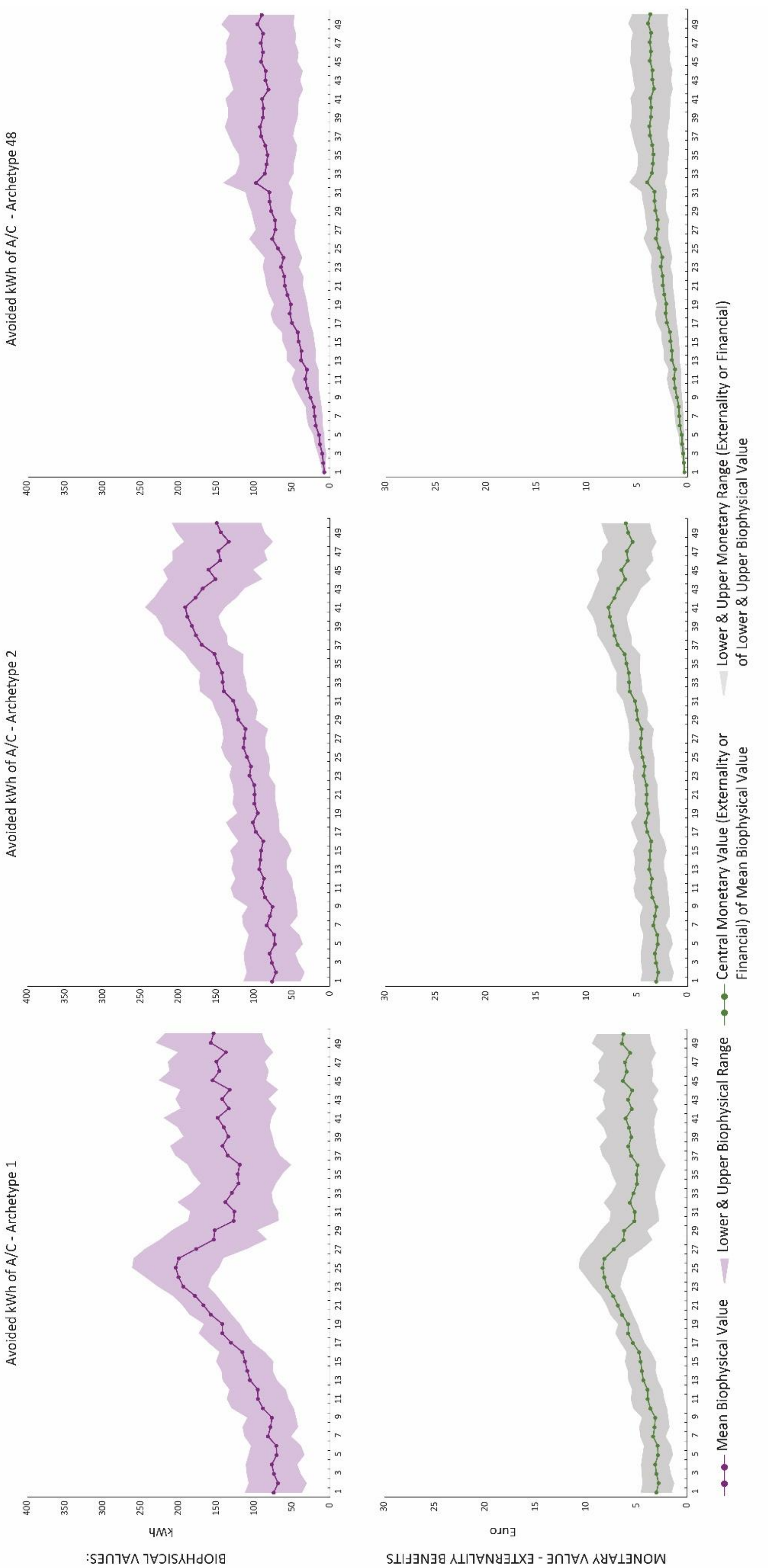

Figure 6.13. Comparison of biophysical and monetary performance over time for “Regulation of temperature and humidity” for archetypes 1, 2 and 48. Results are shown as non-cumulative values (i.e. values represent the ones produced each year)

6.3.2. Evaluation of three hypothetical alternatives for a small urban forest

Three alternatives for a small urban forest were designed and tested to illustrate the value of the synthetic cost and benefit analysis report of NBenefit$. The characteristics of each alternative are summarised in Figure 6.14. Figure 6.15 and 6.16 show synthetic cost-benefit analyses of the alternatives in biophysical and monetary units.

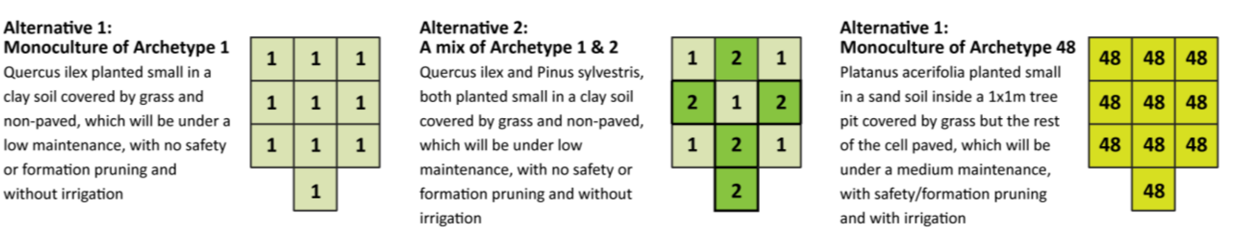

Figure 6.14. Characteristics of three hypothetical alternatives for a small urban forest of 0.1 Ha

For the hypothetical alternatives, archetype 1 (including a broadleaved evergreen species) and archetype 2 (including a coniferous species) are selected because are among the best performers for most of the benefits as well as among the ones that have the least financial and externality costs. In the same sense, archetype 48 (including a broadleaved deciduous species) is selected because it is among the worst performers of the 48 archetypes as well as among the archetypes with higher costs.

As illustrated in Figure 6.15, alternative 3 is clearly the worst option in terms of environmental impacts. In all the stages, it is the one with the highest negative environmental impacts and the lowest positive environmental impacts. In fact, there is only one action, i.e. planting, in which the three alternatives perform equally. This is because the elements used to characterise impacts are assumed to be the same in the underlying model (i.e., the time trees have spent in the nursery, transport distance and planting techniques). However, when comparing alternative 1 and 2, it is not clear if there is a significant difference in performance. Alternative 1 performs clearly better on *Regulation of chemical composition of the atmosphere*, but it contributes more to *Human carcinogenic toxicity*. For the rest of the categories, the numerical differences are minimal, and the graphical comparison of each category against the reference (impact of an average person) reinforces this interpretation. Since the biophysical evaluation does only give one side of the picture about performance, this is one of the situations where monetary valuation can complement the analysis.

In Figure 6.16, it can be seen that alternative 1 has a total net benefit 10% higher than alternative 2 and alternative 3 remains the worst performer, with a negative total net benefit. However, the user can also see that alternative 2 has slightly lower total financial costs, which means a reduced negative cash flow over the NBS life. Given the disaggregation of cost and benefit voices, the user can also identify the structural differences in the economic performance. For instance, Figure 6.16 shows that the largest difference between the net benefits in alternative 1 and 2 is due to *Regulation of chemical composition of the atmosphere*.

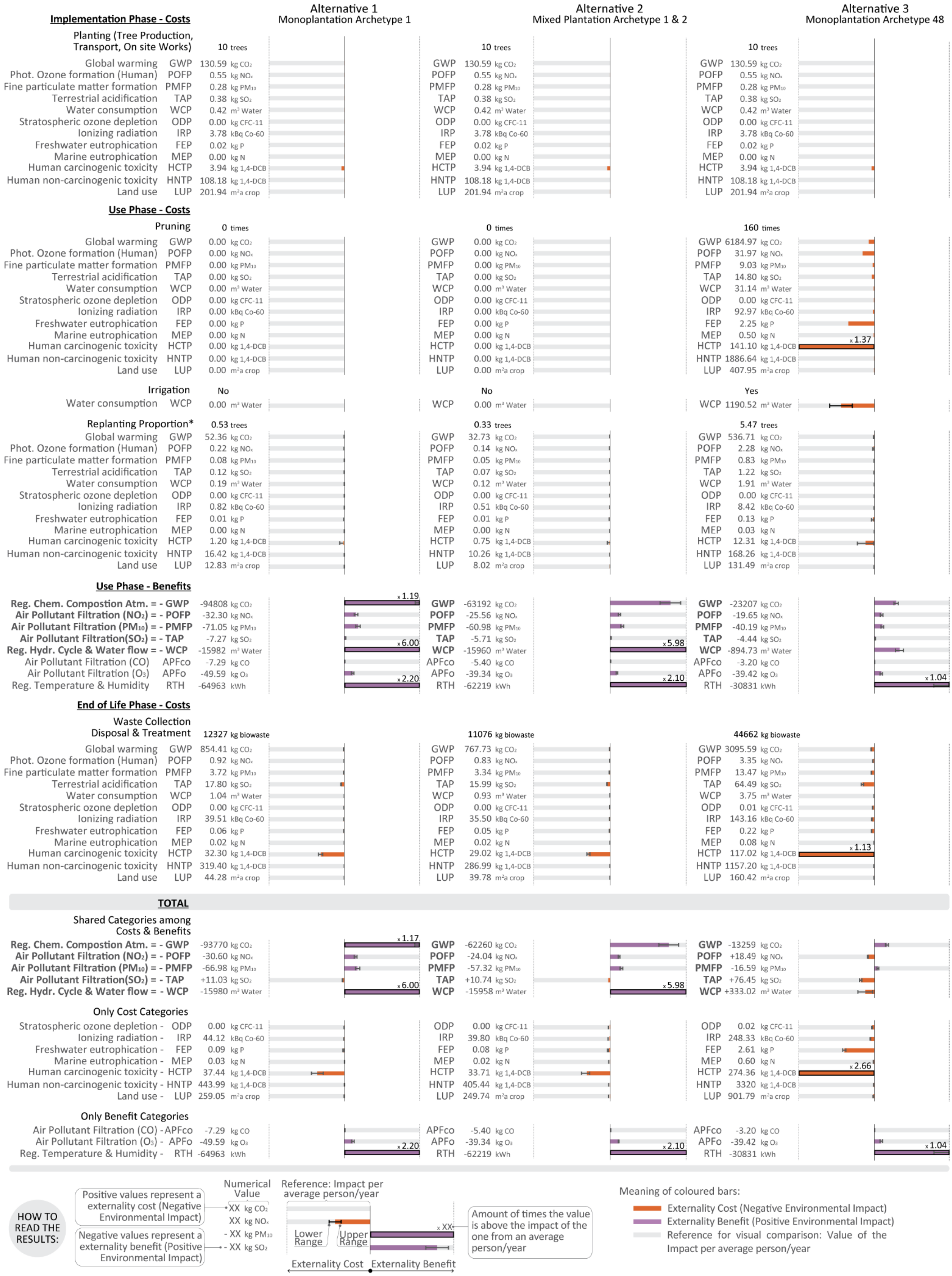

Figure 6.15. Comparison of benefits and costs in biophysical units per life cycle stage and in total. Externality costs are represented in orange and externality benefits in purple.

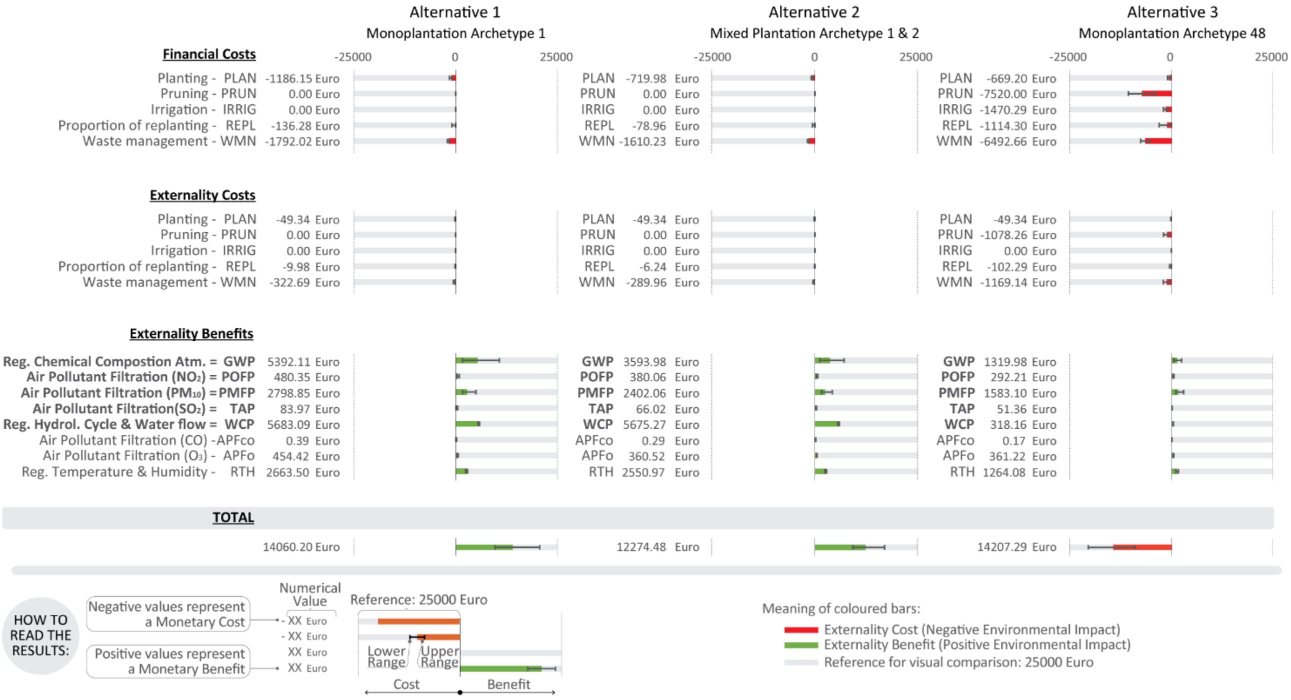

Figure 6.16. Comparison of the benefits (externalities) and costs (financial and externalities) in monetary units per life cycle stage and as a net total value.Costs are represented in red and benefits green and benefits in green.

6.4. Discussion and conclusions

The paper presented the development of NBenefit$, a prototype DSS for the evaluation of NBS projects over their entire life cycle, i.e. from the implementation and operation stages of the NBS, until the end-of-life stages of the solution components. It is described how archetypes of NBS Types for specific environmental conditions can be created, pre-calculated and stored, as well as their posterior utilisation by built environment professionals to evaluate several alternatives for a specific NBS project. It is also illustrated the utilisation of NBenefit$ for an urban forest project, the first NBS Type integrated in the prototype DSS. The following paragraphs discuss limitations and advantages of NBenefit$, and provide a roadmap to move toward a DSS fully operational and accessible to built environment professionals.

The generation of multiple archetypes defined by a specific combination of core attributes provides to the end user an easy form to input variants of a general NBS Type used in a project. The end user is released of tedious tasks and an excessive petition of data, which in some cases (or at some stages of the project) are unknown. However, pre-calculation of new NBS archetypes is a highly time-consuming task from the developer's side. For example, the simulation of the 48 archetypes presented in this paper required around two hours for one replication of the entire set. This means that around one week of time was needed only for the 75 replications, without considering their revision and preparation for the database.

Moreover, the inclusion of new archetypes is dependent upon the availability of data for its pre-calculation in the system dynamics model. For example, in the case of urban forests, tree growth is modelled making use of allometric equations, which need to be already available in published studies or databases. In addition, in some cases the use of a pre-established set of values would not be enough for an end user that needs a high precision in the definition of the solution evaluated. In those cases, the direct use of the offline system dynamics model developed in Chapter 5 might be a suitable alternative.

The balance of environmental impacts of an entire NBS life cycle (in biophysical units) as well as the cost-benefit analysis aggregated at yearly basis are useful for design purposes, but also for communication with a broad range of stakeholders. In fact, as illustrated in Chapter 3, there is still a lack of software and tools that can acts as DSS on urban ES or NBS studies that offer both types of output, are focused on detailed spatial scales, and provide long-term performance outputs understandable by a broad audience.

A variant of Table 3.3, including only relevant tools, is provided in Table 6.3 to support a brief comparison between NBenefit$ and current DSS on urban ES or NBS. The new ARIES and GreenPass still in development during the drafting of the PhD thesis have also been included. Several of the current DSS such as InVEST, ARIES, and LUCI are starting to develop models suitable for urban areas. Despite their proposed models can inform how urbanisation impact the ES supplied by surrounding rural areas and define natural elements making use of land use/cover classes, they do not focus on the detailed spatial and categorical level needed for defining site interventions (e.g. Zank et al. 2016; Trodahl et al. 2017; Balbi et al. 2019; Zhang et al. 2019). Only few DSS tools are suited to detailed urban ES calculation, such as: i-Tree (Hirabayashi, Kroll and Nowak, 2012), specific for urban forests; ENVI-met (Huttner and Bruse, 2009), specific for local climate regulation; and SWMM (Burszta-Adamiak and Mrowiec, 2013), not directly addressing NBS but modelling natural storm-water management solutions. Compared to NBenefit$ some of these tools (ENVI-met, SWMM) offer a detailed temporal resolution, able to work in hourly time steps and inform about the performance of NBS during specific events such as intense storms. Nevertheless, these tools are not focused on modelling long-term performance, they do not explicitly consider negative environmental impacts and negative externalities as part of their calculation and in some cases (e.g. ENVI-met, SWMM) they are not easy to use by low-computer-literate users. NBenefit$ and i-Tree are the only tools offering both biophysical and monetary values, and in a format easy to communicate to stakeholders. However, NBenefit$ also includes standard deviation values to inform the user about the known accumulated uncertainty.

Currently, NBenefit$ is a prototype that have been tested only with urban forest as NBS type. In the future, additional system dynamics models of other NBS have to be incorporated in the tool increasing the utility of the DSS for built environment professionals that develop complex urban green open spaces. In addition, the performance output generated for each alternative does not take into account all possible costs and ES. For example, it does not consider how the planting mix could influence pest vulnerability or aesthetics. Then, in future versions of NBenefit$ calculation of additional costs and ES should be incorporated to provide a more complete picture of NBS cost-effectiveness.

Table 6.3. Synthetic comparison of well-known DSS tool for ES to NBenefit$

Tool	Urban /Rural	Spatial level	Dynamic simulation	Outputs	Uncertainty Represented	Type of value	Online Offline	User	Source
LUCI	Rural	Watershed/ metropolitan	No	ES	No	Biophysical	Offline	Technical user & Decision maker	https://www.lucitools.org
InVEST	Rural	Watershed/ metropolitan	No	ES	Yes	Biophysical	Offline	Technical user & Decision Maker	https://naturalcapitalproject.stanford.edu/invest/
ARIES	Rural	Watershed/ metropolitan	No	ES	No	Biophysical	Offline	Technical User	http://aries.integratedmodelling.org/
i-Tree	Urban	Metropolitan to site level	Yes	ES	No	Biophysical and monetary	Offline	Technical User & Decision Maker	https://www.itreetools.org
ENVI-met	Urban	Metropolitan to Site level	Yes	Like ES	No	Biophysical	Offline	Technical User	https://www.envi-met.com/
SWMM	Urban	Metropolitan to site level	Yes	Like ES	No	Biophysical	Offline	Technical User	https://www.epa.gov/water-research/storm-water-management-model-swmm
GreenPass	Urban	Site level	Yes	Like ES	Unknown	Biophysical	Offline	Built environment Professional. (Non technical expertise required)	https://greenpass.at/
NBenefit$	Urban	Site level	Yes	ES, Externality Costs, and Financial Costs	Yes	Biophysical and monetary	Online	Built Environmental Professional (Non technical expertise required) & Decision Maker	-

Currently, inside each cell, the system dynamics model underneath NBenefit$ is able to represent the interactions of socio-ecological processes to compute ES and operational costs, thus offering the possibility to visualise trade-offs and synergies. However, an advanced version of NBenefit$ should be able to account for cell interaction. At present, NBenefit$ does not acknowledge how adjacent cells influence each other. Moreover, aspects such as space competition between trees in adjacent cells or lateral exchange of soil water are not taken into account. Including a function of cell interaction would allow to create a large number of potential combinations.

In terms of user experience, the current data requirement framework might be tedious in the case of large projects. The integration of additional alternatives of inputting archetypes would improve the experience, especially for advanced users. On the side of outputs, providing an online visualisation of the disaggregated spatialized outputs would help overcome the need of additional steps and the use of GIS software. Similarly, an immediate integration of results into Building Information Modelling (BIM) software could be time saving option for built environment professionals. In that case, it might be adequate to provide the long-term benefits and costs of disaggregated archetypes (or groups of them) in the format of BIM template tables. BIM is more and more used by different built environment professionals and in some countries a certain level of BIM adoption (e.g. United Kingdom) is required for all public-sector works (Landscape Institute 2016). Then, BIM protocols and software are expected to be broadly used by the entire construction sector in the near future. For the abovementioned future works, the collaboration with small groups of built environment professionals of different areas (e.g. landscape architecture, civil engineer, architecture) might be useful. This would help in the further definition of costs of interests that might not be that obvious. It could also be useful to make NBenefit$ more friendly for different groups of professionals.

Despite further work still needs to be developed, the current prototype of NBenefit$ has demonstrated its unprecedented value for assessing NBS and provide support to decision-making. In fact, it is one of the rare examples of DSS on NBS developed to inform built environment professionals with low computer-literacy about the overall long-term cost and benefits of new NBS projects. Ultimately, the software anticipates the cost-effectiveness of NBS projects, which further gives the knowledge basis for more informed implementation of new NBS in urban environments.

Chapter 7

Conclusion

7.1. Summary of the main findings

This section provides a recapitulation of the manuscript, where the thesis statement is discussed. It is followed by a general discussion, Sections 7.2 to 7.5, which includes the strengths and drawbacks of the work presented. It finalises with Section 7.6, where perspectives on key areas of research to further advance the work presented in this thesis are introduced.

The overall aim of this PhD thesis was to develop a coherent and practical methodological procedure for the environmental and economic assessment of nature-based solutions (NBS) in urbanised contexts based on rationales and models underpinning landscape ecology, urban ecology, ecosystem services (ES) and life cycle thinking. The underpinning scope for the development of this methodological procedure was to confirm or reject the thesis statement:

NBS are adequate mechanisms for enhancing sustainability and resilience of urbanised contexts and the combination of methods from landscape ecology, urban ecology, ecosystem services, and life cycle thinking can help to understand to which extent and how effective is their contribution to sustainability and resilience.

In order to confirm the thesis statement, the two research questions from which it derives had to be answered in affirmative.

Question one: Are nature-based solutions suitable interventions to enhance the sustainability and resilience of urban contexts and their hinterland?

Question two: Does a combination of ecosystem services, landscape ecology, urban ecology, and life cycle thinking methods help to move towards a practical and coherent assessment of the contribution of NBS to the sustainability and resilience of urbanised contexts?

Based on the results from Chapter 2, 4 and 5 it can be confirmed that the answer to the first research question is affirmative. As it was demonstrated in a qualitative level in Chapter 2, from a list of 18 common global urban challenges (UC) for sustainability and resilience, NBS are potentially able to mitigate eight[11] of them through the ES that they supply. Depending on their specific implementation model (i.e. underlying governance, finance and business mechanisms), they can contribute to mitigate four[12] additional UC. As it was demonstrated in a semi-qualitative level in Chapter 4, the implementation or maintenance of NBS (e.g. better management of existing ecosystems) in peri-urban areas and in the non-built land covers of the urban regions could act as key patches for the movement of some animal species in the territory, and therefore for the maintenance of biodiversity. This was clearly illustrated in Figure 4.6 with cases such as the one of *Maculinea arion*, where a future housing development proposed by the Plan Sectoriel du Logement of Luxembourg might destroy a key grassland patch for the maintenance of the ecological connectivity of a population of this species in the southern territory of the country. Finally, as it was demonstrated in a quantitative level in Chapter 5 for the case of urban forests (a variant of the NBS woodland-like), NBS are capable to provide a net positive contribution to the sustainability and resilience of urbanised contexts. In other words, as Figure 5.11. showcased, in the long term the positive externalities of NBS (derived from the ES supplied by them) can overpass their negative externalities (derived from the negative environmental impacts) and financial costs, contributing to enhance the sustainability of urbanised contexts. Moreover, as Figure 5.8 exemplified, urban forests can positively contribute to the supply of ES such as *regulation of temperature and humidity* (see Figure 5.8) helping to cope with recurrent stresses such as heatwaves and contributing to the resilience of urbanised contexts. Therefore, this thesis demonstrated that NBS are suitable interventions to enhance the sustainability and resilience of urban contexts and their hinterland.

Furthermore, the results from Chapter 3, 4, 5, and 6 can confirm that the answer to the second research question is affirmative. As it was showcased in Chapter 3, different ES assessment methods have different capacities and are more or less adequate to evaluate ES and the NBS that provide them. The adequacy of the methods depends on the specific ES class of interest, the scope of the assessment (e.g. ES supply or ES demand), the spatial level considered, and the need or not of accounting for temporal changes. As it was also showcased in Chapter 3, methods from life cycle thinking such as life cycle assessment (LCA), and life cycle costing (LCC) can be used to provide a detailed assessment of negative environmental impacts and financial costs of NBS over their entire life cycle, overcoming current limitations. In this sense, the combination of different ES methods would permit an assessment of multiple ES, whilst the inclusion of LCA and/or LCC would permit a detailed consideration of negative impacts (environmental and economic). However, standard LCA and LCC methods by themselves cannot consider the importance of the spatial composition and configuration and neither temporal changes in the assessment of impacts. Instead, as illustrated in Chapter 4, a combination of landscape ecology techniques can take into

[11] These urban challenges are: Urban Environment, Physical Health, Mental Health, Climate Change, Material and Solid Waste Management, Water Management, Energy, and Biodiversity.

[12] These urban challenges are: Socio-spatial Equity, Social Cohesion, Social Vulnerability, and Green and Circular Economy.

account the role of the spatial configuration and composition for the assessment of ecological connectivity, and therefore the evaluation of impacts to local biodiversity. Similarly, as illustrated in Chapter 5, a system dynamics model integrating methods applied in urban ecology, such as urban tree allometric equations, can acknowledge the importance of the temporal dimension overcoming limitations of standard LCA and LCC methods and simple ES methods. In this sense, as Figure 5.2 clearly summarised, a modelling framework combining the abovementioned methods provides a more coherent and complete assessment than each of the single methods and disciplinary approaches alone. Finally, as it was illustrated in Chapter 6, this modelling framework that in principle is quite complex due to the multiple inputs and mathematical relationships, and which therefore could be very time consuming, hampering its practical use, can be adapted into a user-friendly decision support system (DSS). Such DSS can remove the time burdening from the users' side, encouraging decision makers and built environment professionals to integrate the tool in their workflows at different planning/design stages of urban NBS interventions. This means that the prototype DSS, after further development, could become a tool to be used during the resolution of practical urban planning and landscape design problematics related to NBS. Thus, this thesis demonstrated that a combination of ecosystem services, landscape ecology, urban ecology, and life cycle thinking methods help to move towards a practical and coherent assessment of the contribution of NBS to the sustainability and resilience of urbanised contexts.

Consequently, the affirmative replies to both research questions confirmed the thesis statement. However, the findings in this thesis go beyond a black and white statement, and therefore providing the "nuances" is also relevant. As anticipated, the next sections discuss the "nuances" with respect to NBS as a solution for societal challenges, the completeness of the methodological procedure and the DSS.

7.2. The potentialities and limitations of nature-based solutions: breaking with its conceptualisation as a panacea solution for societal challenges

As it was confirmed in the last section, NBS can help to enhance the sustainability and resilience of urbanised contexts. However, this does not mean that they are always the most suitable solutions neither that all of them are adequate to solve all types of societal challenges. The following lines describe key limitations and potentialities of NBS identified in this thesis.

Chapter 2 clarified that NBS by themselves are not capable to solve societal challenges that are mainly rooted in limiting factors of technological or human-social nature. In fact, Chapter 2 also states that the capacity of NBS interventions to provide a net positive contribution to urbanised contexts might be hampered by an inadequate consideration of the UC *governance, public participation* and *public expenditure*. Moreover, beyond the conceptual level, NBS interventions are not neutral or apolitical, as with any type of applied urban planning action a political component is always embedded (Haase *et al.*, 2017; Kotsila *et al.*, 2020). The last statement recalls what is introduced in Chapter 2 as the importance of clearly define *how, where,* and *for whom* NBS are implemented in urbanised contexts. The definition of the previous aspects, which needs to be done integrating social actors with different and sometimes contradictory interests, will ensure that the social scope of specific NBS interventions are explicitly stated in urban agendas before they are implemented.

When planning NBS for specific local contexts, it is also necessary to remind that:

i) urban contexts are hybrid systems under continuous evolution result of human, natural and technological patterns, processes and feedbacks occurring at multiple spatio-temporal levels (Alberti, 2016);
ii) many urban planning/design issues are by default “wicked problems”, i.e. problems difficult to overcome due to incomplete, contradictory and changing requirements difficult to recognise in advance (Prominski and Seggern, 2019)

As a consequence of the above points, what appears as potential beneficial urban planning actions based on past implementations in specific urbanised contexts might not end offering the same results in others. As stated in Chapter 2, this issue can also occur to NBS interventions, and therefore their planning, design and management needs to be sensitive to the specific socio-spatial contexts (result of the abovementioned patterns, processes and feedbacks) in which they are implemented. In this sense, the precautionary principle should be applied before promoting specific NBS in developing nations that have been mostly tested in northern developed countries (Dobbs *et al.*, 2019). Similarly, ensuring the best possible implementation of NBS interventions should go beyond the use of quantitative modelling tools. Local experts and social actors, such as citizen collectives, should always be involved because they might better recognize hidden requirements, contradictions or constraints than an external consultant.

Despite the above drawbacks, NBS can supply multiple ES classes of different ES sections (and their derived benefits), contributing to mitigating multiple societal challenges. This multifunctional character is something generally missing in grey urban solutions (e.g. air conditioners, sewage treatment works) that tend to be designed and optimised to fulfil a unique function. Additionally, compared to grey urban solutions, NBS are the only capable to generate or maintain natural capital in urbanised contexts, and its derived benefits, which according to a strong sustainability perspective, it is not substitutable by other types of capital. Moreover, they are a unique type of solution because as it was illustrated in Chapter 5, their performance and derived value increase over time with their maturity. On the contrary, the performance of many grey solutions tends to decrease over time due to their damage or deterioration. Finally, they are key to maintain biodiversity in our cities and their surrounding areas as showcased in Chapter 4.

To sum up, NBS can contribute to urbanised contexts in forms that grey solutions cannot do. However, it is necessary to keep in mind that as with any type of solution NBS have finite functions, even if they are multiple. Therefore, NBS should never be understood and presented to society as a panacea for all types of societal challenges.

7.3. Advantages and drawbacks of the combined methodological procedure to assess nature-based solutions

As it was confirmed in section 7.1, and previously discussed in Chapter 5, through the integration of methods from different disciplinary approaches the combined methodological procedure can offer a more comprehensive assessment of NBS that each of the individual methods of which it is composed. It goes beyond the consideration of positive environmental impacts incorporating negative environmental impacts, economic values and partially social ones, since most of the benefits quantified are public and therefore represent societal benefits. Other emerging works such as the one published recently by Speak *et al.*, (2020) are also stressing the potentialities of combining ES and LCA methods to quantify positive and negative environmental impacts of urban forests. Similarly, Nowak, Greenfield and Ash (2019) provides a reflection on the potential value, environmental and economic, loss in the USA as a result of non-reutilising biological waste derived from urban trees. Emerging works such as the above ones illustrate the current interest on moving towards more comprehensive assessments of urban NBS that go beyond the consideration of ES and operational phases. On the other side, the life cycle thinking community, which is usually focused on negative environmental impacts, is also working at a conceptual level on how to integrate ES, embracing their consideration of positive impacts beyond end-of-life alternatives such as recycling. For example, recent works such as the one of Rugani *et al.* (2019) have started to investigate the integration of the ES cascade framework into LCA. Despite these emerging works the review of the state of the art could not identified other studies on NBS that accounted for positive and negative environmental impacts, externalities and financial costs concurrently.

Additionally, the development of the methodological procedure as a two-level modelling framework provides a “two velocity” model. The foreground level offers a dynamic modelling at detailed spatio-temporal resolutions for a reduced spatial extent, meanwhile the background level offers a static modelling to calculate outputs for a further spatial extent that otherwise would be too complicated to obtain. In this sense, it takes advantage of dynamic and static modelling approaches to overcome current limitations of both. Emergent studies from life cycle thinking, urban and landscape ecology scientific communities are advocating to move towards more dynamic models. For example, in LCA Levasseur *et al.* (2010, 2012, 2013) have discussed about the poorly consideration of the temporal resolution in LCA. They have identified the need of dynamic life cycle inventories and characterisation factors, which they illustrate focusing on global warming potential and carbon storage. More recently, Pigné *et al.* (2020) go further on this approach and already present a proof of concept tool with time differentiation on the background life cycle inventory. As another example, in landscape ecology a new ecological connectivity tool recently documented in a preprint (Bocedi *et al.*, 2020) includes dynamic changes in the underneath land cover during the simulation to understand how species movement is influenced by future scenarios and when. These emerging works already inform about the growing importance of dynamic assessments, but the mainstreaming of this kind approaches will require much more available data. Meanwhile full dynamic approaches evolve enough to be used in real studies, "two velocity" modelling approaches such as the one presented here could offer a good compromise to built environment professionals.

Despite the above advantages, the methodological procedure presents three main drawbacks that need to be resolved to ensure a more complete integration of methodological approaches in the modelling framework.

First, the integration of the combined use of landscape ecology methods in the modelling framework still needs to move beyond the conceptual level. In principle, the procedure illustrated in Chapter 4 at urban region level could also be applied to urban studies at more detailed spatial scales. In fact, there are already few studies such as the recent one of Balbi *et al.* (2020) that prove the utility of least-cost path analysis (a method used in Chapter 4) to inform the urban planning of NBS networks. As it was the case in Chapter 4, their purpose was to select a method suitable for urban planners and they tested it with multiple species. Despite, the existence of these emergent works, to test the modelling framework integrating the combined use of landscape ecology methods at city or neighbourhood level is still necessary to fully understand its potential or not to inform urban planning/design decisions looking to mitigate urban biodiversity issues.

Second, the modelling framework is only capable to acknowledge changes in positive and negative impacts due to compositional variations of NBS interventions. However, as discussed in Chapter 5 and 6, the foreground level still neglects the importance of the spatial configuration. In practical terms, beyond the conceptual scheme of Figure 5.2, interactions are considered only inside each cell but not between cells. This is a relevant limitation for the calculation of certain ES, such as regulation of hydrological cycle and waterflow. As it was presented in Chapter 3, for fully quantifying this ES, the processes occurring in the service providing units, service connecting units and service benefitting units need to be assessed (Syrbe and Walz, 2012). However, to move towards an approach that fully integrates the importance of the spatial configuration would increase the complexity of the foreground level. It might also require a reconceptualization of how the archetypes are currently defined in the DSS, where it might be necessary to define the properties of the adjacent cells. In the worst scenario, the DSS should move beyond the current modelling architecture that relies on the use of pre-calculated archetypes.

Third, by the fact that up to a certain level the modelling framework considers social values, through the monetary valuation, it does not provide a proper integration of social valuation as it is introduced in Chapter 3. To provide a more complete assessment of the contribution of NBS to the sustainability and resilience of urbanised contexts, the modelling framework should start incorporating elicited social preferences. These preferences might reflect more accurately perceptions result of socio-cultural constructs, as well as contradictions among different social groups, being able to better assess the social acceptance or not of new NBS interventions into existing local socio-ecological systems.

To sum up, the methodological procedure, and specifically the conceptual modelling framework described in Chapter 5, has advanced the current assessment of NBS going beyond the use of static modelling approaches and the consideration of local impacts during operational phases. The methodological procedure has taken advantage of the strengths of different disciplinary approaches to assess concurrently positive externalities, negative externalities and financial costs. Nevertheless, additional applications to case studies are needed as well as further advances in how processes are modelled (integration of the spatial configuration in the foreground level) and which values are considered (social values). Therefore, the conceptual modelling framework should be understood as a work in progress for which this PhD thesis has established the basis, but which still needs to advance further.

7.4. The practical value of the decision support system for the planning and design of nature-based solutions in urbanised contexts

As it was confirmed in section 7.1, and previously discussed in Chapter 6, the DSS called NBenefit$ can embed the conceptual modelling framework into a tool simple to use by built environment professionals with low computer literacy. In this sense, the DSS can be easily integrated into daily planning and design workflows, therefore ensuring its practical value.

Other emergent DSS are also starting to integrate complex models into already commercial easy to use tools to inform urban planning decisions. For example, GreenPass (https://greenpass.at), introduced in Chapter 6 - Table 6.3, integrate the complex micro-climate simulation software ENVI-met (developed by a third-party) into an easy to use tool to inform new urban planning projects. It provides a detailed assessment on the environmental performance on local climate and water flow regulation. It also informs on financial cost during investment and operational phases. However, the focus is different to the one of NBenefit$ because it is centred only on informing professionals such as civil engineers, architects or landscape architects about their urban development proposals, providing much more technical details. Consequently, different to NBenefit$ it does not consider negative externalities over the entire life cycle or outside the spatial extent of the interventions assessed and the results remain still complex for non-experts. Therefore, a DSS such as NBenefit$ can cover a current gap in the sill emergent pool of DSS for NBS by looking to inform experts and a broader audience, as well as including end-of-life processes and externalities into a simple form, meanwhile moving beyond the use of oversimplified methods.

Nevertheless, NBenefit$ still needs to overcome several drawbacks before it can be used to inform the planning and design of real NBS interventions. In the following lines, three main drawbacks, besides improvements in the graphical user interface, are discussed.

First, as described in Chapter 5, due to data constraints many equations or parameters are obtained from non-local sources, in several cases from studies in the USA, and might not well represent conditions of other urban contexts. For example, for most of the tree species the modelling framework makes use of empirical allometric equations obtained from the Urban Tree Database of the USA (McPherson, van Doorn and Peper, 2016). As another example, the modelling of tree mortality is also developed based on statistics obtained from a long-term study in the USA (Nowak, Kuroda and Crane, 2004). Therefore, it is necessary to obtain access to widespread and rich local databases on NBS or their components to customize the model, and as a consequence the DSS to local conditions. For the use of the DSS in practical works this will make a strong difference, ensuring a closer match between model results and reality.

Second, as described in Chapter 5 and 6, the results used to characterise the atmospheric and air pollutant conditions assumed a *ceteris paribus* situation, where nothing changes in 50 years. However, in reality climate is changing and due to technological advances and potential changes in social behaviour (e.g. mobility habits) dynamics in air pollutant concentrations will also variate. Then, the results presented in Chapter 5 and 6 do not reflect well long-term meteorological dynamics which diminishes their credibility as long-term results. In fact, as already stated in Chapter 5, this drawback was known and was corrected in the atmosphere module from the very beginning, but the lack of time impeded to develop a study where this aspect was showcased clearly. It is in fact the reason why the atmosphere module works as a weather generator, instead of simply replicating the historical data collected in a random but coherent form. As explained in Chapter 5, all the meteorological and air pollutant variables are generated based on parameters (e.g. mean, standard deviation, variance) defining their historical statistical distribution and simple correlations between variables (e.g. occurrence of precipitation influences temperature and air pollutant concentration). This means that the atmosphere model can emulate defined scenarios (e.g. regionalised IPCC climate scenarios) in a simple way. For these emulations, the variations over time in the parameters defining the monthly statistical distributions of each variable (to keep capturing seasonality) have to be provided. In fact, other recent works, such as the one of Nadal-Sala, Gracia and Sabaté (2019), are also testing stochastic weather generators to emulate in a simple way daily weather conditions in climate change scenarios to provide weather inputs to other models. Different to recent works, the atmospheric module presented in Chapter 5 also includes the emulation of ambient levels of common air pollutants. The existence of other works in development supports the interest of these kind of models compared to just inputting available meteorological data. This kind of approach provides more flexibility to the ecological modeller, who can test more radical or unknown scenarios, such as those derived from a new technology reducing ambient levels of specific air pollutant species. Therefore, in the future this advantage should be better showcased in the DSS and incorporated first using common climate scenarios such as the regionalised IPCC scenarios already provided by the Spanish Agency of Meteorology (AEMET).

Third, Chapter 5 and 6 illustrated the conceptual modelling framework and the DSS only making use of urban forests (a variation of the NBS woodland-like). However, landscape and urban planning actions are usually quite complex and go beyond a unique NBS, incorporating a multiple set of them in the same green open space. This means that other foreground level models should be developed representing additional types of NBS. For example, in the case of green walls, a system dynamics model such as the one developed by Marchi *et al.* (2015) could be used as a basis and expanded incorporating more outputs than net carbon storage. In fact, some of the existing modules and sub-modules of the urban forest model can be re-utilised, even if some equations might need to be changed. For example, the modelling of biomass growth for non-woody plants would be different than that for woody plants, but their equations of interception, or dry air pollution deposition might not need to be changed. Once a diverse set of urban NBS will be modelled, NBenefit$ can become an attractive practical tool for real urban planning and landscape design projects.

To sum up, NBenefit$ has been developed keeping in mind that it needs to be a practical tool for use in the daily work of built environment professionals. However, as it occurs with the conceptual modelling framework, NBenefit$ is still a work in progress, which should be further advanced before becoming a tool of real practical value.

7.5. Perspectives

In the long-term, three future areas of research have been identified as key for a further advancement of the work developed in this thesis. These three areas of research would help to provide more informed decisions on future NBS implementations in order to enhance their contribution to urban sustainability and resilience.

First, much more data should be made available to move towards accurate assessments capable to represent more realistically the complexity of urbanised system and the NBS placed on them. This data cannot be anymore based on short-term studies that provide limited understanding on long-term dynamics. Otherwise, lack of data would be permanently the main constraint of future studies. Moreover, the monitoring programs should be defined incorporating scientists and professionals from multiple, and if possible, quite different, disciplinary fields in order to maximise the value that can be obtained from all the data collected. For example, it is not enough to monitor changes in the natural features of ecosystems, also changes in the human management actions applied on them should be monitored, including the technology used. It is worth mentioning that long-term monitoring programs already exist, and a strong research community is behind them. In fact, this thesis used data obtained from the urban sites of the Long-Term Ecological Research (LTER) of the USA, since the equations of the Urban Tree Database and statistical mortality data used in the urban forest model comes from the LTER program of the USA. However, when looking at current locations and purposes of the LTER sites (see most LTER site of the world in https://deims.org/about → Explore → Site Map) a strong pattern appears, there are few urban sites focusing on the dynamics of urbanised contexts. As an example, in Europe, it appears that the only LTER focused on urban sites are the ones of Strasbourg

(France), Lodz (Poland), Paris (France) and Venezia (Italy). Consequently, for improving our understanding on NBS in urbanised contexts and how transferable are specific NBS among them more urban LTER sites are needed, which should be selected maximising their future value for a better understanding of the entire network. To make more cost-effective the long-term monitoring, social actors could be actively involved by researchers and public institutions taking profit of citizen science approaches.

Second, beyond the involvement of society through citizen science approaches, the next step for conceptual methodological frameworks such as the one presented in this thesis it is to make them fully participatory. As stated in section 7.2, social actors and local experts could provide knowledge about the system that it is hidden to consultants or scientists. As a result, a segregation science, policy, planning/design, society does not help to fully understand the dynamics of urbanised systems and the NBS placed in them. It does not work either to inform (assess) the development of better policies, plans, and posterior design of specific interventions, such as NBS, for the society that lives on them. This statement partially recalls what was anticipated in Chapter 1 as a pattern:process:design approach (Nassauer and Opdam, 2008). This approach intends to make the scientific knowledge from landscape ecology more applied to design/planning purposes as well as reducing the segregation between scientists and planners/designers making the transfer of knowledge between them more fluid. Here, the next step is advocated, i.e. a pattern:process:design:society approach. The use of visual declarative modelling software such as SIMILE, used during this thesis, and causal loops diagrams of qualitative relationships could facilitate a cooperative building of models among scientists, policy makers, planners/designers and social actors. Consequently, this kind of models might offer a starting basis for building such an approach and testing its feasibility and limits.

Third, and in relation with the other two points raised, NBS interventions or others embedded in complex socio-ecological systems need to move towards assessments that are spatially explicit, consider the temporal dimension, and the interactions between components in a dynamic form. This thesis only treated spatio-temporally explicit and dynamically the generation of waste, few management actions, tree mortality, the supply of ES and when ES are required by society. A further understanding of ES dynamics should also treat the curve of the ES demand in a dynamic way, understanding how changes on it affects the marginal values of the benefits derived from ES. In this thesis, it was assumed that prices were not moving over time due to changes in ES demand and that value transfer was acceptable. It is because the thesis was not focused on representing dynamically the impact of the ES demand on the ES value, and the risk of underestimation was considered acceptable. However, in the long-term future works advancing towards a deep understanding of the socio-economic system side of the ES cascade framework might need to start considering this dynamism. Similarly, standard approaches to LCA and LCC usually consider static models and do not acknowledge how changes in the temporal or spatial dimension could alter results of negative impacts. As stated in section 7.3, researchers such as Pigné *et al.* (2020) are starting to explore how to move towards a dynamic LCA. As we introduced in Chapter 1, others such as Loiseau *et al.* (2013) are exploring how to incorporate better the spatial dimension and the application of LCA to assess territorial structures, which in a broad sense correspond also to NBS. The work in this thesis could contribute to the above emergent movements in the life cycle thinking community. At least it could trigger a research question that seems worth to investigate: whether and how the integration of spatio-temporal explicit and dynamic ES quantifications (as positive impacts) could enhance future life cycle thinking studies on naturalised ecosystems, such as productive woodlands, or complex territorial structures, such as a small municipality or even a neighbourhood.

References

Adhikari, A. R. *et al.* (2011) 'Removal of nutrients and metals by constructed and naturally created wetlands in the Las Vegas Valley, Nevada', *Environmental Monitoring and Assessment*, 180(1–4), pp. 97–113. doi: 10.1007/s10661-010-1775-y.

Adriaensen, F. *et al.* (2003) 'The application of "least-cost" modelling as a functional landscape model', *Landscape and Urban Planning*, 64(4), pp. 233–247. doi: 10.1016/s0169-2046(02)00242-6.

Adyel, T. M., Oldham, C. E. and Hipsey, M. R. (2016) 'Stormwater nutrient attenuation in a constructed wetland with alternating surface and subsurface flow pathways: Event to annual dynamics', *WATER RESEARCH*, 107, pp. 66–82. doi: 10.1016/j.watres.2016.10.005.

Albert, C. *et al.* (2019) 'Addressing societal challenges through nature-based solutions: How can landscape planning and governance research contribute?', *Landscape and Urban Planning*. Elsevier, 182(September 2018), pp. 12–21. doi: 10.1016/j.landurbplan.2018.10.003.

Albertí, J. *et al.* (2017) 'Towards life cycle sustainability assessent of cities. A review of background knowledge', *Science of the Total Environment*. Elsevier B.V., 609, pp. 1049–1063. doi: 10.1016/j.scitotenv.2017.07.179.

Alberti, M. (2016) *Cities that think like planets: complexity, resilience, and innovation in hybrid ecosystems*. University of Washington Press.

Alcamo, J., Bennett, E. M. and (Program), M. E. A. (2003) *Ecosystems and Human Well-being: A Framework for Assessment*, *Millennium Ecosystem Assessment*. Washington, DC: Island Press. doi: Cited By (since 1996) 1\rExport Date 12 August 2012.

Alejandre, E. M., van Bodegom, P. M. and Guinée, J. B. (2019) 'Towards an optimal coverage of ecosystem services in LCA', *Journal of Cleaner Production*, 231, pp. 714–722. doi: 10.1016/j.jclepro.2019.05.284.

Allen, R. G. *et al.* (1998) 'Crop evapotranspiration - Guidelines for computing crop water requirements - FAO Irrigation and drainage paper 56', *Irrigation and Drainage*, pp. 1–15. doi: 10.1016/j.eja.2010.12.001.

Almeida, A. C. and Sands, P. J. (2016) 'Improving the ability of 3-PG to model the water balance of forest plantations in contrasting environments', *Ecohydrology*, 9(4), pp. 610–630. doi: 10.1002/eco.1661.

Almohamad, H., Knaack, A. and Habib, B. (2018) 'Assessing Spatial Equity and Accessibility of Public Green Spaces in Aleppo City, Syria', *Forests*, 9(11), p. 706. doi: 10.3390/f9110706.

van Andel, J. and Aronson, J. (2012) *Restoration ecology: the new frontier*. John Wiley & Sons.

Andersson-Sköld, Y. *et al.* (2018) 'A framework for assessing urban greenery's effects and valuing its ecosystem services', *Journal of Environmental Management*, 205, pp. 274–285. doi: 10.1016/j.jenvman.2017.09.071.

Andersson, K., Dickin, S. and Rosemarin, A. (2016) 'Towards Sustainable Sanitation: Challenges and Opportunities in Urban Areas', *SUSTAINABILITY*, 8(12). doi: 10.3390/su8121289.

Angel, S. *et al.* (2011) 'The dimensions of global urban expansion: Estimates and projections for all countries, 2000-2050', *Progress in Planning*, 75(2), pp. 53–107. doi: 10.1016/j.progress.2011.04.001.

Anguelovski, I. *et al.* (2018) 'Assessing green gentrification in historically disenfranchised neighborhoods: a longitudinal and spatial analysis of Barcelona', *Urban Geography*. Taylor & Francis, 39(3), pp. 458–491.

Arbault, D. *et al.* (2014) 'Integrated earth system dynamic modeling for life cycle impact assessment of ecosystem services', *Sci Total Environ*, 472, pp. 262–272. doi: 10.1016/j.scitotenv.2013.10.099.

Arntzen, J. W. *et al.* (2017) 'Amphibian decline, pond loss and reduced population connectivity under agricultural intensification over a 38 year period', *Biodiversity and Conservation*. Springer Netherlands, 26(6), pp. 1411–1430. doi: 10.1007/s10531-017-1307-y.

Badach, J. *et al.* (2018) 'A case study of odour nuisance evaluation in the context of integrated urban planning', *Journal of Environmental Management*, 213, pp. 417–424. doi: 10.1016/j.jenvman.2018.02.086.

Badampudi, D., Wohlin, C. and Petersen, K. (2015) 'Experiences from using snowballing and database searches in systematic literature studies', (April), pp. 1–10. doi: 10.1145/2745802.2745818.

Bagstad, K. J. *et al.* (2013) 'A comparative assessment of decision-support tools for ecosystem services quantification and valuation', *Ecosystem Services*. Elsevier, 5, pp. 27–39. doi: 10.1016/j.ecoser.2013.07.004.

Balbi, M. *et al.* (2020) 'Least-cost path analysis for urban greenways planning: A test with moths and birds accross two habitats and two cities', *Journal of Applied Ecology*, 00, pp. 1–12. doi: 10.1111/1365-2664.13800.

Balbi, S. *et al.* (2019) 'Human dependence on natural resources in rapidly urbanising South African regions', *Environmental Research Letters*. IOP Publishing, 14(4), p. 044008. doi: 10.1088/1748-9326/aafe43.

Baldocchi, D. D., Hicks, B. B. and Camara, P. (1987) 'A canopy stomatal resistance model for gaseous deposition to vegetated surfaces', *Atmospheric Environment (1967)*, 21(1), pp. 91–101. doi: 10.1016/0004-6981(87)90274-5.

Balmford, A. *et al.* (2008) *Review on the economics of biodiversity loss: scoping the science*, *Cambridge, UK.*

Bani, L. *et al.* (2017) 'Population genetic structure and sex-biased dispersal of the hazel dormouse (Muscardinus avellanarius) in a continuous and in a fragmented landscape in central Italy', *Conservation Genetics*. Springer Netherlands, 18(2), pp. 261–274. doi: 10.1007/s10592-016-0898-2.

Baranyi, G. *et al.* (2011) 'Contribution of habitat patches to network connectivity: Redundancy and uniqueness of topological indices', *Ecological Indicators*. Elsevier Ltd, 11(5), pp. 1301–1310. doi: 10.1016/j.ecolind.2011.02.003.

Baro, F. *et al.* (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, pp. 405–417. doi: 10.1016/j.landusepol.2016.06.006.

Base Paisajismo (2019) *Base de Precios de Paisajismo 2019*. Available at: https://basepaisajismo.com/Paisajismo_WEB/#cap.PTVC.

Bastian, M., Heymann, S. and Jacomy, M. (2009) 'Gephi: An Open Source Software for Exploring and Manipulating Networks', *International AAAI Conference on Web and Social Media; Third International AAAI Conference on Weblogs and Social Media*. Available at: https://www.aaai.org/ocs/index.php/ICWSM/09/paper/view/154.

Bastian, O. (2001) 'Landscape ecology - Towards a unified discipline?', *Landscape Ecology*, 16(8), pp. 757–766. doi: 10.1023/A:1014412915534.

Belussi, L. and Barozzi, B. (2015) 'Mitigation measures to contain the environmental impact of urban areas: a bibliographic review moving from the life cycle approach', *Environ Monit Assess*, 187(12), p. 745. doi: 10.1007/s10661-015-4960-1.

Bencala, E. and Seinfeld, J. H. (1976) 'On frequency distributions of air pollutant concentrations', *Atmospheric Environment*, 10(19624968), pp. 941–950.

Benedek, Z. *et al.* (2011) 'Landscape metrics as indicators: Quantifying habitat network changes of a bush-cricket Pholidoptera transsylvanica in Hungary', *Ecological Indicators*, 11(3), pp. 930–933. doi: 10.1016/j.ecolind.2010.11.007.

Bennett, E. M., Peterson, G. D. and Gordon, L. J. (2009) 'Understanding relationships among multiple ecosystem services', *Ecology Letters*, 12(12), pp. 1394–1404. doi: 10.1111/j.1461-0248.2009.01387.x.

Benoît, C. and Mazijn, B. (2009) 'Guidelines for social life cycle assessment of products, UNEP/SETAC Life Cycle Initiative', *Sustainable Product and Consumption Branch Paris, France*.

Bernath, K. and Roschewitz, A. (2008) 'Recreational benefits of urban forests : Explaining visitors ' willingness to pay in the context of the theory of planned behavior', *Journal of Environmental Management*, 89, pp. 155–166. doi: 10.1016/j.jenvman.2007.01.059.

Bettencourt, L. M. A. *et al.* (2007) 'Growth, innovation, scaling, and the pace of life in cities', *Proceedings of the National Academy of Sciences*, 104(17), pp. 7301–7306. doi: 10.1073/pnas.0610172104.

Bianchini, F. and Hewage, K. (2012) 'Probabilistic social cost-benefit analysis for green roofs : A lifecycle approach', *Building and Environment*, 58, pp. 152–162. doi: 10.1016/j.buildenv.2012.07.005.

Bjørn, A. *et al.* (2018) 'Goal Definition', in Hauschild, M. Z., Rosenbaum, R. K., and Olsen, S. I. (eds) *Life Cycle Assessment: Theory and Practice*. Cham: Springer International Publishing, pp. 67–74. doi: 10.1007/978-3-319-56475-3_7.

Bocchini, P. *et al.* (2014) 'Resilience and Sustainability of Civil Infrastructure : Toward a Unified Approach', *Journal of Infrastructure Systems*, 20(2), pp. 1–16. doi: 10.1061/(ASCE)IS.1943-555X.0000177.

Bocedi, G. *et al.* (2020) 'RangeShifter 2.0: An extended and enhanced platform for modelling spatial eco-evolutionary dynamics and species' responses to environmental changes.', *bioRxiv*. Cold Spring Harbor Laboratory.

Bodin, Ö. and Saura, S. (2010) 'Ranking individual habitat patches as connectivity providers: Integrating network analysis and patch removal experiments', *Ecological Modelling*, 221(19), pp. 2393–2405. doi: 10.1016/j.ecolmodel.2010.06.017.

Bordt, M. and Saner, M. A. (2019) 'Which ecosystems provide which services ? A meta-analysis of nine selected ecosystem services assessments', *One Ecosystem*, 4(e31420). doi: 10.3897/oneeco.4.e31420.

Bosch, J., Beebee, T., Schmidt, B., Tejedo, M., Martinez Solano, I., Salvador, A., García París, M., Recuero Gil, E., Arntzen, J., Díaz-Paniagua, C. & Marquez, L. C. (2016) *Alytes obstetricans (errata version)., The IUCN Red List of Threatened Species: e.T55268A87541047*.

Bottalico, F., Travaglini, D., Chirici, G., Garfì, V., *et al.* (2017) 'A spatially-explicit method to assess the dry deposition of air pollution by urban forests in the city of Florence, Italy', *Urban Forestry & Urban Greening*. Elsevier, 27(August), pp. 221–234. doi: 10.1016/j.ufug.2017.08.013.

Bottalico, F., Travaglini, D., Chirici, G., Garfi, V., *et al.* (2017) 'A spatially-explicit method to assess the dry deposition of air pollution by urban forests in the city of Florence, Italy', *URBAN FORESTRY & URBAN GREENING*, 27, pp. 221–234. doi: 10.1016/j.ufug.2017.08.013.

Boyd, J. and Banzhaf, S. (2007) 'What are ecosystem services? The need for standardized environmental accounting units', *Ecological Economics*, 63(2–3), pp. 616–626. doi: 10.1016/j.ecolecon.2007.01.002.

Brander, L. (2013) *Guidance manual on value transfer methods for ecosystem services*. UNEP.

Breuste, J. *et al.* (2013) 'Urban Ecosystem services on the local level: Urban green spaces as providers', *Ekologia Bratislava*, 32(3), pp. 290–304. Available at: https://www.scopus.com/inward/record.uri?eid=2-s2.0-84893551885&doi=10.2478%2Feko-2013-0026&partnerID=40&md5=e4a1e141ca0a3cc7a529eedf14f1f16d.

Brill, G., Anderson, P. and O'Farrell, P. (2017) 'Methodological and empirical considerations when assessing freshwater ecosystem service provision in a developing city context: Making the best of what we have', *ECOLOGICAL INDICATORS*, 76, pp. 256–274. doi:

10.1016/j.ecolind.2017.01.006.

Brink, E. *et al.* (2016) 'Cascades of green: A review of ecosystem-based adaptation in urban areas', *Global Environmental Change*. Elsevier Ltd, 36, pp. 111–123. doi: 10.1016/j.gloenvcha.2015.11.003.

British Standards (2012) *5837: 2012, Trees in relation to design, demolition and construction-Recommendations UK*. London.

Broun, R. *et al.* (2014) 'Integrated Life Cycle Energy and Greenhouse Gas Analysis of Exterior Wall Systems for Residential Buildings', *Sustainability*, 6(12), pp. 8592–8603. doi: 10.3390/su6128592.

Brouwer, R. and Navrud, S. (2015) 'The Use and Development of Benefit Transfer in Europe', in Johnston, R. J. et al. (eds) *Benefit Transfer of Environmental and Resource Values: A Guide for Researchers and Practitioners*. Dordrecht: Springer Netherlands, pp. 71–83. doi: 10.1007/978-94-017-9930-0_4.

Brown, G. (2013) 'The relationship between social values for ecosystem services and global land cover: An empirical analysis', *Ecosystem Services*. Elsevier, 5, pp. 58–68. doi: 10.1016/j.ecoser.2013.06.004.

Brunori, A. *et al.* (2017) 'Biomass and volume modeling in Olea europaea L . cv "' Leccino '"', *Trees*. Springer Berlin Heidelberg, 31(6), pp. 1859–1874. doi: 10.1007/s00468-017-1592-9.

Bruntland, G. (1987) *Our common future*, *Report of The World Commission on Environment and Development: Our Common Future.*

De Bruyn, S *et al.* (2018) 'Environmental prices handbook 2017-methods and numbers for valuation of environmental impacts', *Delft: CE Delft*, pp. 5–2018.

De Bruyn, S. *et al.* (2018) 'Environmental Prices Handbook EU28 Version', *CE Delft*, p. 175.

Bunn, A. G., Urban, D. L. and Keitt, T. H. (2000) 'Landscape connectivity: A conservation application of graph theory', *Journal of Environmental Management*, 59(4), pp. 265–278. doi: 10.1006/jema.2000.0373.

Burszta-Adamiak, E. and Mrowiec, M. (2013) 'Modelling of green roofs' hydrologic performance using EPA's SWMM', *Water Science and Technology*. IWA Publishing, 68(1), pp. 36–42.

Busch, M. *et al.* (2012) 'Potentials of quantitative and qualitative approaches to assessing ecosystem services', *Ecological Indicators*, 21, pp. 89–103. doi: 10.1016/j.ecolind.2011.11.010.

Bush, J. and Doyon, A. (2019) 'Building urban resilience with nature-based solutions: How can urban planning contribute?', *Cities*. Elsevier, 95(September), p. 102483. doi: 10.1016/j.cities.2019.102483.

Büttner, G. *et al.* (2017) *CLC2018 Technical Guidelines*. Wien.

Buyantuyev, A., Wu, J. and Gries, C. (2010) 'Multiscale analysis of the urbanization pattern of the Phoenix metropolitan landscape of USA: Time, space and thematic resolution', *Landscape and Urban Planning*, 94(3–4), pp. 206–217. doi: 10.1016/j.landurbplan.2009.10.005.

Cabral, P. *et al.* (2016) 'Assessing the impact of land-cover changes on ecosystem services: A first step toward integrative planning in Bordeaux, France', *ECOSYSTEM SERVICES*, 22(B), pp. 318–327. doi: 10.1016/j.ecoser.2016.08.005.

Calabrese, J. M. and Fagan, W. F. (2004) 'A comparison-shopper's guide to connectivity metrics', *Frontiers in Ecology and the Environment*, 2(10), pp. 529–536. doi: 10.1890/1540-9295(2004)002[0529:ACGTCM]2.0.CO;2.

Cambria, D. and Pierangeli, D. (2011) 'A life cycle assessment case study for walnut tree (Juglans regia L .) seedlings production', *International Journal of Life Cycle Assessment*, 16, pp. 859–868. doi: 10.1007/s11367-011-0323-5.

Cariñanos, P. *et al.* (2017) 'Assessing allergenicity in urban parks: A nature-based solution to reduce the impact on public health', *Environmental Research*. Elsevier Inc., 155(September 2016), pp. 219–227. doi: 10.1016/j.envres.2017.02.015.

Caro, T. M. and O'Doherty, G. (1999) 'On the use of surrogate species in conservation biology', *Conservation Biology*, 13(4), pp. 805–814. doi: 10.1046/j.1523-1739.1999.98338.x.

Carter, T. and Keeler, A. (2008) 'Life-cycle cost-benefit analysis of extensive vegetated roof systems', *Journal of Environmental Management*, 87(3), pp. 350–363. doi: 10.1016/j.jenvman.2007.01.024.

Casas, Y. *et al.* (2017) 'Life-cycle greenhouse gas emissions assessment and extended exergy accounting of a horizontal-flow constructed wetland for municipal wastewater treatment : A case study in Chile', *Ecological Indicators*. Elsevier Ltd, 74, pp. 130–139. doi: 10.1016/j.ecolind.2016.11.014.

Cascio, J. (2009) 'Resilience', *Foreign Policy*. Foreign Policy, (172), p. 92.

Castro, A. J. *et al.* (2014) 'Ecosystem service trade-offs from supply to social demand: A landscape-scale spatial analysis', *Landscape and Urban Planning*. Elsevier B.V., 132, pp. 102–110. doi: 10.1016/j.landurbplan.2014.08.009.

Ceirans, A. (2007) 'Microhabitat characteristics for reptiles Lacerta agilis, Zootoca vivipara, Anguis fragilis, Natrix natrix, and Vipera berus in Latvia', *Russian Journal of Herpetology*, 14(3), pp. 172–176.

Chan, K. M. A. *et al.* (2012) 'Where are Cultural and Social in Ecosystem Services? A Framework for Constructive Engagement', *BioScience*, 62(8), pp. 744–756. doi: 10.1525/bio.2012.62.8.7.

Chanes, R. and Castano, P. (1969) 'Arboles y arbustos de jardin en clima templado [1. ed.]'. Editorial Blume.

Chavan, P. V. and Dennett, K. E. (2008) 'Wetland simulation model for nitrogen, phosphorus, and sediments retention in constructed wetlands', *Water, Air, and Soil Pollution*, 187(1–4), pp. 109–118. doi: 10.1007/s11270-007-9501-2.

Chen, A. H. and Warren, J. (2011) 'Sustainable Growth for China', *The Chinese Economy*, 44(5), pp. 86–103. doi: 10.2753/CES1097-1475440505.

Chen, C. *et al.* (2017) 'Incorporating landscape connectivity into household pond configuration in a hilly agricultural landscape', *Landscape and Ecological Engineering*. Springer Japan, 13(1), pp. 189–204. doi: 10.1007/s11355-016-0317-3.

Chen, Y. *et al.* (2017) 'Tree survival and growth are impacted by increased surface temperature on paved land', *Landscape and Urban Planning*, 162, pp. 68–79. doi: 10.1016/j.landurbplan.2017.02.001.

Cheng, X. *et al.* (2019) 'Evaluation of cultural ecosystem services : A review of methods', *Ecosystem Services*. Elsevier B.V., 37(April), p. 100925. doi: 10.1016/j.ecoser.2019.100925.

Childers, D. L. *et al.* (2015) 'An Ecology for Cities: A Transformational Nexus of Design and Ecology to Advance Climate Change Resilience and Urban Sustainability', *Sustainability*, 7, pp. 3774–3791. doi: 10.3390/su7043774.

Coelho, D. and Ruth, M. (2006) 'Seeking a unified urban systems theory'. School of Public Policy, University of Maryland, College Park, United States: WITPress, 93, pp. 179–188. doi: 10.2495/SC060171.

Cohen-Shacham, E. *et al.* (2016) 'Nature-based Solutions to address global societal challenges', *IUCN, Gland, Switzerland. doi*, 10.

Coleman, K. and Jenkinson, D. . (2014) 'RothC - A Model for the Turnover of Carbon in Soil. Model description and windows user guide.', *Evaluation of Soil Organic Matter Models: Using Existing Long-Term Datasets*, I(June), pp. 237–246. doi: 10.1007/978-3-642-61094-3_17.

Coleman, K. and Jenkinson, D. S. (2014) 'RothC - A Model for the Turnover of Carbon in Soil: Model description and Users guide (DOS version; updated June 2014)', *Rothamsted Research*, Rothamsted(June), p. 26pp. doi: 10.1007/978-3-642-61094-3_17.

Cord, A. F. *et al.* (2017) 'Towards systematic analyses of ecosystem service trade-offs and synergies: Main concepts, methods and the road ahead', *Ecosystem Services*. Elsevier B.V., 28, pp. 264–272. doi: 10.1016/j.ecoser.2017.07.012.

Cortinovis, C. (2017) *Integrating Ecosystem Services in urban planning*. Università degli studi di Trento.

Cortinovis, C. and Geneletti, D. (2018a) 'Ecosystem services in urban plans: What is there, and what is still needed for better decisions', *LAND USE POLICY*, 70, pp. 298–312. doi: 10.1016/j.landusepol.2017.10.017.

Cortinovis, C. and Geneletti, D. (2018b) 'Mapping and assessing ecosystem services to support urban planning : A case study on brownfield regeneration in Trento , Italy', *One Ecosystem*, 3(e25477). doi: 10.3897/oneeco.3.e25477.

Cortinovis, C. and Geneletti, D. (2019) 'A framework to explore the effects of urban planning decisions on regulating ecosystem services in cities', *Ecosystem Services*. Elsevier B.V., 38(March), p. 100946. doi: 10.1016/j.ecoser.2019.100946.

Costanza, R. *et al.* (1997) 'The value of the world's ecosystem services and natural capital', *nature*. Nature publishing group, 387(6630), p. 253.

Costanza, R. *et al.* (2014) 'Changes in the global value of ecosystem services', *Global Environmental Change*. Elsevier Ltd, 26(1), pp. 152–158. doi: 10.1016/j.gloenvcha.2014.04.002.

Coulon, A. *et al.* (2015) 'A stochastic movement simulator improves estimates of landscape connectivity', *Ecology*, 96(8), pp. 2203–2213. doi: 10.1890/14-1690.1.

Crow, P. (2005) *The Influence of Soils and Species on Tree Root Depth*.

Davies, H. J. *et al.* (2017) 'Challenges for tree officers to enhance the provision of regulating ecosystem services from urban forests', *Environ Res*, 156, pp. 97–107. doi: 10.1016/j.envres.2017.03.020.

Day, S. D. *et al.* (2010) 'Contemporary Concepts of Root System Architecture of Urban Trees', *Arborculture & Urban Forestry*, 36(July), pp. 149–159.

Delattre, T., Baudry, J. and Burel, F. (2018) 'An onion-like movement corridor? Possible guidelines emerging from small-scale movement rules', *Ecological Informatics*. Elsevier B.V, 45, pp. 48–58. doi: 10.1016/j.ecoinf.2018.03.006.

Dennis, M. and James, P. (2016) 'User participation in urban green commons: Exploring the links between access, voluntarism, biodiversity and well being', *URBAN FORESTRY & URBAN GREENING*, 15, pp. 22–31. doi: 10.1016/j.ufug.2015.11.009.

Denoël, M. *et al.* (2013) 'Similar Local and Landscape Processes Affect Both a Common and a Rare Newt Species', *PLoS ONE*, 8(5), pp. 21–25. doi: 10.1371/journal.pone.0062727.

Dietz, M. *et al.* (2018) 'A small mammal's map: identifying and improving the large-scale and cross-border habitat connectivity for the hazel dormouse Muscardinus avellanarius in a fragmented agricultural landscape', *Biodiversity and Conservation*. Springer Netherlands, pp. 1–14. doi: 10.1007/s10531-018-1515-0.

Dietz, M. and Pir, J. B. (2009) 'Distribution and habitat selection of Myotis bechsteinii in Luxembourg: implications for forest management and conservation.', *Mammal conservation in Europe: status and priorities. Collection of papers from the 5th European Congress of Mammalogy, Siena, Italy, 21-26 September 2007.*, 58(3), pp. 327–340. Available at: http://www.cabdirect.org/abstracts/20103016848.html.

Dietz, S. and Neumayer, E. (2006) 'Weak and strong sustainability in the SEEA : Concepts and measurement', 1. doi: 10.1016/j.ecolecon.2006.09.007.

Dijlkstra, L. and Poelman, H. (2012) *Cities in Europe. The new OCED-EC Definition*.

Dobbs, C. *et al.* (2019) 'Urban ecosystem Services in Latin America: mismatch between global concepts and regional realities?', *Urban Ecosystems*. Urban Ecosystems, 22(1), pp. 173–187. doi: 10.1007/s11252-018-0805-3.

von Döhren, P. and Haase, D. (2015) 'Ecosystem disservices research: A review of the state of the art with a focus on cities', *Ecological Indicators*, 52, pp. 490–497. doi: 10.1016/j.ecolind.2014.12.027.

Dorigo, M., Birattari, M. and Stutzle, T. (2006) 'Ant colony optimization', *IEEE Computational Intelligence Magazine*, 1(4), pp. 28–39. doi: 10.1109/MCI.2006.329691.

Dorst, H. *et al.* (2019) 'Urban greening through nature-based solutions – Key characteristics of an emerging concept', *Sustainable Cities and Society*. Elsevier, 49(January), p. 101620. doi: 10.1016/j.scs.2019.101620.

Duan, N. *et al.* (2011) 'Evaluating the environmental impacts of an urban wetland park based on emergy accounting and life cycle assessment : A case study in Beijing', *Ecological Modelling*, 222, pp. 351–359. doi: 10.1016/j.ecolmodel.2010.08.028.

Edelsparre, A. H., Shahid, A. and Fitzpatrick, M. J. (2018) 'Habitat connectivity is determined by the scale of habitat loss and dispersal strategy', *Ecology and Evolution*, (September 2017), pp. 1–7. doi: 10.1002/ece3.4072.

Edgar, P. and Bird, D. R. (2006) *Action Plan for the conservation of the crested newt Triturus cristatus species complex in Europe*, *Convention on the conservation of European wildlife and natural habitats*.

Eggermont, H. *et al.* (2015) 'Nature-based solutions: New influence for environmental management and research in Europe', *Gaia*, 24(4), pp. 243–248. doi: 10.14512/gaia.24.4.9.

Elliot, N. B. *et al.* (2014) 'The devil is in the dispersers: Predictions of landscape connectivity change with demography', *Journal of Applied Ecology*, 51(5), pp. 1169–1178. doi: 10.1111/1365-2664.12282.

Elliot, T. *et al.* (2019) 'Pathways to modelling ecosystem services within an urban metabolism framework', *Sustainability (Switzerland)*, 11(10). doi: 10.3390/su11102766.

Elliot, T., Babí Almenar, J. and Rugani, B. (2020) 'Impacts of policy on urban energy metabolism at tackling climate change : The case of Lisbon', *Journal of Cleaner Production*. Elsevier Ltd, 276, p. 123510. doi: 10.1016/j.jclepro.2020.123510.

Elmqvist, T. *et al.* (2015) 'Benefits of restoring ecosystem services in urban areas', *Current Opinion in Environmental Sustainability*, 14, pp. 101–108. doi: 10.1016/j.cosust.2015.05.001.

Elmqvist, T. (2017) 'Development: Sustainability and resilience differ', *Nature*. Nature Research, 546(7658), p. 352.

Elmqvist, T. *et al.* (2019) 'Sustainability and resilience for transformation in the urban century', *Nature Sustainability*. Springer US, 2(April). doi: 10.1038/s41893-019-0250-1.

EPA, U. S. (2015) 'National Ecosystem Services Classification System (NESCS): Framework Design and Policy Application. EPA-800-R-15-002. United States Environmental Protection Agency'.

Escobedo, F. J., Kroeger, T. and Wagner, J. E. (2011) 'Urban forests and pollution mitigation: Analyzing ecosystem services and disservices', *Environmental Pollution*. Elsevier Ltd, 159(8–9), pp. 2078–2087. doi: 10.1016/j.envpol.2011.01.010.

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.* Luxembourg: Publications Office of the European Union. Available at: https://doi.org/10.2777/765301.

European Commission (2016) *Policy Topics: Nature-Based Solutions*. Available at:

https://ec.europa.eu/research/environment/index.cfm?pg=nbs (Accessed: 8 January 2020).

European Environmental Agency (2010) *Scaling up ecosystem benefits*. doi: 10.2800/41295.

European Environmental Agency (2011) *Landscape fragmentation in Europe*, *Joint EEA-FOEN report. EEA Report No 2/2011.* doi: 10.2800/78322.

European Environmental Agency (2017) *Landscape fragmentation indicator effective mesh density (seff) - major and medium anthrophogenic fragmentation (FGA2_S_2016)*. Available at: https://www.eea.europa.eu/data-and-maps/data/landscape-fragmentation-indicator-effective-mesh.

European Union (2018) *Copernicus Land Monitoring Service, European Environment Agency (EEA)*. Available at: https://land.copernicus.eu/local/urban-atlas/urban-atlas-2012?tab=download (Accessed: 20 August 2001).

Eurostat (2015) 'Population projections 2015 at national level. In Population on 1st January by age and sex.' Eurostat.

Eurostat (2020a) *Duration of working life - statistics*. Available at: https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Duration_of_working_life_-_statistics (Accessed: 20 October 2020).

Eurostat (2020b) *Mortality and Life Expectancy Statistics*. Available at: https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Mortality_and_life_expectancy_statistics (Accessed: 20 October 2020).

Faivre, N. *et al.* (2017) 'Nature-Based Solutions in the EU: Innovating with nature to address social, economic and environmental challenges', *Environmental Research*, 159(December 2016), pp. 509–518. doi: 10.1016/j.envres.2017.08.032.

FAO (2000) *Definitions of Forest, other land uses, and tree outside forests*, *FRA 2000 ON DEFINITIONS OF FOREST AND FOREST CHANGE*. Available at: http://www.fao.org/3/ad665e/ad665e00.htm#TopOfPage.

Favreau, J. M. *et al.* (2006) 'Recommendations for assessing the effectiveness of surrogate species approaches', *Biodiversity and Conservation*, 15(12), pp. 3949–3969. doi: 10.1007/s10531-005-2631-1.

Fernández, J. E. and Moreno, F. (2008) 'Water Use by the Olive Tree Water Use by the Olive Tree', (2000). doi: 10.1300/J144v02n02.

Fernández Jiménez, M. T., Climent-Font, A. and Sánchez Antón, J. L. (2003) 'Long term atmospheric pollution study at madrid city (spain)', *Water, Air, and Soil Pollution*, 142, pp. 243–260.

Filyushkina, A. *et al.* (2017) 'Preferences for variation in forest characteristics: Does diversity between stands matter?', *Ecological Economics*. Elsevier B.V., 140, pp. 22–29. doi: 10.1016/j.ecolecon.2017.04.010.

Filz, K. J. *et al.* (2013) ‘Missing the target? A critical view on butterfly conservation efforts on calcareous grasslands in south-western Germany’, *Biodiversity and Conservation*, 22(10), pp. 2223–2241. doi: 10.1007/s10531-012-0413-0.

Fisher, B., Turner, R. K. and Morling, P. (2009) ‘Defining and classifying ecosystem services for decision making’, *Ecological Economics*, 68(3), pp. 643–653. doi: 10.1016/j.ecolecon.2008.09.014.

Fleury-Bahi, G. *et al.* (2012) ‘Development and Validation of an Environmental Quality of Life Scale: Study of a French Sample’, *Social Indicators Research*, 113(3), pp. 903–913. doi: 10.1007/s11205-012-0119-4.

Flynn, K. M. and Traver, R. G. (2013) ‘Green infrastructure life cycle assessment: A bio-infiltration case study’, *Ecological Engineering*. Elsevier B.V., 55, pp. 9–22. doi: 10.1016/j.ecoleng.2013.01.004.

Folke, C. (2006) ‘Resilience: The emergence of a perspective for social-ecological systems analyses’, *Global Environmental Change*, 16(3), pp. 253–267. doi: 10.1016/j.gloenvcha.2006.04.002.

Foltête, J. C., Clauzel, C. and Vuidel, G. (2012) ‘A software tool dedicated to the modelling of landscape networks’, *Environmental Modelling and Software*, 38, pp. 316–327. doi: 10.1016/j.envsoft.2012.07.002.

Forman, R. T. T. (2014) *Urban Ecology: Science of Cities*. Cambridge University Press. Available at: https://books.google.it/books?id=gQSuAgAAQBAJ.

Forrester, D. I. *et al.* (2017) ‘Generalized biomass and leaf area allometric equations for European tree species incorporating stand structure, tree age and climate’, *Forest Ecology and Management*. The Authors, 396, pp. 160–175. doi: 10.1016/j.foreco.2017.04.011.

Foudi, S. *et al.* (2017) ‘The climatic dependencies of urban ecosystem services from green roofs: Threshold effects and non-linearity’, *Ecosystem Services*, 24, pp. 223–233. doi: 10.1016/j.ecoser.2017.03.004.

Francis, L. F. M. and Jensen, M. B. (2017) ‘Benefits of green roofs: A systematic review of the evidence for three ecosystem services’, *Urban Forestry and Urban Greening*, 28(April), pp. 167–176. doi: 10.1016/j.ufug.2017.10.015.

Frantzeskaki, N. *et al.* (2019) ‘Nature-based solutions for urban climate change adaptation: Linking science, policy, and practice communities for evidence-based decision-making’, *BioScience*, 69(6), pp. 455–466. doi: 10.1093/biosci/biz042.

Fry, G. *et al.* (2009) ‘The ecology of visual landscapes: Exploring the conceptual common ground of visual and ecological landscape indicators’, *Ecological Indicators*, 9(5), pp. 933–947. doi: 10.1016/j.ecolind.2008.11.008.

Fujiwara, M. and Takada, T. (2017) ‘Environmental Stochasticity’, in *eLS*. Chichester, UK: John Wiley & Sons Ltd. doi: 10.1002/9780470015902.a0021220.pub2]02/9780470015902.a0021220.pub2.

Futter, M. N. *et al.* (2014) ‘PERSiST : a flexible rainfall-runoff modelling toolkit for use with the INCA family of models’, *HYdrological and Earth System Sciences*, 18, pp. 855–873. doi:

10.5194/hess-18-855-2014.

Gemechu, E. D. *et al.* (2016) ‘Environmentally extended input-output analysis for sustainable regional development’, in Massari, S., Sonnemann, G., and Balkau, F. (eds) *Life cycle approaches to sustainable regional development*. 1st edn. Taylor & Francis, pp. 53–58.

Geneletti, D. (2015) ‘A conceptual approach to promote the integration of ecosystem services in strategic environmental assessment’, *Journal of Environmental Assessment Policy and Management*. World Scientific, 17(04), p. 1550035.

Goh, B. H. and Sun, Y. (2016) ‘The development of life-cycle costing for buildings The development of life-cycle costing for buildings’, *Building Research & Information*, 44(3), pp. 319–333. doi: 10.1080/09613218.2014.993566.

Golinkoff, J. (2010) ‘Biome BGC version 4.2: Theoretical framework of Biome-BGC’, *Respiration*, (Thornton 1998), pp. 1–71.

Gomes Miguez, M. *et al.* (2017) ‘Urban Flood Simulation Using MODCEL — An Alternative Quasi-2D Conceptual Model’, *Water*, 9(445), pp. 1–28. doi: 10.3390/w9060445.

Graca, M. *et al.* (2018) ‘Assessing how green space types affect ecosystem services delivery in Porto, Portugal’, *LANDSCAPE AND URBAN PLANNING*, 170, pp. 195–208. doi: 10.1016/j.landurbplan.2017.10.007.

Graça, M. S. *et al.* (2017) ‘Assessing mismatches in ecosystem services proficiency across the urban fabric of Porto (Portugal): The influence of structural and socioeconomic variables’, *Ecosystem Services*. Elsevier, 23(December 2016), pp. 82–93. doi: 10.1016/j.ecoser.2016.11.015.

Grace, K. *et al.* (2016) ‘A review of methods , data , and models to assess changes in the value of ecosystem services from land degradation and restoration’, *Ecological Modelling*. Elsevier B.V., 319, pp. 190–207. doi: 10.1016/j.ecolmodel.2015.07.017.

Grafius, D. R. *et al.* (2016) ‘The impact of land use/land cover scale on modelling urban ecosystem services’, *Landscape Ecology*. Springer Netherlands, 31(7), pp. 1509–1522. doi: 10.1007/s10980-015-0337-7.

Grandison, A. (2020) ‘Determining Confidence Intervals , and Convergence, for Parameters in Stochastic Evacuation Models’, *Fire Technology*. Springer US, 56(5), pp. 2137–2177. doi: 10.1007/s10694-020-00968-0.

Gray, M. *et al.* (2016) ‘Landscape feature-based permeability models relate to puma occurrence’, *Landscape and Urban Planning*. Elsevier B.V., 147, pp. 50–58. doi: 10.1016/j.landurbplan.2015.11.009.

Grêt-Regamey, A. *et al.* (2017) ‘Review of decision support tools to operationalize the ecosystem services concept’, *Ecosystem Services*. Elsevier B.V., 26(October 2016), pp. 306–315. doi: 10.1016/j.ecoser.2016.10.012.

de Groot, Rudolf. *et al.* (2010) ‘Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making’, *Ecological Complexity*. Elsevier B.V., 7(3), pp. 260–272. doi: 10.1016/j.ecocom.2009.10.006.

de Groot, Rudolf *et al.* (2010) 'Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation', *The economics of ecosystems and biodiversity: The ecological and economic foundations*. doi: 10.1017/s1355770x11000088.

de Groot, R. *et al.* (2012) 'Global estimates of the value of ecosystems and their services in monetary units', *Ecosystem Services*. Elsevier, 1(1), pp. 50–61. doi: 10.1016/j.ecoser.2012.07.005.

de Groot, R. S., Wilson, M. A. and Boumans, R. M. J. (2002) 'A typology for the classification, description and valuation of ecosystem functions, goods and services', *Ecological Economics*. doi: 10.1016/S0921-8009(02)00089-7.

Gurrutxaga, M., Lozano, P. J. and del Gabriel, B. (2010) 'Assessing highway permeability for the restoration of landscape connectivity between protected areas in the Basque country, Northern Spain', *Landscape Research*, 35(5), pp. 529–550. doi: 10.1080/01426397.2010.504915.

Gutleben, C. and Goumot, P. (2015) *Recensement du patrimoine arboré des alignements urbains et de la répartition des essences*.

Haase, A. (2017) 'The contribution of nature-based solutions to socially inclusive urban development–some reflections from a social-environmental perspective', in Kabisch, N. et al. (eds) *Theory and Practice of Urban Sustainability Transitions. Nature-based Solutions to Climate Change Adaptation in Urban Areas: Linkages between science, policy and practice*. Springer, pp. 221–236. doi: 10.1007/978-3-319-56091-5_3.

Haase, D. *et al.* (2014) 'A quantitative review of urban ecosystem service assessments: Concepts, models, and implementation', *Ambio*, 43(4), pp. 413–433. doi: 10.1007/s13280-014-0504-0.

Haase, D. *et al.* (2017) 'Greening cities – To be socially inclusive? About the alleged paradox of society and ecology in cities', *Habitat International*, 64, pp. 41–48. doi: 10.1016/j.habitatint.2017.04.005.

Haigh, S. L. (1998) 'Stem Diameter-Age Relationships of Tamarix ramosissima on Lake Shore and Stream Sites in Southern Nevada', *The Southwestern Naturalist*. Southwestern Association of Naturalists, 43(4), pp. 425–429. Available at: http://www.jstor.org/stable/30054078.

Haines-Young, R. and Potschin, M. (2010) 'The links between biodiversity, ecosystem services and human well-being', *Ecosystem Ecology: a new synthesis*. Cambridge University Press Cambridge, pp. 110–139.

Haines-Young, R. and Potschin, M. (2014) 'Typology / Classification of Ecosystem Services', *OpenNESS Ecosystem Services Reference Book*, (2), pp. 1–8. Available at: http://www.xn--opennessproject-7s9h.eu/library/reference?book.

Haines-Young, R. and Potschin, M. (2017) '2.4. Categorisation systems: The classification challenge', *Mapping Ecosystem Services*. Pensoft Publishers, Sofia, p. 42.

Haines-Young, R. and Potschin, M. B. (2018) *Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure*. Available at: www.cices.eu.

Haines-Young, R., Potschin, M. and Kienast, F. (2012) 'Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs', *Ecological Indicators*, 21, pp. 39–53. doi: 10.1016/j.ecolind.2011.09.004.

Hargreaves, G. H. and Samani, Z. A. (1985) 'Reference crop evapotranspiration from temperature', *Applied engineering in agriculture*. American Society of Agricultural and Biological Engineers, 1(2), pp. 96–99.

Harris, J. and Tewdwr-Jones, M. (2010) 'Ecosystem Services and Planning', *Town & Country Planning*, pp. 222–226.

Hassine, K. *et al.* (2014) 'Measuring Quality of Life in the Neighborhood: The Cases of Air-Polluted Cities in Tunisia', *Social Indicators Research*, 119(3), pp. 1603–1612. doi: 10.1007/s11205-013-0565-7.

Hastings, D. A. and Dunbar, P. K. (1993) 'Global land one-kilometer base elevation (GLOBE)'.

Heink, U. *et al.* (2016) 'Requirements for the selection of ecosystem service indicators–the case of MAES indicators', *Ecological Indicators*. Elsevier, 61, pp. 18–26.

Hellies, M., Deidda, R. and Viola, F. (2018) 'Retention performances of green roofs worldwide at different time scales', (November 2017), pp. 1940–1952. doi: 10.1002/ldr.2947.

Hirabayashi, S. (2013) 'i-Tree Eco Precipitation Interception Model Descriptions', p. 21. Available at: https://www.itreetools.org/eco/resources/iTree_Eco_Precipitation_Interception_Model_Descriptions.pdf.

Hirabayashi, S. (2016) 'Air Pollutant Removals , Biogenic Emissions and Hydrologic Estimates for i-Tree Applications'.

Hirabayashi, S., Kroll, C. N. and Nowak, D. J. (2012) 'i-Tree eco dry deposition model descriptions', *Citeseer*.

Hirabayashi, S., Kroll, C. N. and Nowak, D. J. (2015) 'i-Tree Eco Dry Deposition Model Descriptions', p. 35. Available at: http://www.itreetools.org/eco/resources/iTree_Eco_Dry_Deposition_Model_Descriptions.pdf.

Hobbs, R. J., Higgs, E. and Harris, J. A. (2009) 'Novel ecosystems: implications for conservation and restoration', *Trends in Ecology and Evolution*, 24(11), pp. 599–605. doi: 10.1016/j.tree.2009.05.012.

Hodgson, J. A. *et al.* (2012) 'The Speed of Range Shifts in Fragmented Landscapes', *PLoS ONE*, 7(10). doi: 10.1371/journal.pone.0047141.

Hollberg, A. *et al.* (2018) 'Designing Sustainable Technologies, Products and Policies', in *Designing Sustainable Technologies, Products and Policies*. Springer International Publishing. doi: 10.1007/978-3-319-66981-6.

Holling, C. S. (1973) 'Resilience and stability of ecological systems', *Annual review of ecology and systematics*. Annual Reviews 4139 El Camino Way, PO Box 10139, Palo Alto, CA 94303-0139, USA, 4(1), pp. 1–23.

Holling, C. S. (1996) 'Engineering resilience versus ecological resilience', *Engineering within ecological constraints*, 31(1996), p. 32.

Hong, W. and Guo, R. (2017) 'Indicators for quantitative evaluation of the social services function of urban greenbelt systems: A case study of shenzhen, China', *ECOLOGICAL INDICATORS*, 75, pp. 259–267. doi: 10.1016/j.ecolind.2016.12.044.

Hoogmartens, R. *et al.* (2014) 'Bridging the gap between LCA, LCC and CBA as sustainability assessment tools', *Environmental Impact Assessment Review*. Elsevier Inc., 48, pp. 27–33. doi: 10.1016/j.eiar.2014.05.001.

Huang, C., Geiger, E. L. and Kupfer, J. A. (2006) 'Sensitivity of landscape metrics to classification scheme', *International Journal of Remote Sensing*, 27(14), pp. 2927–2948. doi: 10.1080/01431160600554330.

Huijbregts, M. A. J. *et al.* (2017) 'ReCiPe 2016 v1.1'. Available at: www.rivm.nl/en.

Huttner, S. and Bruse, M. (2009) 'Numerical modeling of the urban climate–a preview on ENVI-met 4.0', in *7th international conference on urban climate ICUC-7, Yokohama, Japan*.

I-Tree (2020) *i-Tree Eco: User's Manual v6.0*.

IADB (2019) *The Emerging and Sustainable Cities Program of the Inter-american Development Bank*. Available at: https://www.iadb.org/en/urban-development-and-housing/emerging-and-sustainable-cities-program.

Ingram, D. L. (2012) 'Life cycle assessment of a field-grown red maple tree to estimate its carbon footprint components', *International Journal of Life Cycle Assessment*, 17, pp. 453–462. doi: 10.1007/s11367-012-0398-7.

Ingram, D. L. (2013) 'Life Cycle Assessment to Study the Carbon Footprint of System Components for Colorado Blue Spruce Field Production and Use', *J.AMER.SOC.HORT.SCI*, 138(1), pp. 3–11.

Ingram, D. L. and Fernandez, R. T. (2012) 'Life Cycle Assessment: A Tool for Determining the Environmental Impact of Horticultural Crop Production', *HORTTECHNOLOGY*, 22(June), pp. 275–279.

ISO (2017) *ISO 15686-5:2017, Buildings and constructed assets -- Service life planning -- Part 5: Life-cycle costing*.

ISO 14040 (1997) *ISO 14040: Environmental management-Life cycle assessment-principles and framework*.

ISO 14044 (2006) *Environmental management: Life cycle assessment; requirements and guidelines*. ISO Geneva.

ISO 14049 (2000) '14049 Environmental Management–Life Cycle Assessment Examples of Application of ISO 14041 to Goal and Scope Definition and Inventory Analysis', *International Organization for Standardization, Geneva*.

IVE (2019) *Base de Precios del Instituto Valenciano de la Edificación*. Available at: https://www.five.es/productos/herramientas-on-line/visualizador-2019/.

Jaeger, J. A. G. (2000) ‘Landscape division, splitting index, and effective mesh size: New measures of landscape fragmentation’, *Landscape Ecology*, 15(2), pp. 115–130. doi: 10.1023/A:1008129329289.

Januchowski-Hartley, S. R. *et al.* (2013) ‘Restoring aquatic ecosystem connectivity requires expanding inventories of both dams and road crossings’, *Frontiers in Ecology and the Environment*, 11(4), pp. 211–217. doi: 10.1890/120168.

Jenkinson, D. S. and Coleman, K. (2008) ‘The turnover of organic carbon in subsoils. Part 2. Modelling carbon turnover’, *European Journal of Soil Science*, 59(2), pp. 400–413. doi: 10.1111/j.1365-2389.2008.01026.x.

Jerez, M., Quevedo, A. and Moret, A. (2015) ‘Simulador de crecimiento y secuestro de carbono para plantaciones de teca en Venezuela : una aplicación en SIMILE Growth and carbon sequestration simulator for teak plantations in Venezuela : An application in SIMILE’, *Bosque*, 36(3), pp. 519–530. doi: 10.4067/S0717-92002015000300018.

Jim, C. Y. and Chan, M. W. H. (2016) ‘Urban greenspace delivery in Hong Kong: Spatial-institutional limitations and solutions’, *URBAN FORESTRY & URBAN GREENING*, 18, pp. 65–85. doi: 10.1016/j.ufug.2016.03.015.

Jim, C. Y. and Chen, W. Y. (2008) ‘Assessing the ecosystem service of air pollutant removal by urban trees in Guangzhou (China)’, *Journal of Environmental Management*, 88(4), pp. 665–676. doi: 10.1016/j.jenvman.2007.03.035.

Johnson, D. and Geisendorf, S. (2019) ‘Are Neighborhood-level SUDS Worth it? An Assessment of the Economic Value of Sustainable Urban Drainage System Scenarios Using Cost-Benefit Analyses’, *Ecological Economics*. Elsevier, 158(July 2018), pp. 194–205. doi: 10.1016/j.ecolecon.2018.12.024.

JRC-IES (2010) *General guide for Life Cycle Assessment-Detailed guidance, International Reference Life Cycle Data System (ILCD) Handbook*. Edited by J. R. C.-I. for E. and Sustainability. European Commission.

Jujnovsky, J. *et al.* (2017) ‘Water assessment in a peri-urban watershed in Mexico City: A focus on an ecosystem services approach’, *ECOSYSTEM SERVICES*, 24, pp. 91–100. doi: 10.1016/j.ecoser.2017.02.005.

Kabisch, N *et al.* (2016) ‘Nature-based solutions to climate change mitigation and adaptation in urban areas: perspectives on indicators, knowledge gaps, barriers, and opportunities for action’, *Ecology and Society*, 21(2), p. 39. doi: 10.5751/ES-08373-210239.

Kabisch, Nadja *et al.* (2016) ‘Nature-based solutions to climate change mitigation and adaptation in urban areas: perspectives on indicators, knowledge gaps, barriers, and opportunities for action’, *ECOLOGY AND SOCIETY*, 21(2). doi: 10.5751/ES-08373-210239.

Kale, R. V and Sahoo, B. (2011) ‘Green-Ampt Infiltration Models for Varied Field Conditions : A Revisit’, *Water Resources Management*, 25, pp. 3505–3536. doi: 10.1007/s11269-011-9868-0.

Kassomenos, P. A. *et al.* (2014) ‘Study of PM 10 and PM 2 . 5 levels in three European cities : Analysis of intra and inter urban variations’, *Atmospheric Environment*, 87, pp. 153–163.

Keeler, B. L. *et al.* (2019) 'Social-ecological and technological factors moderate the value of urban nature', *Nature Sustainability*. Springer US, 2(1), pp. 29–38. doi: 10.1038/s41893-018-0202-1.

Kendall, A. and Mcpherson, E. G. (2012) 'A life cycle greenhouse gas inventory of a tree production system', *International Journal of li*, 17, pp. 444–452. doi: 10.1007/s11367-011-0339-x.

Kennedy, C., Pincetl, S. and Bunje, P. (2011) 'The study of urban metabolism and its applications to urban planning and design', *Environmental Pollution*. Elsevier Ltd, 159(8–9), pp. 1965–1973. doi: 10.1016/j.envpol.2010.10.022.

Kent, M. (2009) 'Biogeography and landscape ecology: The way forward - gradients and graph theory', *Progress in Physical Geography*, 33(3), pp. 424–436. doi: 10.1177/0309133309338119.

Kim, G., Miller, P. and Nowak, D. (2016) 'The Value of Green Infrastructure on Vacant and Residential Land in Roanoke, Virginia', *SUSTAINABILITY*, 8(4). doi: 10.3390/su8040296.

Kindlmann, P. and Burel, F. (2008) 'Connectivity measures: A review', *Landscape Ecology*, 23(8), pp. 879–890. doi: 10.1007/s10980-008-9245-4.

Klar, N. *et al.* (2008) 'Habitat selection models for European wildcat conservation', *Biological Conservation*, 141(1), pp. 308–319. doi: 10.1016/j.biocon.2007.10.004.

Klar, N. *et al.* (2012) 'Between ecological theory and planning practice: (Re-) Connecting forest patches for the wildcat in Lower Saxony, Germany', *Landscape and Urban Planning*, 105(4), pp. 376–384. doi: 10.1016/j.landurbplan.2012.01.007.

Ko, Y. *et al.* (2015a) 'Factors affecting long-term mortality of residential shade trees: Evidence from Sacramento, California', *Urban Forestry and Urban Greening*. Elsevier GmbH., 14(3), pp. 500–507. doi: 10.1016/j.ufug.2015.05.002.

Ko, Y. *et al.* (2015b) 'Long-term monitoring of Sacramento Shade program trees: Tree survival, growth and energy-saving performance', *Landscape and Urban Planning*. Elsevier B.V., 143, pp. 183–191. doi: 10.1016/j.landurbplan.2015.07.017.

Koeser, A. K. *et al.* (2014) 'Factors influencing urban tree planting program growth and survival in Florida, United States', *Urban Forestry and Urban Greening*. Elsevier GmbH., 13(4), pp. 655–661. doi: 10.1016/j.ufug.2014.06.005.

Koivusalo, H. and Kokkonen, T. (2003) 'Modelling runoff generation in a forested catchment in southern Finland', *Hydrologic*, 17, pp. 313–328. doi: 10.1002/hyp.1126.

Kool, J. T., Moilanen, A. and Treml, E. A. (2013) 'Population connectivity: Recent advances and new perspectives', *Landscape Ecology*, 28(2), pp. 165–185. doi: 10.1007/s10980-012-9819-z.

Korhonen, J. and Seager, T. P. (2008) 'Beyond eco-efficiency: a resilience perspective', *Business Strategy and the Environment*. Wiley Online Library, 17(7), pp. 411–419.

Korhonen, J. and Snäkin, J. P. (2015) 'Quantifying the relationship of resilience and eco-efficiency in complex adaptive energy systems', *Ecological Economics*. Elsevier B.V., 120, pp. 83–92. doi: 10.1016/j.ecolecon.2015.09.006.

Kosareo, L. and Ries, R. (2007) ‘Comparative environmental life cycle assessment of green roofs’, *Building and Environment*, 42, pp. 2606–2613. doi: 10.1016/j.buildenv.2006.06.019.

Kotsila, P. *et al.* (2020) ‘Nature-based solutions as discursive tools and contested practices in urban nature’s neoliberalisation processes’, *Environment and Planning E: Nature and Space*, 0(0), pp. 1–23. doi: 10.1177/2514848620901437.

Kottek, M. *et al.* (2006) ‘World map of the Köppen-Geiger climate classification updated’, *Meteorologische Zeitschrift*, 15(3), pp. 259–263. doi: 10.1127/0941-2948/2006/0130.

Krzeminski, P. *et al.* (2019) ‘Performance of secondary wastewater treatment methods for the removal of contaminants of emerging concern implicated in crop uptake and antibiotic resistance spread : A review’, *Science of the Total Environment*. Elsevier B.V., 648, pp. 1052–1081. doi: 10.1016/j.scitotenv.2018.08.130.

Kuhlman, T. and Farrington, J. (2010) ‘What is Sustainability?’, pp. 3436–3448. doi: 10.3390/su2113436.

Kumar, M. and Kumar, P. (2008) ‘Valuation of the ecosystem services: A psycho-cultural perspective’, *Ecological Economics*, 64(4), pp. 808–819. doi: 10.1016/j.ecolecon.2007.05.008.

Landscape Institute (2016) *BIM for Landscape*. Taylor & Francis. Available at: https://books.google.it/books?id=suIeDAAAQBAJ.

Larsen, R. I. (1971) *A mathematical model for relating air quality measurements to air quality standards*. Environmental Protection Agency, Office of Air Programs.

Lausch, A. and Herzog, F. (2002) ‘Applicability of landscape metrics for the monitoring of landscape change: Issues of scale, resolution and interpretability’, *Ecological Indicators*, 2(1–2), pp. 3–15. doi: 10.1016/S1470-160X(02)00053-5.

Lechner, A. M. *et al.* (2015) ‘From static connectivity modelling to scenario-based planning at local and regional scales’, *Journal for Nature Conservation*. Elsevier GmbH., 28, pp. 78–88. doi: 10.1016/j.jnc.2015.09.003.

Lee, E. R., Mostaghimi, S. and Wynn, T. M. (2002) ‘a Model To Enhance Wetland Design and Optimize Nonpoint Source Pollution Control’, *Journal of the American Water Resources Association*, 38(1), pp. 17–32. doi: 10.1111/j.1752-1688.2002.tb01531.x.

Lehmann, I. *et al.* (2014) ‘Urban vegetation structure types as a methodological approach for identifying ecosystem services – Application to the analysis of micro-climatic effects’, *Ecological Indicators*, 42, pp. 58–72. doi: 10.1016/j.ecolind.2014.02.036.

Leitao, A. B. and Ahern, J. (2002) ‘Applying landscape ecological concepts and metrics in sustainable landscape planning’, *Landscape and Urban Planning*, 59(2), pp. 65–93.

Levasseur, A. *et al.* (2010) ‘Considering time in LCA: Dynamic LCA and its application to global warming impact assessments’, *Environmental Science and Technology*, 44(8), pp. 3169–3174. doi: 10.1021/es9030003.

Levasseur, A. *et al.* (2012) ‘Valuing temporary carbon storage’, *Nature Climate Change*. Nature Publishing Group, 2(1), pp. 6–8. doi: 10.1038/nclimate1335.

Levasseur, A. *et al.* (2013) 'Biogenic Carbon and Temporary Storage Addressed with Dynamic Life Cycle Assessment', *Journal of Industrial Ecology*, 17(1), pp. 117–128. doi: 10.1111/j.1530-9290.2012.00503.x.

Li, H., Chen, W. and He, W. (2015) 'Planning of green space ecological network in urban areas: An example of Nanchang, China', *International Journal of Environmental Research and Public Health*, 12(10), pp. 12889–12904. doi: 10.3390/ijerph121012889.

Limburg, K. E. *et al.* (2002) 'Complex systems and valuation', *Ecological Economics*, 41(3), pp. 409–420. doi: 10.1016/S0921-8009(02)00090-3.

Lin, B., Meyers, J. and Barnett, G. (2015) 'Understanding the potential loss and inequities of green space distribution with urban densification', *Urban Forestry & Urban Greening*. Elsevier, 14(4), pp. 952–958.

Lin, S. *et al.* (2018) 'Spatial trade-offs and synergies among ecosystem services within a global biodiversity hotspot', *Ecological Indicators*. Elsevier, 84(September 2017), pp. 371–381. doi: 10.1016/j.ecolind.2017.09.007.

Liquete, C. *et al.* (2016a) 'Integrated valuation of a nature-based solution for water pollution control. Highlighting hidden benefits', *Ecosystem Services*. Elsevier, 22(September), pp. 392–401. doi: 10.1016/j.ecoser.2016.09.011.

Liquete, C. *et al.* (2016b) 'Integrated valuation of a nature-based solution for water pollution control. Highlighting hidden benefits', *Ecosystem Services*. Elsevier, 22(June), pp. 392–401. doi: 10.1016/j.ecoser.2016.09.011.

Liski, J., Tuomi, M. and Rasinmäki, J. (2009) 'Yasso07 user-interface manual', *University of Helsinki*, (March). Available at: www.environment.fi/syke/yasso.

Liu, X. *et al.* (2012) 'An integrated approach of remote sensing, GIS and swarm intelligence for zoning protected ecological areas', *Landscape Ecology*, 27(3), pp. 447–463. doi: 10.1007/s10980-011-9684-1.

Lizarralde, G. *et al.* (2015) 'Sustainability and resilience in the built environment: The challenges of establishing a turquoise agenda in the UK', *Sustainable Cities and Society*. Elsevier B.V., 15, pp. 96–104. doi: 10.1016/j.scs.2014.12.004.

Lobo, G. P. *et al.* (2015) 'Catena Evaluation and improvement of the CLIGEN model for storm and rainfall erosivity generation in Central Chile', *Catena*. Elsevier B.V., 127, pp. 206–213. doi: 10.1016/j.catena.2015.01.002.

Loiseau, E. *et al.* (2013) 'Adapting the LCA framework to environmental assessment in land planning', *International Journal of Life Cycle Assessment*, 18(8), pp. 1533–1548. doi: 10.1007/s11367-013-0588-y.

Loiseau, E. *et al.* (2014) 'Implementation of an adapted LCA framework to environmental assessment of a territory: Important learning points from a French Mediterranean case study', *Journal of Cleaner Production*, 80, pp. 17–29. doi: 10.1016/j.jclepro.2014.05.059.

Loiseau, E. *et al.* (2018) 'Territorial Life Cycle Assessment (LCA): What exactly is it about? A proposal towards using a common terminology and a research agenda', *Journal of Cleaner Production*, 176, pp. 474–485. doi: 10.1016/j.jclepro.2017.12.169.

López-Serrano, F. R. *et al.* (2000) 'LAI estimation of natural pine forest using a non-standard sampling technique', *Agricultural and Forest Meteorology*, 101, pp. 95–111.

Lopsik, K. (2013) 'Life cycle assessment of small-scale constructed wetland and extended aeration activated sludge wastewater treatment system', *International Journal of Environmental Science and Technology*, pp. 1295–1308. doi: 10.1007/s13762-012-0159-y.

Lorilla, R. S. *et al.* (2018) 'Assessment of the spatial dynamics and interactions among multiple ecosystem services to promote effective policy making across Mediterranean island landscapes', *Sustainability (Switzerland)*, 10(9). doi: 10.3390/su10093285.

Loro, M. *et al.* (2015) 'Ecological connectivity analysis to reduce the barrier effect of roads. An innovative graph-theory approach to define wildlife corridors with multiple paths and without bottlenecks', *Landscape and Urban Planning*. Elsevier B.V., 139, pp. 149–162. doi: 10.1016/j.landurbplan.2015.03.006.

Lowenstein, D. M., Matteson, K. C. and Minor, E. S. (2015) 'Diversity of wild bees supports pollination services in an urbanized landscape', *Oecologia*. Springer Berlin Heidelberg, 179(3), pp. 811–821. doi: 10.1007/s00442-015-3389-0.

Lozano, J. (2010) 'Habitat use by European wildcats (Felis silvestris) in central Spain: what is the relative importance of forest variables ?', *Animal Biodiversity and Conservation*, 33(2), pp. 143–150.

Luck, T. *et al.* (2014) 'Assessing the contribution of arboriculture operations to anthropogenic greenhouse gas emissions : A case study of a UK tree surgery company', *The International Journal of Urban Forestry*. Taylor & Francis, 36(2), pp. 89–102. doi: 10.1080/03071375.2014.921483.

Luederitz, C. *et al.* (2015) 'A review of urban ecosystem services: Six key challenges for future research', *Ecosystem Services*, 14, pp. 98–112. doi: 10.1016/j.ecoser.2015.05.001.

Lundy, L. and Wade, R. (2011) 'Integrating sciences to sustain urban ecosystem services', *Progress in Physical Geography*, 35(5), pp. 653–669. doi: 10.1177/0309133311422464.

Luo, L., Hamilton, D. and Han, B. (2010) 'Estimation of total cloud cover from solar radiation observations at Lake Rotorua , New Zealand', *Solar Energy*. Elsevier Ltd, 84(3), pp. 501–506. doi: 10.1016/j.solener.2010.01.012.

Lupp, G. *et al.* (2016) 'Assessing the Recreation Value of Urban Woodland Using the Ecosystem Service Approach in Two Forests in the Munich Metropolitan Region', *SUSTAINABILITY*, 8(11). doi: 10.3390/su8111156.

Lyytimäki, J. and Sipilä, M. (2009) 'Hopping on one leg - The challenge of ecosystem disservices for urban green management', *Urban Forestry and Urban Greening*, 8(4), pp. 309–315. doi: 10.1016/j.ufug.2009.09.003.

Madadi, H. *et al.* (2017) 'Degradation of natural habitats by roads: Comparing land-take and noise effect zone', *Environmental Impact Assessment Review*, 65(October 2016), pp. 147–155. doi: 10.1016/j.eiar.2017.05.003.

Madrid City Council (2018) *Estrategia de Residuos*. Madrid.

Maes, J. *et al.* (2013) *Mapping and assessment of forest ecosystems and their services – An* *analytical framework for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020*. doi: 10.2779/12398.

Maes, J. *et al.* (2016) *Mapping and Assessment of Ecosystems and their Services. Urban Ecosystems*. doi: 10.2779/625242.

Maes, J. and Jacobs, S. (2017) 'Nature-Based Solutions for Europe's Sustainable Development', *Conservation Letters*, 10(1), pp. 121–124. doi: 10.1111/conl.12216.

Manyena, S. B. (2006) 'The concept of resilience revisited', *Disasters*, 30(4), pp. 433–450. doi: 10.1111/j.0361-3666.2006.00331.x.

Marchi, M. *et al.* (2015) 'Carbon dioxide sequestration model of a vertical greenery system', *Ecological Modelling*. Elsevier B.V., 306, pp. 46–56. doi: 10.1016/j.ecolmodel.2014.08.013.

Marron Institute of Urban Management (New York University) (2018) *100RC HANDBOOK: Planning for Resilient Urban Growth Tools for Proactively Managing Rapid Urban Growth*. Edited by Rockefeller Foundation. Rockefeller Foundation. Available at: http://www.100resilientcities.org/urban-areas-expand-proactive-planning-key/.

Martín-López, B. *et al.* (2014) 'Trade-offs across value-domains in ecosystem services assessment', *Ecological Indicators*. Elsevier Ltd, 37(PART A), pp. 220–228. doi: 10.1016/j.ecolind.2013.03.003.

Mcconville, J. R. (2006) 'Applying Life Cycle Thinking to International Water and Sanitation Development Projects : An assessment tool for project managers in sustainable development work', p. 8.

McDonough, K. *et al.* (2017) 'Analysis of publication trends in ecosystem services research', *Ecosystem Services*. Elsevier B.V., 25, pp. 82–88. doi: 10.1016/j.ecoser.2017.03.022.

McGarigal, K., Cushman, S. A. and Ene, E. (2012) *FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical and Continuous Maps*, *Computer software program produced by the authors at the University of Massachusetts, Amherst.* Available at: http://www.umass.edu/landeco/research/fragstats/fragstats.html (Accessed: 10 March 2018).

McPhearson, T., Haase, D., *et al.* (2016) 'Advancing understanding of the complex nature of urban systems', *Ecological Indicators*, 70(April), pp. 566–573. doi: 10.1016/j.ecolind.2016.03.054.

McPhearson, T., Pickett, S. T. A., *et al.* (2016) 'Advancing Urban Ecology toward a Science of Cities', *BioScience*, 66(3), pp. 198–212. doi: 10.1093/biosci/biw002.

McPherson, E. G., van Doorn, N. and Peper, P. (2016) *Urban Tree Database and Allometric Equations*. doi: 10.13140/RG.2.2.35769.98405.

Mcpherson, E. G. and Kendall, A. (2014) 'A life cycle carbon dioxide inventory of the Million Trees Los Angeles program', *International Journal of Life Cycle Assessment*, 19, pp. 1653–1665. doi: 10.1007/s11367-014-0772-8.

Mcpherson, E. G., Kendall, A. and Albers, S. (2015) 'Life cycle assessment of carbon dioxide for different arboricultural practices in Los Angeles , CA', *Urban Forestry & Urban Greening*.

Elsevier GmbH., 14(2), pp. 388–397. doi: 10.1016/j.ufug.2015.04.004.

Mcrae, B. H. *et al.* (2008) 'Using Circuit Theory To Model Connectivity in Ecology, Evolution, and Conservation', *Ecology*, 89(10), pp. 2712–2724. doi: 10.1890/07-1861.1.

McRae, B. H., Shah, V. B. and Mohapatra, T. K. (2013) 'Circuitscape 4 User Guide'. The Nature Conservancy. Available at: http://www.circuitscape.org.

McVittie, A. and Hussain, S. (2013) 'The Economics of Ecosystems & Biodiversity Valuation Database - Manual', p. 26.

MEA (2005) *Ecosystems and human well-being: General Synthesis*, *Millennium Ecosystem Assessment*. Washington, D.C: Island Press.

Meerow, S. and Newell, J. P. (2016) 'Urban resilience for whom, what, when, where, and why?', *Urban Geography*. Routledge, 00(00), pp. 1–21. doi: 10.1080/02723638.2016.1206395.

Meerow, S., Newell, J. P. and Stults, M. (2016) 'Defining urban resilience: A review', *Landscape and Urban Planning*. Elsevier B.V., 147, pp. 38–49. doi: 10.1016/j.landurbplan.2015.11.011.

Mimet, A., Clauzel, C. and Foltête, J. C. (2016) 'Locating wildlife crossings for multispecies connectivity across linear infrastructures', *Landscape Ecology*, 31(9), pp. 1955–1973. doi: 10.1007/s10980-016-0373-y.

Mitsch, W. J. (2012) 'What is ecological engineering ?', *Ecological Engineering*. Elsevier B.V., 45(October), pp. 5–12. doi: 10.1016/j.ecoleng.2012.04.013.

Mohammed, A. and Babatunde, A. O. (2017) 'Modelling heavy metals transformation in vertical flow constructed wetlands', *Ecological Modelling*. Elsevier B.V., 354, pp. 62–71. doi: 10.1016/j.ecolmodel.2017.03.012.

Moltesen, A. *et al.* (2018) 'Social Life Cycle Assessment: An Introduction', in Hauschild, M. Z., Rosenbaum, R. K., and Olsen, S. I. (eds) *Life Cycle Assessment: Theory and Practice*. Cham: Springer International Publishing, pp. 401–422. doi: 10.1007/978-3-319-56475-3_16.

Montero, G., Ruiz-peinado, R. and Muñoz, M. (2005) *Producción de biomasa y fijación de CO2 por los bosques españoles*. Madrid.

Morgan, R. P. C. (2013) *Soil Erosion and Conservation*, *Environmental Modelling: Finding Simplicity in Complexity: Second Edition*. doi: 10.1002/9781118351475.ch22.

Mörtberg, U. and Wallentinus, H.-G. (2000) 'Red-listed forest bird species in an urban environment — assessment of green space corridors', *Landscape and Urban Planning*, 50(4), pp. 215–226. doi: 10.1016/S0169-2046(00)00090-6.

Mortelliti, A., Santulli Sanzo, G. and Boitani, L. (2009) 'Species' surrogacy for conservation planning: Caveats from comparing the response of three arboreal rodents to habitat loss and fragmentation', *Biodiversity and Conservation*, 18(5), pp. 1131–1145. doi: 10.1007/s10531-008-9477-2.

Moseley, D. *et al.* (2017) 'Developing an indicator for the physical health benefits of recreation in woodlands', *Ecosystem Services*. doi: 10.1016/j.ecoser.2017.12.008.

Moss, D. (2014) *EUNIS habitat classification–a guide for users*, *European Topic Centre on Biological Diversity*. Citeseer.

Moss, J. L. *et al.* (2019) 'Influence of evaporative cooling by urban forests on cooling demand in cities', *Urban Forestry and Urban Greening*. Elsevier, (July 2017), pp. 0–1. doi: 10.1016/j.ufug.2018.07.023.

Müller, A. *et al.* (2018) '"Wild" in the city context: Do relative wild areas offer opportunities for urban biodiversity?', *Landscape and Urban Planning*, 170(February 2017), pp. 256–265. doi: 10.1016/j.landurbplan.2017.09.027.

Nadal-Sala, D., Gracia, C. A. and Sabaté, S. (2019) 'The RheaG Weather Generator Algorithm: Evaluation in Four Contrasting Climates from the Iberian Peninsula', *Journal of Applied Meteorology and Climatology*, 58(1), pp. 55–69.

Nassauer, J. I. and Opdam, P. (2008) 'Design in science: extending the landscape ecology paradigm', *Landscape Ecology*, 23(6), pp. 633–644. doi: 10.1007/s10980-008-9226-7.

Neitsch, S. . *et al.* (2011) 'Soil & Water Assessment Tool Theoretical Documentation Version 2009', *Texas Water Resources Institute*, pp. 1–647. doi: 10.1016/j.scitotenv.2015.11.063.

Nesshover, C. *et al.* (2017) 'The science, policy and practice of nature-based solutions: An interdisciplinary perspective', *Sci Total Environ*, 579, pp. 1215–1227. doi: 10.1016/j.scitotenv.2016.11.106.

Nesshöver, C. *et al.* (2017) 'The science, policy and practice of nature-based solutions: An interdisciplinary perspective', *Science of the Total Environment*. The Authors, 579, pp. 1215–1227. doi: 10.1016/j.scitotenv.2016.11.106.

Nicks, A. D. (1975) *Stochastic generation of hydrologic model inputs*. The University of Oklahoma.

Nicks, A. D. and Harp, J. F. (1980) 'STOCHASTIC GENERATION OF TEMPERATURE AND SOLAR RADIATION DATA', *Journal of Hydrology*, 48, pp. 1–17.

Ninements, Ü. and Valladares, F. (2006) 'Tolerance to Shade , Drought , and Waterlogging of Temperate Northern Hemisphere Trees and Shrubs', *Ecological Monographs*, 76(4), pp. 521–547. Available at: http://mobot.mobot.org/W3T/Search/foc.htmli.

Nitschelm, L. *et al.* (2016) 'Spatial differentiation in Life Cycle Assessment LCA applied to an agricultural territory : current practices and method development', *Journal of Cleaner Production*, 112, pp. 2472–2484. doi: 10.1016/j.jclepro.2015.09.138.

Nocco, M. A., Rouse, S. E. and Balster, N. J. (2016) 'Vegetation type alters water and nitrogen budgets in a controlled, replicated experiment on residential-sized rain gardens planted with prairie, shrub, and turfgrass', *URBAN ECOSYSTEMS*, 19(4), pp. 1665–1691. doi: 10.1007/s11252-016-0568-7.

La Notte, Alessandra, D'Amato, Dalia, Mäkinen Hanna, Parachini, Mari a Luisa, Liquete, Camino, Egoh, Benis, Geneletti, Davide, Crossman and Neville. (2017) 'Ecosystem services classification: A systems ecology perspective of the cascade framework', *Ecol Indic*, 74, pp.

392–402. doi: 10.1016/j.ecolind.2016.11.030.

Nowak, D. J. (2000) 'UFORE Methods', *Diversity*, 21.

Nowak, D. J. *et al.* (2008) 'A Ground-Based Method of Assessing Urban Forest Structure and Ecosystem Services', *Arboriculture & Urban Forestry*, 34(6), pp. 347–358. doi: 10.1039/b712015j.

Nowak, D. J., Greenfield, E. J. and Ash, R. M. (2019) 'Annual biomass loss and potential value of urban tree waste in the United States', *Urban Forestry & Urban Greening*. Elsevier, 46(September), p. 126469. doi: 10.1016/j.ufug.2019.126469.

Nowak, D., Kuroda, M. and Crane, D. (2004) 'Tree mortality rates and tree population projections in Baltimore, Maryland, USA', *Urban Forestry & Urban Greening*, 2, pp. 139–147. doi: 10.1078/1618-8667-00030.

Ode, Å., Tveit, M. S. and Fry, G. (2008) 'Capturing Landscape Visual Character Using Indicators: Touching Base with Landscape Aesthetic Theory', *Landscape Research*, 33(1), pp. 89–117. doi: 10.1080/01426390701773854.

Ode, Å., Tveit, M. S. and Fry, G. (2010) 'Advantages of using different data sources in assessment of landscape change and its effect on visual scale', *Ecological Indicators*, 10(1), pp. 24–31. doi: 10.1016/j.ecolind.2009.02.013.

OECD (2019) *Urban population by city size (indicator)*. doi: 10.1787/b4332f92-en.

Olguin, E. J. *et al.* (2017) 'Long-term assessment at field scale of Floating Treatment Wetlands for improvement of water quality and provision of ecosystem services in a eutrophic urban pond', *SCIENCE OF THE TOTAL ENVIRONMENT*, 584, pp. 561–571. doi: 10.1016/j.scitotenv.2017.01.072.

Olguín, E. J. *et al.* (2017) 'Long-term assessment at field scale of Floating Treatment Wetlands for improvement of water quality and provision of ecosystem services in a eutrophic urban pond', *Science of the Total Environment*, 584–585, pp. 561–571. doi: 10.1016/j.scitotenv.2017.01.072.

Oliver, A. and Pearl, D. S. (2018) 'Rethinking sustainability frameworks in neighbourhood projects: a process-based approach', *Building Research and Information*. Taylor & Francis, 46(5), pp. 513–527. doi: 10.1080/09613218.2017.1358569.

Olsson, M. (1997) 'Determinants of breeding dispersal in the sand lizard, Lacerta agilis,(Reptilia, Squamata)', *Biological Journal of the …*, 60, pp. 243–256. Available at: http://onlinelibrary.wiley.com/doi/10.1111/j.1095-8312.1997.tb01494.x/abstract.

Opdam, P. (2010) 'Learning science from practice', *Landscape Ecology*, 25(6), pp. 821–823. doi: 10.1007/s10980-010-9485-y.

Opher, T., Shapira, A. and Friedler, E. (2017) 'A comparative social life cycle assessment of urban domestic water reuse alternatives', *The International Journal of Life Cycle Assessment*. doi: 10.1007/s11367-017-1356-1.

Othoniel, B. *et al.* (2016) 'Assessment of Life Cycle Impacts on Ecosystem Services: Promise, Problems, and Prospects', *Environ Sci Technol*, 50(3), pp. 1077–1092. doi: 10.1021/acs.est.5b03706.

Ottelé, M. *et al.* (2011) 'Comparative life cycle analysis for green façades and living wall systems', *Energy and Buildings*, 43(12), pp. 3419–3429. doi: 10.1016/j.enbuild.2011.09.010.

Ouyang, Y. *et al.* (2007) 'Simulating uptake and transport of TNT by plants using STELLA', *Chemosphere*, 69(8), pp. 1245–1252. doi: 10.1016/j.chemosphere.2007.05.081.

Palmer, S. C. F., Coulon, A. and Travis, J. M. J. (2011) 'Introducing a "stochastic movement simulator" for estimating habitat connectivity', *Methods in Ecology and Evolution*, 2(3), pp. 258–268. doi: 10.1111/j.2041-210X.2010.00073.x.

Pappalardo, V. *et al.* (2017a) 'The potential of green infrastructure application in urban runoff control for land use planning: A preliminary evaluation from a southern Italy case study', *Ecosystem Services*. Elsevier B.V., 26(May), pp. 345–354. doi: 10.1016/j.ecoser.2017.04.015.

Pappalardo, V. *et al.* (2017b) 'The potential of green infrastructure application in urban runoff control for land use planning: A preliminary evaluation from a southern Italy case study', *Ecosystem Services*. Elsevier B.V., (June 2016), pp. 1–10. doi: 10.1016/j.ecoser.2017.04.015.

Paramasivam, C. R. and Venkatramanan, S (2019) 'Chapter 3 - An Introduction to Various Spatial Analysis Techniques', in Venkatramanan, Senapathi, Prasanna, M. V., and Chung, S. Y. B. T.-G. I. S. and G. T. for G. S. (eds). Elsevier, pp. 23–30. doi: https://doi.org/10.1016/B978-0-12-815413-7.00003-1.

Park, J. and Kim, J. (2019) 'Economic impacts of a linear urban park on local businesses: The case of Gyeongui Line Forest Park in Seoul', *Landscape and Urban Planning*. Elsevier, 181(August 2018), pp. 139–147. doi: 10.1016/j.landurbplan.2018.10.001.

Parque Tecnológico de Valdemingomez (2018) *Memoria de Actividades del Parque Tecnologico de Valdemingomez - 2018*.

Pascual, U. *et al.* (2010) 'Chapter 5 The economics of valuing ecosystem services and biodiversity', *The Economics of Ecosystems and Biodiversity. Ecological and economic foundations*, (March), pp. 183–255. doi: 10.4324/9781849775489.

Pataki, D. E. *et al.* (2011) 'Coupling biogeochemical cycles in urban environments: Ecosystem services, green solutions, and misconceptions', *Frontiers in Ecology and the Environment*, 9(1), pp. 27–36. doi: 10.1890/090220.

Pauleit, S. *et al.* (2017) 'Nature-Based Solutions and Climate Change – Four Shades of Green BT - Nature-Based Solutions to Climate Change Adaptation in Urban Areas: Linkages between Science, Policy and Practice', in Kabisch, N. et al. (eds). Cham: Springer International Publishing, pp. 29–49. doi: 10.1007/978-3-319-56091-5_3.

Peh, K. S. H. *et al.* (2013) 'TESSA: A toolkit for rapid assessment of ecosystem services at sites of biodiversity conservation importance', *Ecosystem Services*, 5, pp. 51–57. doi: 10.1016/j.ecoser.2013.06.003.

Pelorosso, R. *et al.* (2016) 'Evaluation of Ecosystem Services related to Bio-Energy Landscape Connectivity (BELC) for land use decision making across different planning scales', *ECOLOGICAL INDICATORS*, 61(1, SI), pp. 114–129. doi:

10.1016/j.ecolind.2015.01.016.

Pelorosso, R. *et al.* (2017a) 'PANDORA 3.0 plugin: A new biodiversity ecosystem service assessment tool for urban green infrastructure connectivity planning', *ECOSYSTEM SERVICES*, 26(B), pp. 476–482. doi: 10.1016/j.ecoser.2017.05.016.

Pelorosso, R. *et al.* (2017b) 'PANDORA 3 . 0 plugin : A new biodiversity ecosystem service assessment tool for urban green infrastructure connectivity planning', (June 2016). doi: 10.1016/j.ecoser.2017.05.016.

Pereboom, V. *et al.* (2008) 'Movement patterns, habitat selection, and corridor use of a typical woodland-dweller species, the European pine marten (Martes martes), in fragmented landscape', *Canadian Journal of Zoology*, 86(9), pp. 983–991. doi: 10.1139/Z08-076.

Pereira, J. (2018) 'Multi-node protection of landscape connectivity: habitat availability and topological reachability', *Community Ecology*, 19(2), pp. 176–185. doi: 10.1556/168.2018.19.2.10.

Pereira, J., Saura, S. and Jordán, F. (2017) 'Single-node vs. multi-node centrality in landscape graph analysis: key habitat patches and their protection for 20 bird species in NE Spain', *Methods in Ecology and Evolution*. Edited by F. Parrini, 8(11), pp. 1458–1467. doi: 10.1111/2041-210X.12783.

Peri, G. *et al.* (2012a) 'The cost of green roofs disposal in a life cycle perspective : Covering the gap', *Energy*. Elsevier Ltd, 48(1), pp. 406–414. doi: 10.1016/j.energy.2012.02.045.

Peri, G. *et al.* (2012b) 'The cost of green roofs disposal in a life cycle perspective: Covering the gap', *Energy*. Elsevier Ltd, 48(1), pp. 406–414. doi: 10.1016/j.energy.2012.02.045.

Perini, K. and Rosasco, P. (2013) 'Cost-benefit analysis for green façades and living wall systems', *Building and Environment*, 70, pp. 110–121. doi: 10.1016/j.buildenv.2013.08.012.

Petit-Boix, A. *et al.* (2017) 'Application of life cycle thinking towards sustainable cities: A review', *Journal of Cleaner Production*, 166, pp. 939–951. doi: 10.1016/j.jclepro.2017.08.030.

Petri, A. C. *et al.* (2016) 'How Green Are Trees ? — Using Life Cycle Assessment Methods to Assess Net Environmental Benefi ts 1', *Journal of Environmental Horticulture*, 34(December), pp. 101–110.

Petucco, C. *et al.* (2018) *Development of a monetary value scale in MIMES: Deliverable 4.2 of Nature Based Solutions for re-naturing cities: knowledge diffusion and decision support platform through new collaborative models*.

Pickett, S. T. A. *et al.* (2016) 'Evolution and future of urban ecological science : ecology in , of , and for the city', *Ecosystem Health and Sustainability*, 2(7), p. e01229. doi: 10.1002/ehs2.1229.

Pierce, J. C., Budd, W. W. and Lovrich Jr, N. P. (2011) 'Resilience and sustainability in US urban areas', *Environmental Politics*, 20(4), pp. 566–584. doi: 10.1080/09644016.2011.589580.

Pigné, Y. *et al.* (2020) 'A tool to operationalize dynamic LCA , including time differentiation on the complete background database', *The International Journal of Life Cycle Assessment*. The International Journal of Life Cycle Assessment, 25, pp. 267–279.

Pincetl, S. (2012) 'Nature, urban development and sustainability - What new elements are needed for a more comprehensive understanding?', *Cities*. Center for Sustainable Urban Systems, Institute of the Environment and Sustainability, La Kretz Hall, Suite 300, Los Angeles, CA 90095-1496, United States, 29(SUPPL.2), pp. S32–S37. doi: 10.1016/j.cities.2012.06.009.

Plummer, M. L. (2009) 'Assessing benefit transfer for the valuation of ecosystem services', *Frontiers in Ecology and the Environment*, 7(1), pp. 38–45. doi: 10.1890/080091.

Poodat, F. *et al.* (2015) 'Prioritizing Urban Habitats for Connectivity Conservation: Integrating Centrality and Ecological Metrics', *Environmental Management*. Springer US, 56(3), pp. 664–674. doi: 10.1007/s00267-015-0520-2.

Potschin-young, M. *et al.* (2018) 'Understanding the role of conceptual frameworks : Reading the ecosystem service cascade', *Ecosystem Services*. The Authors, 29, pp. 428–440. doi: 10.1016/j.ecoser.2017.05.015.

Potschin, M. *et al.* (2015) 'Nature-based solutions', *OpenNESS Ecosystem Service Reference Book. OpenNESS Synthesis Paper. Available at: http://www. openness-project. eu/library/reference-book/sp-NBS*.

Potschin, M. *et al.* (2016) 'Nature-based-solutions', in *OpenNESS Ecosystem Services Reference Book*.

PRéConsultants (2020) 'SimaPro, LCA software.' Amersfoort, The Netherlands: © PRé Consultants,. Available at: ttp://www.pre-sustainability.com/simapro.

Prominski, M. and Seggern, H. (2019) *Design Research for Urban Landscapes: Theories and Methods*. Taylor & Francis. Available at: https://books.google.lu/books?id=NmOXDwAAQBAJ.

Pulford, I. D. and Watson, C. (2003) 'Phytoremediation of heavy metal-contaminated land by trees - A review', *Environment International*, 29(4), pp. 529–540. doi: 10.1016/S0160-4120(02)00152-6.

Pulselli, F. M. *et al.* (2008) *The road to sustainability: GDP and future generations*. WIT Press.

Qian, Y. *et al.* (2017) 'Risk assessment and interpretation of heavy metal contaminated soils on an urban brownfield site in New York metropolitan area', *Environmental Science and Pollution Research*. Environmental Science and Pollution Research, 24(30), pp. 23549–23558. doi: 10.1007/s11356-017-9918-0.

R Core Team (2020) 'R: A language and environment for statistical computing.' Vienna, Austria.: R Foundation for Statistical Computing,. Available at: https://www.r-project.org/.

Raymond, C.M., Berry, P., Breil, M., Nita, M.R., Kabisch, N., de Bel, M., Enzi, V., Frantzeskaki, N., Geneletti, D., Cardinaletti, M., Lovinger, L., Basnou, C., Monteiro, A., Robrecht, H., Sgrigna, G., Munari, L. and Calfapietra, C. (2017) *An impact evaluation framework to support planning and evaluation of nature-based solutions projects. Report* *prepared by the EKLIPSE Expert Working Group on Nature-based Solutions to Promote Climate Resilience in Urban Areas*. doi: 10.13140/RG.2.2.18682.08643.

Raymond, C. M. *et al.* (2017) 'A framework for assessing and implementing the co-benefits of nature-based solutions in urban areas', *Environmental Science and Policy*. Elsevier, 77(July), pp. 15–24. doi: 10.1016/j.envsci.2017.07.008.

Reynolds, C. *et al.* (2017) 'Does "Greening" of Neotropical Cities Considerably Mitigate Carbon Dioxide Emissions? The Case of Medellin, Colombia', *Sustainability*, 9(5), p. 785. doi: 10.3390/su9050785.

Reynolds, C. C. *et al.* (2017) 'Does "Greening''' of Neotropical Cities Considerably Mitigate Carbon Dioxide Emissions? The Case of Medellin, Colombia"', *SUSTAINABILITY*, 9(5). doi: 10.3390/su9050785.

RFSC (2016) *The Reference Framework for European Sustainable Cities*. Available at: http://rfsc.eu/ (Accessed: 1 March 2019).

RIBA (2013) *RIBA Plan of Work 2013*. Available at: https://www.ribaplanofwork.com/PlanOfWork.aspx (Accessed: 31 October 2020).

Ribeiro, J. W. *et al.* (2017) 'LandScape Corridors (lscorridors): a new software package for modelling ecological corridors based on landscape patterns and species requirements', *Methods in Ecology and Evolution*. doi: 10.1111/2041-210x.12750.

Richardson, C. W. (1981) 'Stochastic simulation of daily precipitation, temperature, and solar radiation', *Water Resources Research*, 17(1), pp. 182–190. doi: 10.1029/WR017i001p00182.

Rödder, D. *et al.* (2016) 'Coupling Satellite Data with Species Distribution and Connectivity Models as a Tool for Environmental Management and Planning in Matrix-Sensitive Species', *Environmental Management*, 58(1), pp. 130–143. doi: 10.1007/s00267-016-0698-y.

Rödger, J.-M., Kjær, L. L. and Pagoropoulos, A. (2018) 'Life Cycle Costing: An Introduction', in Hauschild, M. Z., Rosenbaum, R. K., and Olsen, S. I. (eds) *Life Cycle Assessment: Theory and Practice*. Cham: Springer International Publishing, pp. 373–399. doi: 10.1007/978-3-319-56475-3_15.

Roman, L. A., Battles, J. J. and McBride, J. R. (2014) 'The balance of planting and mortality in a street tree population', *Urban Ecosystems*, 17(2), pp. 387–404. doi: 10.1007/s11252-013-0320-5.

Roman, L. A. and Scatena, F. N. (2011) 'Street tree survival rates: Meta-analysis of previous studies and application to a field survey in Philadelphia, PA, USA', *Urban Forestry and Urban Greening*. Elsevier GmbH., 10(4), pp. 269–274. doi: 10.1016/j.ufug.2011.05.008.

Rooney, R. C. *et al.* (2015) 'Replacing natural wetlands with stormwater management facilities: Biophysical and perceived social values', *Water Research*. Elsevier Ltd, 73, pp. 17–28. doi: 10.1016/j.watres.2014.12.035.

Rosasco, P. and Perini, K. (2018) 'Evaluating the economic sustainability of a vertical greening system: A Cost-Benefit Analysis of a pilot project in mediterranean area', *Building* *and Environment*. Elsevier Ltd, 142, pp. 524–533. doi: 10.1016/j.buildenv.2018.06.017.

Rosenbaum, R. K. *et al.* (2018) 'Life Cycle Impact Assessment', in Hauschild, M. Z., Rosenbaum, R. K., and Olsen, S. I. (eds) *Life Cycle Assessment: Theory and Practice*. Cham: Springer International Publishing, pp. 167–270. doi: 10.1007/978-3-319-56475-3_10.

Rossman, L. A. and Huber, W. C. (2016) *Storm Water Management Model Reference Manual Volume I–Hydrology (Revised)*, *US Environmental Protection Agency: Cincinnati, OH, USA*.

Roussel, F. *et al.* (2017) 'Testing the applicability of ecosystem services mapping methods for peri-urban contexts: A case study for Paris', *ECOLOGICAL INDICATORS*, 83, pp. 504–514. doi: 10.1016/j.ecolind.2017.07.046.

Rugani, B. *et al.* (2019) 'Towards integrating the ecosystem services cascade framework within the Life Cycle Assessment (LCA) cause-effect methodology', *Science of the Total Environment*. Elsevier B.V., 690, pp. 1284–1298. doi: 10.1016/j.scitotenv.2019.07.023.

Ruiz-González, A. *et al.* (2014) 'Landscape genetics for the empirical assessment of resistance surfaces: The European pine marten (Martes martes) as a target-species of a regional ecological network', *PLoS ONE*, 9(10). doi: 10.1371/journal.pone.0110552.

Russell, L. (2012) *The conservation and landscape genetics of the Sand Lizard (Lacerta agilis)*. University of Sussex.

Russo, A. *et al.* (2017) 'Edible green infrastructure: An approach and review of provisioning ecosystem services and disservices in urban environments', *Agriculture, Ecosystems & Environment*, 242, pp. 53–66. doi: 10.1016/j.agee.2017.03.026.

Sacristan, D., Peñarroya, B. and Recatala, L. (2015) 'Increasing the Knowledge on the Management of Cu-Contaminated Agricultural Soils by Cropping Tomato (Solanum Lycopersicum L.)', *Land Degradation and Development*, 26(6), pp. 587–595. doi: 10.1002/ldr.2319.

Saiz, S. *et al.* (2006) 'Comparative Life Cycle Assessment of Standard and Green Roofs', *Environmental Science and Technology*, 40(13), pp. 4312–4316. doi: 10.1021/es0517522.

Sala, A., Smith, S. D. and Devitt, D. A. (1996) 'Water Use by Tamarix Ramosissima and Associated Phreatophytes in a Mojave Desert Floodplain Author (s): Anna Sala , Stanley D . Smith and Dale A . Devitt Published by : Wiley on behalf of the Ecological Society of America Stable URL : http://www.jstor.c', *Ecological Applications*, 6(3), pp. 888–898.

Sala, S., Farioli, F. and Zamagni, A. (2013) 'Progress in sustainability science: Lessons learnt from current methodologies for sustainability assessment: Part 1', *International Journal of Life Cycle Assessment*, 18(9), pp. 1653–1672. doi: 10.1007/s11367-012-0508-6.

Salvador, P. *et al.* (2011) 'Spatial and temporal variations in PM10 and PM2 . 5 across Madrid metropolitan area in 1999-2008', *Procedia Environmental Sciences*, 4, pp. 198–208. doi: 10.1016/j.proenv.2011.03.024.

Santos-Martin, F. *et al.* (2018) 'Creating an operational database for Ecosystems Services Mapping and Assessment Methods', *One Ecosystem*, 3, p. e26719. doi: 10.3897/oneeco.3.e26719.

Santos-martín, F. *et al.* (2018) *Report on Social Mapping and Assessment Methods for Ecosystem Services*. doi: 10.13140/RG.2.2.30102.24644.

Saura, S. and Pascual-Hortal, L. (2007) 'A new habitat availability index to integrate connectivity in landscape conservation planning: Comparison with existing indices and application to a case study', *Landscape and Urban Planning*, 83(2–3), pp. 91–103. doi: 10.1016/j.landurbplan.2007.03.005.

Saura, S. and Torné, J. (2012) *Conefor 2.6 user manual*, *Universidad Politécnica de Madrid*. Available at: www.conefor.org. (Accessed: 10 March 2018).

Saxton, K. E. and Rawls, W. J. (2006) 'Soil water characteristic estimates by texture and organic matter for hydrologic solutions', *Soil Science Society of America Journal*, 70(5), pp. 1569–1578. doi: 10.2136/sssaj2005.0117.

Schäfer, K. V. R. *et al.* (2014) 'Carbon dioxide fluxes of an urban tidal marsh in the Hudson-Raritan estuary', *Journal of Geophysical Research: Biogeosciences*, 119, pp. 557–566. doi: 10.1002/2013JG002433.

Schaubroeck, T. (2017) 'A need for equal consideration of ecosystem disservices and services when valuing nature; countering arguments against disservices', *Ecosystem Services*. Elsevier B.V., 26, pp. 95–97. doi: 10.1016/j.ecoser.2017.06.009.

Schaubroeck, T., Petucco, C. and Benetto, E. (2019) 'Resources , Conservation & Recycling Evaluate impact also per stakeholder in sustainability assessment , especially for fi nancial analysis of circular economy initiatives', *Resources, Conservation & Recycling*. Elsevier, 150(August), p. 104411. doi: 10.1016/j.resconrec.2019.104411.

Schelhaas, M. J. *et al.* (2004) 'CO2FIX V 3.1 - A modelling framework for quantifying carbon sequestration in forest ecosystems', *Evolution*, 48(1068), pp. 965–978. doi: 10.2307/2410359.

Schneider, C. and Fry, G. (2005) 'Estimating the consequences of land-use changes on butterfly diversity in a marginal agricultural landscape in Sweden', *Journal for Nature Conservation*, 13(4), pp. 247–256. doi: 10.1016/j.jnc.2005.02.006.

Scholte, S. S. K., van Teeffelen, A. J. A. and Verburg, P. H. (2015) 'Integrating socio-cultural perspectives into ecosystem service valuation: A review of concepts and methods', *Ecological Economics*, 114, pp. 67–78. doi: 10.1016/j.ecolecon.2015.03.007.

Scolozzi, R. and Geneletti, D. (2012a) 'A multi-scale qualitative approach to assess the impact of urbanization on natural habitats and their connectivity', *Environmental Impact Assessment Review*, 36, pp. 9–22. doi: 10.1016/j.eiar.2012.03.001.

Scolozzi, R. and Geneletti, D. (2012b) 'Assessing habitat connectivity for land-use planning: a method integrating landscape graphs and Delphi survey', *Journal of Environmental Planning and Management*. Routledge, 55(6), pp. 813–830. doi: 10.1080/09640568.2011.628823.

SEEA (2012) 'System of Environmental-Economic Accounting 2012: Central Framework (United Nations, New York)'.

Selmi, W. *et al.* (2016) 'Air pollution removal by trees in public green spaces in Strasbourg city, France', *URBAN FORESTRY & URBAN GREENING*, 17, pp. 192–201. doi: 10.1016/j.ufug.2016.04.010.

Seppelt, R. *et al.* (2011) 'A quantitative review of ecosystem service studies: Approaches, shortcomings and the road ahead', *Journal of Applied Ecology*, 48(3), pp. 630–636. doi: 10.1111/j.1365-2664.2010.01952.x.

Shackleton, C. M. *et al.* (2016) 'Unpacking Pandora's Box: Understanding and Categorising Ecosystem Disservices for Environmental Management and Human Wellbeing', *Ecosystems*, 19(4), pp. 587–600. doi: 10.1007/s10021-015-9952-z.

Sharpley, A. N. and Williams, J. R. (1990) *Erosion productivity impact calculator: 1 Model Documentation (EPIC)*.

Shekhar, S. and Xiong, H. (2008) 'Soil and Water Assessment Tool "SWAT"', *Encyclopedia of GIS*, pp. 1068–1068. doi: 10.1007/978-0-387-35973-1_1231.

Shen, Y., Sun, F. and Che, Y. (2017) 'Public green spaces and human wellbeing: Mapping the spatial inequity and mismatching status of public green space in the Central City of Shanghai', *Urban Forestry & Urban Greening*. Elsevier, 27, pp. 59–68.

Sherrouse, B. C., Clement, J. M. and Semmens, D. J. (2011) 'A GIS application for assessing, mapping, and quantifying the social values of ecosystem services', *Applied Geography*, 31(2), pp. 748–760. doi: 10.1016/j.apgeog.2010.08.002.

Shirato, Y. and Yokozawa, M. (2006) 'Acid hydrolysis to partition plant material into decomposable and resistant fractions for use in the Rothamsted carbon model', *Soil Biology and Biochemistry*, 38(4), pp. 812–816. doi: 10.1016/j.soilbio.2005.07.008.

Sielezniew, M., Włostowski, M. and Dziekańska, I. (2010) 'Myrmica schencki (Hymenoptera: Formicidae) as the primary host of Phengaris (Maculinea) arion (Lepidoptera: Lycaenidae) at heathlands in Eastern Poland', *Sociobiology*, 55(1 B), pp. 95–106. doi: www.csuchico.edu/biol/Sociobiology/sociobiologyindex.html.

Silvennoinen, S. *et al.* (2017) 'Monetary value of urban green space as an ecosystem service provider: A case study of urban runoff management in Finland', *ECOSYSTEM SERVICES*, 28(A), pp. 17–27. doi: 10.1016/j.ecoser.2017.09.013.

Simpkins, C. E. *et al.* (2018) 'Assessing the performance of common landscape connectivity metrics using a virtual ecologist approach', *Ecological Modelling*. Elsevier B.V., 367, pp. 13–23. doi: 10.1016/j.ecolmodel.2017.11.001.

Singh, N. P. and Santal, A. R. (2015) 'Phytoremediation of Heavy Metals: The Use of Green Approaches to Clean the Environment BT - Phytoremediation: Management of Environmental Contaminants, Volume 2', in Ansari, A. A. et al. (eds). Cham: Springer International Publishing, pp. 115–129. doi: 10.1007/978-3-319-10969-5_10.

Skelton, A. *et al.* (2011) 'Mapping flows of embodied emissions in the global production system', *Environ Sci Technol*, 45(24), pp. 10516–10523. doi: 10.1021/es202313e.

Smith, A. C. *et al.* (2017) 'How natural capital delivers ecosystem services : A typology derived from a systematic review', *Ecosystem Services*, 26, pp. 111–126. doi: 10.1016/j.ecoser.2017.06.006.

Smith, J *et al.* (2010) 'Model to Estimate Carbon in Organic Soils – Sequestration and Emissions (ECOSSE).', *Carbon*, 44(August), pp. 1–73.

Soares, J. V and Almeida, A. C. (2001) 'Modeling the water balance and soil water fluxes in a fast growing Eucalyptus plantation in Brazil', *Journal of Hydrology*, 253, pp. 130–147.

Speak, A. *et al.* (2020) 'Total urban tree carbon storage and waste management emissions estimated using a combination of LiDAR , fi eld measurements and an end-of-life wood approach', *Journal of Cleaner Production*. Elsevier Ltd, 256, p. 120420. doi: 10.1016/j.jclepro.2020.120420.

Spitzer, L. *et al.* (2009) 'The Large Blue butterfly, Phengaris [Maculinea] arion, as a conservation umbrella on a landscape scale: The case of the Czech Carpathians', *Ecological Indicators*, 9(6), pp. 1056–1063. doi: 10.1016/j.ecolind.2008.12.006.

Sproul, J. *et al.* (2014) 'Economic comparison of white, green, and black flat roofs in the United States', *Energy and Buildings*. Elsevier B.V., 71, pp. 20–27. doi: 10.1016/j.enbuild.2013.11.058.

Sun, X. *et al.* (2017) 'Assessment of the denitrification process in alluvial wetlands at floodplain scale using the SWAT model', *Ecological Engineering*. Elsevier B.V., 103, pp. 344–358. doi: 10.1016/j.ecoleng.2016.06.098.

Sutherland, I. J. *et al.* (2018) 'Undervalued and under pressure : A plea for greater attention toward regulating ecosystem services', *Ecological Indicators*. Elsevier, 94(June 2017), pp. 23–32. doi: 10.1016/j.ecolind.2017.06.047.

Svoboda, J. A. and Dorf, R. C. (2003) *Introduction To Electric Circuits*. 6th Editio, *IEEE Transactions on Education*. 6th Editio. New York, USA: John Wiley and Sons. doi: 10.1109/TE.1997.572332.

Swarr, T. E. *et al.* (2011) 'Environmental life-cycle costing: a code of practice', *The International Journal of Life Cycle Assessment*, 16(5), pp. 389–391. doi: 10.1007/s11367-011-0287-5.

Syrbe, R.-U. and Walz, U. (2012) 'Spatial indicators for the assessment of ecosystem services: Providing, benefiting and connecting areas and landscape metrics', *Ecological Indicators*, 21, pp. 80–88. doi: 10.1016/j.ecolind.2012.02.013.

Szücs, L., Anders, U. and Bürger-Arndt, R. (2015) 'Assessment and illustration of cultural ecosystem services at the local scale - A retrospective trend analysis', *Ecological Indicators*. Elsevier Ltd, 50, pp. 120–134. doi: 10.1016/j.ecolind.2014.09.015.

Ta, M., Tardieu, L. and Levrel, H. (2020) *Specifying preference heterogeneity regarding natural attributes of urban green spaces to inform renaturation policies*, *CIRED Working Paper*. No. 2020-78. Paris, France.

Tassielli, G., Renzulli, P. A. and Notarnicola, B. (2016) 'Life Cycle Assessment', in Massari, S., Sonnemann, G., and Balkau, F. (eds) *Life cycle approaches to sustainable regional development*. 1st edn. Taylor & Francis, pp. 65–70.

Taylor, P. D. *et al.* (1993) 'Connectivity Is a Vital Element of Landscape Structure', *Oikos*. [Nordic Society Oikos, Wiley], 68(3), pp. 571–573. doi: 10.2307/3544927.

TECNALIA *et al.* (2018) *NBS Implementation Models typology*. Available at: https://www.nature4cities.eu/n4c-publications-and-results.

Termorshuizen, J. W. and Opdam, P. (2009) 'Landscape services as a bridge between landscape ecology and sustainable development', *Landscape Ecology*, 24(8), pp. 1037–1052. doi: 10.1007/s10980-008-9314-8.

The Mersey Forest *et al.* (2018) 'GI-Val: the green infrastructure valuation toolkit.' Available at: https://bit.ly/givaluationtoolkit.

Thompson, P. L., Rayfield, B. and Gonzalez, A. (2017) 'Loss of habitat and connectivity erodes species diversity, ecosystem functioning, and stability in metacommunity networks', *Ecography*, 40(1), pp. 98–108. doi: 10.1111/ecog.02558.

Thornton, I. *et al.* (2008) 'Urban geochemistry: Research strategies to assist risk assessment and remediation of brownfield sites in urban areas', *Environmental Geochemistry and Health*, 30(6), pp. 565–576. doi: 10.1007/s10653-008-9182-9.

Tian, Y. *et al.* (2011) 'Landscape ecological assessment of green space fragmentation in Hong Kong', *Urban Forestry & Urban Greening*. Elsevier GmbH., 10(2), pp. 79–86. doi: 10.1016/j.ufug.2010.11.002.

Tischendorf, L. and Fahrig, L. (2000) 'On the usage and measurement of landscape connectivity', *Oikos*, 90(1), pp. 7–19. doi: 10.1034/j.1600-0706.2000.900102.x.

Titeux, N., Mestdagh, X. and Cantú-Salazar, L. (2013) *Reporting under Article 17 of the Habitats Directive in Luxembourg (2007-2012): conservation status of species listed in Annexes II, IV and V of the European Council Directive on the Conservation of Habitats, Flora and Fauna (92/43/EEC)*.

Toxopeus, H. S. and Polzin, F. H. J. (2017) 'Characterizing nature-based solutions from a business model and financing perspective.' Naturvation.

Triest, L., Stiers, I. and Van Onsem, S. (2016) 'Biomanipulation as a nature-based solution to reduce cyanobacterial blooms', *Aquatic Ecology*. Springer Netherlands, 50(3), pp. 461–483. doi: 10.1007/s10452-015-9548-x.

Trodahl, M. I. *et al.* (2017) 'Investigating trade-offs between water quality and agricultural productivity using the Land Utilisation and Capability Indicator (LUCI)–A New Zealand application', *Ecosystem Services*. Elsevier B.V., 26(November 2016), pp. 388–399. doi: 10.1016/j.ecoser.2016.10.013.

Troy, A. and Wilson, M. A. (2006) 'Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer', *Ecological economics*. Elsevier, 60(2), pp. 435–449.

De Troyer, N., Mereta, S., *et al.* (2016) 'Water Quality Assessment of Streams and Wetlands in a Fast Growing East African City', *Water*, 8(4), p. 123. doi: 10.3390/w8040123.

De Troyer, N., Mereta, S. T., *et al.* (2016) 'Water Quality Assessment of Streams and Wetlands in a Fast Growing East African City', *WATER*, 8(4). doi: 10.3390/w8040123.

Turner, M. G. (2005) 'Landscape Ecology : What Is the State of the Science ?', *Annual Review of Ecology, Evolution, and Systematics*, 36, pp. 319–344. doi:

10.1146/annurev.ecolsys.36.102003.152614.

Turner, R. K. *et al.* (2003) 'Valuing nature: lessons learned and future research directions', *Ecological economics*. Elsevier, 46(3), pp. 493–510.

Tveit, M., Ode, Å. and Fry, G. (2006) 'Key concepts in a framework for analysing visual landscape character', *Landscape Research*, 31(3), pp. 229–255. doi: 10.1080/01426390600783269.

Tyllianakis, E. and Skuras, D. (2016) 'The income elasticity of Willingness-To-Pay (WTP) revisited: A meta-analysis of studies for restoring Good Ecological Status (GES) of water bodies under the Water Framework Directive (WFD)', *Journal of Environmental Management*. Elsevier Ltd, 182, pp. 531–541. doi: 10.1016/j.jenvman.2016.08.012.

UKNEA (2011) *The UK National Ecosystem Assessment: Synthesis of the Key Findings*, *UNEP-WCMC, Cambridge*.

UKNEA (2013) 'The UK national ecosystem assessment: Synthesis of the key findings. Cambridge: UNEP-WCMC'.

Ulubeyli, S. and Arslan, V. (2017) 'Economic viability of extensive green roofs through scenario and sensitivity analyses: Clients' perspective', *Energy and Buildings*. Elsevier B.V., 139, pp. 314–325. doi: 10.1016/j.enbuild.2017.01.042.

UN (2015) *Transforming our world: the 2030 agenda for sustainable development*.

UNDP (2016) *Sustainable Urbanization Strategy UNDP's Support to Sustainable, Inclusive and Resilient Cities in the Developing World*. New York, NY.

Unglaub, B. *et al.* (2015) 'Linking habitat suitability to demography in a pond-breeding amphibian', *Frontiers in Zoology*. ???, 12(1), pp. 1–10. doi: 10.1186/s12983-015-0103-3.

United Nations (2017) *New Urban Agenda*. Edited by U. Nations.

United Nations (2018) *The 2017 Revision of World Population Prospects*. Department of Economic and Social Affairs, Population Division. Available at: https://population.un.org/wpp/ (Accessed: 2 March 2019).

USDA (2020) *Plants Database*. Available at: https://plants.sc.egov.usda.gov/java/ (Accessed: 2 February 2020).

Uuemaa, E., Mander, Ü. and Marja, R. (2013) 'Trends in the use of landscape spatial metrics as landscape indicators: A review', *Ecological Indicators*. Elsevier Ltd, 28, pp. 100–106. doi: 10.1016/j.ecolind.2012.07.018.

Vaccari, F. P. *et al.* (2013) 'Carbon dioxide balance assessment of the city of Florence (Italy), and implications for urban planning', *Landscape and Urban Planning*. Elsevier B.V., 120, pp. 138–146. doi: 10.1016/j.landurbplan.2013.08.004.

Vacek, P., Struhala, K. and Mat, L. (2017) 'Life-cycle study on semi intensive green roofs', *Journal of Cleaner Production*, 154, pp. 203–213. doi: 10.1016/j.jclepro.2017.03.188.

Valdivia, S. *et al.* (2011) *Towards a life cycle sustainability assessment-making informed choices on products*.

Valdivia, S. *et al.* (2013) 'A UNEP/SETAC approach towards a life cycle sustainability assessment - Our contribution to Rio+20', *International Journal of Life Cycle Assessment*, 18(9), pp. 1673–1685. doi: 10.1007/s11367-012-0529-1.

Vaz, A. S. *et al.* (2017) 'Integrating ecosystem services and disservices: insights from plant invasions', *ECOSYSTEM SERVICES*, 23, pp. 94–107. doi: 10.1016/j.ecoser.2016.11.017.

Vázquez-Rowe, I. *et al.* (2013) 'Application of three independent consequential LCA approaches to the agricultural sector in Luxembourg', *International Journal of Life Cycle Assessment*, 18, pp. 1593–1604. doi: 10.1007/s11367-013-0604-2.

Vineyard, D. *et al.* (2015) 'Comparing Green and Grey Infrastructure Using Life Cycle Cost and Environmental Impact: A Rain Garden Case Study in Cincinnati, OH', *Journal of the American Water Resources Association*, 51(5), pp. 1342–1360. doi: 10.1111/1752-1688.12320.

Vogt, J., Hauer, R. J. and Fischer, B. C. (2015) 'The Costs of Maintaining and Not Maintaining the Urban Forest : A Review of the Urban Forestry and Arboriculture Literature', *Arborculture & Urban Forestry*, 41(November), pp. 293–323.

Vuorio, V., Reunanen, P. and Tikkanen, O.-P. (2016) 'Spatial Context of Breeding Ponds and Forest Management Affect the Distribution and Population Dynamics of the Great Crested Newt', *Annales Zoologici Fennici*, 53(1–2), pp. 19–34. doi: 10.5735/086.053.0202.

Walker, B. *et al.* (2004) 'Resilience, Adaptability and Transformability in Social – ecological Systems', *Ecology and Society*, 9(2), p. 5. doi: 10.1103/PhysRevLett.95.258101.

Wallace, K. J. (2007) 'Classification of ecosystem services: Problems and solutions', *Biological Conservation*, 139(3–4), pp. 235–246. doi: 10.1016/j.biocon.2007.07.015.

Wamsler, C. *et al.* (2016) 'Operationalizing ecosystem-based adaptation: Harnessing ecosystem services to buffer communities against climate change', *Ecology and Society*, 21(1). doi: 10.5751/ES-08266-210131.

Wang, J. and Banzhaf, E. (2018) 'Towards a better understanding of Green Infrastructure: A critical review', *Ecological Indicators*. Elsevier, 85(September 2017), pp. 758–772. doi: 10.1016/j.ecolind.2017.09.018.

Wang, T. *et al.* (2018) 'Assessment of a Field Tidal Flow Constructed Wetland in Treatment of Swine Wastewater : Life Cycle Approach', *Water*, 10(573). doi: 10.3390/w10050573.

Watts, K. *et al.* (2010) 'Targeting and evaluating biodiversity conservation action within fragmented landscapes: An approach based on generic focal species and least-cost networks', *Landscape Ecology*, 25(9), pp. 1305–1318. doi: 10.1007/s10980-010-9507-9.

Wei, X. *et al.* (2012) 'Allometric Equations for Predicting Above-ground Biomass of Tamarix in the Lower Colorado River Basin', *Desert Plants*, 28(2), pp. 6–16.

White, R. *et al.* (2000) 'Developing an urban land use simulator for European cities', *Proceedings of the Fifth EC GIS Workshop: GIS of Tomorrow*, pp. 179–190. Available at: http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.499.9771&rep=rep1&type=pdf .

Wickham, J. D. and Riitters, K. H. (1995) 'Sensitivity of landscape metrics to pixel size', *International Journal of Remote Sensing*. Taylor & Francis, 16(18), pp. 3585–3594.

Wolfram, M. and Frantzeskaki, N. (2016) 'Cities and Systemic Change for Sustainability: Prevailing Epistemologies and an Emerging Research Agenda', *Sustainability*, 8(2), p. 144. doi: 10.3390/su8020144.

Wong, N. H. *et al.* (2003) 'Life cycle cost analysis of rooftop gardens in Singapore', *Building and Environment*, 38(3), pp. 499–509. doi: 10.1016/S0360-1323(02)00131-2.

World Bank (2019) *Country classifications by income level: 2018-2019*. Available at: https://blogs.worldbank.org/opendata/new-country-classifications-income-level-2018-2019 (Accessed: 4 February 2019).

World Health Organisation (2005) *WHO Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide: Global update 2005 - Summary of risk assessment*.

Wu, J. (2010) 'Urban sustainability : an inevitable goal of landscape', *Landscape Ecology*, 25, pp. 1–4. doi: 10.1007/s10980-009-9444-7.

Wu, J. *et al.* (2013) 'Urban Landscape Ecology : Past , Present , and Future', in Fu, B. and Jones, K. B. (eds) *Landscape Ecology for Sustainable Environment and Culture*. Springer Science+Business Media, pp. 37–53. doi: 10.1007/978-94-007-6530-6.

Wu, J. (2014) 'Urban ecology and sustainability: The state-of-the-science and future directions', *Landscape and Urban Planning*, 125, pp. 209–221. doi: 10.1016/j.landurbplan.2014.01.018.

Wu, J. *et al.* (2018) 'Dismantling the fence for social justice? Evidence based on the inequity of urban green space accessibility in the central urban area of Beijing', *Environment and Planning B: Urban Analytics and City Science*. SAGE Publications Sage UK: London, England, p. 2399808318793139.

Wu, J. (Jingle) (2008) 'Making the Case for Landscape Ecology', *Landscape Journal*, 27, pp. 1–8.

Xing, Y., Jones, P. and Donnison, I. (2017) 'Characterisation of Nature-Based Solutions for the Built Environment', *Sustainability*, 9(1), p. 149. doi: 10.3390/su9010149.

Xu, C. *et al.* (2017) 'Life cycle environmental and economic assessment of a LID-BMP treatment train system: A case study in China', *Journal of Cleaner Production*, 149, pp. 227–237. doi: 10.1016/j.jclepro.2017.02.086.

Yang, G. *et al.* (2015) 'Using ecosystem service bundles to detect trade-offs and synergies across urban-rural complexes', *Landscape and Urban Planning*. Elsevier B.V., 136, pp. 110–121. doi: 10.1016/j.landurbplan.2014.12.006.

Yang, J. *et al.* (2017) 'Assessing the impacts of urbanization-associated green space on urban land surface temperature: A case study of Dalian, China', *URBAN FORESTRY & URBAN GREENING*, 22, pp. 1–10. doi: 10.1016/j.ufug.2017.01.002.

Yang, X., Zheng, X.-Q. and Lv, L.-N. (2012) 'A spatiotemporal model of land use change based on ant colony optimization, Markov chain and cellular automata', *Ecological Modelling*, 233, pp. 11–19. doi: https://doi.org/10.1016/j.ecolmodel.2012.03.011.

Zamagni, A. (2012) ‘Life cycle sustainability assessment’, *The International Journal of Life Cycle Assessment*, 17(4), pp. 373–376. doi: 10.1007/s11367-012-0389-8.

Zamagni, A., Pesonen, H.-L. and Swarr, T. (2013) ‘From LCA to Life Cycle Sustainability Assessment: concept, practice and future directions’, *The International Journal of Life Cycle Assessment*, 18(9), pp. 1637–1641. doi: 10.1007/s11367-013-0648-3.

Zanin, G. and Bortolini, L. (2018) ‘Assessing Stormwater Nutrient and Heavy Metal Plant Uptake in an Experimental Bioretention Pond’, (Lid), pp. 1–16. doi: 10.3390/land7040150.

Zank, B. *et al.* (2016) ‘Modeling the effects of urban expansion on natural capital stocks and ecosystem service flows: A case study in the Puget Sound, Washington, USA’, *Landscape and Urban Planning*. Elsevier B.V., 149, pp. 31–42. doi: 10.1016/j.landurbplan.2016.01.004.

Zardo, L. *et al.* (2017) ‘Estimating the cooling capacity of green infrastructures to support urban planning’, *ECOSYSTEM SERVICES*, 26(A), pp. 225–235. doi: 10.1016/j.ecoser.2017.06.016.

Zeigler, S. L. and Fagan, W. F. (2014) ‘Transient windows for connectivity in a changing world’, *Movement Ecology*, 2(1), pp. 1–10. doi: 10.1186/2051-3933-2-1.

Zemanova, M. A. *et al.* (2017) ‘Impact of deforestation on habitat connectivity thresholds for large carnivores in tropical forests’, *Ecological Processes*. Ecological Processes, 6(1), pp. 1–11. doi: 10.1186/s13717-017-0089-1.

Zetterberg, A., Mörtberg, U. M. and Balfors, B. (2010) ‘Making graph theory operational for landscape ecological assessments, planning, and design’, *Landscape and Urban Planning*, 95(4), pp. 181–191. doi: 10.1016/j.landurbplan.2010.01.002.

Zhang, D. *et al.* (2019) ‘Planning urban landscape to maintain key ecosystem services in a rapidly urbanizing area: A scenario analysis in the Beijing-Tianjin-Hebei urban agglomeration, China’, *Ecological Indicators*. Elsevier, 96(June 2018), pp. 559–571. doi: 10.1016/j.ecolind.2018.09.030.

Zhou, W., Pickett, S. T. A. and Cadenasso, M. L. (2016) ‘Shifting concepts of urban spatial heterogeneity and their implications for sustainability’, *Landscape Ecology*, 32(1), pp. 15–30. doi: 10.1007/s10980-016-0432-4.

Zölch, T. *et al.* (2016) ‘Using green infrastructure for urban climate-proofing: An evaluation of heat mitigation measures at the micro-scale’, *Urban Forestry and Urban Greening*. Elsevier GmbH., 20, pp. 305–316. doi: 10.1016/j.ufug.2016.09.011.

Zölch, T. *et al.* (2017a) ‘Regulating urban surface runo ff through nature-based solutions – An assessment at the micro-scale’, *Environmental Research*. Elsevier Inc., 157(April), pp. 135–144. doi: 10.1016/j.envres.2017.05.023.

Zölch, T. *et al.* (2017b) ‘Regulating urban surface runoff through nature-based solutions – An assessment at the micro-scale’, *Environmental Research*. Elsevier Inc., 157(November 2016), pp. 135–144. doi: 10.1016/j.envres.2017.05.023.

ANNEXES - CONTENT

The annexes for each chapter are included in Zenodo Online Repository:

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