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Ecosystem Services In Practice: Well-Being And Vulnerability Of Two European Urban Areas

by Yaella Depietri

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About the author

Yaella Depietri is an Environmental Scientist and has been in the field since her bachelor degree which she started back in 1999 at the Ca' Foscari University of Venice, Italy. In January 2015, she earned a PhD in Environmental Sciences at the Institute of Environmental Sciences and Technology (ICTA), Autonomous University of Barcelona (UAB) (Spain).

The PhD was carried out in collaboration with the United Nations University, Institute for Environment and Human Security (UNU-EHS) in Bonn (Germany) and was largely funded by the MOVE (Methods for the Improvement of Vulnerability Assessment in Europe) EU project. Before that, from 2006 to 2008, she worked as a young researcher at the Fondazione Eni Enrico Mattei (FEEM) (Italy) on EU projects related to the socio-economics of water resources management, in particular on decision support systems, public participation and adaptation.

In 2006, Yaella obtained a Master's degree in "Sustainable Development and Mediterranean Environments" from The Université de Perpignan, France and in 2005 a Master's degree in "Biogeochemical Oceanography" from The Université Aix-Marseille II, France. Her current interests are urban social-ecological systems, ecosystem services, disaster risk reduction and vulnerability assessment

ECOSYSTEM SERVICES IN PRACTICE: WELL-BEING AND VULNERABILITY OF TWO EUROPEAN URBAN AREAS

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PhD Thesis
Programme in Environmental Sciences
Autonomous University of Barcelona

Supervisors: Prof. Giorgos Kallis
Dr. Fabrice Renaud
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COVER IMAGE: Aerial view of Barcelona, Spain

Kol Ha'olam kulo
Gesher Tsar me'od
Gesher Tsar me'od
Gesher Tsar me'od
[…]
(Hebrew song)

The whole world / is a very narrow bridge / a very narrow bridge / a very narrow bridge […] (Own translation)

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Abstract

The large majority of the population in Europe lives in cities and the improvement of the quality of life in urban areas is a policy objective of primary importance. Cities are typically affected by crowding, traffic, air and noise pollution and present features that make them more vulnerable to natural hazards. This Thesis explains how ecosystem services help improve the livability of cities, with a focus on European medium-sized urban areas, using empirical, integrated (e.g. qualitative and quantitative or social-ecological) and policy relevant studies. It explores how and why urban areas can benefit, in terms of well-being and reduction of vulnerability to hazards, from a broader range of policies when ecological aspects are properly accounted for at different geographical scales.

The literature reviews of this Thesis find that the regulating functions of ecosystems often provide efficient, cost-effective alternatives or complementary solutions to hard infrastructures for the well-being of populations and disaster risk reduction in urban areas. Empirical research in the city of Cologne (Germany) shows that environmental variables play a central role in shaping the vulnerability of the urban social-ecological system to heat waves. While, in Barcelona (Spain), the assessment of regulating and cultural services supplied by the Collserola peri-urban Natural Park, demonstrates how a compact city benefits from the presence of a vast adjacent green area that enables it to reach an environmental performance and a hazards regulation potential comparable to that of other greener, western urban areas. Thus, besides the well-explored social, economic and physical dimensions of vulnerability, information about the ecological dimension needs also to be taken into account in urban vulnerability studies. In this respect, two features are found to be essential with respect to the vulnerability of urban areas: the impacts of urbanization on the local and surrounding ecosystems, which tend to further increase the damages that the social-ecological system can cause when a hazard strikes; and the connections of the urban core with surrounding green areas.

The research presented in this Thesis also yields theoretical conclusions about the nature of the interactions and coupling between the social and the ecological systems in the vulnerability assessment to natural hazards. It highlights how the social system depends on the ecosystem for exposure reduction and increased resilience. Detailed ecological information on ecosystem health should also be included. The ecosystem, especially if degraded, might not be effective or fail in supplying services when a hazard strikes, exacerbating the vulnerability of the human population.

Tracing back the steps that led to the present configuration and distribution of green areas in and around Cologne and especially in Barcelona, the research contributes to the characterization of the historical and political dimension of ecosystem services, and thus of vulnerability, in urban areas. Ecosystem services, besides being determined by the biophysical features of the territory, are the

outcome of the controversy between different and more often than not antagonistic social articulations of value. Conflicts over the management of green areas are mostly evident in urban areas due to the highly contested nature of space. Ecosystem services can thus be seen as socially constructed and need not only to be assessed and valued, but also to become part of a broader, participatory decision making process in which conflicting interests are discussed.

Finally, in each of the chapters focusing on different scales of the dependence of the urban population on ecosystems (from the properly urban to the watershed), the research shows that, despite the traditional focus on the local scale, the urban ecosystem presents different nested levels which are complementary in terms of the ecosystem services they provide to the city's inhabitants. This broader definition of urban ecosystems has implications in terms of an enlarged set of more targeted policies that can benefit urban areas.

Resumen

La gran mayoría de la población europea habita en ciudades y el mejoramiento de la calidad de vida en zonas urbanas constituye un objectivo primordial de las políticas gubernamentales. Las ciudades se ven típicamente afectadas por apiñamientos, tráfico intenso, contaminación atmosférica y acústica, y factores que aumentan su vulnerabilidad con respecto a riesgos naturales. Esta Tesis explica cómo los ecosistemas contribuyen a mejorar la habitabilidad de las ciudades, con foco en zonas urbanas europeas medianas, sirviéndose para ello de estudios empíricos integrados (por ejemplo cualitativos y cuantitativos o socio-ecológicos), así como estudios relevantes con respecto a las medidas a tomar por las autoridades gubernamentales. Explora cómo y porqué las zonas urbanas pueden beneficiarse, en términos del bienestar de sus habitantes y de la reducción de las vulnerabilidad frente a riesgos naturales, de una más amplia gama de medidas y soluciones cuando se tienen en cuenta en forma apropriada los aspectos ecológicos utilizando diferentes escalas geográficas.

La revisión de la literatura consultada para esta Tesis comprueba que las funciones reguladoras de los sistemas ecológicos a menudo proveen alternativas eficientes y rentables o soluciones complementarias para las infraestructuras que contribuyen a asegurar el bienestar de la población urbanas y reducir el riesgo de desastres. La investigación empírica en la ciudad de Colonia (Alemania) demuestra que las variables medioambientales juegan un rol central y decisivo, ya que determinan las características de la vulnerabilidad del sistema socio-ecológico urbano por ejemplo cuando es afectado por olas de calor. A su vez, en Barcelona (España), el asesoramiento de servicios reguladores ejemplifica cómo una ciudad compacta se beneficia al estar rodeada de una vasta superficie verde

adyacente, que le permite alcanzar un nivel de rendimiento ecológico y un potencial regulador de amenazas naturales comparables con otras ciudades más “verdes” de la Europa occidental. Por lo tanto, aparte de las dimensiones sociales, económicas y físicas bien exploradas de la vulnerabilidad, también su dimensión ecológica debe ser tenida en cuenta en el estudio de las zonas urbanas. En este sentido, dos aspectos revelan ser esenciales cuando se analiza la fragilidad urbana: los impactos de la urbanización en los sistemas ecológicos locales que tienden a agudizar aún más los daños causados por el sistema social-ecológico cuando azota un riesgo; y las conexiones del centro de la urbe con las zonas verdes circundantes.

La investigación presentada en esta Tesis provee asimismo conclusiones teóricas respecto a la naturaleza de las interacciones y el acoplamiento de los sistemas sociales y ecológicos al asesorar la vulnerabilidad relacionada con amenazas naturales. Subraya cómo el sistema social depende del ecosistema para reducir la exposición a tales riesgos y aumentar la resiliencia. Para ello se requiere también la inclusión de datos ecológicos detallados concernientes la salud del ecosistema. El ecosistema, especialmente en caso de degradación, puede perder efectividad o fallar en la provisión de servicios en caso de amenazas, exacerbando así la vulnerabilidad de la populación humana.

Volviendo atrás los pasos que llevaron en el pasado a la configuración y distribución actual de zonas verdes en y alrededor de la ciudad de Colonia y en especial de Barcelona, la investigación contribuye a caracterizar las dimensiones históricas y políticas de los ecosistema pertinentes, y por lo tanto, de la vulnerabilidad en zonas urbanas. Los ecosistemas, a más de ser determinados por características biofísicas del territorio, son el producto del enfrentamiento de diferentes articulaciones sociales valuatorias que son a menudo antagónicas. Los conflictos referentes a la gestión de zonas verdes son evidentes en su mayor parte en las zonas urbanas dado el alto valor del espacio y la lucha por obtenerlo. Los ecosistemas por lo tanto pueden ser vistos como construcciones sociales que no solamente requieren ser asesoradas y valorizadas, sino también ser incluídas en un proceso más amplio de toma de decisiones en el que se discuten los conflictos de intereses en forma participative.

Finalmente, en cada uno de los capítulos que enfocan los diferentes grados de dependencia de la población urbana de los servicios proporcionados por los ecosistemas (tanto en el ámbito urbano propiamente dicho como en cuencas y vertientes), la investigación demuestra que, a pesar del foco tradicional concentrado en la escala local, el ecosistema urbano alberga en si mismo diferentes niveles anidados que complementan los servicios prestados a los habitantes de la ciudad. Esta definición más comprensiva de lo que los ecosistemas urbanos abarcan implica la ampliación del conjunto de medidas y soluciones políticas que tienen como objetivo favorecer las zonas urbanas.

PREFACE

This thesis is submitted for the doctoral degree in the field of Environmental Science at the Institute of Environmental Science and Technology (ICTA), Autonomous University of Barcelona (UAB). It was largely funded by the MOVE (Methods for the Improvement of Vulnerability Assessment in Europe) EC Funded Project (7^{th} Framework Programme, contract number: 211590) at the United Nations University, Institute for Environment and Human Security (UNU-EHS). Two of the dissertation chapters have been published in peer-reviewed scientific journals, one chapter has been published in a working paper series and one is intended to be submitted to a scientific journal with the required amendments.

Working on an emerging field of research on the role of ecosystems for disaster risk reduction, in close contact with renowned specialists in the field of vulnerability assessment at UNU-EHS and within the MOVE project, highly fuelled my interest in the field and the ambition to make an original contribution to it. In fact, little work has been done so far to shed light on the nature of the ecological component of vulnerability to hazards, and even less so in urban areas, which made it a very appealing subject to work on. Progressing with the work, I also got interested in the question of the scales involved in the functioning of urban ecosystems, especially with respect to regulating services. This inspired me to offer, in parallel, a reflection on the definition of urban ecosystems based on practical cases to document a revised and enlarged characterization of such systems. Finally, the high concentration of human activities in and around urban areas made it a particularly appropriate frame to explore the socio-political nature of ecosystem services. As a result, this Thesis has become a journey throughout some of the less explored aspects of the integrated, empirical and theoretical study of urban social-ecological systems seen through the lens of the ecosystem services concept.

Accompanying papers to this Thesis are two published book chapters which I co-authored. One focuses on the vulnerability, mainly to floods, in Cologne (Germany), published by Elsevier, and the other is on ecosystems and disaster risk in urban areas, published by the UNU Press.

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Essential has been the collaboration with my co-authors in the various chapters: my supervisors, Torsten Welle, Francesc Baró, Lorenzo Guadagno and Margaretha Breil. At UNU-EHS, Tobias Raphael Blätgen, Divya Rajeswari and Philipp Koch greatly contributed to the successful work on the Cologne (Germany) case study. I thank Martí Boada, Erik Gómez Baggethun, Mariano Barriendos, Jordi Honey-Roses, Beatriz Rodriguez Labajos and Heleni Munujos Vinyoles for the rich discussions on the Barcelona (Spain) case study as well as the anonymous reviewers of both the published papers of Chapters 1 and 2.

To Carlota Schneider for her constant support and care, and, not least, for proof reading large parts of the Thesis and for translating the abstract. I feel fortunate I could count on her.

Many thanks go to the Fondazione Eni Enrico Mattei (FEEM) of Venise and Carlo Carraro for hosting me, again I'd say, for some months and, this time, as a visiting researcher.

Yet, this work would not have been possible without the material and often much needed moral support of my parents, my brother and his wife Anna and of my wonderful family at large. I owe them a lot. I'm particularly grateful also to Ramy and Shachar Banin and to all the Chabad of Venice for their care, the teachings and for making my weeks so much more agreeable and meaningful. To my friends on whom, wherever they were, in Venice, Bonn, Barcelona, Paris, London, Ispra, Sydney, Israel, Tunisia, Egypt… I could count on.

INTRODUCTION

Background

At present, more than 50% of the world population lives in urban areas and the proportion reaches 75% in Europe where it is projected to increase to about 80% by 2020 (EEA, 2010a). A trend mainly driven by the wider and increasing range of opportunities which urban agglomerations offer, as well as the inhabitants' proximity to businesses and services (EEA, 2014a). However, urban stressors such as noise from traffic, fear of crime, pollution and crowding are current problems of European cities which may encourage people to look for greener areas in the suburbs (EEA, 2010a; van den Berg et al., 2007). The expansion of cities which accompanies population growth brings new challenges to the well-being of urban population themselves by altering and fragmenting the local and surrounding ecosystems, besides diminishing efficiency in energy and resources use (EEA, 2014a). In Europe, urban sprawl (or unplanned incremental urban expansion) is growing faster than the population and is undermining urban sustainability by consuming land and further increasing energy use, noise, air pollution and emissions of greenhouses gases (EEA, 2006). Angel et al. (2010) found that, in the built up areas of developed countries, population density was on average of 70 persons per ha (excluding land-rich developed countries such as the US, Canada, Australia and Russia), which is low if compared with cities of developing nations presenting on average a density of 135 persons per ha. Furthermore, while in the developing world low access to sanitation, water scarcity and epidemics are some of the main environmental risk factors in urban areas (MA, 2005), the main threats to city inhabitants of affluent societies are air pollution (notably particular matter, PM_{10}, and ozone, O_3), high exposure to noise, and natural and anthropogenic hazards, susceptible to be magnified in intensity and frequency by climate change (EEA, 2012).

Facing these threats, Europe has implemented various measures, for instance to cut emissions of several air pollutants in recent decades which greatly reduced them including the exposure to other harmful substances to human health and ecosystems (EEA, 2013, 2010a). However, hazards and disastrous consequences are still on the rise in Europe (EEA, 2010b). According to the International Emergency Disasters Database (EM-DAT, http://www.emdat.be), between 1950 and 2014 the number of people affected continued to be relatively small in Europe, especially if compared to other continents such Asia, but the total damages incurred, estimated in US$, increased significantly. Hydrological, meteorological and climate related hazards, in particular, are of increasing concern (see Figure 0.1).

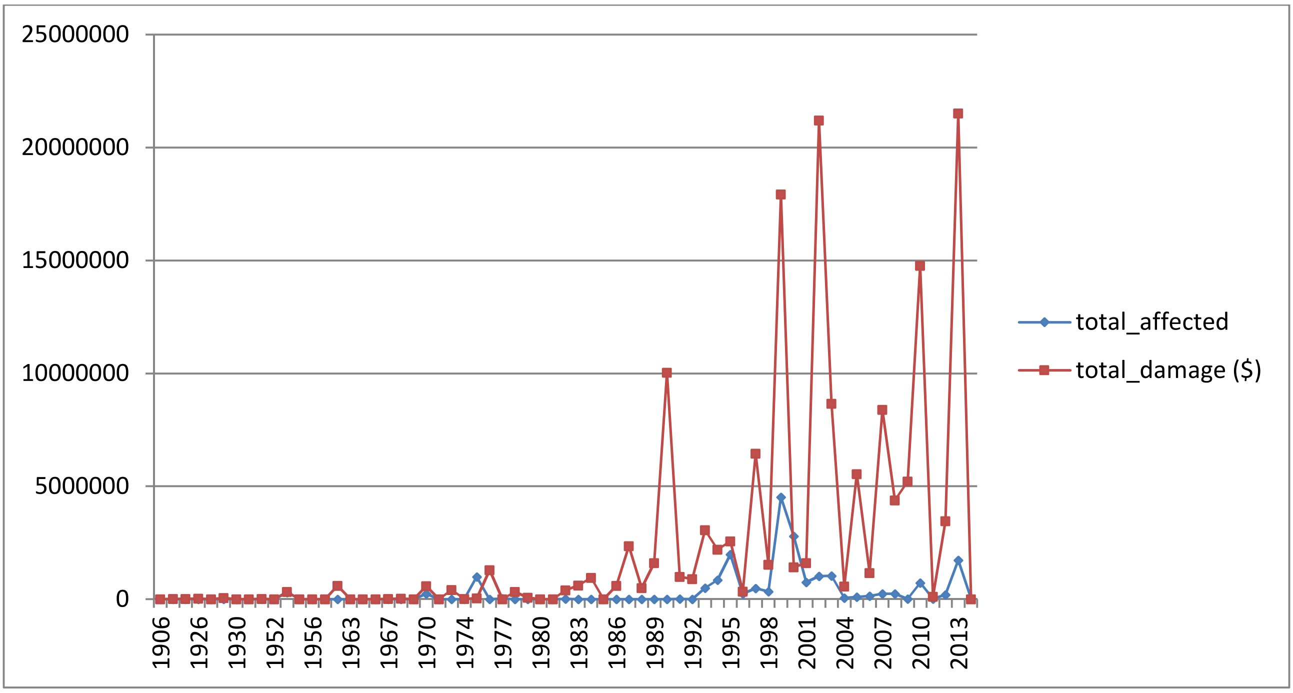

Figure 0.1. Total number of persons affected and total damages (US$) due to climatological, meteorological and hydrological hazards in Europe between 1900 and 2014
(Data source: International Emergency Disasters Database, EM-DAT, http://www.emdat.be/).

Between 1998 and 2009, hydro-meteorological hazards have been the most prominent natural hazards in Europe: extreme temperature events were the cause of major casualties (i.e. 70.000 in summer 2003), while floods were, along with storms, the most important in terms of economic losses (e.g. in the Elbe Basin in 2002 these amounted to € 20 billion, and in Italy, France and the Swiss Alps in 2000 to around € 12 billion) (EEA, 2010b). Although climate change further challenges the life quality of European cities (EEA, 2010a), the increase in damages measured so far is most likely due to changes in exposure. These can be associated to changes in the physical, technological and human/social systems (Barredo, 2010; EEA, 2010b; IPCC, 2012), more than to an increase of frequency and intensity of hazards, as it has been demonstrated for the case of floods (Barredo, 2009).

Cities, in particular, are especially susceptible to flash floods due to the imperviousness of land surfaces which facilitates surface runoff. But also riverine floods are of concern for European cities due to the spread of low density suburbs and the occupation of floodplains by homes and industries (Mitchell, 2003). Urban areas are also more affected by heat waves than their surrounding rural zones due to the higher population density and the Urban Heat Island (UHI) effect which maintains temperatures high also at night time. Badly planned and managed urban areas and declining ecosystems are in fact two of the major drivers of hazard risk worldwide (UNISDR, 2013). Overall,

the increase in hazards impacts in European cities can mainly be traced back to changes in the human and biophysical environment involving the replacement of vegetation with sealed surfaces.

Well preserved ecosystems in and around urban areas contribute in different ways to the well-being of urban populations (Bolund and Hunhammar, 1999). Peoples' quality of life in urban areas depends in fact to a great extent on the state of ecosystems as these provide goods (such as food, water, medicines and energy) and services (such as the dilution and transformation of waste, the regulation of the water cycle, air purification, climate regulation, urban cooling, noise reduction, carbon sequestration, the maintenance of biodiversity and recreation) that sustain and satisfy human life (MA, 2005). However, ecosystems have been degraded globally (MA, 2005) and their functions traditionally overlooked or supplanted by the construction of hard (grey or engineering) infrastructures. Regarding hazard risk, engineering works not only do not tackle the root causes of risk but also potentially increase the vulnerability of populations in the long run. It has been shown that grey infrastructures encourage people to settle in unsafe areas due to a false sense of security, further increasing the long term vulnerability of the social-ecological system (Pielke, 1999; Tobin, 1995). When a hazard of higher magnitude than average affects the area or if grey infrastructures fail, the losses are very high and magnified (e.g. Katrina in New Orleans or the flooding in England in the summer of 2007) (EEA, 2010a; UNISDR, 2013). Improvements in the well-being of the urban population might therefore increasingly rely on the preservation and healthiness of the local and surrounding ecosystems and the services these provide.

Objectives and research questions

This Thesis explores some aspects of the most pressing issues related to human well-being, ecosystem services (ES) and urban sustainability. It aims at showing, through different cases, how the concept of ES can be employed to improve the sustainability of cities especially in Europe, going beyond a mere exercise of assessment or valuation. Concentrating mainly on regulating services[1], this research points at providing empirical evidences, policy relevant conclusions and theoretical insights into the role of ecosystems for urban populations well-being and vulnerability reduction in Europe. It aims at adding for instance to the large literature on the social (Cannon, 2008; Cutter et al., 2003; Cutter and Finch, 2008; Wisner et al., 2004) or the physical (Fuchs et al., 2007; Papathoma-Köhle et al., 2011) dimensions of vulnerability to hazards already available by focusing on its less explored ecological dimension. Overall, it is at the hybrid space between the social and ecological dimensions

[1] The Millennium Ecosystem Assessment distinguishes between provisioning, regulating, cultural, and supporting services (MA, 2005).

of urban studies that the present Thesis aims at shedding light on. Furthermore, it intends to provide insights on the multiple scales involved in the supply of services to urban areas.

Particularly, it seeks to answer the following research questions:

1) How do ecosystems and their services contribute to shaping the vulnerability of urban populations to hydro-meteorological hazards in Europe? What are the theoretical as well as the policy implications?
2) How are ES embedded in socio-economic and political processes, especially in and around urban areas?
3) At what corresponding relevant scale each of the different ES analysed is supplied and more effectively contributes to the well-being of the urban population?

Each one of these questions finds answers across the chapters, which are then summarized in the conclusions.

Study areas

The focus of the Thesis is on European urban areas, not least because this work has been initiated and framed within the area of study of the MOVE project. However, cities in Europe present some features which make them particularly suitable to explore environmental related aspects of urban sustainability. Despite urban expansion, which culminated in the second half of the last century, European urban areas have a long tradition in including in their urban plans green open spaces and woodlands in proximity to the city core (Beatley, 2000). These often take the form of green fingers or green belts. The two case studies analysed in this Thesis show how the historically traceable inclusion of green areas in city planning in and around cities provides responses to some of the most pressing issues that these areas face at present.

The first case study is the **Cologne** urban area, located in the Federal State of North Rhine Westphalia (NRW), in Germany (Figure 0.2). Cologne can benefit from a well-managed forest distributed along two circular green belts (*Grüngürtel*) which expand concentrically, the outer belt following the old fortification perimeter. Both green rings are also partially connected radially through green corridors. The presence of well distributed green areas allows assessing the role of the vegetation in Disaster Risk Reduction (DRR) across different neighbourhoods. The green belts were planned and created between 1921-1924 (Sennott, 2004), a fact that also permits to consider the historical path leading to the present configuration of vulnerability. Cologne is surrounded by agricultural land, additionally

making it a suitable case to account for the role played by peri-urban ecosystems for the urban inhabitants' well-being.

The Cologne area has particularly suffered during the 2003 heat wave. In fact, of the 7295 excess deaths recoded in Germany in 2003 August alone (an 11% increase), most concentrated in north-western Germany (Robine et al., 2008). The region is also projected to increasingly suffer from heat stress in the coming decades (http://www.eea.europa.eu/data-and-maps/figures/increase-in-the-number-of, retrieved on 1st October 2014). In NRW, the increase in elderly population aggravates the hazard's impacts, while, specifically in Cologne, previous studies documented that the UHI effect seems to be the most important factor of risk to extreme heat when comparred to other cities in the region (Lissner et al., 2012).

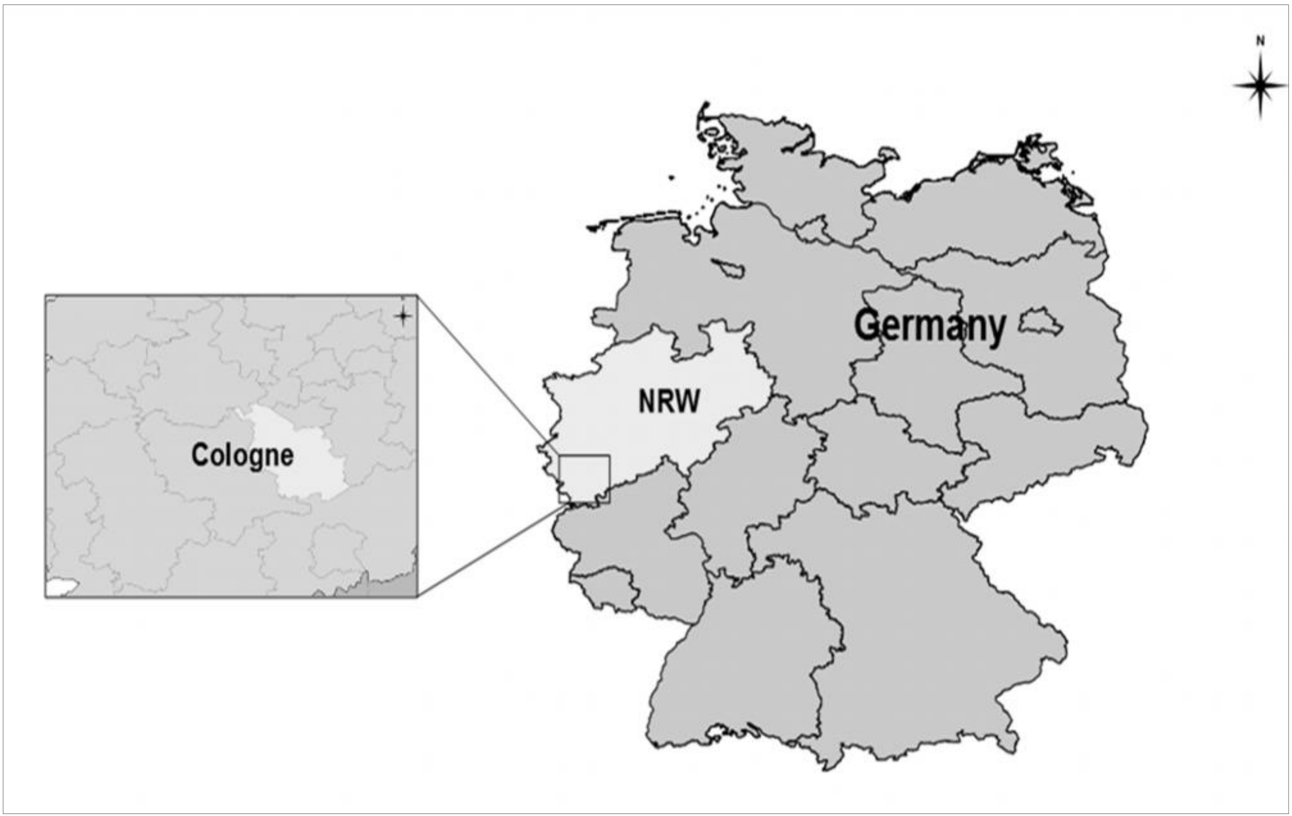

Figure 0.2. Geographical location of the Municipality of Cologne (Germany)
(Source: own map; data source: http://www.gadm.org/ , retrieved on 31st July 2014).

The second case study is the **Barcelona** municipality, in Spain (Figure 0.3) and the adjacent Collserola Natural Park (Figure 0.4). The municipality and the park are located in a dense metropolitan area which has increased its urban population since industrialization when high waves of immigration occurred in the area and culminated in the 1960s-1970s (Parés et al., 2013). It was in fact with the liberalization in 1959, after a long period of isolationism brought by Franco, that Spain

knew an unprecedented level of economic growth. At that time, the pattern of urban expansion in Barcelona led to the colonization of marginal areas situated in the periphery of the city and illegal construction, which often resulted in conflicts (Calavita and Ferrer, 2000). Barcelona has therefore a long tradition of urban social movements, the birth of which coincided with this period and quickly became an alternative forum for the discussion of urban affairs (Calavita and Ferrer, 2000).

Rapidly expanding, Barcelona has become the largest and most densely populated Metropolitan area in the Mediterranean region (Marulli and Mallarach, 2005). At present, it has a dense road traffic which is the major source of air pollution, high concentrations of industrial activities in its surroundings, with two gas power stations (Besòs and Sant Adrià) and two city waste incinerators (Sant Adrià and Montcada) (Querol, 2001). Furthermore, the Municipality of Barcelona, with a typical Mediterranean climate, has constantly been exposed to flash floods (Barrera et al., 2006), and intense and long periods of extreme heat which attained 43 consecutive days in 2003 (D'Ippoliti et al., 2010).

On the other hand, confining with the densely urbanized area of Barcelona extends a vast forest now protected as natural park. Since a century and a half ago, the Collserola forest was "produced" through both spontaneous as well as planned reforestation after the abandonment of agriculture. However, it became increasingly under the pressure of continuous urban expansion from the surrounding metropolitan area (Sotoca Garcia and Carracedo García-Villalba, 2011). The location of the park, the urban expansion which occurred in the sourrounding areas, the numerous planning steps towards its protection and the long tradition of urban social movements allow to analyse the dynamics and tension between the interests of an ever expanding urban area, on one side, and the socio-political forces for the preservation of a peri-urban park on the other. As detailed in Chapter 3, this process has led nonetheless to secure the supply of most needed ES, especially regulating and cultural, to the municipality of Barcelona.

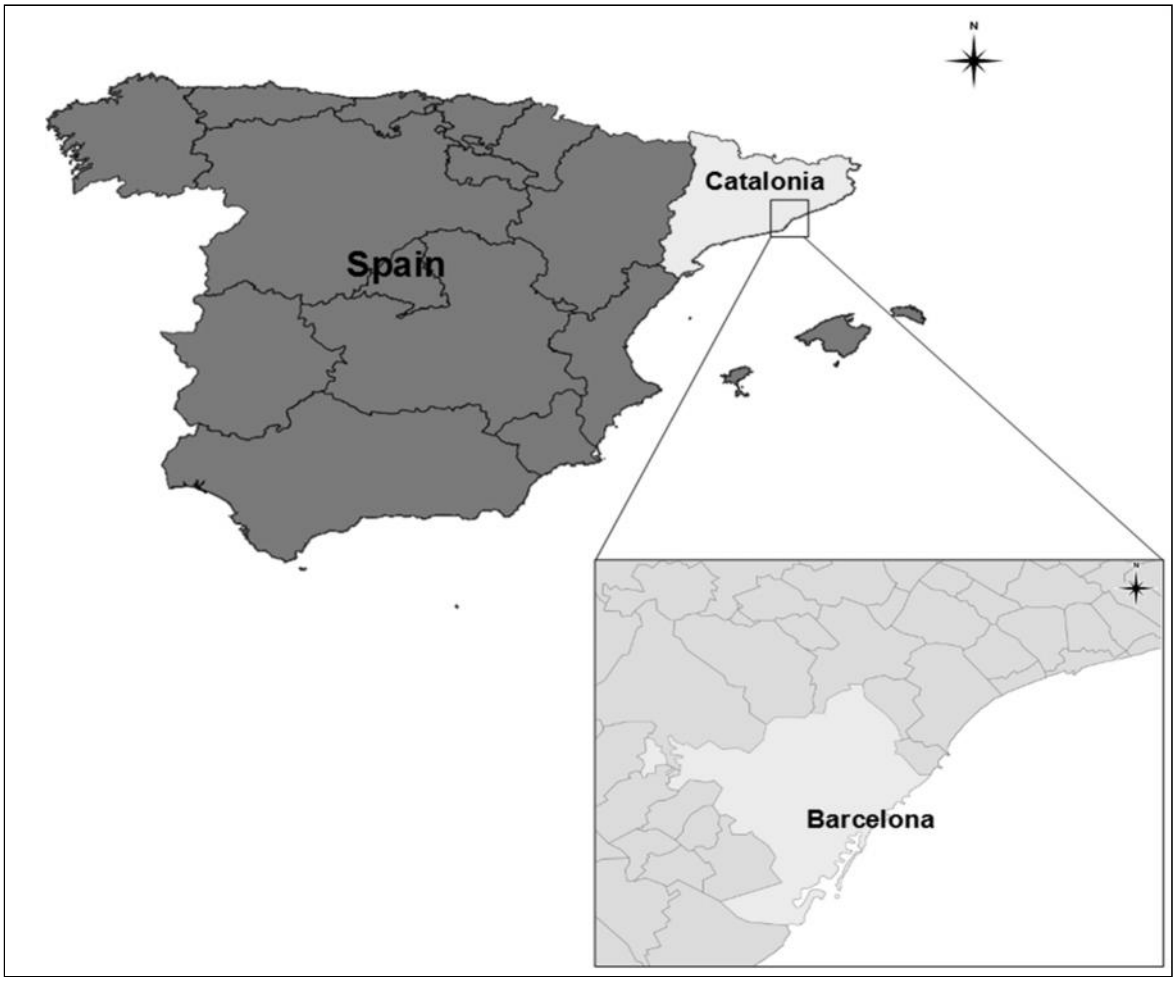

Figure 0.3. Geographical location of the Municipality of Barcelona (Spain)
(Source: own map; data source: http://www.gadm.org/ , retrieved on 31st July 2014)

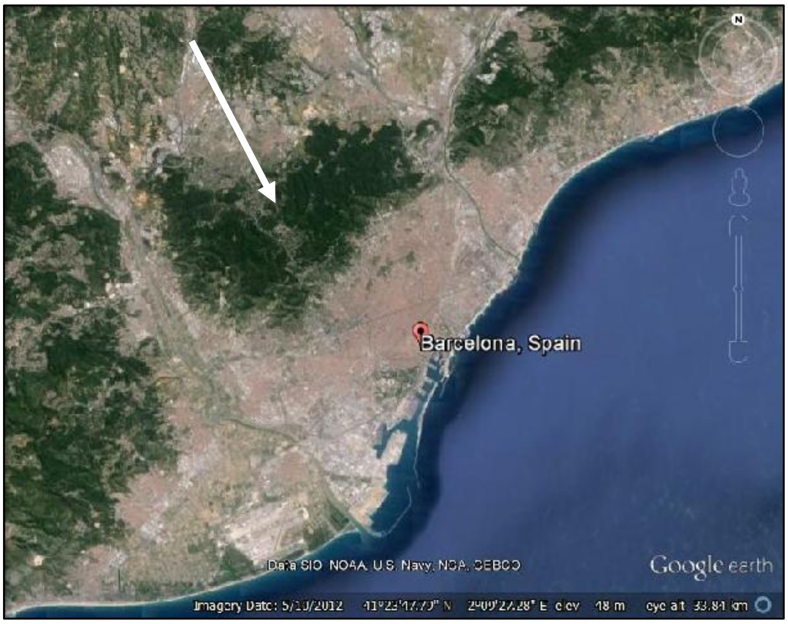

Figure 0.4. The Collserola Natural Park adjacent to the Barcelona Municipality
(Source: Google Earth).

Methodology

For a study to be qualified as integrated "it must reach beyond the bounds of a single discipline and consider more than one sector or one aspect of the problem under consideration" (Rothman and Robison, 1997). It must move the focus of the research from advancing knowledge for its intrinsic value to that of informing policy and decision making, integrating quantitative and qualitative approaches or pass from linear to more complex chains of analysis (Rothman and Robison, 1997). As mentioned, this Thesis brings examples of integrated studies. It does this in different ways by: targeting policy relevant conclusions; combining the social and ecological dimension of urban sustainability and analysing it at multiple scales; or adopting quantitative and qualitative integrated methods for the collection and analysis of data.

Regarding the gathering and analysis of data, quantitative assessment is generally widely used in the study of biophysical systems while qualitative approaches are more common in the humanities as these explore power relations, governance, social organization and institutions. Mixed quantitative and qualitative approaches are however increasingly applied especially in the social sciences (Bryman, 2006). These often provide complementary information (Amaratunga et al., 2002) and multiply the likelihood to obtain unanticipated outcomes (Bryman, 2006). Quantitative spatial, GIS based approaches with ArcGIS 9.3 and 10 (ESRI) have been applied for the assessment of ES, while qualitative approaches (e.g. through experts interviews and historical analysis) have been used to frame and complement this information in the case studies chapters. The qualitative information collected has been analysed through the Atlas.ti (GmbH) software which allows coding and synthesizing transcripts and texts transversally throughout the different sources. Overall, this mixed approach highly benefitted the comprehensiveness of the results obtained. Detailed methodological information is provided in the Chapters 2 and 3.

The following sections of the *introduction* present the state of the art of the literature on urban ecosystems and their services, as well as areview of concepts and methods on vulnerability assessment to hazards with a focus on the ecological dimension and introduce to the political component of the study of ES in urban areas.

Theoretical Background

Urban areas, environmental degradation and ecosystem services

Urban ecosystems are those socio-natural systems in which people live at high densities (Pickett et al., 2001) or those with extensive impervious surface areas (Wu, 2014). Brought about by their

features, ecosystem performance declines with increasing urban density (Sanford et al., 2009), but this decrease is variable from case to case and there generally is substantial scope for enhancing the ecological performance of urban areas (Tratalos et al., 2007).

Environmental degradation in and around cities has been constantly accompanying urban development and growth and can be traced back to the first settlements in Mesopotamia 4000 years ago when overexploitation of the surrounding land for the supply of agricultural goods to cities was widespread (Elmqvist et al., 2013; Grimm et al., 2000). Given the accrued number of people living in urban areas and the global impact of human activities, it is now more than ever before that urban environmental performance needs to be improved. Nowadays, cities highly contribute in modifying the environment as is the case for the impaired ecology of urban riparian zones (Groffman et al., 2003), the modification of the local and regional climate (Seto et al., 2013), losses of native biodiversity (Müller et al., 2013) and the increase in alien species (Handel et al., 2013), the abiotic stresses such as fragmentation and the suppression of natural disturbances which hampers the regeneration of the ecosystems through early succession stages (Handel et al., 2013), or increased surface run-off due to soil sealing (Scalenghe and Marsan, 2009).

As mentioned, physical urban expansion is also growing faster than urban population growth and is now one of the major urban trends of the urbanization process (Kronenberg et al., 2013; Seto et al., 2013). This is happening especially in medium-sized cities (with a population of 1 to 5 million) and directly threatens the surrounding ecosystems through processes of sub-urbanization accompanied by fragmentation and habitat isolation (Seto et al., 2013). An opposite phenomenon, which takes place especially in developed countries, is the shrinking of city cores which demands to reinterpret space and offer opportunities for reinserting urban nature (Haase, 2013). Overall, the quality of life in cities ultimately depends on the capacity to contain these threats. For instance, at the local and regional levels, preventing or restricting urban growth where this threatens ecosystems, such as watershed systems or ecologically fragile areas, need to be better taken into account in urban planning (MA, 2005).

However, historically, the urban and rural or surrounding areas have been seen as separate realities, a view concealing the links existing between urban activities and the environment. It is at least since the beginning of the 19th century, for instance with the influential work of the Chicago School, that urban theory excluded aspects of rural life from the study of cities (Elmqvist et al., 2013). This duality failed to appropriately accommodate the complexity of urban environments. As a consequence, the field of Human Ecology was started in the course of the 20th century to overcome this divide by looking at the ecological (or biophysical) aspects of the quality of human life, while including social

organization and spatial distribution of human groups and communities (Lawrence, 2003). A sub-discipline of it is the field of Urban Ecology which is an integrative science aiming at exploring not only the built and social components of cities but also their biological and physical features (Childers et al., 2014; Pickett et al., 2008). Urban Ecology sees cities as complex ecological entities in which humans are the dominant component and behave according to emergent properties (Alberti and Marzluff, 2004). According to Wu (2014), the field can itself be subdivided in: the ecology *in* cities (i.e. distribution and abundance of plants and animals in and around cities); the ecology *of* cities (i.e. cities as biophysical as well as socio-economic systems) (e.g. Pickett et al., 1997); and the urban sustainability approach (i.e. focusing on coupled social-ecological systems with an emphasis on ES and human well-being in urban areas).

This Thesis focuses on urban sustainability which stresses and explores the links between urban areas and local and more distant ecosystems through the concept of ES. ES as well as ecosystem disservices in urban areas have been explored and defined in various studies (Bolund and Hunhammar, 1999; Gómez-Baggethun et al., 2013; Gómez-Baggethun and Barton, 2013; Lyytimäki et al., 2008; Lyytimäki and Sipilä, 2009) and have been given policy relevance especially with the report on The Economics of Ecosystems and Biodiversity (TEEB) for Local and Regional Policy Makers (TEEB, 2010). Overall, the study of urban areas and ES is a growing field of research (Gómez-Baggethun et al., 2013).

Some characteristics distinguish urban ecosystems from other types of environments in terms of, for instance, the range of specific services. According to various authors, properly urban are services such as microclimate regulation, air pollution removal, water supply and regulation, noise reduction and recreation (Bolund and Hunhammar, 1999; Gómez-Baggethun et al., 2013; Gómez-Baggethun and Barton, 2013). However, there is still little empirical evidence that supports the benefits provided by urban green areas (Gómez-Baggethun et al., 2013), especially for regulating services such as air purification (Pataki et al., 2011) or those linked to risk in urban areas (Guadagno et al., 2013).

The role of cities in the restoration and preservation of healthy ecosystems becomes even more important when considering that urban areas are drivers of change and sites of intellectual ferment. The ES concept offers a tool to highlight issues and set policies. However, most of the studies on ES have focused on their classification and valuation in economic terms (Boyd and Banzhaf, 2007; Costanza et al., 2006, 1997; de Groot et al., 2002; Farber et al., 2002, 2002; Fisher et al., 2009) and less on practical, planning and policy implications. Therefore, especially in Chapter 1 and 2, I look at ES for hazard regulation and policy implications.

Notes on urban ES assessment and valuation

ES can be assessed in biophysical terms or through preference base methods, both extensively described in the TEEB (2012). A comprehensive list of biophysical proxies as well as of economic valuation methods for urban areas can be found in Gómez-Baggethun and Barton (2013) and in Gómez-Baggethun et al. (2013). In this Thesis, ES have been assessed in biophysical terms and specifically through proxies of landscape functions, "defined as the capacity or potential of a landscapes to provide services" (Bolliger and Kienast, 2010). These methods would, amongst other, largely suffice to respond to the research objectives of this Thesis. The results obtained in this way provide in fact information both for policy oriented assessments (Willemen et al., 2008) or as a background information for theoretical assumptions. Some proxies for the calculation of landscape functions relevant for urban areas and related to services analysed in this Thesis are presented in Table 0.1.

Table 0.1. List of proxies of landscape functions for some ES relevant at the urban scale and assessed in this Thesis.

Ecosystem service	Proxy	Description	Reference
Air purification	Leaf area index	Total amount of pollutants removed via dry deposition on leaves (ton ha^{-1} $year^{-1}$) for land cover classes	(Knote et al., 2009)
	Dry deposition velocity per pollutant concentration	Deposition velocity is the inverse sum of three resistances. The main ecosystem based parameters affecting deposition velocity are the height of the vegetation (related to the roughness length of the land) and the leaf area index	(Maes et al., 2011) http://www.ceip.at/
Urban cooling	Land surface thermal emissions or surface emissivity	Total amount of energy emitted by a surface (Landsat 7 ETM+ thermal band 6.1)	(Haase et al., 2012; Schwarz et al., 2011)
	Surface air temperature	Derived by a thermal scan of land surface temperatures	(Haase et al., 2012)
	Evapotranspiration	f-value for evapotranspiration potential of a land use class	(Larondelle and Haase, 2013; Schwarz et al., 2011)
Flood regulation	Multiple proxies for water infiltration capacity of soils	Includes percent vegetation cover, percent agricultural cover, flow distance from 100-y floodplain, percent vegetation cover within riparian zone (whose width depends on stream order) [unit-less score]	(Chan et al., 2006)

Ecosystem service	Proxy	Description	Reference
	Riparian areas	Regional ecosystems vegetation and land use datasets for riparian areas: "green zones" which lie between stream channels and uplands	(Pert et al., 2010)
	Ground water recharge	Percentage (%) contribution of groundwater to base-flows	(Egoh et al., 2008)
	Ground water recharge	Millions of cubic meters of groundwater recharge per 1-km^2 grid cell	(Reyers et al., 2009)
	% sealed soil	Soil infiltration capacity	(Haase and Nuissl, 2007)
Recreation	% green cover	Available green areas	(Jim and Chen, 2006)
	Green space per capita [m^2 / person]	Total available green space per raster cell divided by the number of people living in the same area	(Haase et al., 2012)
	Provision of attractive landscapes for tourism	Occurrence of rural tourist accommodations or the amount of tourists counted in a certain region + land cover (pattern), accessibility and landscape characteristics	(Verburg et al., 2009)
	Visual amenity	The spatial distribution of the viewing population (using travel data to present both local inhabitants and visitors), and actual and potential amount of viewed woodland or landscape	(Gimona and Horst, 2007)
	On-site recreation	Expected number of visitors at any grid cell	(Gimona and Horst, 2007)
	Outdoor recreation	Proximity to major roads, weighted per population density, level of public access, amount of land in natural or agricultural cover	(Chan et al., 2006)
	Tourism	Areas that tourists can see from the major tourist driving routes	(Reyers et al., 2009)
	Leisure cycling function	Potential leisure cycling population	(Willemen et al., 2008)

(Table 0.1 continued)

Some remarks refer however to monetary ecosystem valuation, an issue dealt with in more detail in Chapter 4 by presenting some examples at the watershed scale. The ES concept emerged in fact in the 70s to increase the interest in ecosystem conservation, propagated by the field of ecological economics to counter the neoclassical economic theory belief that technical advancement decouples economic growth from natural resources (Dempsey and Robertson, 2012). However, it soon served as a basis for ecosystem valuation especially with monetary methods (e.g. Costanza et al., 2006, 1997) (see also Gómez-Baggethun et al., 2010 for an historical analysis of the concept).

Economic, especially monetary, valuation is affected by numerous drawbacks and all the methods so far developed have important limitations (see Chapter 4). These badly depict or do not take into

account the complexity and non-linearity of ecosystem functions for the supply of services (Chee, 2004; Farber et al., 2002). Furthermore, while some ES have markets (such as food supply) most of them cannot be traded because they are not “private” in nature (Farber et al., 2002). This leads economic valuation to be based on preferences which however change spatially and over time (Farber et al., 2002) and often do not contain appropriate ecological information. In ecological economics it is in fact denied that there exists a set of “ecologically correct prices” (Martinez-Alier, 2003). Additionally, the ecological understanding behind the supply of ES (e.g. the loss of which species leads to community changes and to a rapid damage of functions) is still poor (Kremen, 2005; Kremen and Ostfeld, 2005; Norgaard, 2010). Valuation targeting sustainability could therefore be structured within conceptual frameworks, such as the one proposed by Kallis et al. (2013), which acknowledges that the ecological goals of the project, along with the social and economic ones, are also attained.

To note is that, valuing ES in urban areas, what appears striking is the higher value attributed to the services, due to the concentration of population in addition to the little share of green areas, as well as the high spatial heterogeneity of values, due to the high variation of both social and environmental factors (Gómez-Baggethun and Barton, 2013). Additionally, to highlight is that - for hazard mitigation services - what it often accounted for is their insurance value in relation to avoided damages (Farber et al., 2002; Gómez-Baggethun and Barton, 2013).

Urban form, scales and ecological performance

Regarding urban structure, a study by Alberti and Marzluff (2004) indicates that landscapes characterized by a mixture of sealed and forested land may be more resilient than extensive, well connected natural areas or areas of widespread urban sprawl, as these cannot sustain simultaneously human and ecological systems. Colding (2007) also suggests to design urban areas (especially suburbs) by clustering together different types of urban green zones “to increase available habitats for species, to promote landscape complementation/supplementation functions, and to nurture key ecosystem processes essential for the support of biodiversity”.

While acknowledging that the urban ecology is interdisciplinary and multi-scale (Gómez-Baggethun et al., 2013), most of the studies, such as those cited in the previous section, focus on the local, city scale. However, the impacts of cities and their demand in services go well beyond the municipal boundaries and may affect entire regions. A study by Folke et al. (1997) showed that the ecosystem-appropriation by cities in many cases reaches the global scale. As the amount of services produced at the local administrative scale is often small, an increasing number of studies focus on the city core as

well as on the surrounding built up areas (e.g. Haase et al., 2012). Different types of ES might in fact be produced at different scales for a same urban area. For instance, at the peri-urban scale, most of the studies look at recreational, intangible services (Vejre et al., 2010) or peri-urban agriculture (Zasada, 2011), while carbon sequestration and storage is a service more relevant at the global scale. While decoupling between city dynamics and ecosystem can occur for certain ES at the local scale (Gómez-Baggethun et al., 2013), urban areas ultimately depend on well-functioning local as well as on regional ecosystems to guarantee a good quality of life, security and the well-being of its inhabitants. The water components of local and regional ecosystems, for instance, contribute to the supply of ES in all of the MA categories, strengthening the view that urban water bodies and watersheds should be seen as multifunctional components of the urban space (Lundy and Wade, 2011; Pickett et al., 1997). Specific policies for these broader, urban units, different from those targeting the rural or the urban context, are however needed (Allen, 2003).

While little work has been done to link the concept of ES to urban planning policy making at multiple scales, ES assessment might for instance be a promising tool to settle the century long argument between the visions of the pioneers of the Planning Movement, such as Howard's Garden City or Geddes and Mumford's Regional Planning, as opposed to the city with high density centre with tall buildings and increased road traffic surrounded by a residential area of Le Corbusier (Hall, 1988), as I explore in more detail in Chapter 3.

Urban systems and disaster risk

While natural hazards are generally part of the functioning of biophysical systems, disasters[2] can ultimately be seen as a social construction as they are the result of the interactions between human and ecological systems (e.g. Pelling, 1999). Oliver-Smith (2004) considers that "disasters come into existence in both the material and social world and, perhaps, in some space between them". It is in fact increasingly acknowledged that the vulnerability to natural hazards of populations is the result of the socio-economic processes that characterise a population and is thus socially constructed (Oliver-Smith, 1999). This is even more so in urban areas where the environment is highly modified by physical infrastructures and socio-economic activities.

Cities are centres of interchange of knowledge, cultures, innovations and goods. To facilitate exchanges, these are often located in the proximity of rivers and seas making them exposed to a

[2] i.e. the potential disaster losses, in lives, health status, livelihoods, assets and services, which could occur to a particular community or a society over some specified future time period

number of hazards such as storms, flooding, cyclones, coastal erosion and sea level rise (Sherbinin et al., 2007). The interactions between the hazards and the system, due to the elevated concentration of infrastructures, government institutions, population and economic activities, often lead to potentially high impacts. In urban areas, the frequency and intensity of hazards are in fact often socially produced as combined with “poor urban management, inadequate planning, high population density, inappropriate construction, ecological imbalances and infrastructure dependency” (Jacobs, 2005).

Romero-Lankao and Qin (2011) review and cluster urban vulnerability to hazards under different research themes: from a focus on the features of the hazard which, interacting with the urban environments, increase its impacts; to the characteristics of the urban population which make it more vulnerable (e.g. Ishigami et al., 2008 who looked at social factors, such as age and sex, behind heat related mortality in three European cities); to the political economy (or ecology) theme which looks at how different populations or different zones of the city are more or less vulnerable and why (e.g. O’Neill, 2005 on the links between race, heat mortality and the presence of air conditioning); and to the urban resilience related to the notions of tipping points and path dependency, focusing more on response-capacity building (see also Leichenko, 2011). In Chapter 2 I analyse mainly the first and second aspects of the interactions between the social and ecological components of Cologne’s vulnerability to heat waves.

Research on urban vulnerability has concentrated mainly on heat waves (Harlan et al., 2006; Ishigami et al., 2008; Jenerette et al., 2011; O’Neill, 2005; Romero-Lankao et al., 2012; Semenza et al., 1996; Uejio et al., 2011; Vaneckova et al., 2008; Wilhelmi and Hayden, 2010) as this is a typical urban hazard, but also on earthquakes as damages in cities are also a direct function of human behaviour (Barbat et al., 2010; Rashed and Weeks, 2003), or to coastal hazards in the Low Elevation Coastal Zone (LECZ)[3] due to the high concentration of population in these areas and the high frequency of hazards (Comfort, 2006; McGranahan et al., 2007; Sterr, 2008). However, overall, the vulnerability to climate change and natural hazards in urban areas is understudied (Kallis, 2008; Romero-Lankao and Qin, 2011). Furthermore the role of ecosystem-based DRR measures in cities has so far been given little attention (Guadagno et al., 2013).

The next sections of the introduction focus mainly on reviewing and framing the little explored environmental dimension of vulnerability to hazards to set the path towards its integration into social-ecological studies.

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[3] coast that is less than 10 metres above sea level

The ecological dimension of vulnerability

Environmental vulnerability to anthropogenic degradation

The term “ecological” or “ecosystem” vulnerability is relatively recent. The link between ecosystem and risk needs to be traced back in the field of eco-toxicology where the effects of toxic compounds on living organisms are traditionally assessed (De Lange et al., 2010; Ippolito et al., 2010). This field of studies has then been extended to the level of ecological species and at the level of populations, communities, ecosystems and landscapes, which progressively demanded to carry out research outside the laboratory and into the field. Here vulnerability is defined according to its components such as exposure, sensitivity of the organisms/species and the recovery potential (at the population or community level) (De Lange et al., 2010).

However, numerous studies refer to (eco-) environmental vulnerability as the level of degradation of ecosystems themselves, both terrestrial and marine (Furlan et al., 2011; Halpern et al., 2007; Li et al., 2006; Teck et al., 2010; Tran et al., 2010; S.-Y. Wang et al., 2008; X. D. Wang et al., 2008; Wilson et al., 2005). This research area generally provides background information for conservation purposes (see Wilson et al., 2005 for a review). Nonetheless, some integration between social and ecologic variables takes also place here. Tran et al. (2010), for instance, developed a set of environmental variables classified as environmental and social stressors (e.g. count of exotic aquatic – fish and mussels – species; percent of impervious surface by land-cover) and resources (e.g. count of native aquatic – fish and mussels – species; percent of area classified as wetlands) and applied it to the system of watersheds located in the U.S. Mid-Atlantic region. Similarly, Wang et al. (2008) carried out an environmental vulnerability assessment of the Tibetan Plateau but with a reduced number of variables (e.g. topography, soil erosion, population density). Other, more integrated studies look at the links between environmental vulnerability (in terms of degradation) and poverty (Scott, 2006).

Within this domain, environmental degradation is secondarily approximated to the measure of the incapacity of the system to sustain an external stress. It may relate to the domain of risk through the concept of “environmental criticality”. This refers to an environment where “the quantity and quality of those uses and/or the well-being of the population cannot be sustained, given feasible socio-economic and/or technological responses”; this also applies to “a decrease in the capacity of the environment as managed to meet its user demands” (Brookfield et al., 1995). But it is still considered as distinct from the definition of vulnerability (Adger, 2000).

Environmental vulnerability to natural or man-made hazards

Environmental vulnerability in relation to natural or anthropogenic hazards is defined as an estimate of the inability of an ecosystem (in all its levels of organization) to tolerate stressors over time and space (Villa and McLeod, 2002). De Lange et al. (2010) reviewed ecosystem vulnerability to hazards in general (but mainly to chemical compounds), looking at the ecological traits of the species, like genetic diversity, physiology, reproductive output of a species or heritable variation, and describe ecosystem vulnerability according to the three components mentioned above. Some other authors specifically look at the vulnerability of ecosystems to climate change (Gritti et al., 2006; Heide-Joergensen and Johnsen, 1998; Hughes et al., 2003; Lassiter et al., 2000; Scholze et al., 2006; Schröter et al., 2005; Williams et al., 2008), to land use change (Metzger et al., 2006), or carry out more broad scenario analyses for changes in ecosystem processes (e.g. wildfire frequency, photosynthesis, plant respiration, or runoff) (e.g. Hurd et al., 1999; Scholze et al., 2006). It should be noted that, at least in Europe, the larger impacts of ecosystems are caused by technological hazards (i.e. oil and toxic spills and industrial accidents) (EEA, 2010b).

The approach developed by the United Nations Economic Commission for Latin America and the Caribbean (ECLAC) classifies the effects of large scale natural phenomena on physical (e.g. erosion, soil destabilization, landslides), biotic (e.g. effects on fauna and flora) and perceptual environment (i.e. to the landscape, or to scientific and cultural resources) (ECLAC, 2003). ECLAC includes in the methodology environmental damage assessment and defines direct damages (i.e. environmental changes caused by disaster impacts on resources or on structures built to use them) and indirect losses (i.e. when environmental services are reduced, diminished in quality or made more expensive) (ECLAC, 2003). The impact scale is based on six classes going from “Zero Impact” (i.e. when environmental recovery can occur and to a low cost) to “Total Impact” (i.e. when recovery is not possible and the options to use resources in the future is lost). In this case, the vulnerable object is the ecosystem itself, whereby its links with the socio-economic system are often not straightforward.

Another example is the Environmental Vulnerability Index (EVI), a biologically oriented approach developed by the South Pacific Applied Geoscience Commission (SOPAC) and the United Nations Environment Programme (UNEP) (UNEP and SOPAC, 2005). It addresses through 50 indicators the state of the biophysical systems, which include ecosystems, habitats, populations and communities as well as physical and biological processes, productivity and energy flows, diversity, and ecological resilience (UNEP and SOPAC, 2005; Villa and McLeod, 2002). These entities and processes can be affected by human and natural hazards which can lead to losses of diversity, extent, quality and function of ecosystems (Kaly et al., 2004). The indicators are combined by simple averaging, and are

designed to reflect the extent to which the natural environment of a country is prone to damage and degradation. The scale of application is national planning, but it can also be used for regions and provinces. According to Kaly et al. (2004), the social relevance of this method lies in the fact that the more degraded the ecosystem of a country is, the more vulnerable the social-ecological system of that country. The method was also applied in Greece at the wide national scale (Skondras et al., 2011), however, Barnett et al. (2008) contest the applicability of EVI at this broad scale, due to the very local relevance of the indicators used. The researchers also have criticized EVI for adopting a reductionist approach which confines complex issues to a series of indicators defined by common numerical scales as well as due its too indirect social relevance.

Other authors make use of the term environmental vulnerability but their focus is rather on the vulnerability of people towards various environmental threats (e.g. air and water quality, noise level, smell) (Drori and Yuchtman-Yaar, 2002) or hazards (Boruff and Cutter, 2010) not acknowledging the role of, and the impacts on, the ecosystem.

Finally, Kelman (2007), describes Disaster or Risk Ecology, terms first used by Lewis (1980), as a discipline derived from Human Ecology looking at the relationship and interactions between organisms (in this case humans), or the society, and disasters. It specifically looks at what are the positive and negative social changes brought by environmental disturbances. Looking at the ecology of vulnerability, this branch aims at stressing that hazards can also bring benefits or positive changes other than bringing destruction and this perspective applies to all the measures taken by a society to resist change driven by hazards (Kelman and Lewis, 2005).

In Chapters 1 and 2 I explore the vulnerability of urban populations with respect to the state and characteristics of the ecosystem, aiming at bridging, through the concept of ES, the social component of vulnerability and the ecological one. The next sub-sections of the introduction give background information at this scope.

Social-ecological vulnerability to natural and man-made hazards

The concept of mutuality

The traditional dichotomy between “natural” and “human” often leads to see social and natural systems as completely distinct or independent, obfuscating, in this way, the multiple interactions, dependences and feedbacks that take place between the two systems (Demeritt, 2002; Oliver-Smith, 2004). As mentioned, according to Oliver-Smith (2004), the very concepts of risk and vulnerability

refer to a multidimensional problem as these are conceptually located at the intersection between nature and culture, expressed by the concept of “mutuality” (i.e. multiple interactions) between the two systems. Humans shape the environment as much as they are shaped by it (Zimmer, 2010) and it is in periods of stress that this interplay between social and ecological systems is the most evident.

Social and ecological processes mutually affect each other but these interactions cannot be easily assessed by a common metric system. In sustainability science, in fact, the properties of socio-economic or of ecological systems are weakly comparable, that is to say that they are comparable but without recurring to a single type of value due to the multidimensional nature of the criteria (Martinez-Alier et al., 1998). The goal of integrated approaches is nonetheless to resume and make explicit the links and the multiple processes and feedbacks through which social and ecological systems interact and change in the face of a pressure. Social knowledge, values, economics and institutions, have in fact an impact on natural systems: they can enhance the diversity and productivity and improve the quality of the services provided by the environment as well as cause environmental degradation or increase vulnerability to hazards. Social processes and systems, vice versa, are also bounded by and depend on the functioning of ecosystems at large. On the side of the social system, human interactions with biophysical systems and their effects have been often known through processes of trial and error, as explained in co-evolutionary theory (Kallis, 2007; Kallis and Norgaard, 2010), or by learning processes in adaptive co-management of social-ecological systems, in resilience theory (Olsson et al., 2004).

Overall, due to its complexity and to the properties of social-ecological systems (e.g. emergence), their functioning is not fully understood so far (Manson, 2001). However, their study, by turning to complexity theory, should not conceal the diversity as well as the mutuality existing and taking place between social and ecological systems. The ES concept remarkably helps in framing the interactions between ecological and social systems. It is in fact through the concept of ES that most of the interactions and mutualities within ecosystems can be expressed and elicited (Adger, 2000). This is true also for the assessment of disaster risk (see for example Renaud, 2006). The vulnerability of coupled systems could in fact be expressed through the nature and the quality of the dependence of communities and their economic activities on ecosystems (Adger, 2000; Renaud et al., 2010).

Ecosystem services and DRR

The coupled social-ecological approach is gaining in influence in the natural hazards field. An expanding field of research, working especially at the international level (e.g. the work of the

Partnership for Environment and Disaster Risk Reduction – PEDRR – http://pedrr.org/), aims, for instance, at studying and demonstrating how the implementation of ecosystem based approaches[4], green infrastructures[5] or mixed solutions can provide viable alternatives to hard infrastructures for DRR. Initial works have shown that well managed ecosystems and their regulating services can contribute, to some extent, to the reduction of risk and are very often cost-effective, multifunctional and win-win solutions, especially in the long run (EEA, 2014b; Renaud et al., 2013; Sudmeier-Rieux, 2013), while securing an important source of livelihood for local communities (Sudmeier-Rieux et al., 2006).

An example about the benefits of the ecosystem based approach to DRR is the case study of flood regulation policies in The Netherlands. Investments in alternative flood control policies, such as land use changes and floodplain restoration, resulted to be justified when including the additional ecological and socio-economic benefits in the long term perspective (Brouwer and van Ek, 2004). Hoang Tri et al. (1998) also found that both direct and indirect benefits to local communities make mangrove rehabilitation a desirable option from an economic perspective to reduce vulnerability to tropical storms of three costal districts of northern Vietnam.

For what concerns urban areas, Guadagno et al. (2013) gathered and reviewed a wide range of case studies demonstrating the effectiveness of promoting ecosystem management for DRR in urban areas, not least for its reduced economic costs. Other available examples are the positive role played by ecosystems with respect to heat waves risk in Phoenix, US (Jenerette et al., 2011) or the flood mitigation potential of an urban ecosystem provided by the Sanjay Gandhi National Park Mumbai, India which protected the city from floods. In July 2005, in fact, Mumbai experienced a rainstorm of unprecedented proportions (994 mm of rain fell in the first 24 hours alone) that resulted in extensive flooding (Trzyna, 2014). The park, the largest permeable surface remaining in the metropolitan area, helped to prevent an even worse situation (Trzyna, 2014). Gill et al. (2007) explored the potential of green infrastructures in adapting cities to climate change using as a case study the conurbation of Greater Manchester. The authors found that maintaining and enhancing urban green areas would be one possible adaptation strategy to the increasing temperatures, while it might be a less effective one to reduce surface runoff due to increase in precipitations. Similar results have been documented for other urban areas in Europe, such as in Slovakia (Hudeková, 2011). Overall, although natural capital

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[4] "A strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way"(http://www.cbd.int/ecosystem/ , retrieved on 1st October 2014)

[5] "Green Infrastructure is the interconnected network of natural areas including some agricultural land, such as greenways, wetlands, parks, forest preserves and native plant communities, and marine areas that naturally regulate storm flows, temperatures, flooding risk, and water, air and ecosystem quality" (EC, 2009)

might not be the only resource necessary to cope with natural hazards, it can be a relevant one amongst others such as social, built or economic capital (Pérez-Maqueo et al., 2007).

On the other hand, anthropogenic environmental change, by affecting the functioning of ES through land use and climate changes, is one of the main drivers of the increasing impacts of a number of natural hazards (Kaly et al., 2004) such as floods (Lahmer et al., 2000), coastal hazards (Adger, 2005) or climate change (Metzger et al., 2008; Metzger and Schröter, 2006).

From a policy level, the ecosystem based strategy to achieve DRR is acknowledged as one option to adapt to climate change in the White Paper of the European Commission (2009). But, despite the recognition of soft options as “low regret” measures for DRR and climate change adaptation, also at the international level (IPCC, 2012; UNISDR, 2005), this perhaps remains the most disregarded component of DRR (Renaud et al., 2013).

The role of ecosystems in mitigating hazards has led so far to contradictory results or has been overemphasized in some cases (Renaud et al., 2013). The available evidence is still scarce and in some cases contentious (Balmford et al., 2008) which, together with the lack of concrete studies on the role of ecosystems for human well-being, might be one of the main obstacles explaining the little implementation of ecosystem based solutions for hazard mitigation in general but also in Europe (Sudmeier-Rieux, 2013) and in cities worldwide (Guadagno et al., 2013).

Resilience thinking and DRR

There is a large field of research which goes under the term “resilience thinking” and which was started by Holling (1973) in the ecological field. This concerns the behaviour of ecological systems and refers to notions of panarchy and the adaptive cycle framework. The term has also been adopted to describe social-ecological systems and focuses on the capacity of the system to learn and renew, re-organize and develop, in addition to the capacity of the system to absorb disturbance while retaining structure (Folke, 2006; Walker et al., 2004).

In the DRR community the concept of resilience has been reviewed by Alexander (2013). According to the author, despite widespread use of the term across disciplines, it presents some limitations. The focus of the term on keeping structure of the system rather than on transformation and change, which is an important process in DRR, might be a drawback rather than facilitating it. Resilience, in fact, might be better described as the capacity of the system to keep adapting rather than on its propensity to return to a given point of equilibrium (Alexander, 2013). This perspective is partially raised by

integrated studies of the vulnerability of social-ecological systems for which the resilience concept aims at being a shift from the control of change in systems, to managing the capacity of social-ecological systems to cope with, learn, adapt to, and shape change (Folke, 2006). We will see in the MOVE framework for hazard vulnerability assessment (Figure 0.5), how the term resilience is meant to account for the capacity of the system to anticipate, cope and recover from a hazard in the short term, rather than on its resistance or its capacity to retain structure.

Furthermore, social-ecological systems have been characterized by complexity and non-linearity, but, so far, the multiplication of research aiming at the identification of thresholds and tipping points has acknowledged the difficulty or impossibility in predicting such ecosystem changes (Renaud et al., 2010), and even less how these can be monetarily valued (Chee, 2004). Ecosystem management might then benefit from the improvement of knowledge about the mutuality between social and ecological components on the top of the, nonetheless important, study of complex behaviours of the system.

Perhaps what should be underlined here is that most of the study of ecosystems functioning under external stresses has gone under the category of the study of resilience, often erroneously considered as the flip side of vulnerability in hazard risk. Based on this literature, measures of ecosystem health, such as the dimension and connectivity of habitats (EEA, 2014b) or system biodiversity (Folke et al., 2004), could be integrated to measure the vulnerability to hazards of the degraded ecosystem. However, data to assess these indicators are difficultly available at the urban scale and ES supply failure has been explored in Chapter 2 through expert judgment by identifying which ecosystems, with their services, would to be potentially affected by heat waves. A further example of application of this approach, but to riverine floods, is presented in Welle et al. (2014).

<u>Towards integrated, social-ecological frameworks for the assessment of vulnerability to natural hazards</u>

The study of the impacts of natural hazards has moved from the focus on the geophysical, climatological or hydro-meteorological phenomena by considering first physical vulnerability (i.e. exposure and fragility of the exposed elements) and then the socio-economic, institutional and cultural factors determining risk (Cardona, 2004). The coupling or ecological component of risk was integrated only later on. According to Damm (2010), integrated studies considering both the social and ecological dimension have moved from place-based vulnerability studies, to multi-dimensional vulnerability, and finally to coupled social-ecological vulnerability, levelling in this way the simplistic and dualistic representation of the human-environment systems. With it, also frameworks

for the analysis of risk and vulnerability have evolved. Last in this series is perhaps the MOVE Generic Framework (Figure 0.5), described in Birkmann et al. (2013). This builds on previous integrated vulnerability frameworks such as the Holistic framework to disaster risk assessment (Cardona and Barbat, 2000), the Turner framework on social-ecological coupling (Turner et al., 2003a) and the BBC framework based on sustainable development principles (Birkmann, 2006). The MOVE framework acknowledges both the multidimensional nature of various components of vulnerability and stresses the presence of processes of coupling between the hazards and the social-ecological systems. The cultural and ecological dimensions of vulnerability are here taken into account while they had received little attention in past approaches.

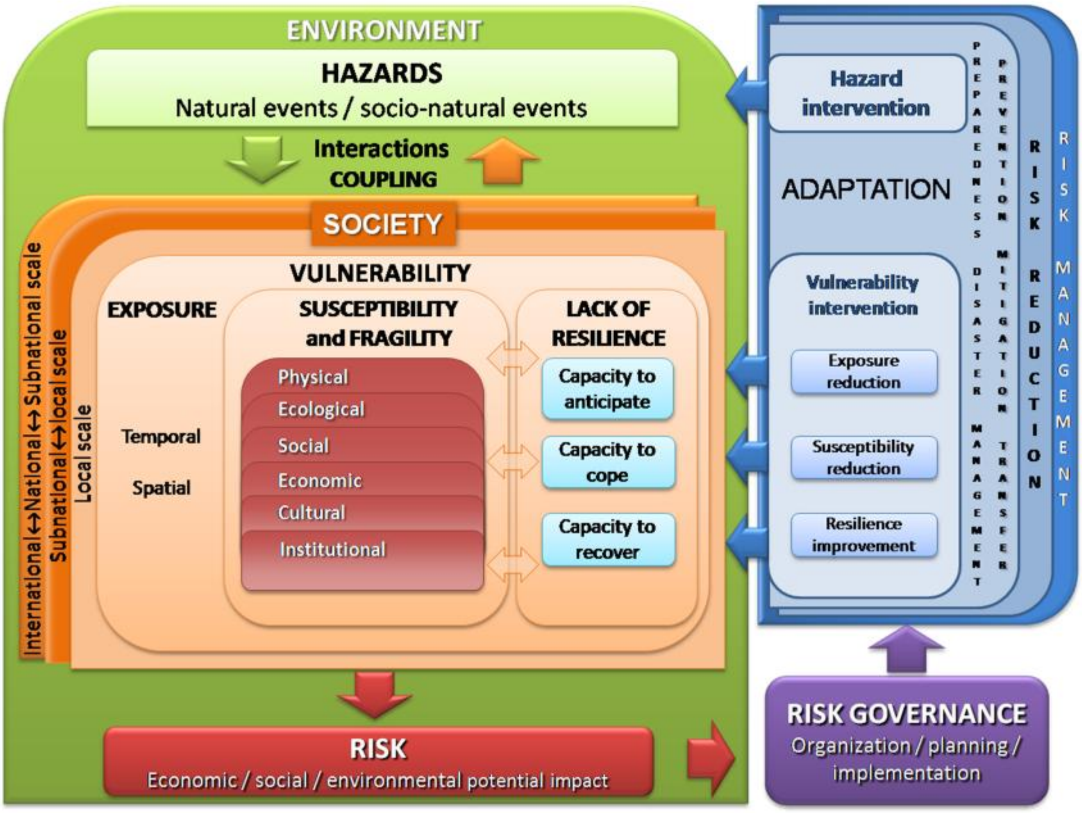

Figure 0.5. The MOVE generic framework
(Source: Birkmann et al., 2013).

An example of the application of coupled social-ecological frameworks to vulnerability assessments is that by Turner et al. (2003b), who analyse processes of coupling by discursively eliciting how socio-economic processes, policies and strategies engender changes in the environment further increasing the vulnerability of communities living next to and depending on ecosystems. Also Damm (2010) modified Turner’s framework and applied it quantitatively through a mixed set of indicators (both socio-economic and environmental) to the case of flood vulnerability in Germany. Another example of multidisciplinary vulnerability assessment is provided by Adger et al. (2005), who

qualitatively explore the resilience of social-ecological systems to coastal hazards by stressing ecosystem degradation as a further source of risk.

Overall, little work has been done on framing the use of the concept of ES in this field. There is therefore the need to better incorporate the ecological component of vulnerability in integrated assessments. The lack of common understanding of the different links between healthy ecosystems, hazards and human vulnerability might have in fact additionally hampered its recognition so far.

Political ecology in urban areas

Proving and acknowledging the role of ecosystems for human well-being and DRR would probably not suffice if the historical dimension and political nature of ES are also not recognized. Political ecology offers the conceptual framework in which to look at how differently nature and societal structure determine each other and at how constructed concepts of society and nature shape the availability and access to resources through power relations.

Urban studies in political ecology are a growing domain of research but which have emerged rather recently with respect to the traditional focus on rural areas (Zimmer, 2010). A characteristic of nature in urban areas is that its effects are highly contested and often unevenly distributed (Swyngedouw and Heynen, 2003). For instance, Collins (2010) shows how in “Paso del Norte” (a city between two countries: El Paso County, USA and Ciudad Juárez, Mexico) the impacts of floods, which occurred between July and September 2006, were overall of an order of greater magnitude in the Mexican part of the city due to unequal power relations expressed through the economic system.

Furthermore, most of the political ecology work focusing on cities concerns urban water cycles (Heynen, 2014). It is in fact through processes of metabolism discursively and politically translated that different power relations in cities can originate (Swyngedouw and Heynen, 2003). Such is the case of Barcelona and its conflicts and struggles over water resources (Masjuan et al., 2008), or the case of Guayaquil, Ecuador in which water distribution leads to high levels of marginalization (Swyngedouw, 1997), that of droughts and constructed water scarcity in Athens (Kaika, 2003), or that of water scarcity and a Catalan elite suburb (Otero et al., 2011).

About cities and green areas, some studies investigate the production of urban natures. Domene and Saurí (2007) look for instance at urban vegetable gardens as “socio-natures” in the Metropolitan areas of Barcelona, created to counteract an ever expanding urbanization which sees them as non-urban. Heynen (2006) looks at changes in households income and changes in household canopy cover to highlight their interdependence and the production of inequalities in the case of the Indianapolis

inner-city urban forest. This literature recognizes that, since urban natures are socially produced, they often lead to an uneven distribution of the green and of its benefits.

Additionally, while there is a wide literature on the political ecology of protected areas and the effects of conservation especially on local populations in developing countries (such as Adams and Hutton, 2007; Brown, 1998; Neumann, 1995, 1992; Peluso, 1993), few looked at parks in urban areas. Amongst these, most are focused on the interrelations between gender, race and politics, stressing the human health and ecological consequences of the uneven spatial distribution of green space within cities (Brownlow, 2006; Byrne et al., 2009). To note is that overall ecological dynamics in the production of urban environments have been neglected in political ecology because of the focus on the social component (Heynen, 2003).

Regarding the political ecology of ES, most studies look at the effects of neoliberal policies and the commodification of the services. Kosoy and Corbera (2010) show, amongst other, how processes involved in schemes of payments for ES are characterized by power asymmetries which increase rather than diminish a fair distribution of resources, while Robertson (2004) looks at the particular case of the commercialization of a wetland and the colonization of ecological knowledge by the economic system. Corbera et al. (2007) show how markets for ES in the forests of Chiapas (Mexico) actually reinforce existing power structures and vulnerabilities. Due to commodification, markets for ES have also been criticized as obscuring ecosystem functions (Peterson et al., 2010)

In the third Chapter of this Thesis, I argue, similarly to Ernstson (2013), who focuses however on environmental distributive justice, or to Ernstson et al. (2008), who look at ES and social movements, that ES, especially in the proximity of urban areas, are socially constructed and the result of a series of processes and decisions often highly politicized. The supply of ES should therefore not be taken for granted as it ultimately is the result of socio-economic and political processes which lead to a certain configuration of the environment guaranteeing or enhancing some ecosystem functions and their benefits, possibly to the detriment of the supply of others. This is ever more true in urban areas where the proximity and the concentration of people, economic activities and infrastructures lead to intense and continuous transformation of land through highly politicized and contested processes (Ernstson, 2013; Ernstson and Sörlin, 2013; Swyngedouw, 1997).

Finally, as an emerging field of research, political ecology is far from being a grand theory (Schubert, 2005). Despite that, some main key arguments have been identified: that of degradation and marginalization; conservation and control; environmental identity and social movements (Robbins,

2004). In the last part of Chapter 3 I look at the role of social movements as a successful social actor against the continuous appropriation of peri-urban ecological system by urbanization.

Structure of the Thesis

This Thesis, besides the above introduction, presents four chapters and a conclusion.

The *first chapter* is a literature review on the functions of ecosystems in mitigating the vulnerability of urban areas to hydro-meteorological hazards, targeting mainly questions 1 and 3 listed above. While reporting a series of evidences about the benefits provided by ecosystems in this domain, the study leads to the design of a policy-relevant framework for the analysis and policy intervention to restore urban ecosystems for DRR. The review found that urban areas would benefit from well managed ES for the reduction of vulnerability to hydro-meteorological hazards but that policies need to address not only ecosystem health but also threats from urbanization and potential impacts of the hazards on the ecosystem itself.

The *second chapter* is an assessment of the social and ecological dimensions of vulnerability to heat waves of the Cologne urban areas in Germany. It also aims at answering to the research question 1 with an empirical case and touches upon the theme of question 3. Relevance is in fact given to the role of the environment in shaping the vulnerability of the urban population through an application of the MOVE generic framework. The results stress the prominent role played by properly managed urban ecosystems for the reduction of vulnerability to heat waves in Cologne and suggest paying increased attention to the dynamics that take place at the peri-urban interface.

The *third chapter* makes use of concepts developed in the field of political ecology to show how ES, especially in and around urban areas, are ultimately the result of a political and historically traceable process, and are thus not to be taken for granted. The chapter targets question 2 but has also implications for a revised characterization of urban ecosystems which includes surrounding natural areas as an important, nearby source of services for city well-being (question 3).

The *fourth chapter*, through the analysis of water related services for urban well-being, locates cities within the larger unit of the watershed. It in fact demonstrates through a wide range of case studies that it is at this scale that important regulating services for urban areas originate. This perspective has policy implications as urban well-being can benefit from a wider range of policies which would for instance facilitate the transfer of resources (especially economic) traditionally concentrated in urban areas to more peripheral zones. The chapter aims at answering to question 3 by analysing the higher

regional scale and providing complementary information to the findings of the previous two chapters on the urban and peri-urban scales.

All the chapters show in different ways how the concept of ES can be applied in integrated studies. Besides bringing policy relevant results in Chapter 1 and Chapter 4, this Thesis offers two examples of multidisciplinary, integrated studies: in Chapters 2, by focusing both on the social and ecological dimension of vulnerability and, in Chapter 3, by focusing on environmental as well as on the socio-political factors determining the well-being of city inhabitants (see Figure 0.6).

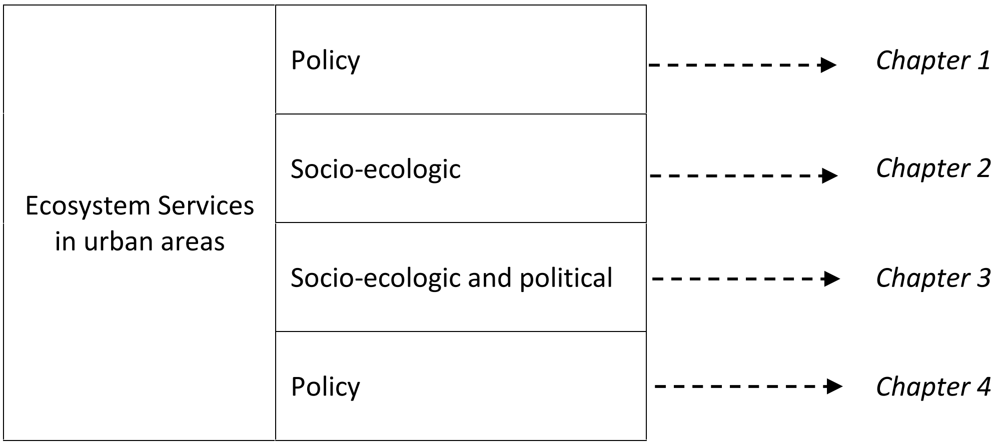

Figure 0.6. Integrated approaches to the study of ES in urban areas detailed per chapter.

Finally, excluding the global scale, each core chapter of the Thesis focuses on a different scale of the urban ecosystem (see Figure 0.7), thus allowing to draw relevant conclusions for multi-scalar study as well as with respect to the characterization of urban ecosystems.

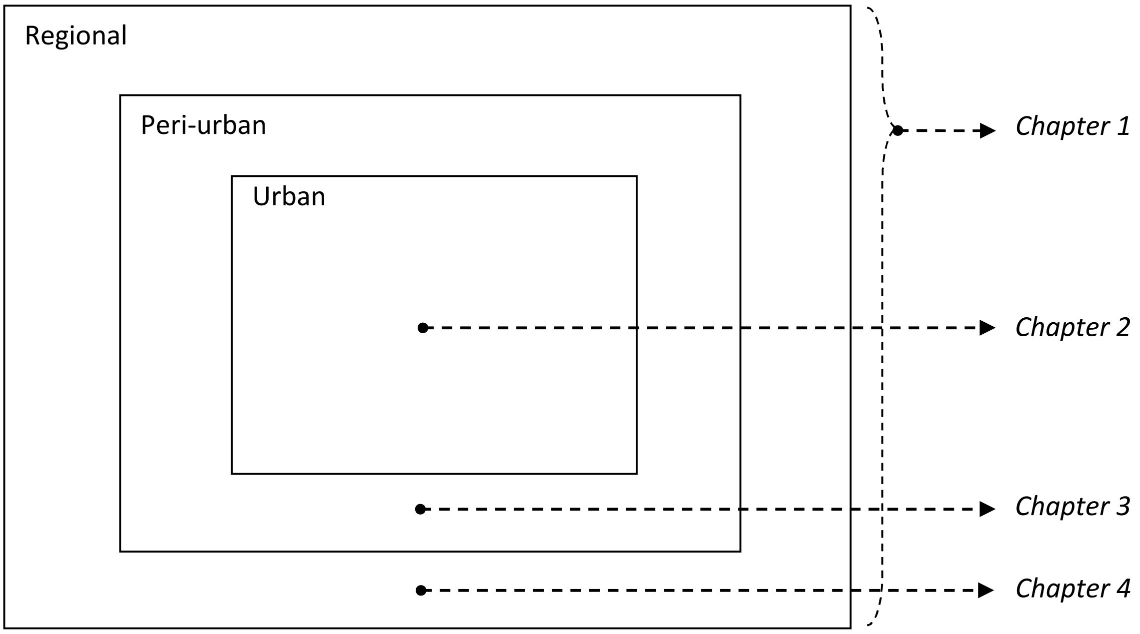

Figure 0.7. Scales of the urban ecosystem and focus of the chapters of the Thesis.

The *conclusions* section, by linking the findings of the chapters and the introductory section, delineates relevant contributions to integrated studies of urban ecosystems for human well-being and highlights how the chapters contribute by responding to the above three research questions. The concluding section also provides suggestions for further research.

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CHAPTER 1 . Heat waves and floods in urban areas: a policy-oriented review of ecosystem services

Abstract

Urbanisation is increasing and today more than a half of the world's population lives in urban areas. Cities, especially those where urbanisation is un-planned or poorly planned, are increasingly vulnerable to hydro-meteorolog ical hazards such as heat waves and floods. Urban areas tend to degrade the environment, fragmenting and isolating ecosystems, compromising their capacity to provide ser vices. The regulating role of ecosystems in buffering hydro-meteorological hazards and reducing urban vulnerability has not received adequate policy attention until now. Whereas there is a wide body of studies in the specialised biological and ecological literature about particular urban ecosystem features and the impacts of hazards upon people and infrastructures, there is no policy-driven overview looking holistically at the ways in which eco system features can be managed by cities to reduce their vulnerability to hazards. Using heat waves and floods as examples, this review article identifies the aggravating factors related to urbanisation, the various regulating ecosystem services that buffer cities from hydro-meteorological impacts as well as the impacts of the hazards on the ecosystem. The review also assesses how different cities have attempted to manage related ecosystem services and draws policy-relevant conclusions.

Keywords: heat wave, floods, ecosystem services, urban areas, inland water systems, environmental vulnerability

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1.1 Introduction

Nowadays, more than 50% of people live in urban areas with some 3.5 billion people having settled in cities throughout the world (Guzmán et al., 2009). This urbanisation trend continues today and is likely to continue in the coming decades (Marshall, 2007). As a consequence, large changes in exposure to natural hazards and vulnerability of social-ecological systems are taking place in urban areas. At the local and regional levels, consumption of natural assets and disruption of ecosystems[6] by cities have contributed to the modification of the surrounding environment. Urban development fragments, isolates and degrades natural habitats and disrupts hydrological systems (Alberti, 2005). The occupation of floodplains, land conversion, deforestation and loss of ecosystems are anthropogenic factors that contribute to the loss of buffering capacity of ecosystems to hazards. For instance, the impairment of soil functions in urban areas causes the loss of water permeability (i.e. soil sealing), which increases the impacts of potential floods and the likelihood of urban floods. Besides this, ecosystems can also be affected by hazards that can be an important determinant of vulnerability when communities have high dependencies on specific ecosystem services in and around cities. Extreme, large-scale weather events are likely to trigger ecosystem level disturbances, which may affect the organisation (species composition and diversity) and the functional attributes (Parmesan et al., 2000). Overall, ecological effects of extreme events have been identified as one of the main gaps of knowledge in community ecology (Agrawal et al., 2007).

While there is growing literature on climate change and vulnerability assessment (Adger, 1999; Füssel and Klein, 2006; Handmer et al., 1999; O'Brien et al., 2004), comparatively little of it concerns cities (Kallis, 2008) and even fewer address the importance of urban ecosystem services (Niemelä et al., 2010). There is an extended literature on ecosystem services[7], classification and valuation (Costanza et al., 1997; de Groot et al., 2002; Fisher et al., 2009; MA, 2005a), but likewise little of it focuses on the contribution of buffering hydro-meteorological hazards[8], and much less in cities. This gap is covered by the present review. Our starting point is that there is a large volume of relevant material in specialised literatures, not least ecology, about particular components of the urban ecosystem, but no overarching interdisciplinary and policy-oriented synthesis targeting disaster risk reduction. To ground our analysis we focus on two important hazards, especially for European cities: heat waves and floods.

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[6] Ecosystems are the dynamic complexes of plant, animal, and microorganism communities and the nonliving environment interacting as a functional unit, including humans (MA 2005, p. 27).

[7] Ecosystem services are the benefits people obtain from the ecosystem (MA 2005, p. 27).

[8] Hydro-meteorological hazards are processes or phenomena of atmospheric, hydrological or oceanographic nature that may cause loss of life, injury or other health impacts, property damage, loss of livelihoods and services, social and economic disruption, or environ-mental damage (UNISDR 2009, p. 18).

Section 1.2 presents the conceptual framework we used for the review. Section 1.3 discusses concepts related to ecosystem services and urban systems. Sections 1.4 and 1.5 review heat waves and floods, respectively. Both sections address the following components of the framework presented in Figure 1.1: a description of the hydro-meteorological hazard, the aggravating factors related to urbanisation, the regulating services (e.g. climate regulation, air quality regulation and water regulation) and the impacts of the hazard on the ecosystem. Section 1.5 reviews related policy initiatives for the protection of urban ecosystem services. The review ends with a concluding discussion and policy recommendations (Section 1.6).

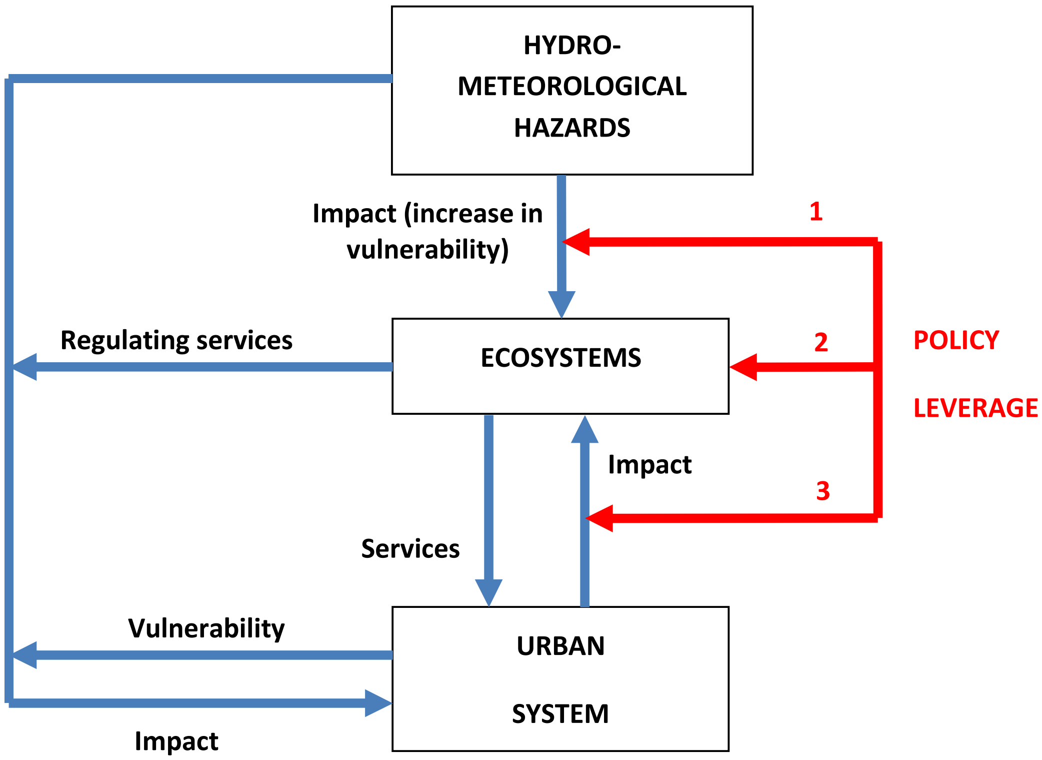

Figure 1.1. Conceptual framework highlighting the relationships between hydro-meteorological hazards, ecosystems and urban systems.

1.1.1 Conceptual framework

Figure 1.1 provides a conceptual framework for understanding the relationships between urban systems, ecosystems and hydro-meteorological hazards. While we recognise more integrated definitions of urban ecosystems that include both human and non-human elements in their inter-relation (e.g. Pickett et al., 2001), for the sake of this analysis we maintain a distinction between the human ("urban systems") and ecosystem components. As Figure 1.1 indicates, urban areas are vulnerable to hydro-meteorological hazards but ecosystems offer services that can buffer these

potential impacts. However, urbanisation as a process erodes these ecosystem services, as do hazard impacts, especially in a context of climate change and intensifying extremes, as indicated in Figure 1.1. Urbanisation fragments ecosystems, which diminishes their regulating capacity and increases the vulnerability of urban areas themselves. As shown in Figure 1.1, we are also interested in the vulnerability of ecosystems to hazards, defined in relation to the human component, i.e. their capacity to withstand stresses and maintain important regulating and other services. The vulnerability of the urban system derives therefore from the effect of the impacts of the hydro-meteorological hazard on the ecosystem and on the urban system combined with the potential role of regulating services. It is possible to act in different ways to reduce this vulnerability. Conventionally policy has focussed on the urban system component in terms of adaptation policies seeking to reduce the vulnerability of infrastructures or particular segments of the population to hydro-meteorological hazards. This article shifts attention instead to policies that can act on different leverages, such as urban ecosystem restoration and preservation by protecting ecosystems themselves from hazards directly (arrow 1 in Figure 1.1), ecosystems restoration and preservation (arrow 2) or indirectly by reducing urbanisation pressures (arrow 3).

1.1.2 Urban areas and ecosystem services

An urban area is defined as ‘‘a set of infrastructures, other structures, and buildings that create an environment to serve a population living within a relative small and confined geographic area’’(Albala-Bertrand, 2003, p. 75). Cities can be additionally defined as settlements that are permanent. The definition of urban areas adopted in national census varies from country to country. Three main classifications of localities as urban can be identified according to: (1) the size of the population (e.g. civic district which is in general greater than 2,000, 2,500 or 5,000 inhabitants); (2) the proportion of population of a civic district engaged in agriculture and the predominance of non-agricultural workers; and (3) administrative, legal criteria (e.g. type of local government) (2007 United Nations Demographic Yearbook). Delimitating urban areas, from a multidisciplinary perspective, is not a straightforward task (Pelling, 2003). The city can be considered as a single ecosystem or composed of individual ecosystems: all natural green and blue areas in the city, including street trees and ponds; seven ‘‘natural ecosystems’’ are identified: street trees, lawns/parks, urban forests, cultivated land, wetlands lakes/sea and streams (Bolund and Hunhammar, 1999). These ecosystems provide a variety of services including climate regulation, air purification, water regulation and carbon dioxide (CO_2) sequestration. For instance, urban green areas can buffer extreme events such as heat waves and floods by reducing temperatures, increasing ventilation, storing water and reducing run-off (EEA, 2010a). The Millennium Ecosystem Assessment (MA) has identified the

following classes of ecosystem services: ‘‘provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other nonmaterial benefits’’(MA, 2005a, p. 27). The supply of these services are the result of the functioning of ecosystems, representing the products of processes that occur within every ecosystem and, because the processes depend on organisms and the organisms are linked by their interactions, the services themselves are also linked (Fitter et al., 2010). For the sake of this review we focus mainly on regulating services and the multiple benefits they provide for the buffering of heat waves and floods in urban areas. Of all ecosystem services, regulating services are amongst the least investigated and assessed, and regulating service indicators are weaker (i.e. low ability to convey information and data availability) overall than provisioning service indicators (Layke, 2009). The analysis is based on the classification of ecosystem services proposed by the MA and the relevant services for each hazard have been identified. For heat waves we considered: air quality regulation and climate regulations, while for floods we analysed water regulation.

1.2 Heat waves

1.2.1 Heat waves as a hazard

Heat waves are extreme events associated with particularly hot sustained temperatures able to produce notable impacts on human mortality and morbidity, regional economies and ecosystems (Koppe et al., 2004; Meehl and Tebaldi, 2004). In Europe, heat waves have been the most prominent hazard with regards to human fatalities in the last 10 years (EEA, 2010b). One well-documented example is the European 2003 heat wave when more than 70,000 excess deaths were reported during the summer (EEA, 2010b) and 15,000 excess deaths in France alone (Fouillet et al., 2006). A large precipitation deficit during spring 2003 contributed to a rapid loss of soil moisture (Ciais et al., 2005; Fischer et al., 2007; Zaitchik et al., 2006). As a result, the summer 2003 was by far the hottest since 1500 AD in Europe (Luterbacher, 2004), and it seems that heat waves will become more intense, longer lasting and/or more frequent in future warmer climates (Luber and McGeehin, 2008; Meehl and Tebaldi, 2004).

1.2.2 Urbanization as an aggravating factor

In urban areas, the impacts of heat waves are aggravated and the vulnerability of ecosystems and urban communities are increased (see Figure 1.1). Urban development modifies land surface, leading to the creation of distinct urban climates (Grimmond, 2004). Urbanisation has quickly transformed

ecosystems to infrastructures and buildings that increase thermal-storage capacity (Luber and McGeehin 2008). Built up and impervious surfaces are stronger absorbers and the radiation is then slowly re-emitted as long-wave radiation that is responsible of warming up the boundary layer of the atmosphere within the urban canopy layer (Oke, 1988), producing the so called “Urban Heat Island” (UHI) effect. The UHI effect concerns the magnitude of the difference in temperature between cities and their surrounding rural regions and the temperature difference increases with the number of inhabitants and the building density: in Europe, the maximum UHI goes from 2 to 12°C (Koppe et al., 2004). Due to this effect, the highest morbidity and mortality associated with extreme heat appear to occur in cities (Clarke, 1972). Harlan et al. (2006) examined the relation among the microclimate of urban neighbourhoods, population characteristics, thermal environments that regulate microclimates and the resources people have to cope with climate conditions in Phoenix, AZ. Neighbourhoods with few open and green spaces, which have been proven to have cooling functions, contribute to increasing the impacts of heat in cities. Heat waves’ mortality rates, neighbourhoods’ environmental quality and population characteristics are thus spatially correlated (Harlan et al., 2006).

1.2.3 Climate regulation

As mentioned, temperatures in cities are higher than in the surroundings, which cause higher impacts of extreme heat events. Ecosystems in urban areas contribute to reducing the UHI effect (Bolund and Hunhammar, 1999). However, urban forests, a common term to characterise all of the vegetation of an urban region (McPherson et al., 1994), play a particularly important role in regulating climate, energy and water between the land surface and the atmosphere (Zaitchik et al., 2006). According to various authors, greening can cool the environment at least at the local scale (Akbari et al., 2001; Bowler et al., 2010; Oke et al., 1989), providing a climate-regulating service that can buffer the impacts of heat waves. This is because plants and trees regulate their foliage temperature by evapo-transpiration, leading to a reduction of the air temperature. In addition, green vegetation absorbs up to 90% of the photosynthetically active radiation while reflecting up to 50% of the near infrared radiation (Braun and Herold, 2003), thus absorbing less heat than built infrastructures. The size of the green area contributes to the magnitude of the cooling effect, although it is not clear if there is a minimum size threshold or if there is a simple linear relationship between these two factors: on average an urban park would be around 1°C cooler than a non-green site (Bowler et al., 2010). Gomez et al. (1998) observed that, in green areas, there was a drop of 2.5°C with respect to the city of Valencia (Spain) maximum temperature. Wong and Yu (2005) observed a maximum difference of 4.01°C between well planted area and the central business district area of Singapore, while according to Hamada and Ohta (2010) the temperature difference between urban and green areas in Nagoya

(Japan) was large in summer and small in winter. The maximum air temperature difference was 1.9°C in July 2007, and the minimum was -0.3°C in March 2004. Renaud and Rebetez (2009) compared open-site and below-canopy climatic conditions from 14 different sites in Switzerland during the 11-day August 2003 heat wave. Maximum temperatures were cooler under the canopy and, the warmer the temperature, the stronger the impact of the forest. For maximum temperature, the difference was higher in deciduous and mixed forests compared to coniferous forests. For minimum temperature, in contrast, the discrepancy was higher in coniferous forests (Renaud and Rebetez, 2009). Similarly it is worth noting that, during heat wave days, the increase in sensible heat flux is initially much larger over forests than over grasslands (Teuling et al., 2010). In the long term, however, grasslands become the main heat source due to the fact that elevated evaporative cooling accelerates soil moisture depletion (Teuling et al., 2010).

According to Alexandri and Jones (2008), although parks manage to lower temperatures within their vicinity, they are incapable of significantly cool the surrounding areas where people live. Therefore, the authors suggest that it would be more effective to place vegetation within the built space of the urban fabric; thus raised urban temperatures can decrease within the human habitats themselves and not only in the detached spaces of parks. For single trees, evapotranspiration and tree shading are important control measures in heat-island mitigation in Tel-Aviv (Shashua-Bar and Hoffman, 2000). According to these researchers, the cooling effect depends mainly on the amount and extent of the partial shaded area. For instance in Athens, during a short exceptionally hot weather period in 2007, the highest cooling effect of 2.2°C was found to be reached in a street with high tree shaded area and minimal traffic load (Tsiros, 2010). These results imply the passive cooling potential of shade trees. Akbari et al. (2001) estimated that 20% of the cooling demand of the USA can be avoided through the implementation of heat island mitigation measures for instance by planting trees. In general, frequentation and use of green spaces could generate benefits and well being on people, especially during heat waves periods, and this could be explained by the capacity of green spaces to provide better thermal comfort (Lafortezza et al., 2009).

1.2.4 Air quality regulation

Air quality regulation refers to the role ecosystems play in regulating the gaseous portion of nutrient cycles that affect atmospheric composition. Air quality plays an important role during heat waves and can be a source of human illnesses during these extreme events. During a heat wave in urban areas, hot days are often followed by hot nights because of the heat island effect. These conditions can produce a high degree of heat and air pollution stress, especially for people with cardiovascular and respiratory disorders (Piver et al., 1999). For instance, in The Netherlands, 1,000–1,400 deaths were

estimated because of the hot temperatures that occurred during the 2003 summer period, and of these, the number of deaths attributable to the ozone (O_3) and particular matter (PM_{10}) concentrations in the period June–August were estimated at around 400–600 deaths (Fischer et al., 2004). In France, the relative contribution of O_3 and temperature in the high mortality during the 2003 heat wave was heterogeneous among cities (Filleul et al., 2006). For the nine cities considered in their study, the excess risk of death for an increase of 10 $\mu g/m^3$ in O_3 level is significant (Filleul et al., 2006). In particular, between 3 and 17 August 2003, the excess risk of deaths linked to O_3 and temperatures together ranged from 10.6% in Le Havre to 174.7% in Paris, while the contribution of O_3 alone varied, ranging from 2.5% in Bordeaux to 85.3% in Toulouse (Filleul et al., 2006). In Croatia, a significant part of excess mortality, during the same period, was attributed to PM_{10} and O_3 in the air (Alebić-Juretić et al., 2007).

Due to their large leaf areas and their physical properties trees can act as biological filters. These can remove large numbers of airborne particles and hence improve the quality of air in polluted environments (Beckett et al., 1998; Brack, 2002; Escobedo and Nowak, 2009; Jim and Chen, 2008; Nowak et al., 2000). In particular, trees can be effective in reducing the impacts of damaging forms of particulate pollution such as PM_{10} or gasses such as sulphur dioxide (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO) and CO_2, and are effective in reducing O_3 concentrations in cities (Nowak et al., 2000). The effectiveness of this ecosystem service varies according to plant species, canopy area, type and characteristics of air pollutants, and local meteorological environment. Larger trees extract and store more CO_2 from the atmosphere and their greater leaf area traps air pollutants, casts shade and intercepts rainfall run-off (Brack, 2002). Uptake happens mainly through dry deposition, a mechanism by which gaseous and particulate pollutants are transported to and absorbed into plants mainly through their surfaces. In urban areas, districts with more extensive urban trees capture more pollutants from the air, and this capacity is increased as trees gradually reach final dimensions (Jim and Chen, 2008). In general, the effectiveness of uptake by trees of particles >5 μm is increased if their leaf and bark surfaces are rough or sticky (Beckett et al., 1998). For smaller particles the most effective uptake happens in the needles of conifers. Due to the larger total surface area of needles, coniferous trees have a larger filtering capacity than trees with deciduous leaves, with pines (*Pinus spp.)* capturing significantly more material than cypresses (*Cupresses spp.*) (Beckett et al., 2000). In addition, this capacity is also greater because the needles are not shed during the winter, when the air quality is usually worst (Bolund and Hunhammar, 1999). According to Jim and Chen (2008), most removal occurs in the winter months mainly due to the higher pollutants concentrations. However, coniferous trees are also more sensitive to air pollution and deciduous trees are better at absorbing gasses (Bolund and Hunhammar, 1999). Veteran trees (i.e. trees that have lived a long time and are

significant elements of the landscape) often contribute substantially more benefits to society relative to other (smaller) trees in the landscape (Nowak, 2004). For instance, veteran trees will store, due to their increased size, larger amounts of carbon in their tissues. Interception of particles by vegetation seems also to be much greater for street trees, due to their location in proximity to high road (Beckett et al., 1998). Trees situated close to a busy road capture significantly more material, especially larger particles, than those situated in a rural area (Beckett et al., 1998). In Chile, it has been demonstrated that Santiago's urban forests are effective at removing PM_{10} (Escobedo and Nowak, 2009). In 1991, trees in the city of Chicago (11 percent tree cover) removed an estimated 15 metric tons of CO, 93 tons of SO_2, 98 tons of NO_2, 210 tons of O_3 and 234 tons of PM_{10} (McPherson et al., 1994). Similarly, the peri-urban vegetation of the Madrid region constitutes a sink of O_3, with evergreen broadleaf and deciduous tree species removing more atmospheric O_3 than conifer forests. Nowak et al. (2000) modelled the effects of increased urban tree cover on O_3 concentrations (13–15 July 1995) from Washington, DC (USA), revealing that urban trees generally reduce O_3 concentrations in cities.

Jim and Chen (2008) assessed the capability and monetary value of the removal by urban trees of air pollution in Guangzhou city in South China. The researcher found that an annual removal of SO_2, NO_2 and total suspended particulates of about 312.03 mg, and the benefits were valued at RMB 90.19 thousand. Overall, there are few known studies that analyse differences in urban forest structure and air pollution removal in sub-regions of a city, and there are even fewer studies that link a city's urban forest structure and socioeconomic activity with site-specific pollution dynamics through time (Escobedo and Nowak, 2009).

1.2.5 Impacts of heat waves on ecosystems in and around urban areas

As showed in Figure 1.1, hydro-meteorological hazards can also affect ecosystems, such as forests and water systems, and their services in urban and peri-urban areas. In the summer 2003, the drought experienced by the vegetation was worsened by the length of the period with scarce precipitations and humidity, by the heat during the summer and the longer duration of the sunshine period (Rebetez et al., 2006). In the course of a drought, the gradually decreasing rate of passage of either water vapour or CO_2 through the stomata, or small pores of the plant, assimilation and growth are observed (Leuzinger et al., 2005). Fires can also affect vegetation during heat waves or periods of drought. During the 2003 heat waves, small-scale forest fires were observed all over central Europe and the western Mediterranean (Fink et al., 2004).

Trees can be directly affected by air pollutants depending on the types and concentrations. The most common effects of plant exposure to O_3 are modifications of stomata behaviour, which leads to a reduced photosynthesis and an increased respiration. As gas, NO_x and SO_2 damage cuticles and

stomata and, most importantly, they penetrate through stomata and alter tissues. SO_2 can cause both acute (e.g. cell plasmolysis) and chronic injury (e.g. reduced gas exchange, chlorophyll degradation, chloroplast swelling, and alteration of cellular permeability). As mentioned above, after a certain threshold is reached trees can be affected by nutrient stress, reduced photosynthetic or reproductive rate, predisposed to entomological or microbial stress, or direct disease induction (Smith, 1974).

In the urban context, water bodies and wetlands are often beneficial for transportation, recreation, dilution and purification. On the other hand, urbanisation is thought to cause: ‘‘reduced baseflows, increased frequency and magnitude of peak discharges, increased sediment loads, impaired water quality, reduction in channel and floodplain complexity’’ (LeBlanc et al., 1997). Water quality may further deteriorate to critical values during periods of prolonged low-flow conditions in combination with high water temperatures. In the Meuse river basin, which flows through the cities of `Namur and Liege in Belgium, during the 1976 and 2003 heat waves, deterioration of water quality by high water temperatures, eutrophication, increased concentrations of major elements and some metals or metalloids (selenium, nickel and barium) were measured (van Vliet and Zwolsman, 2008). However, concentrations of nitrate and some heavy metals with high affinity for adsorption onto suspended solids (i.e. lead, chrome, mercury and cadmium) decreased, which also positively affected chemical water quality during drought (van Vliet and Zwolsman, 2008). Overall eutrophication, following hot spells, causes major impacts on the water system.

During a heat wave event, temperature of lakes can rise until reaching record temperatures (increase varies between 1 and 3°C on average) (Jankowski et al., 2006). Jankowski et al. (2006) investigated the consequences of the 2003 European heat wave for lake temperature profiles, thermal stability (i.e. suppressing downward turbulent mixing) and hypolimnetic oxygen depletion. Warming of water bodies can restrict lake overturning and lead to anaerobic conditions, thus potentially seriously impacting aquatic ecosystems, leading, in the summer, to the development of harmful cyanobacterial blooms. Oxygen depletion may lead to negative ecological consequences such as phosphorous dissolution from the sediments leading to internal loading and algal blooms (Jankowski et al., 2006). Surface blooms of toxic cyanobacteria in eutrophic lakes may lead to mass mortalities of fish and birds, and affects cattle, pets, and humans (Jöhnk et al., 2008). This may have impacts on recreational activities, fishing and surface water sports. In sport fisheries, increased water temperature has been associated with decreased activity and movement to deeper cooler waters, which reduces fish catches. Concerning water quantity, most of the studies agree that wetlands reduce the flow of water in downstream rivers during dry periods. In fact evapotranspiration from wetlands is shown to be higher than from other portions of the catchment during these periods (Bullock and Acreman, 2003). For groundwater resources, prolonged heat stress may lead to lowered water table levels. In urban areas,

the impairment of soils aggravates the magnitude of this impact because the reduced infiltration reduces the water table. Lower water table levels were measured in urban areas (Scalenghe and Marsan, 2009)

1.3 Floods

1.3.1 Floods as a hazard

The European Union Floods Directive defines a flood as a temporary covering by water of land not normally covered by water. According to Few et al. (2004), flood disasters and their mortality impacts are heavily concentrated in Asia, where there are high population concentrations in floodplains, such as the Ganges, Brahmaputra, Mekong and Yangtze basins, and in cyclone-prone coastal regions such as around the Bay of Bengal and the South China Sea. Floods affecting urban areas can be either generated locally or in other locations in the watershed and basin. Urban areas often generate impacts on watershed-wide ecosystems such as through land use changes and infrastructure development, which affects watercourses. When dealing with floods, it is therefore important to consider the role of ecosystems not only within the urban areas themselves, but also in the entire landscape of the watershed and the influences urban areas have on them (PEDRR, 2011). These two scales of analyses are considered in this section with a focus on urban areas.

Floods are the result of meteorological and hydrological factors, but anthropogenic modifications can also play a role in defining the magnitude of the event. Therefore, floods in urban areas are the result of natural and manmade factors. Although these influences are very diverse, they generally tend to aggravate flood hazards by accentuating flood peaks. As a result of different combinations of factors, urban floods can basically be divided into four categories: local floods, riverine floods, coastal floods and flash floods. Floods in urban areas can be attributed to one or a combination of the above types. The main cause of urban flooding is a severe thunderstorm, which is generally preceded by a long but moderate rainfall that saturates the soil. Therefore floods in urban areas are, in general, flash floods that expand both on impaired surfaces and in surrounding parks and streets (Andjelkovic, 2011). Other causes of urban floods are: ‘‘inadequate land use and channelization of waterways; failure of the city protection dikes; inflow from the river during high stages into urban drainage systems; surcharge due to blockage of drains and street inlets; soil erosion generating material that clogs drainage system and inlets; inadequate street cleaning practice that clogs street inlets’’ (Andjelkovic, 2011).

Major destructive flooding events occurred in Europe in the last few years: floods in the Elbe basin in 2001 that produced losses of over 20 billion Euros; floods in Italy, France and the Swiss Alps in

2000 costing around 12 billion Euros and a series of floods in the UK during summer 2007 accumulating losses of more than 4 billion Euros (EEA, 2010b). Losses as a consequence of floods have increased in the past decades in Europe. While there is no evident trend over time in respect to the number of fatalities and observations do not show a clear increase in flood frequency (Mudelsee et al., 2003), increases in population and wealth in the affected areas are the main factors contributing to the increase in losses (EEA, 2010b).

1.3.2 Impacts of urbanization on ecosystems

When degraded by a variety of human activities and changing climatic conditions, hydrologic regimes typically have increased frequency and severity of flooding, lowered water tables and reduced groundwater recharge compared to previous, more ‘‘natural’’ conditions (Cai et al., 2011).

At the river basin level, urbanisation directly affects catchments hydrology by changes in surface runoff, groundwater runoff, groundwater levels and water quality. The introduction of impervious surfaces inhibits infiltration and reduces surface retention (Packman, 1980). Thus the proportion of storm rainfall that goes to surface runoff is increased, and the proportion that goes to evapotranspiration, groundwater recharge and base flow is reduced. This increased surface runoff is combined with an increase in the speed of response and increased peak discharge, which can lead to floods (Nirupama and Simonovic, 2007; Packman, 1980). For instance, in the Upper Thames River watershed in Canada, urban areas have increased to 22% of the total watershed area in the year 2000 compared to only 10% in 1974, enhancing the risk from river flooding (Nirupama and Simonovic, 2007). On the other hand, in the Dead Run watershed (14.3 km^2) in the Baltimore, MD, region, the current tree cover is 13.2% with an impervious cover of 29%. Increasing tree cover in the watershed to 71% is estimated to reduce total runoff in the watershed by about 5% for the simulation period of the year 2000 (Nowak, 2006). Li and Wang (2009) analysed the urban expansion of St. Charles County, a suburb of St. Louis, MO, located in the Dardenne Creek watershed. A rapid increase of urban areas in the watershed took place from 3.4% in 1982 to 27.3% in 2003, and model simulations suggested an increase in more that 70% in average direct runoff in the watershed from 1982 to 2003, correlated with urban expansion.

At the urban scale, more radical changes in surface characteristics and soil sealing (i.e. the covering of soil for housing, roads and parking lots, etc.) increase the impermeability of soils, drainage and water run-off and lead to rapid precipitation run-off, flooding, erosion and impervious surfaces cause: “local decreases in infiltration, percolation and soil moisture storage, reductions in natural interception and depression storage and increases in runoff” (Brun and Band, 2000). Severe storms may also yield discharges exceeding the capacity of the sewer system, causing choking of the flow

and increased attenuation in localised ponding. The soft ground of vegetated areas allows water to seep through, and the vegetation takes up water and releases it into the air through evapo-transpiration (Bolund and Hunhammar, 1999). Urban sprawl with moderate to high soil sealing over a large area reduces the infiltration potential of the soil and increases the flood risk of urban areas (EEA, 2010a). Soil sealing also impacts the porosity of soils by reducing it or by modifying its pattern, which reduces water infiltration (Scalenghe and Marsan, 2009). In the surroundings of urban areas the amount and the speed of flooding water that arrives on unsealed surfaces are increased and increase the risk of ponding and erosion (Scalenghe and Marsan, 2009). In these areas, when human population increases with urban sprawl, wetland's functions can easily be affected with pollution or recreational activities (Mitsch and Gosselink, 2000). In these conditions, wetlands can no longer effectively reduce floods, sequester pollutants or host different biota (Mitsch and Gosselink, 2000). Thus wetland value appears to be maximum when close to the river system and distributed spatially across an environment that is not dominated either by cities or agriculture, but one that balances nature and human aspects (Mitsch and Gosselink, 2000).

1.3.3 Water regulation

The role of forests and soil

On the local scale, forests and forest soils are capable of reducing runoff generally as the result of enhanced infiltration and storage capacities. At the river basin scale, this holds true for small-scale rainfall events in small catchments, which are not directly responsible for severe flooding in downstream areas (FAO and CIFOR, 2005). The geomorphology of the area and the preceding rainfall seem to be the two most important factors in determining the magnitude of the flooding event: the amount of storm flow is most directly linked to the area in the watershed and volume of precipitation or snowmelt deposited on the site, stored or transported to the stream (Eisenbies et al., 2007). Overall, forests seem not to be able to stop large-scale floods, which are caused by severe meteorological events (Eisenbies et al., 2007).

According to the physical properties of soils, some, not heavily affected by human activities, have a large capacity to store water, facilitate transfer to groundwater, and prevent and reduce flooding (MA, 2005b). The initial conditions of soil saturation in a basin determine the manifestation and the intensity of a flood event. In conditions of low soil saturation, water percolates in the soil and flood risk is diminished (Lahmer et al., 2000). While in conditions of high soil saturation, additional precipitation is rapidly transferred to the river through surface runoff or by interflow (Lahmer et al., 2000). When the soil becomes saturated and loses its ability to store further water, there does not

seem to be any evidence that forests and their soil have a noticeable effect in regulating floods for the extreme rainfall conditions that lead to major flooding events (Balmford et al., 2008). However, evidence indicates that the key factor linking land use and flood regulation is soil condition rather than the trees, and that much of the soil degradation associated with deforestation results from poor land use practices (e.g. soil compaction during road building, overgrazing, litter removal, destruction of organic matter, clean weeding) (Balmford et al., 2008).

The role of wetlands

Wetland, floodplain, lake and reservoir ecosystems play an important role in the regulation of floods in inland systems and provide protection from the adverse consequences of natural hazards to humans, even for urban areas. In particular, wetlands are significant in altering water cycle and perform hydrological functions (Bullock and Acreman, 2003). Floodplain wetlands reduce or delay floods; on the other hand most of the studies show that wetlands located upstream in a watershed tend to be quickly saturated and increase the risk of flash floods (Bullock and Acreman, 2003). If wetlands are too small, functions, such as storage of floodwater (for mitigation of floods), no longer exist: it has been assessed that 3–7% of the area of a watershed in temperate zones should be maintained as wetlands to provide both adequate flood control and water quality improvement functions (Mitsch and Gosselink, 2000). Effective control is more often the result of the combined effect of a series of wetlands within a catchment area instead of single units. Some authors argue that, to maintain the pulse control function of wetlands, a greater number of wetlands in the upper reaches of a watershed is preferable to fewer larger wetlands in the lower reaches [Loucks (1989) cited in Mitsch and Gosselink (2000)]. A modelling effort on flood control suggested the opposite: the usefulness of wetlands in decreasing flooding increases with the distance of the wetland downstream [Ogawa and Male (1986) cited in Mitsch and Gosselink (2000)]. Further research is therefore needed in order to determine the real effects of wetlands and their position in the landscape in terms of buffering urban areas from floods.

1.3.4 Impacts of floods on ecosystem services

While floods provide a series of benefits (e.g. nutrients deposition), these hazards can also affect ecosystems especially when the environment has been degraded. For instance, flooding and submergence are responsible for major abiotic stresses and, together with water shortage, salinity and extreme temperatures are the major factors that determine species distribution (Visser, 2003). Overall, only pioneering species are able to locate in zones close to the river (Blom and Voesenek, 1996), and

floods have a greater impact on plant species during the growing season (Kozlowski, 1997). Plant responses to flooding during the growing season include: ‘‘injury, inhibition of seed germination, vegetative growth, and reproductive growth, changes in plant anatomy, and promotion of early senescence and mortality’’ (Kozlowski 1997). Adaptive strategies and flood tolerance of plants depend on plant species and genotype, age of plants, frequency, duration, timing and conditions of flooding (i.e. soil flooding, water logging, total submergence of vegetation) (Kozlowski, 1997; Vervuren et al., 2003). Underwater light intensity and depth of the water column may also affect survival during periods of submergence (Vervuren et al., 2003).

Flooding also changes the physical status of soils, which may severely affect peri-urban agricultural practices. For instance, water logging causes the breakdown of large aggregates into smaller particles, deflocculation of clay and destruction of cementing agents. As the water level declines, these small parts of soil are redistributed in a new, more dense structure, creating: ‘‘smaller soil pore diameters, higher mechanical resistance to root penetration, low O_2 concentrations and the inhibition of resource use’’ (Blom and Voesenek, 1996) and the accumulation of CO_2 in soils (Kozlowski, 1997). Gas diffusion is severely inhibited during flooding (Blom and Voesenek, 1996): oxygen remains in the soils and is consumed by aerobic processes (roots and soil organisms), nutrient availability for plants strongly decreases and anaerobic processes take place producing toxic substances for plants [lowering of soil redox potential (Eh)]. Chemical changes also include the increased solubility of mineral substances, reduction of Fe, Mn and S, and anaerobic decomposition of organic matter. The reduction condition (i.e. low soil Eh) is also a major factor in wetland ecosystems that influences plant survival, growth and productivity (Pezeshki, 2001). Floods can also cause secondary impacts, for instance affecting industries and in particular petrochemical ones, causing the release of toxic chemicals, which further impact the environment.

1.4 Policy initiatives for the protection of urban ecosystem services for disaster risk reduction

At the international level, policy initiatives that target any of the three points of policy leverage identified in Fig. 1, are rare. The United Nations International Strategy for Disaster Reduction’s (UNISDR) Hyogo Framework for Action 2005–2015 lists ‘‘sustainable ecosystems and environmental management’’ as one of the main pillars for reducing underlying risk factors (UNISDR, 2005). Calling for an improved management of ecosystems and their services for disaster risk reduction, this initiative directly targets arrow 2 of the framework in Figure 1.1. Paragraph 19 of the Hyogo Framework can be situated in correspondence to arrow 3 of the framework as it establishes the Priority for Action 4: ‘‘Reduce the underlying risk factors’’, which include: ‘‘Incorporate disaster

risk assessments into urban development planning and management of disaster-prone human settlements'' … ''rural development'' … ''major infrastructure'' … ''including considerations based on social, economic and environmental impact assessments''. In addition the UNISDR campaign on Making Cities Resilient proposes a 10-point checklist to serve as a guide for commitment by Mayors (UNISDR, 2010). Point 8 makes explicit the need to consider the environmental dimension: ''Protect ecosystems and natural buffers to mitigate floods, storm surges and other hazards to which your city may be vulnerable. Adapt to climate change by building on good risk reduction practices'' (UNISDR, 2010, p. 9).

At the local and regional levels, policy initiatives are very few for the incorporation of ecosystem preservation for disaster risk reduction in urban areas. At the European level, the Communication from the Commission to the European Parliament ''Options for an EU vision and target for biodiversity beyond 2010'' identifies four policy options to halt the loss of biodiversity and ecosystem services by 2020 but with no reference to disaster risk reduction (EC, 2010). For European countries, notably the UK, The Netherlands and Germany, affected by severe flooding in recent years, have made policy shifts to ''making space for water'', represented by arrow 2 of the framework (Figure 1.1). New risk management policies and practices favour a more holistic approach to flood risk management based on River Basin Management Plans, Integrated Coastal Zone Management to enhance natural processes.

Regarding heat waves, the London local government authorities have developed the ''Right Trees for a Changing Climate'' database and website to provide advice on planting the right trees in the right place, based on the fact that planting more trees, alongside increasing other green cover, is one of the ways in which London can adapt to climate change (http://www.right-trees.org.uk/default.aspx), using vegetation to keep the city cool. The city has set ambitious targets to: increase tree cover by 5% by 2025; increase greenery in the centre of London by 5% by 2030 and a further 5% by 2050; create 100,000 m^2 of new green roofs by 2012; and enhance 280 hectares of green space by 2012. Stuttgart has planned to exploit the role of natural wind patterns and dense vegetation in reducing problems of overheating and air pollution. A Climate Atlas was developed for the Stuttgart region, presenting the distribution of temperature and cold air flows according to the city's topography and land use. Based on this information, a number of planning and zoning regulations are recommended that aim to preserve open space and increase the presence of vegetation in densely built-up areas (Kazmierczak and Carter, 2010). In Stuttgart the preservation of the natural environment in urban areas is principally guided by the Federal Nature Conservation Act (BNatSchG), which prohibits the modification or impairment of protected green spaces, or changing land use in these protected areas (i.e. ''zones in settlement areas, parks, cemeteries, significant gardens, single trees, lines of trees, avenues or groves

in settled or unbuilt areas; and some plantings and protective wood outside forests'') (Kazmierczak and Carter, 2010). London and Stuttgart have thus put in place policies that focus on arrow 2 and 3 of the framework presented in Fig. 1, meaning that they call for ecosystem conservation and an improvement of urban planning.

Other examples of the implementation of projects and policies to protect cities from hydro-meteorological hazards through the restoration and management of ecosystem services come from the US and Lao PDR. The ''Grow Boston Greener (GBG)'' is a collaborative effort of the city of Boston (USA) and its partners to increase the urban tree canopy cover in the city by planting 100,000 trees by 2020. The planting of these trees will increase Boston's tree canopy cover from 29 to 35% by 2030 as the planted trees mature. The goals are to: ''increase the tree canopy cover in low canopy areas; mitigate the urban heat island effect and reduce energy consumption through the appropriate placement of trees on residential and commercial properties; improve air quality; and improve storm water management through strategic neighbourhood plantings'' (http://people.tribe.net/phoenix_fire_nectar/blog/bfacb2b9300a-4a28-9ac9-48681de04b07). This case is interesting as it considers the multiple benefits and services that can be derived by the same ecosystem, in this case the urban forest.

''Integrating Wetland Ecosystem Values into Urban Planning: The Case of That Luang Marsh'' is an economic assessment of the goods and services provided by the marshes in an attempt to examine the economic value of urban wetland biodiversity and its importance to people living around the wetland as well as the larger urban area of Vientiane (Gerrard, 2004). Wetlands and marsh areas in and around the city are important physical features and provide hydrological functions such as flood control. There are currently 175 flood-prone areas within the city limits, 70 of which are located in the city's core area. Flooding occurs at least six times a year but in many cases flood-prone areas will flood every time it rains. In the urban area of Vientiane flooding is not deep, but frequent flooding causes damage to buildings and roads, and interrupts transportation. The value of the regulating ecosystem service has been measured as the annual value of flood damages avoided in these areas and it will amount at close to US$ 3 million by the year 2020 (Gerrard, 2004).

1.5 Conclusions

Basing our analysis on the conceptual framework of Figure 1.1, we reviewed how locally and regionally generated and well-managed ecosystem services can contribute to lower the vulnerability of urban communities to hydro-meteorological hazards such as heat waves and floods. We also reviewed studies that quantify, when possible, the role of these ecosystem services.

Urban areas will continue to attract people who want to settle in them because of the many economic advantages they provide. It is therefore critical for urban areas to provide safe livelihoods to their populations, and although urbanisation will always imply a change in ecosystems and the services they provide, urban planning should systematically consider the role of ecosystems in buffering and mitigating the effects of environmental hazards such as heat waves and floods. In this respect, urban areas would benefit from the restoration and adequate management of ecosystems. Urban planners and managers should take into consideration the role of ecosystems in reducing risks and vulnerabilities (arrow 2 of Figure 1.1). When establishing urban development plans the presentation of landscape plans and green open space structure plans should be taken into account (arrow 2 and 3 of Figure 1.1). Although urban areas will generally create similar types of effects when it comes to heat waves and flooding, the specific local climatic conditions (e.g. urban heat island effect) will dictate the magnitude and frequency of the hazards and the urban ecosystems the potential to buffer these. There are therefore no generic sets of solutions to address the problems globally, but adapted solutions need to be sought regionally or locally. On the other hand, while designing strategies that make use of ecosystem services to buffer cities, it is important to take into consideration the effects of hazards on the ecosystem itself (arrow 1 of Figure 1.1). For instance, the right type of trees (i.e. in general native species) or the right succession that is more resistant to the impacts of hydro-meteorological hazards should be identified to be planted. Overall we have sufficient ecological knowledge on how cities can be buffered by ecosystem services with respect to the reviewed hydro-meteorological hazards. But we know little on how to measure ecosystem services for hazard regulation, which is an obstacle towards the design and implementation of appropriate policies and plans. We propose here a series of general recommendations to fill this gap:

- ecosystem conservation and restoration play an important role in lowering the vulnerability of urban areas to hydro-meteorological hazards. A meta-analysis of 89 restoration assessments in a wide range of ecosystem types across the globe indicates that ecological restoration increased provision of biodiversity and ecosystem services by 44 and 25%, respectively (Benayas et al., 2009);
- the vulnerability of urban communities and ecosystems to hydro-meteorological hazards is influenced and aggravated by the impacts of these hazards on ecosystems and urbanisation. It is therefore necessary to integrate disaster risk reduction, appropriate urban planning and ecological restoration. For instance, in areas where land-use pressure is considerable, ecosystem services can be secured by leaving green and blue areas in close proximity to one another, so that these areas form larger nature and landscape entities (Niemelä et al., 2010);

- one ecosystem my provide services for the regulation of more than one type of hydro-meteorological hazard, as is the case for green areas in cities. It is therefore recommended to adopt a multiple-hazard approach;
- generally cities are located in a watershed. The management of ecosystems for the protection of cities from hydro-meteorological hazards at this scale, which transcends administrative boundaries, can provide important elements of comparison of the vulnerability of different cities.

Urban ecosystems provide essential services to cities and city dwellers that are exposed to heat waves and floods. Given the rapid urbanisation throughout the world and the likely impacts of climate change in terms of these two hazards, it is urgent to manage these ecosystems in a better way than has been done in the past through integration of ecosystem management in urban planning and disaster risk reduction.

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CHAPTER 2 . Social vulnerability assessment of the Cologne urban area (Germany) to heat waves: links to ecosystem services

Abstract

More than three quarters of the European population live in urban areas and this proportion is increasing, leading, in some cases, to increased vulnerability of cities to environmental hazards. The health impacts of heat waves are aggravated in cities due to the high density of buildings, the fragmentation of green areas and the higher concentrations of air pollutants. Ecosystems can provide important benefits that mitigate the impacts of heat waves but at the same time can themselves be affected by the hazard, thus limiting their services. The objective of this study was to assess the vulnerability of the Cologne urban population to heat waves, taking into consideration a range of social and ecological variables. Based on the MOVE framework, indicators were developed and GIS applications were used to spatially assess the relative vulnerability of the 85 districts of Cologne to heat waves. The insights gained were integrated and corroborated with the outcomes of stakeholders' interviews. As environmental factors play a major role in this assessment, it is suggested that ecosystem management in Cologne and its surroundings be improved. In addition, though vulnerability is higher in central districts, attention needs to be paid to the periphery where the most susceptible groups reside.

Keywords: vulnerability assessment, heat waves, ecosystem services, urban areas, GIS

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2.1 Introduction

In Europe, 75% of the population live in urban areas and the proportion is increasing (EEA, 2006). However, absolute population growth is not the major contributor to the increase of the disaster potential in cities (Mitchell, 1995). Rather, the shift in the location of industries and homes, driven by economic factors and lifestyles, plays a significant role in the conversion of rural lands near European cities (Mitchell, 2003) which in turn alters ecosystems, affecting their services through e.g. the compaction of soil and the impairment of its functions. These changes increase the loss of water permeability (soil sealing), compromise the availability of water supply in terms of groundwater recharge and fragment green cover which is accompanied by an increase in resources and energy consumption (EEA, 2006). In addition, under scenarios of climate change, geographical areas that were less affected by heat spells are likely to become at higher risk of extreme hydro-meteorological events (IPCC, 2012a). The vulnerability of urban populations to hazards is in this way further exacerbated.

The impacts of heat waves on the ageing segment of the population that lives in the highly modified ecosystem of urban areas are of increasing concern for European cities. In fact, urbanisation affects climate locally, as cities tend to be warmer than their surroundings, producing the so-called Urban Heat Island (UHI) effect. This manifests itself especially at night and principally as a consequence of the properties and density of built infrastructures, low albedo, low green cover and low moisture availability in cities. Air quality is also degraded in cities, mainly due to the higher concentration of road traffic and industrial activities which require fuel combustion leading to the emission of air pollutants dangerous for human health. As a result, during heat waves, the rates of heat-related morbidity and mortality are often higher in cities than in their surroundings (Yardley et al., 2011). This is especially true for densely populated areas, where heat-retaining buildings, few and fragmented green areas with a lower cooling capacity, and higher levels of air pollution due to higher road traffic, amplify the impacts of the hazard (Conti et al., 2005; Filleul et al., 2006; Fischer et al., 2004; Gabriel and Endlicher, 2011; Tan et al., 2010; Tressol et al., 2008). It is therefore suggested that excess deaths occurring in urban areas during periods of extreme heat can be significantly reduced through appropriate urban land cover planning (Solecki et al., 2005). Land use patterns are in fact related to the capacity of urban ecosystems to provide regulating services which can be assessed through landscape functions (i.e. the capacities of a landscape to provide goods and services to the society) (MA, 2005; Willemen et al., 2008). It should however be emphasized that social and institutional considerations (e.g. early warning systems, the adoption of appropriate behaviours, facilitating tighter social networks) remain paramount while dealing with this type of vulnerability.

The objective of this study is to assess the social vulnerability of the 85 districts of the Cologne urban area as part of the MOVE (Methods for the improvement of Vulnerability Assessment in Europe) project funded by the European Commission. In the following introductory sections, vulnerability assessment, social vulnerability to heat waves, the role of ecosystem services in mitigating the impacts of heat waves, as well as the assessment of ecosystem services as landscape functions are briefly reviewed and defined. Section 2.2 presents the methodology developed, Section 2.3 presents the results, which are then discussed in Section 2.4.

2.1.1 Vulnerability assessment

An extensive review of the vulnerability terminology was carried out by Thywissen (2006) and includes a comprehensive list of definitions that primarily differ according to the school of thought in which these are developed and in use. According to these different schools, which can mainly be clustered into "political economy", "social-ecology", "holistic vulnerability and disaster reduction assessment" and "climate change science" (Birkmann et al., 2013), various approaches and frameworks have been developed to assess vulnerability. The MOVE project was intended to overcome these differences by producing a generic Framework (Figure 2.1) that would bring together and be applicable both in the domain of disaster risk reduction and climate change adaptation. The MOVE framework is intended to be a guiding tool more than a close representation of reality. By assembling the main elements of the vulnerable social-ecological system (i.e. a coupled system of biophysical and social components that interact and evolve according to complex dynamics) at multiple scales, and representing the risk factors, the framework closes the loop through the adaptation section. Adaptation is actually considered as a central element in shaping vulnerability in the long term within the risk governance frame. The framework is also intended to facilitate the integration of different indices and methodologies to contribute to a more integrated assessment. To respond to the need of holistic approaches, using quantitative, qualitative and participatory methods at different scales, it is in effect a challenge to combine different methodologies (Birkmann, 2006).

The present study also applies the multidimensional concepts of vulnerability assessment developed within the MOVE project. Vulnerability is defined as "the propensity of exposed elements such as physical or capital assets, human beings and their livelihoods to experience harm and suffer damage and loss when impacted by single or compound hazard events" (Birkmann et al., 2013). In the Framework, vulnerability is considered to be the result of the contributions and interactions of three components: exposure, susceptibility (or fragility) and lack of resilience. Exposure defines "the extent

to which a unit of assessment falls within the geographical range of a hazard event"; susceptibility "describes the predisposition of elements at risk to suffer harm"; while lack of resilience is defined by the "limitation in access to and mobilisation of the resources of a community or a social-ecological system in responding to an identified hazard", compromising the capacity to anticipate, to cope and to recover of the system (Birkmann et al., 2013). Six dimensions of vulnerability are considered within MOVE to characterise the susceptibility and the lack of resilience of the system: physical, ecological, social, economic, cultural and institutional. While socio-economic and institutional aspects of vulnerability are well explored in the literature, other dimensions of vulnerability, such as the ecological or the cultural ones, are currently not well integrated in vulnerability assessments.

The present study, through the application of the MOVE generic framework and definitions, aims at illustrating the operationalization of these theoretical concepts for practical use in the decision-making context. The assessment presented in this work focuses on the core part of the framework, which considers the main interactions between the three components of vulnerability, and this from the social as well as the ecological dimensions.

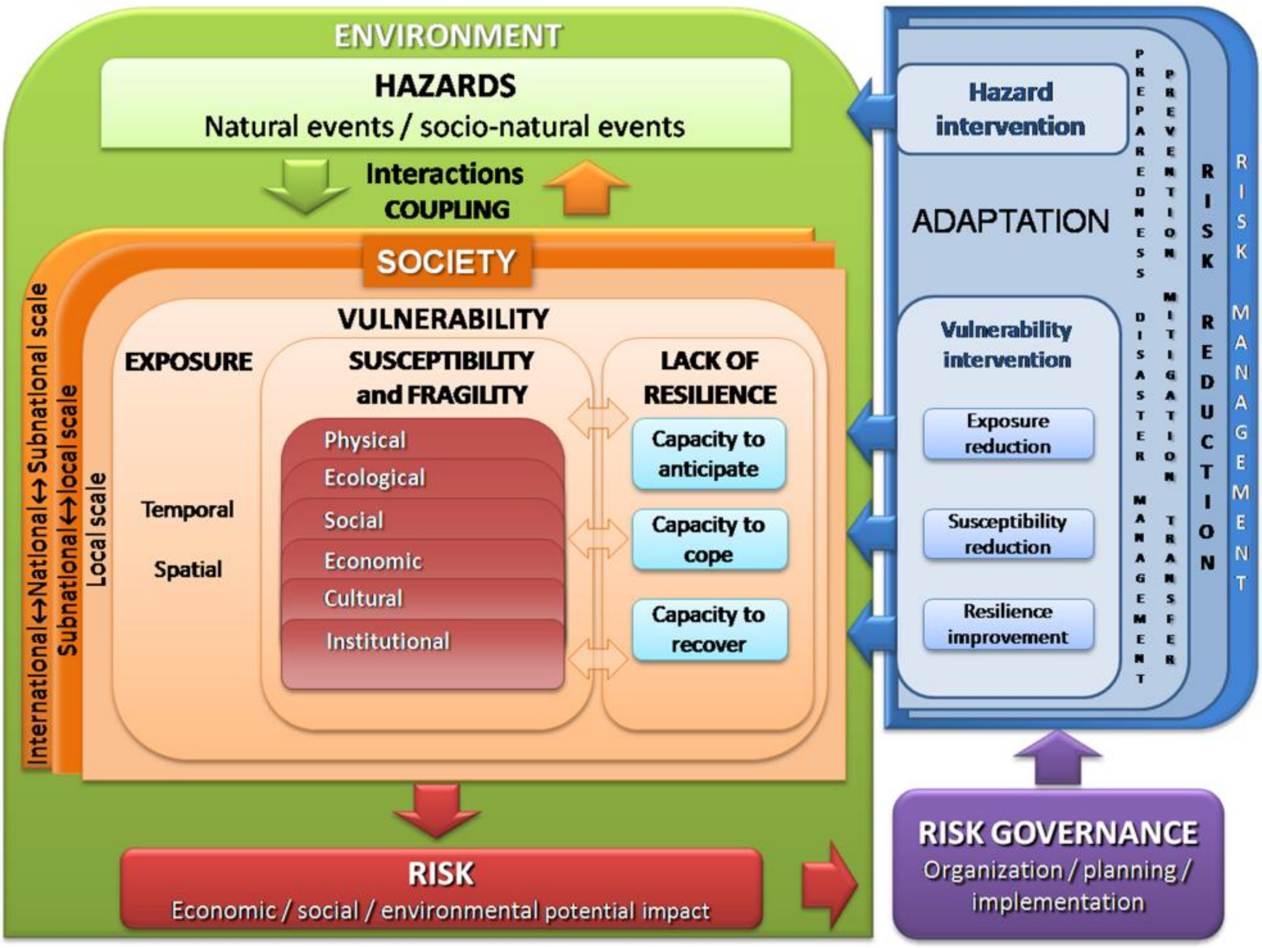


Figure 2.1. The MOVE Generic Framework.

2.1.2 Heat waves and social vulnerability

A heat wave is considered to be a continued and intensive period of heat stress, which may be accompanied by high humidity, and that directly affects human health (Koppe et al., 2004). Various morbidity and mortality impacts on the human population have been assessed for past(Brown and Walker, 2008; Brücker, 2005; Ebi, 2008; Fouillet et al., 2006; Semenza et al., 1996; Wilhelmi and Hayden, 2010). It is estimated that during the heat wave which affected Europe in 2003, more than 70,000 people died (Robine et al., 2008). Similarly, the 2010 heat wave that affected Russia claimed the lives of ca. 55,000 people (CRED, EM-DAT, 2012). Also, extreme heat events are expected to increase in number, length and intensity in the future (IPCC, 2012a).

Koppe et al. (2004) note that “skin eruptions, heat fatigue, heat cramps, heat syncope, heat exhaustion and heat stroke are traditionally considered as heat related illnesses”. Those most likely to die or be affected by heat are the elderly people, the chronically ill and the isolated (Kosatsky, 2005). In Chicago, people older than 65 accounted for 72% of the heat-related deaths due to the mid-July 1995 heat wave (Whitman et al., 1997) and in the 1999 Chicago heat wave, the strongest risk factor for heat-related death was living in isolation (Naughton et al., 2002). In the elderly people, physiological responses to changes in the environment are less acute and medications may interact with thermoregulation and risk perception, further increasing their vulnerability to heat (Oudin Åström et al., 2011). Fouillet et al. (2006) found excess mortality during the 2003 heat wave in France to be higher for people living at home and in retirement institutions than for those in hospitals, and that the mortality of widowed, single and divorced individuals was greater than that of married people. In a review article, Bouchama et al. (2007) found that being confined to bed, not leaving home on a daily basis, and being unable to care for oneself were associated with the highest risk of death during heat waves. Additionally, pre-existing psychiatric illness tripled the risk of death, followed by cardiovascular and pulmonary illness, while using home air-conditioning, visiting cool environments, and increasing social contact were strongly associated with reduced mortality (Bouchama, 2007). A further increase in losses in central European regions was due to a higher vulnerability of the population as this was located where hot spells are relatively infrequent (Kovats and Ebi, 2006). The presence of strong social networks may increase the resilience of the system when the hazard strikes. Although, direct positive relationship between tight social networks and resilience for elderly people exposed to heat waves in two UK cities was not found (Wolf et al., 2010), Yardley et al. (2011), reviewing both environmental and socio-economic factors that may determine the health and mortality impacts of heat waves, found in several studies that lack of social contact is a major risk factor, as people might only become aware of their condition and seek help when it is already too late

(Klinenberg, 2002; Yardley et al., 2011). Ethnicity is also considered in studies, especially in the US where non-white[9] people had a higher death rate (sometime double) than white people. This was linked to poverty rates, showing that socio-economic factors (i.e. income, education), more than ethnicity, play an important role in heat related mortality (Yardley et al., 2011). The researchers concluded that a socio-ecological approach, able to take into account the multiple factors that play a role in heat mortality risk and the different local circumstances, is required.

2.1.3 Heat waves and ecosystem services

Mortality risk increases by between 0.2-5.5% for every 1°C rise in temperature above a location-specific threshold (EEA, 2008), though it is unlikely to be a truly linear trend. Therefore, zones of the city where the UHI effect is stronger are those where the risk of illness or death during a heat wave is generally higher.

Green cover and trees in streets make important contributions to the improvement of urban climate, especially during summer months and periods of heat stress (Bernatzky, 1982; Bowler et al., 2010; Dimoudi and Nikolopoulou, 2003; Hamada and Ohta, 2010; Katayama et al., 1993; Oliveira et al., 2011; Shashua-Bar and Hoffman, 2000). Various studies have analysed how vegetation influences the thermal microclimate of urban areas and mitigates the UHI effect. Depietri et al. (2011) reviewed some of these studies, reporting data taken from measurements in different cities and showing how the cooling potential of green areas, while being considerable, varies from one urban area to another depending on the local conditions. On average, an urban green area is 1-2 °C cooler if compared with a non-vegetated zone (Bowler et al., 2010; Taha, 1997) and a 0.8°C reduction in ambient air temperature should follow a 10% increase in the ratio green/built up area in a city (Dimoudi and Nikolopoulou, 2003). Studies also stress the importance of placing vegetation (e.g. street trees) within the urban fabric and consider visits to green areas as a good coping strategy in case of heat stress (Depietri et al., 2011). Vegetation structure for improving the microclimate is a significant factor: maximum benefits seem to be obtained by planting two or three rows of trees with a relatively high density and adequate ventilation (Jim and Chen, 2008; Scott et al., 1998) .

Higher concentrations of air pollutants during heat waves can lead to an increase of excess death (Filleul et al., 2006; Fischer et al., 2004; Rainham, 2003; Vautard et al., 2007). There is increasing evidence for a synergic effect on mortality between high temperatures and ozone (O_3) concentrations

[9] "White" is one of the race categories used in the US census, which includes both racial and national-origin groups.

(WHO, 2007). Similar, but less pronounced differences, have been found for other pollutants such as particulate matter less than 10 µm (PM_{10}), black smoke, nitrogen dioxide (NO_2), and sulphur dioxide (SO_2) (WHO, 2007). The mortality increase due to the combined effect of heat and air pollution can be reduced by decreasing exposure to O_3 and PM_{10} on hot days (EEA, 2008). Interestingly, ground level O_3 and PM_{10} are the pollutants whose concentrations declined the least in Europe between 1990 and 2009 and that directly affect human (EEA, 2011). European ecosystems are also most affected by substances that cause acidification, eutrophication and vegetation damage (i.e. resulting from O_3 exposure) (EEA, 2011).

As reviewed in Depietri et al. (2011), urban trees can improve air quality in different ways such as by intercepting atmospheric particles and absorbing various gaseous pollutants. Additional information is provided by Nowak et al. (Nowak et al., 2006) who estimate, for instance, that the total annual air pollution removal by US urban trees amounts to 711,000 metric tons[10] ($3.8 billion value). To contribute to better air quality in cities it is important to plant appropriately selected tree species which are more tolerant to air pollutants and more effective in their removal.

On the other hand, heat waves may have a series of impacts on cities' ecosystems and services that would amplify the vulnerability of the urban population. For instance, sources of water supply (i.e. surface and groundwater) for agriculture, drinking water, water treatment plants and cooling of hydropower plants may experience shortages or may fail due to a sudden increase in demand (McGregor et al., 2007). Food production and distribution, as well as forestry, may be affected when peri-urban agricultural land sees its productivity diminished. This in turn can have implications for employment in the rural sector (McGregor et al., 2007). More broadly, the well-being of the urban population may diminish when recreational services are affected by heat stress: vegetation in parks may be damaged by heat and the higher concentration of air pollutants; the use of small watercourses, ponds and lakes may be interrupted when high levels of eutrophication leads to algal blooms and hypoxia. At the wildland-urban interface, the risk of forest fires may also increase. Some of these impacts have been described in detail in Depietri et al.(2011). However, most of the information available from the literature concerns the role of ecosystems in mitigating the impacts of the hazard, and much less refers to indirect impacts, namely those that, affecting the urban and peri-urban ecosystems, could increase the magnitude of the impact on the human population.

[10] The pollutants considered in the study are: carbon monoxide (CO), NO_2, O_3, PM_{10} and SO_2.

2.1.4 Measuring landscape functions

Flows of ecosystem services remain poorly characterized at local-to-regional scales mainly because there is no direct relation between land cover and functionality of ecosystems (Chan et al., 2006; de Groot et al., 2010). Verburg et al. (2009) state that, "a proper representation of land function will always require additional data beyond land cover observations": information on the spatial distribution of landscape functions generally needs additional intensive field observations or cartographic work (de Groot et al., 2010). Also, according to Burkhard et al. (2010), using quantitative and qualitative assessment data in combination with Land Use (LU) and Land Cover (LC) information originating from remote sensing and GIS, the state of ecosystem services can be evaluated.

Some studies have assessed the state of ecosystem services through land use and land cover features at different scales, from the global to the national/regional scales and to the local(e.g. Egoh et al., 2008; La Rosa and Privitera, 2013; Metzger et al., 2006; Naidoo et al., 2008; Raymond et al., 2009; Reyers et al., 2009). Most of them make use of additional quantitative biophysical information to build proxies for landscape functions for a wide range of services at different scales. The advantage of this approach is that, besides providing an estimate of the ecosystem services considered, it conveys information which has a spatial component and facilitates the presentation of the results to policy and decision makers. In this study, we contribute to this field focusing on the local level to assess the potential use of land cover and land use as a proxy for regulating services.

2.2 Methods

2.2.1 Case study: the Cologne urban area

The case study is the Cologne urban area (50°57'N and 6°58'E), situated within a floodplain along the river Rhine, in central-western Germany and in the Federal State of North Rhine Westphalia (NRW). Cologne is the fourth largest city in Germany and the largest one both in NRW and within the Rhine-Ruhr Metropolitan Area, with around 1 million inhabitants. It is also considered to be the warmest city in the country with a sub-Atlantic climate with traits of mild oceanic to mild continental climate due to the surrounding relief and its position in the landscape (Sabovljević and Sabovljević, 2009). On a daily basis, the winds that enter the city at night prior to the onset of the Rhine Valley Wind (from 01:00 CET), namely the country breeze and the down-slope wind, are insufficient to adequately ventilate the city centre (Kutter et al., 1998). Heat wave events have affected the region in the past and elevated temperatures have characterised the recent summers. The highest temperature

ever reached in NRW was recorded on the 12 August 2003 and was about 38.8 °C[11]. The health department of Cologne reported an increase of 16.5% of deaths in the month of August 2003 compared to the mean values in August for the previous three years (i.e. 775 deaths per month) (Koppe et al., 2003).

From a geographical point of view, the urban area includes industrial sites, inner harbours, historic city centres and residential areas with different vegetation portions and fallow land (Braun and Herold, 2004). In the environs of Cologne and along the river Rhine, the land use pattern is composed mainly of agricultural land with maize, sugar beet and forests as well as several artificial lakes (Braun and Herold, 2004). Hills to the northeast and southwest comprise mixed cultivations of both meadows and maize fields. Historically, the spatial structure of Cologne was significantly influenced by perpendicular alignments dating back to the Roman occupation of the area (Curdes, 1998). It developed radially from the centre around rings arranged in a semicircle along the Roman city wall. At the beginning of the 20th century, while the city was expanding outwards, certain spaces were protected to ensure environmental quality (Curdes, 1998). In fact, during the term of office of Konrad Adenauer (Cologne City Mayor between 1917–1933), the city's former fortifications were converted into a greenbelt. The initial plan of the outer greenbelt by the German urbanist Fritz Schumacher (who collaborated with Konrad Adenauer) was put in place in the early 1920s and was further developed throughout the century. The external green ring was planned to act as a buffer between the urbanized city and its peripheral industrial areas (Gritching, 2006). At present, the main green areas of the city are composed of an outer ring and inner ring connected radially by green axes. A plan to develop open green spaces along the River Rhine was initiated in 1978. Overall, Cologne has abundant natural land: some 230 km^2 covering 57% of the urban area (Siemens, 2012).

From a socio-economic point of view, the vast majority of the lowest wealth neighbourhoods are clustered in two parts of the city: the larger one is located east of the river Rhine and the second one is located in the north-western part of the city (Wolf, 2002). Compact suburbs with subsidized housing settlements and public transport systems for low-income groups were built during the baby boom in the 1960s after 60% of the city was destroyed during the Second World War (Carus, 2010). The city districts with the highest social status are more dispersed, though most of them are located at the border of the city (Wolf, 2002). In the surrounding 'green peripheral areas', wealthier families developed larger, individual plots during the 1980s (Carus, 2010).

[11]http://www.dwd.de/bvbw/generator/DWDWWW/Content/Oeffentlichkeit/KU/KUPK/Wetterrekorde/absolute__hoechsttemperaturen__brd__en,templateId=raw,property=publicationFile.pdf/absolute_hoechsttemperaturen_brd_en.pdf

2.2.2 Data Used

For the quantitative part of the study, four main datasets were used: socio-economic data regarding the population of the city of Cologne (i.e. statistical values), remote sensing data in the form of thermal infrared imagery, land use (LU) and land cover (LC) classification maps, and a map of the forest cover. All of the socio-economic data for each of the city districts were obtained from the Statistical Department of Cologne for the year 2001. Data about elderly people living alone per district were available only for the year 2005. The Environmental Department of Cologne provided the thermal imagery. The dataset was captured using a thermal infrared camera airborne at 3000 m altitude which delivered an image with a resolution of 7.5 m. The thermal scans were conducted on 30 June 1993 at 9 p.m. (reflecting the heat accumulated during the day; Figure 2.2) and on 1 July 1995 at 4 a.m. (reflecting night temperature; Figure 2.3). The LU and LC data were obtained from the Centre for Remote Sensing of Land Surfaces (ZFL) of the University of Bonn (Germany). Based on the Landsat TM satellite image of 2001, the LU and LC class information were categorised with a resolution of 30 m. The classes defined include both sealed (i.e. low, middle, high and other areas) and unsealed (i.e. grassland, coniferous forest, deciduous forest, mixed forest, agricultural land and water bodies) surfaces (Figure 2.4). The shape file of the Cologne urban forest measured in 2003 and provided by the Office of Landscaping and Green Spaces of Cologne was used. For the administrative subdivision of Cologne the 85 districts were used (see Figure 2.5).

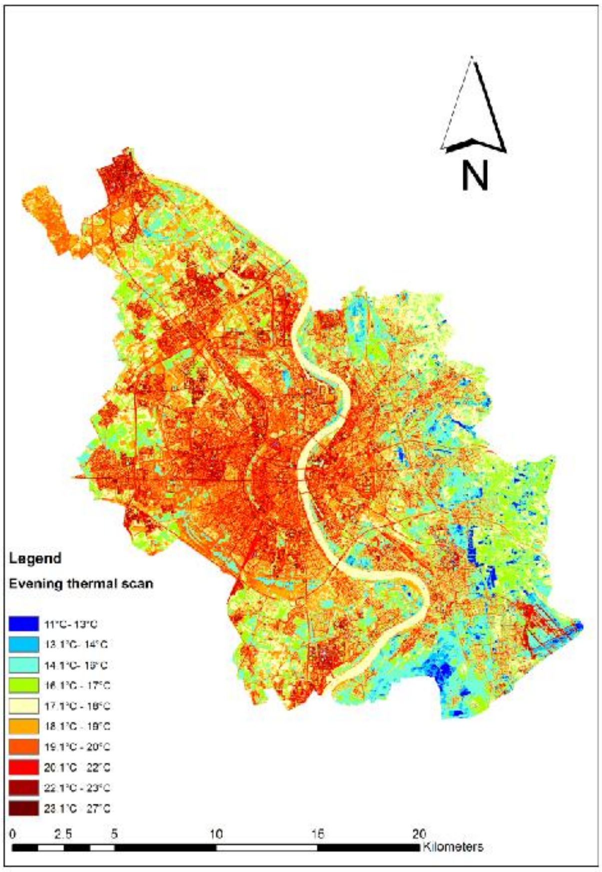


Figure 2.2. Evening thermal scan of Cologne (June 30th 1993 at 9 p.m.)
(Source: Environmental Department of Cologne)

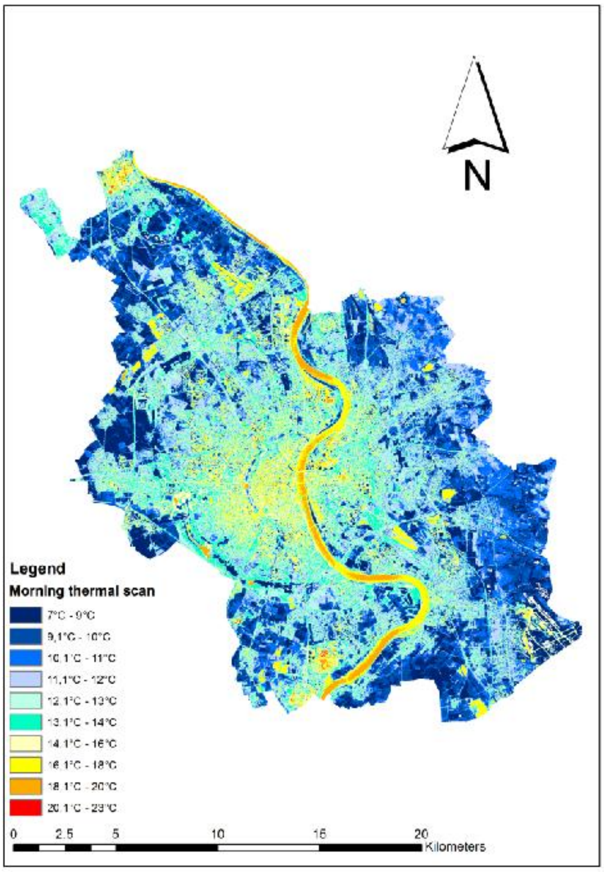


Figure 2.3. Morning thermal scan of Cologne (July 1st 1995 at 4 a.m.)
(Source: Environmental Department of Cologne)

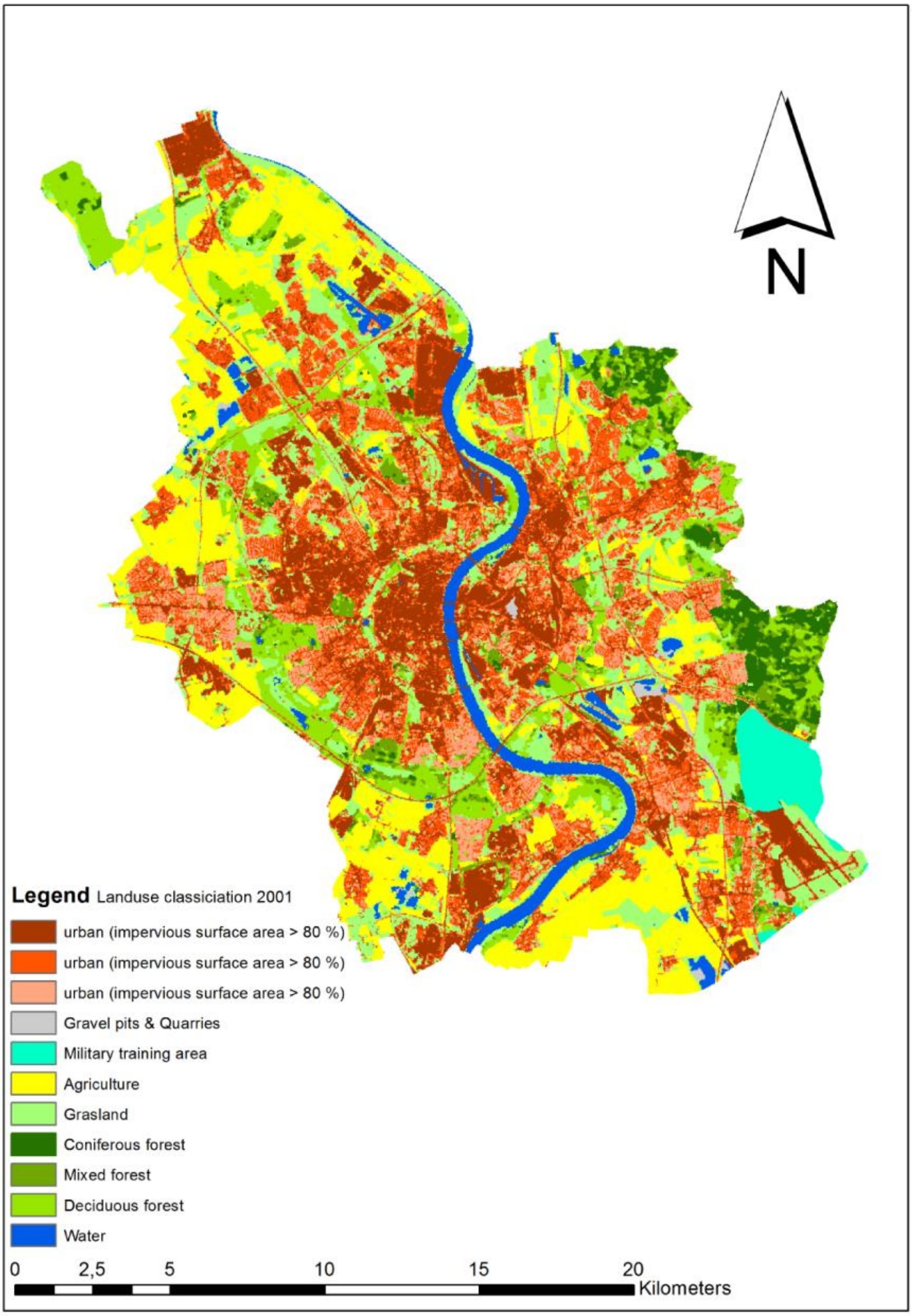


Figure 2.4. Land use and land cover map of Cologne.
(Source: Centre for Remote Sensing of Land Surfaces (ZFL), University of Bonn (Germany))

Figure 2.5. Districts of Cologne.

We considered as negligible the chronological mismatch between the datasets used. Based on the LU and LC maps for the years 1984, 2001 and 2005, also provided by ZFL, most of the major land use

changes in Cologne that took place between 1993 and 2005 involved the marked decrease in grassland and a symmetrical increase of urbanised areas characterised by low fractions of impervious surfaces (<40%) (ZFL, Univ. Bonn). The maximal Leaf Area Indexes (LAI) (providing information on evapotranspiration capacity) for the two CORINE classes, discontinuous urban fabric and natural grasslands, do not differ significantly (Knote et al., 2009). In addition, city areas occupied by high impervious surfaces and forests, which most influence the intensity of the UHI, have changed little during the same period. Therefore, the thermal images of 1993 and 1995 can be considered as representative of the environmental conditions in which the people of Cologne lived between 2001 and 2005.

Stakeholders' interviews were carried out between December 2011 and January 2012 to investigate the perception of relevant local authorities regarding the capacity of a range of ecosystems to mitigate the impacts of heat waves, and to gather their opinion on past or potential indirect impacts of heat waves on the urban and peri-urban ecosystem. Interviewees were identified at the city level amongst those institutions in charge or contributing to the planning and management of urban ecosystems and those responsible or active in the sector of human health at the city level. In particular, a list of 25 institutions and organisation working in a field relevant to our focus at the city level was drawn up. The list was discussed internally, benefiting from our previous experience of working on disaster risk in Cologne. We then contacted each one of the institutions via mail and/or phone to find out about their interest in participating in an interview. Some of them refused as they didn't consider their daily work to be strictly relevant for our study, but, nevertheless, some of these pointed out other local authorities that they thought would be more appropriate to be involved in our analysis. We finally ended up with a list of 7 institutions willing to be interviewed which covered exactly the set of dimensions we considered in our study (see Table 2.1).

Table 2.1. Characterisation of the stakeholders interviewed

Interviewee	Male (M) / Female (F)	Position	Type of institution	Sector
1	M	Project manager	Municipal	Environment
2	M	Head of department	Municipal	Landscape and urban green areas
3	M	Head of department	Municipal	Public health
4	F	Head of department	Municipal	Urban planning
5	M	Head of department	Municipal	Water management
6	M	Professor	University Institute	Public health
7	M	Project manager	Local branch of a national NGO	Forest management

The interviews were carried out based on a questionnaire composed of open questions. Following the above-mentioned aim of the qualitative assessment, the questionnaires were divided into two sections: the first set of questions focused on the perceptions of the interviewees with respect to the role of the Cologne urban and peri-urban green and blue areas in regulating city climate and air quality; while the second focused on past and potential impacts of heat waves on the local ecosystem and its services (e.g., water supply, recreational activities and peri-urban agriculture and gardening) and thus, indirectly on the urban population. Each interview lasted from 40 to 60 minutes, was recorded and then transcribed. The transcribed text was analysed making use of the Atlas.ti (ATLAS.ti Scientific Software Development GmbH, Berlin) software which supports qualitative data analysis. By allowing coding of the text and insertion of quotations, the software facilitates the cross comparison of the information contained in the transcripts and the generation of knowledge, while avoiding reducing the complexity contained in the data.

2.2.3 Indicators selection and development

Based on the MOVE framework and the definitions presented in Sect. 1.1, an extensive literature review, stakeholders' involvement and data availability, relevant indicators were identified to characterise the three components of vulnerability (i.e. exposure, susceptibility and lack of resilience). The indicators used and the composite indicators developed were then assessed and spatially represented through the application of the GIS ArcMap 10 software (ESRI, Redlands, CA).

The exposure component (E) was defined in time and space as the social and material context, represented by persons, resources, infrastructure, goods, services and ecosystems that may be affected by the hazard. In this study, it was measured as the number of people per city district differently exposed to heat waves due to the UHI effect which aggravates the intensity of the hazard. For Cologne it was not possible to investigate the effect of the spatial distribution of air pollutants and the role of air purification capacity of urban green areas on the exposure: while data on emissions for road traffic, households and industries are abundant and spatially detailed, few measurement points were available for the concentration of pollutants in the air.

With the objective of exploring different methods and assessing the opportunity to use each one of them according to the data availability, exposure is calculated in two ways named respectively E_1 and E_2: E_1 is obtained by multiplying the number of inhabitants per city district (I) with the normalized mean surface temperature (normalized using the Min-Max Normalization method) per city district (T) derived from the thermal infrared satellite images (see Eq. (1)); while E_2 is obtained by

multiplying I with one minus the percentages of different LU/LC types per city district (L_n where n goes from 1 to 8 as listed in Table 2) weighted by specific coefficients (c_n) (see Table 2) which indicate the capacity of the LU/LC cover types to provide climate regulating services (Eq. (2)). Following the literature presented in Sect. 1.4, c_n were calculated as the average between the values presented in Burkhard et al. (2010), who assign, through expert judgment, coefficients from 0 to 5 to the capacity of CORINE land cover types to provide ecosystem services (0 = no relevant capacity, 1 = low relevant capacity, 2 = relevant capacity, 3 = medium relevant capacity, 4 = high relevant capacity and 5 = very high relevant capacity), and the values obtained through the stakeholder interviews of the local authorities (see Table 2). The CORINE land cover classes were homogenised and translated into the classes used by ZFL, Univ. Bonn.

$$E_1 = I \times T \tag{1}$$

$$E_2 = I \times (1 - \sum_{n=1}^{8} L_n \times c_n) \tag{2}$$

Table 2.2. Matrix of the coefficients (c_n) which estimate the capacities of different LU/LC types to provide climate regulation services, as derived in Burkhard et al. 2010 and local stakeholders interviews.

ES coefficients / LU/LC type		c_n		
n	**Name**	**Burkhard et al. 2010**	**SHs interviews**	**Average**
1	Continuous urban fabric	0.00	0.00	0.00
2	Discontinuous urban fabric	0.00	1.60	0.80
3	Agricultural land	2.00	3.10	2.55
4	Deciduous forest	5.00	4.40	4.70
5	Coniferous forest	5.00	4.00	4.50
6	Mixed forest	5.00	4.40	4.70
7	Natural grassland	2.00	3.70	2.85
8	Water bodies	2.00	4.30	3.15

Indicators measuring susceptibility translate the predisposition of a society (and ecosystems) to suffer harm resulting from the levels of fragilities of settlements, disadvantageous conditions and relative weaknesses. As shown in the literature reviewed in Sect. 1.2, which mainly reflects on the extreme heat events in Chicago of 1995 and in Europe of 2003 (thus on the features of vulnerability to heat waves in richer countries as relevant to the location of our case study), the age and health conditions

of the population, followed by socio-economic and socio-cultural factors, are the main drivers that shape susceptibility to heat waves. The elderly, the unemployed and the immigrant are considered to be the most susceptible groups to suffer harm in the case of extreme heat events. Based on these findings and on the discussions held during expert workshops, the following indicators were chosen as representative: the percentage of the population per city district older than 65 years (*El*); and the percentage of unemployed per city district (*U*) as a proxy for low income. Initially, the number of immigrants per city district was also considered as a proxy for low income and of disadvantageous conditions due to difficulties in understanding warning messages. However, it proved to be highly correlated with the number of unemployed per city district (r=0,979) and *U* was kept as it facilitates the comparison with relevant and related studies. *El* and *U* are both derived from census data on the population of Cologne and percentages are calculated with respect to the total population of each city district. The composite indicator of susceptibility (*S*) is obtained, according to Eq. (3), by normalising and by equally weighting the two single indicators.

$$S = 1/2 \times El + 1/2 \times U \qquad (3)$$

Lack of resilience (*LoR*) describes the limitations in access to and mobilization of resources of the social-ecological system and its incapacity to respond by absorbing the impact. This component of vulnerability includes the capacity to anticipate, cope and recover in the short term. In our assessment, it is calculated as a composite indicator of two single indicators. During a heat wave, the majority of deaths generally occurs amongst the elderly who live alone as they are less able to promptly recognize, seek help and be assisted in case of malaise (see literature presented in Sect. 1.2). It is therefore assumed that the percentage of elderly living alone per city district (El_a) is a proxy for the lack of coping capacity of the population. Second, the vicinity of a household to urban parks and forests encourages and facilitates visits. The most susceptible groups can gain relief in case of extreme heat, benefiting from the cooler microclimate and cleaner air. The percentage of the surface of Cologne covered by urban forest per city district (*Cf*) is used as a proxy, thus city districts with a low percentage of or no urban forest indicate a lack of coping capacity. The composite indicator for *LoR* is then calculated by normalising, giving equal weights, and aggregating these two indicators according to Eq. (4).

$$LoR = 1/2 \times El_a + 1/2 \times (1 - Cf) \qquad (4)$$

For all three components. equal weightings were given to provide, as a first step, a more generalised approach that can be applicable in a wide range of situations. On another occasion, the weightings can beallocated with local experts to tailor the analysis to specific conditions.

Finally, the vulnerability (V) of the city of Cologne to heat waves is calculated by normalising and aggregating the composite indicators of the three components through Eq. (5):

$$V = E \times [\ (S + LoR) / 2\ ] \tag{5}$$

This formulation takes into account that with no elements exposed, there would be no vulnerability.

For the spatial representation and mapping of the single and composite indicators shown in Figure 2.6 to Figure 2.19 (see next Section), the values obtained were grouped into five classes using the quantile method, which is a predefined function of the ArcGIS 10 software (ESRI, Redlands, CA). With this method, each class contains an equal number of features, thus all classes differ in their values ranges. To facilitate comparison, the qualitative labels "very high, high, medium, low, very low" are used in the legends.

2.3 Results

2.3.1 Vulnerability assessment to heat waves

In this section, we present and describe the results obtained through the spatial analysis for all single and composite indicators used for the calculation of the final map of vulnerability.

First, the spatial distribution of the UHI effect on exposure is calculated through two methods (E_1 and E_2) and presented in Figure 2.6 and Figure 2.7. Figure 2.6 is based on the thermal infrared images and shows the mean surface temperature per city district while Figure 2.7 is based on the capacity of different land cover types to cool the environment. The first map clearly shows that the stronger effect of the UHI is concentrated in the central districts, where building density is higher and it then diminishes departing from the city centre towards less densely populated and more green/rural areas. Figure 2.7, shows a less clear pattern in the UHI effect. While reflecting the fact that more densely built districts are affected by higher temperatures, the presence of forest areas and parks appears to counterbalance this effect and takes an excessively high weight in the equation. In fact, the almost central districts on the western side of Cologne, although crossed by the green belt, actually present high average surface temperatures possibly due to their vicinity to the dense centre and the still significantly elevated percentage of high sealed areas. On the other hand, in the southeast, the local and peripheral concentration of sealed surfaces does not seem to significantly affect the surface temperature (Figure 2.6). The discrepancies in the results between the two methods are shown in Figure 2.8 which compares Figure 2.6 and Figure 2.7. These differences appear to be significant throughout all the urban area, with relatively higher values of UHI measured through the thermal

scans compared with the land cover classification, the opposite being true for the more peripheral districts.

The one million inhabitants of Cologne are principally distributed in the western-central districts and along the river Rhine (see Figure 2.9). Multiplying separately the spatial values obtained with the two methods to calculate the UHI with the spatial distribution of the population we obtain respectively E1 (Figure 2.10) and E2 (Figure 2.11). Both show a higher degree of exposure for the districts at the city core which hosts most of the population and for which the UHI is higher. The differences between the results obtained with the two methods are also mitigated when calculating the composite indicators (Figure 2.12) but still remain high for highly populated districts such as Lindental and Sülz. Some additional considerations and quantitative data are needed to derive an adequate estimation of the UHI based on the land cover types. The vicinity to and the concentration of districts with high or low sealed surfaces need, for instance, to be integrated into the calculation.

Consequently, and given the reliability of the data input for the two methods, E_1 is used in the final calculation of vulnerability.

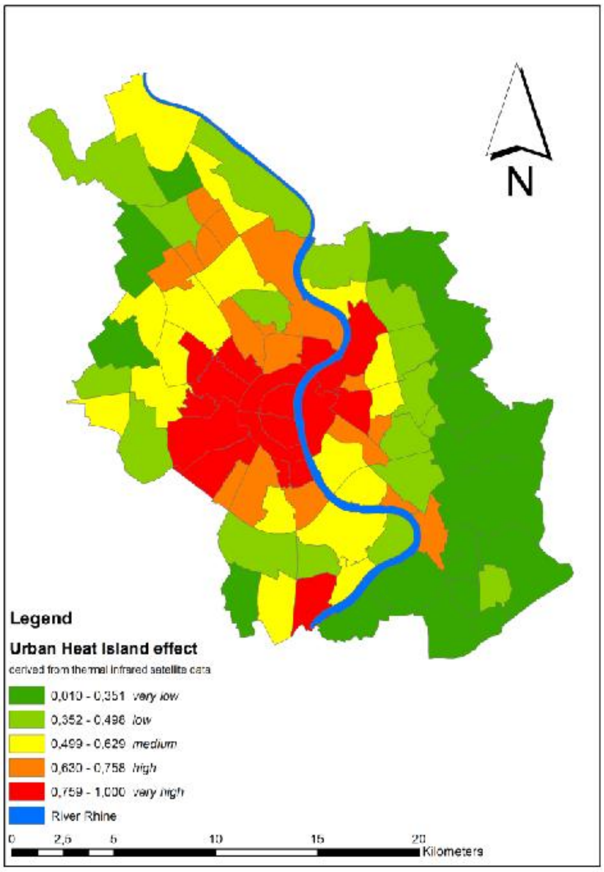


Figure 2.6. Degree to which each district is exposed to heat waves, based on mean surface temperatures derived from thermal infrared satellite data.

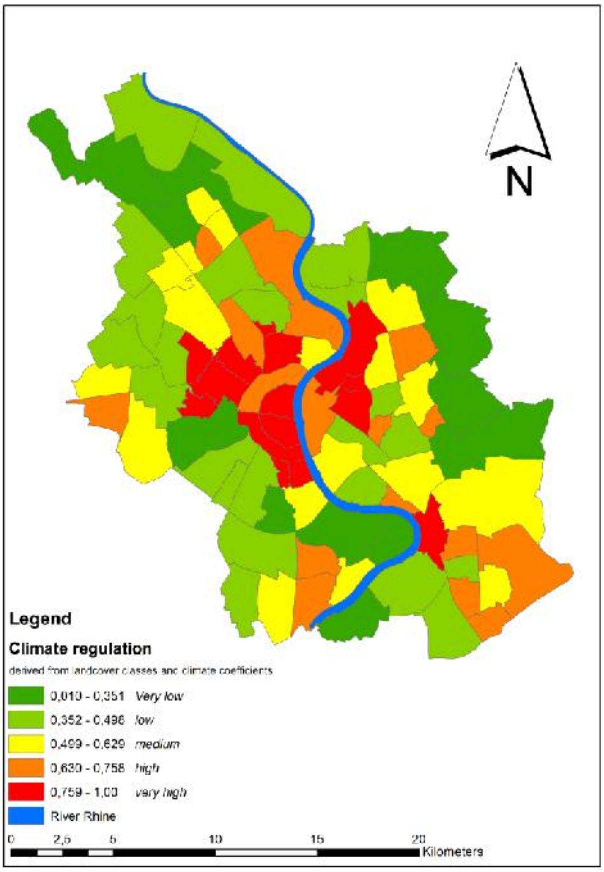


Figure 2.7. Degree to which Cologne districts are exposed to heat waves based on the capacity of different land covers to regulate the urban microclimate.

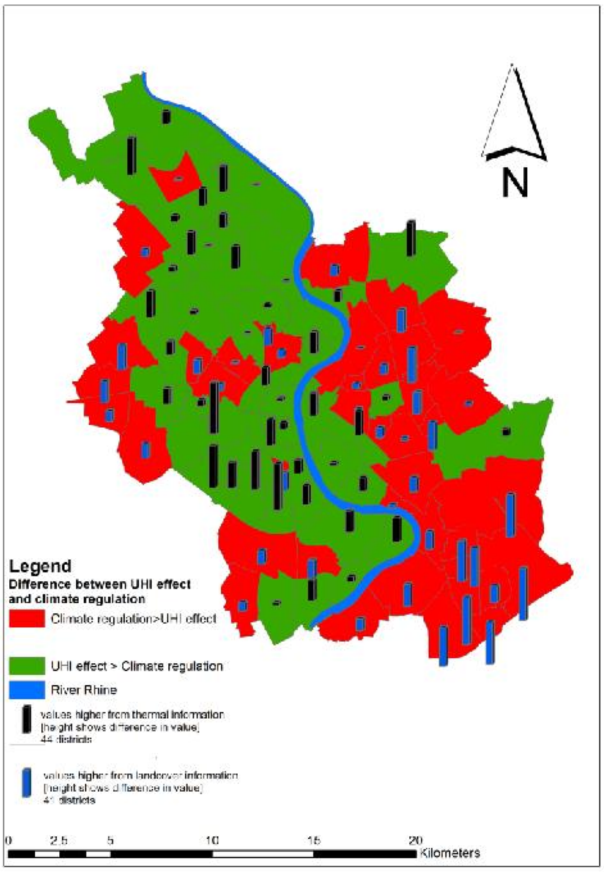


Figure 2.8. Difference between the UHI effect calculated through the mean surface temperatures per city district (Figure 2.6) and through the land cover capacity to cool the environment (Figure 2.7).

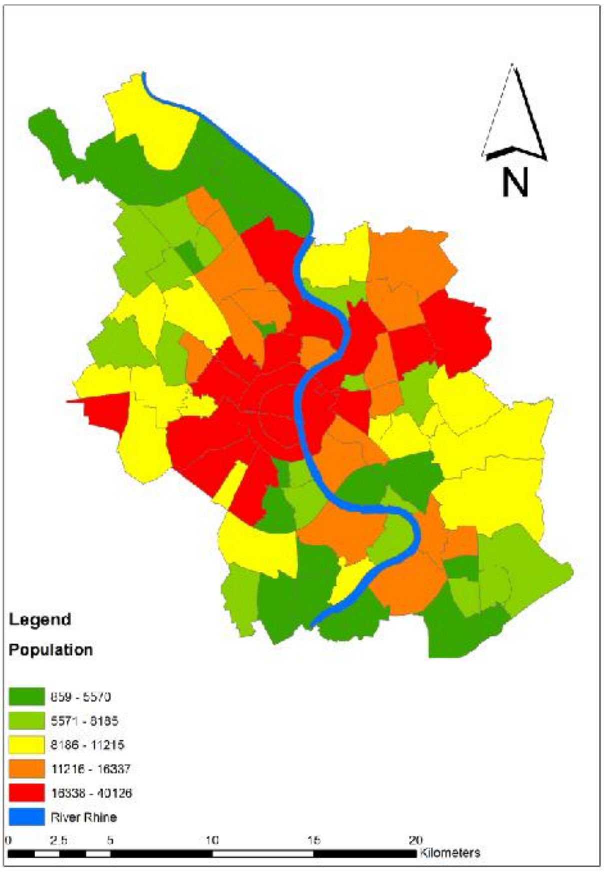
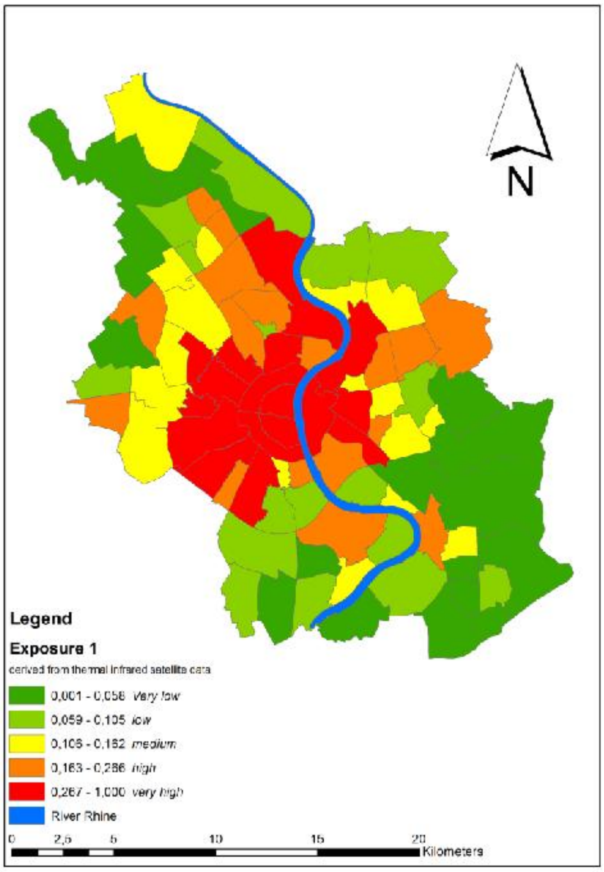


Figure 2.9. Population per city district.



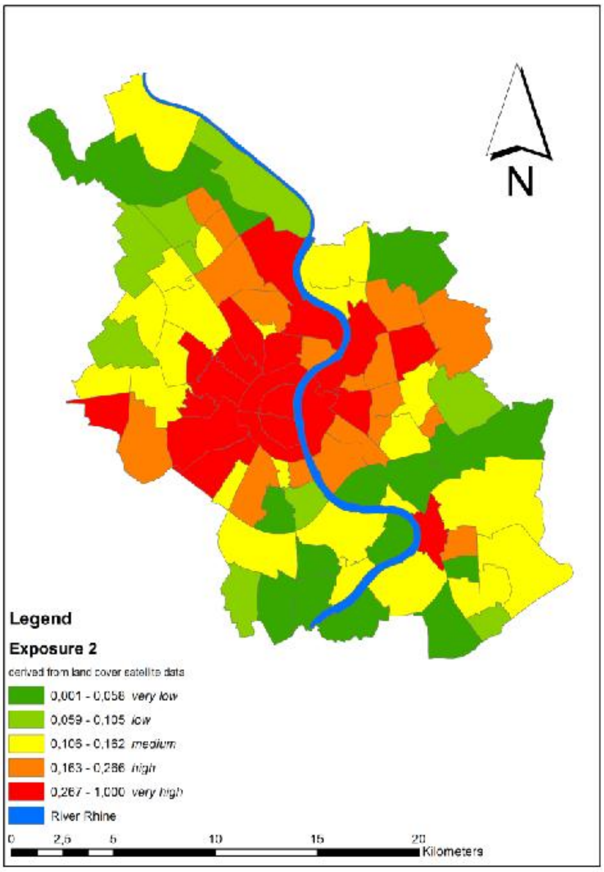


Figure 2.10. Exposure of the Cologne population to heat waves based on surface temperatures distribution (E_1).

Figure 2.11. Exposure of the Cologne population to heat waves based on the capacity of different land cover types to regulate microclimate (E_2)

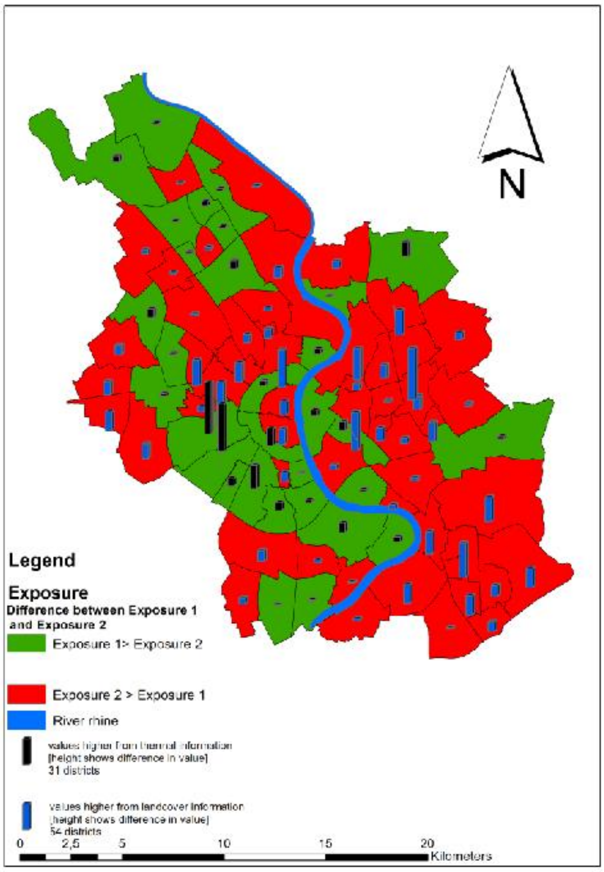


Figure 2.12. Difference between the exposure based on temperature distribution (E_1) and exposure based on the capacity of different land covers to regulate microclimate (E_2).

For the calculation of susceptibility, two indicators, elderly per city district and unemployed, were measured and spatially represented. The elderly are sparsely distributed in often isolated districts, all around the city centre (Figure 2.13). The unemployed are mainly concentrated in two zones of the city: one towards the north-west around Ossendorf and one close to centre but on the eastern side of the river Rhine (Figure 2.14), as also described in Wolf [70]. As a result of the combination of these two indicators, Cologne presents hotspots of susceptibility in districts situated all around the city centre (e.g Zollstock and Raderthal, Bocklemünd/Mengenich and Vogelsang, Rodenkirchen, Longerich, Flittard) with a concentration of susceptible areas on the eastern side of the river Rhine, around Ostheim (Figure 2.15). The most exposed central districts are thus not densely inhabited by the most susceptible groups.

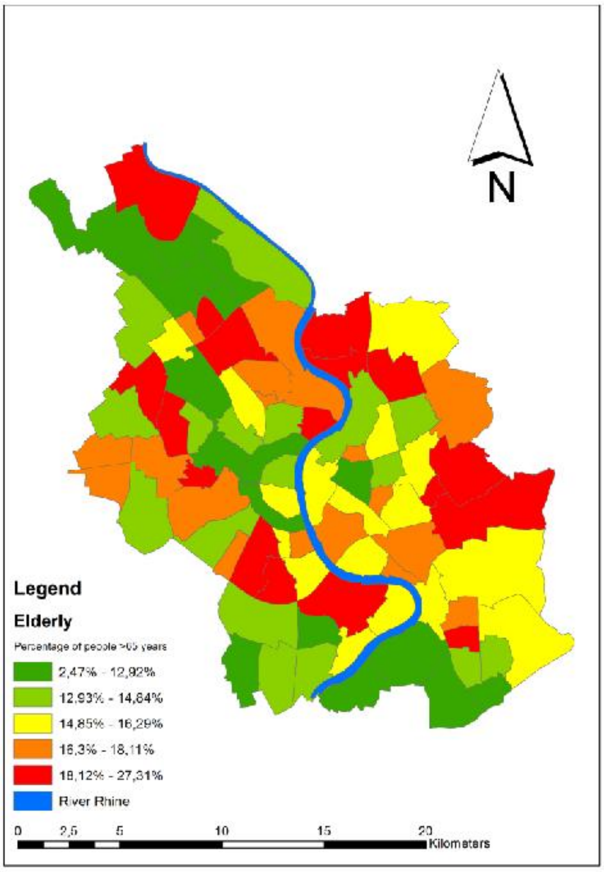


Figure 2.13. Percentage of elderly (older than 65 years) per city district.

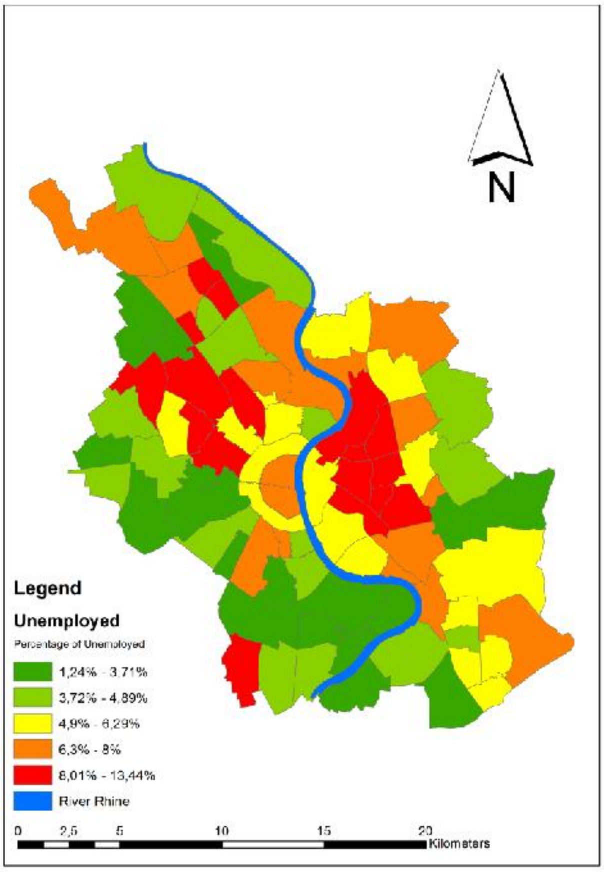


Figure 2.14. Percentage of unemployed per city district.

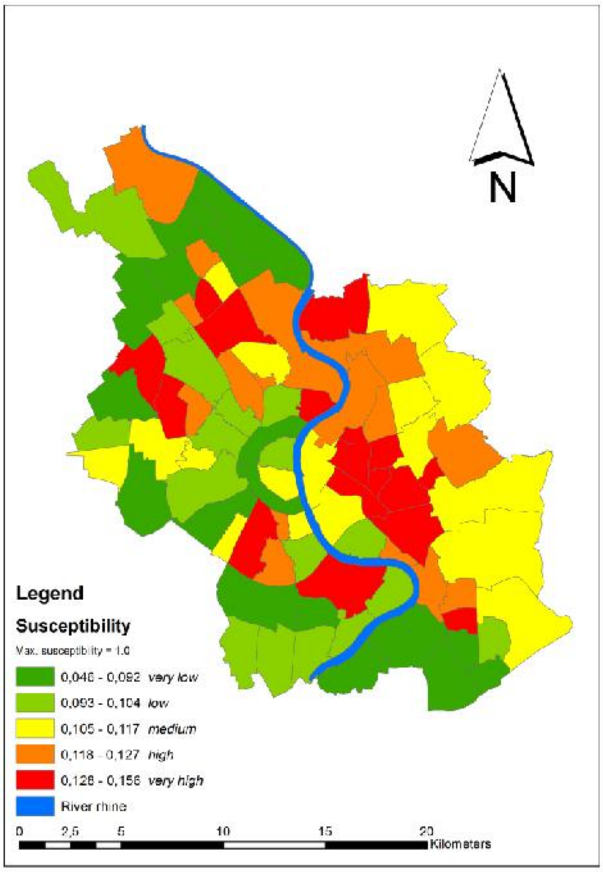


Figure 2.15. Susceptibility of the population of Cologne to heat waves.

For the calculation of the lack of resilience we used two indicators: elderly people living alone and green cover per city district. Elderly people living alone, who form the group with least capacity to be promptly assisted in case the hazard becomes an event, are principally distributed in the central areas of the city on the western side of the Rhine (Figure 2.16). High percentages of elderly people living alone are also located north to the city centre, on both sides of the river. Green cover per city district links the lack of resilience to environmental components as it gives a measure of the capacity to cope with the event by having access to nearby cooler areasv. As mentioned, urban parks are in fact at least 1-2 °C cooler then surrounding built up areas (Bowler et al., 2010; Taha, 1997), thus this might not affect significantly the thermal condition of the surrounding areas (Depietri et al., 2011). Interestingly, some of the districts with very high percentages of elderly people living alone are crossed by the outer green ring of the Cologne urban forest which provides an opportunity to benefit from cooler places and a healthier environment (Figure 2.17). As a result, the lack of resilience is higher in very central districts of Cologne and with high values for some single districts such as

Lövenich in the west, Porz, Ensen and Libur in the south (Figure 2.18). The distribution of the Cologne forest seems to play an important role in this component.

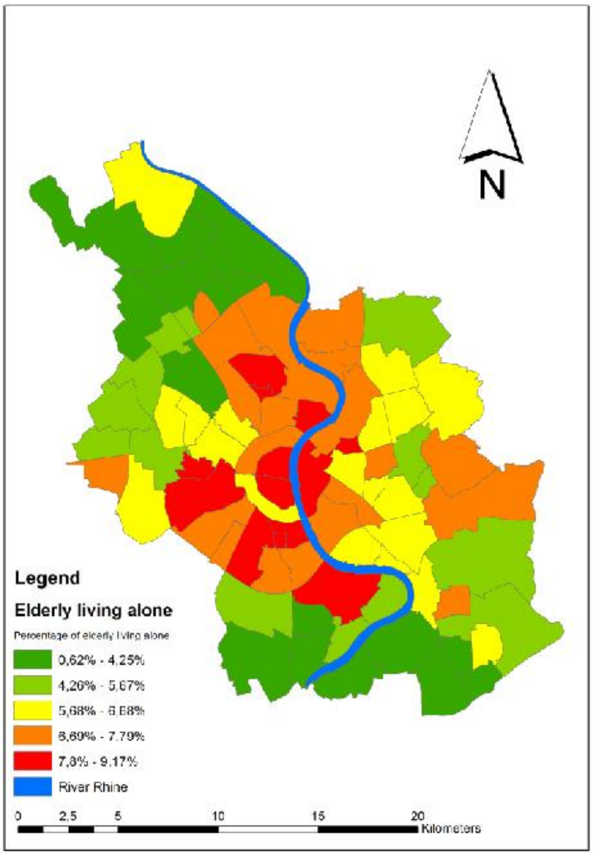


Figure 2.16. Percentage of elderly living alone per city district.

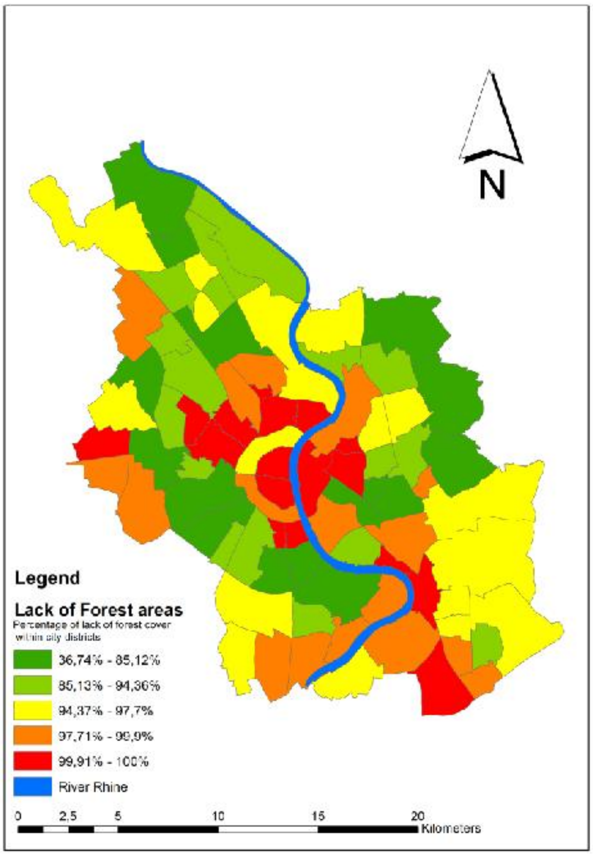


Figure 2.17. Percentage of forest cover per city district.

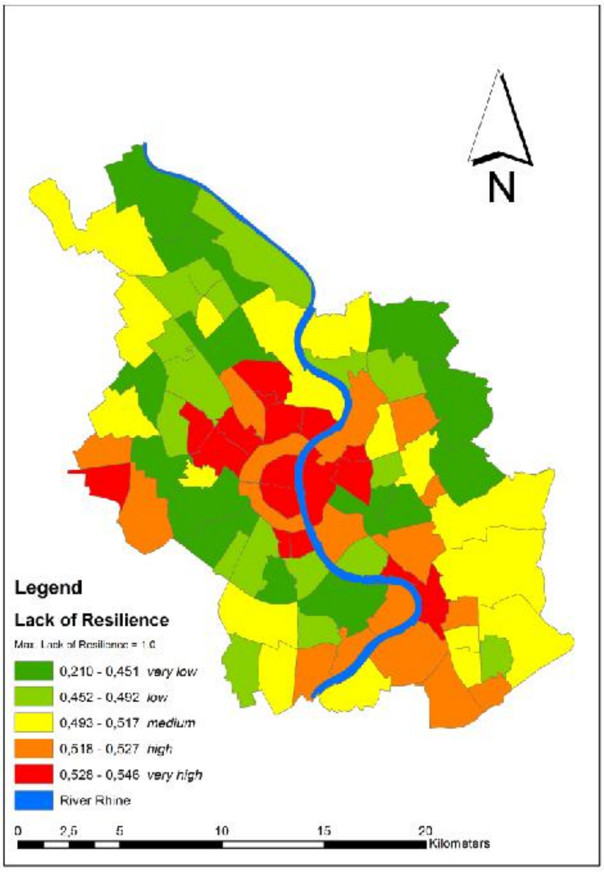


Figure 2.18. Spatial distribution of the lack of resilience of the population of Cologne to heat waves per city district.

As a result of the combination of the three composite indicators, the highest vulnerability of Cologne to heat waves affects the central districts on the western side of the river. In the assessment, exposure has a strong influences illustrated by the sensitivity analysis below. This explains the high degrees of vulnerability which affect also the wealthy districts crossed by the external green belt. The vulnerability map is presented in Figure 2.19.

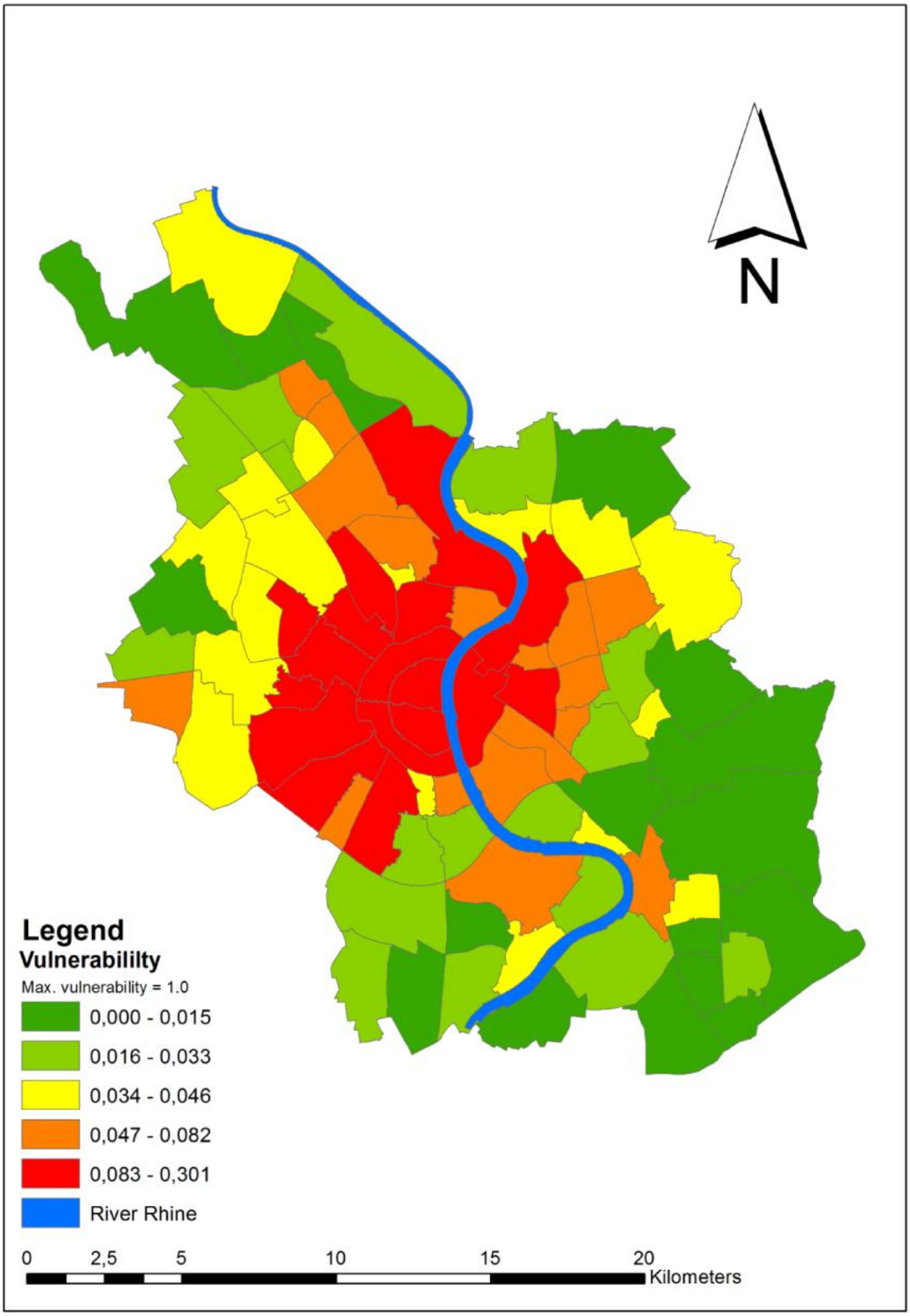


Figure 2.19. Map of the vulnerability of the population of Cologne to heat waves.

2.3.2 Validation of the results using a sensitivity analysis

A sensitivity analysis was performed with “IBM SPSS Statistics” (IBM, New York) in order to assess the impact of each indicator on the model output. This analysis examines the sources of variation in a model and can therefore be used to determine input variables largely contributing to the variation and those with low influence on the outputs (Saltelli, 2000). The results of the analysis are shown in Figure 2.20. The Figure consists of 3 parts (a, b and c). Figure 2.20a shows the effect of each indicator as a curve. On the x-axis are the original input data for each indicator scaled between -0.5 and +0.5, and the y-axis shows the variance of these indicators scaled between 0 and 1 in terms of overall response on the final index of vulnerability. The stronger the influence, the steeper is the curve. Figure 2.20b shows a boxplot with the different indicators on the x-axis and the sensitivity on the y-axis. On the x-axis, the indicator names are in the same order as they are ranked in Figure 2.20a. The size of the box explains the degree of dispersion of the value of each indicator in influencing the index. The smaller the box, the more distinct is the influence on the index. The bold line in each box describes the median, whereas high values on the y-axis explain the strength of the influence of each indicator to the overall index. Figure 2.20c shows the influence and interaction of each indicator with the other in the case of changes of one indicator. This could lead to the following effects e.g. the total sensitivity index of one indicator would be y=0 meaning that this indicator has no influence on the model output and thus could be neglected whereas a high median represents a non-substitutable and meaningful indicator.

Overall Figure 2.20 shows that the exposure has the highest impact on the model output, followed by the percentage of forest cover as a measure of the coping capacity of the population.

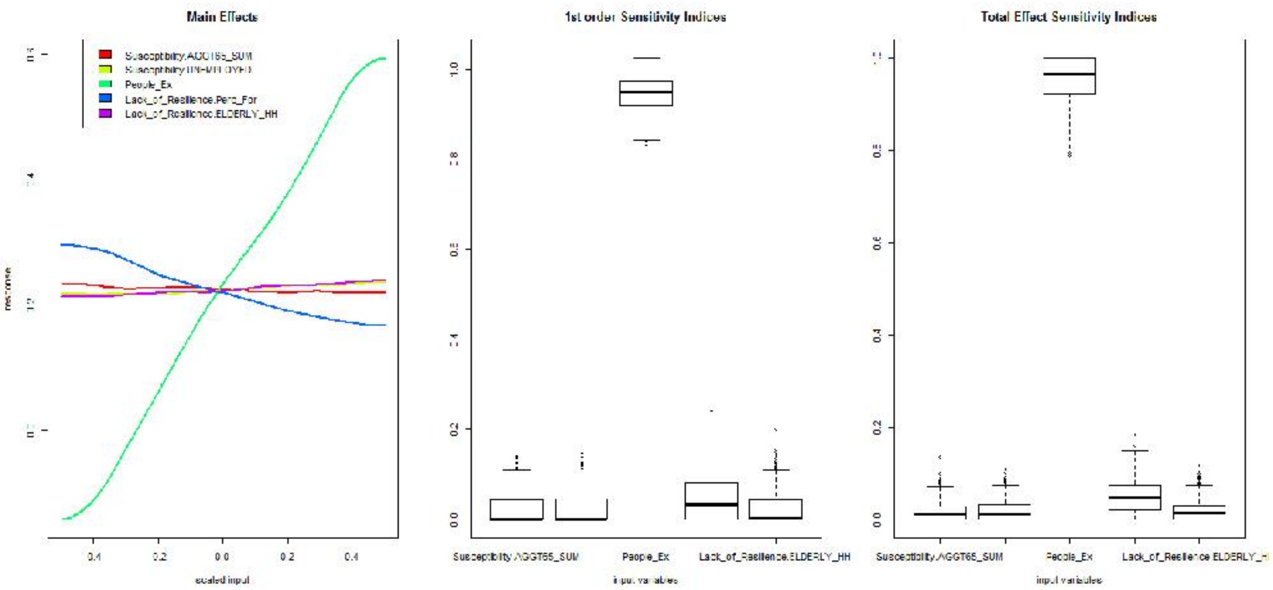


Figure 2.20 a, b and c. Sensitivity analysis

2.3.3 Experts' interviews

The results of the stakeholders' interviews are summarized in Table 2.3 and Table 2.4. The first table presents the gathered opinions on the role urban and peri-urban ecosystems may have in mitigating the impacts of heat waves in Cologne. Most of the respondents agreed on the need to conserve and have well managed green and blue areas in the city as these can contribute to mitigating the impacts of extreme heat. In particular, this is due to the cooling functions performed by the urban vegetation of Cologne. A mix of indigenous species, resistant to droughts but also presenting adequate levels of evapotranspiration is thought to be suitable for this purpose. A smaller emphasis was put on the effectiveness of green areas to remove pollutants and thus reduce the impact on health of extreme heat. To mitigate this impact a reduction in emissions is primarily necessary. It is recommended that the full range of services provided by urban ecosystems is taken into account in planning and management. Decisions on the allocation of green areas and street trees should not be based only on their aesthetic value. An additional expectation and suggestion made was to have ecosystems as much connected as possible through the urban fabric to form corridors that would bring fresh air into the city center from the surrounding cooler areas. In this way, obstacles encountered in the process of reconversion of built-up areas into green and blue areas spaces can be partially overcome.

The second table, Table 2.4, presents the perceptions and previous experience of stakeholders regarding the impacts of heat waves on urban / peri-urban ecosystems and their services in and around Cologne which would indirectly further increase the vulnerability of the city. Negative impacts of

heat waves would mainly hit agricultural land and gardening, diminishing their productivity. In this regard, strategies to adapt to increasing warm conditions (such as the selection of more resistant crops) are already being taken by farmers. Recreational activities can also be affected by the hazard due to the deterioration of small water bodies and of the vegetation in urban parks and forests distributed in the surrounding areas. The risk of forest fire should be monitored, especially in periods of extreme heat. Local mixed forests have shown to be the most appropriate to cope with these impacts. All respondents agreed that the provision of drinking water is not affected at present by the impacts of extreme heat, thanks to the high quantity of groundwater available in the region.

Table 2.3. Stakeholders perceptions on the capacity of the Cologne ecosystem to regulate climate and air quality and thus mitigate the impacts of heat waves.

Ecosystem	Negative opinion	Positive opinion	What cannot/should not be done	What could/should be done
Green areas		Important for their cooling capacity, especially for the high evapotranspiration rates of plants. Broad woodlands are the most effective in this sense	To broaden urban green areas is thought to be unrealistic as it would imply the reconversion of buildings and streets over large areas and would be in contrast with the spreading concept of compact city	Appropriate management of existing green spaces and increase of the number of trees along streets
				Green roofing. To date there are no shared guidelines, only ad-hoc projects
				The cooling function of peri-urban grasslands and agricultural areas should also be taken into account
				Opening wind corridors (both green corridors and large streets) to build connections between green areas. This would bring cooler air into the city center from the surrounding woodand agricultural lands
			It is often problematic to reserve space for street trees as these compete for space mainly with parking for cars	Street tress should be considered in urban planning for their cooling function in addition to their aesthetic value. This is valid also for parks and urban forests
			While planting Mediterranean species has become a trend in the last years because they consume less water, these also evapotranspirate less reducing their cooling functions	Selected species should be able to cope with 2-3 °C increase in temperature. Indigenous beech forests and oak trees are considered to be the appropriate species to cope with the condition of projected climates
	Visits to parks are generally not considered as a relevant coping strategy in periods of heat waves			Parks should however be well distributed amongst the city to allow easy access of the most vulnerable (i.e. elderlies, poor, ill)
	Green areas have minor positive effects on the reduction of air pollutants. Urban parks and street trees are thought to be the less	Urban and peri-urban forests are the most effective in purifying the air	Trees positioned along streets with narrow spacing can form a dense canopy that traps pollutants paradoxically worsening air	A reduction of air pollutants at the source is needed. Zones at the center of the city

Ecosystem	Negative opinion	Positive opinion	What cannot/should not be done	What could/should be done
	effective in delivering this service and can themselves be affected by high concentrations of air contaminants in inner parts of the city		quality	restricted to traffic should be delineated
Blue areas	The cooling effect of water bodies is noticeable on a seasonal rather than on a daily basis. This constrains their effect in mitigating the impacts of heat waves in urban areas		Most of urban water bodies have been buried under streets and their conversion is complicated	Small water bodies should be restored throughout the city. Fountains should also be reactivated through the city

(Table 2.3 continued)

Table 2.4. Stakeholders' perceptions on the potential impacts of heat waves on the urban ecosystem and its services to the inhabitants of Cologne.

Ecosystem/system	Direct impacts	Indirect impacts	Potential/undertaken actions	Notes
Surrounding agricultural land / gardening	Negative impacts on crops growth and gardening	Decrease in local agricultural productivity	Farmers are already aware of these impacts and are starting to select and grow alternative, more resistant crops	Dry periods occurring in spring times preceding a heat wave are the main responsible for the reduction in productivity rather than the heat waves itself. It can become an even more important issue in the future due to climate warming
Sources of drinking water	Generally not relevant. In some cases, when the water table is low, the flow of water can change and get exposed to chemical compounds			It is mainly due to the high rate of infiltration in the region which allows the city to rely on sufficient groundwater resources. In future climates drinking water might also be affected due to an increase in surface runoff which leads to the decrease in groundwater recharge

Ecosystem/system	Direct impacts	Indirect impacts	Potential/undertaken actions	Notes
Small water bodies	The quality can be deteriorated due to eutrophication and consequent decrease in the concentration of oxygen. Some smaller ponds may even dry-up	On recreational activities around the city		
Water treatment plants	Not relevant		In case of low levels of the river Rhine due to dry spells the concentration of effluents should be kept under control	
Power plants	Not relevant. These are not much present along the river Rhine so there is no risk of increase in temperature of the cooling water			
Vegetation	Negative. These have also occurred in the past: trees lose their leaves, photosynthesis is reduced which leads to a reduction in the production of oxygen. Grasslands are the most affected while mixed forest and indigenous vegetation have shown to be the most resistant to droughts. There might also be a risk of forest fires, especially in the future with the increase of heat extremes.	On recreational activities in and around the city	A good mixture of native plants, avoiding monoculture or grassland, increases the resilience of the forest system to fires and should be preferred	The risk of forest fires also depends on the type of trees. Beech, oak, ash and lime trees are thought to be quite resistant while coniferous forests are considered to be more at risk

(Table 2.4 continued)

2.4 Discussion and conclusions

The assessment of the vulnerability of Cologne to heat waves presented in this paper is based on the MOVE project generic framework and integrates both quantitative and qualitative data as well as the social and ecological dimensions of vulnerability. This allowed consideration of a broader set of drivers and elements that come into play in determining morbidity and mortality from heat waves in the urban environment. In fact, though the vulnerability of Cologne to heat waves is expected to be low compared to that of other municipalities in NRW due to the relatively low percentage of elderly population (Lissner et al., 2012), it represents a relevant case to investigate the role of the extended set of variables that shape urban vulnerability to heat waves, particularly the environmental ones.

From the spatial quantitative analysis, vulnerability was higher overall in the central and western districts where most of the population resides and where the percentage of sealed surfaces is high and contributes to the UHI. This is further accentuated by the fact that most of the elderly people living alone live in these central districts. On the other hand, the measurement and representation of each single indicator allowed highlighting a different geography in which the most susceptible groups are sparsely distributed towards the periphery of Cologne and on the eastern side of the Rhine. The urban forest also plays a relevant role in our assessment. Distributed along circular green belts, it contributes to reducing the vulnerability of susceptible groups (mainly the elderly people) in some of the more peripheral areas.

These results are directly linked to the historical dimension of the vulnerability of Cologne. Its present distribution clearly appears to be the result of processes that occurred through the centuries but that culminated in the last century when planning decisions more strongly influenced the assessed patterns of vulnerability. Two of these urban developments should be underscored: the allocation of space for the greenbelts in and around the city at the beginning of the century, along which the wealthiest segments of the population are located; and the intense sub-urbanisation which took place during the reconstruction period following World War II and associated with the baby boom of the 60's. In these latter parts of the city, low income, more susceptible groups are still concentrated (see Section 2.1).

The results of the quantitative assessment, on the other hand, suggest that, to effectively tackle the vulnerability of Cologne to heat waves, local city authorities in charge of urban planning, environmental management and health should collaborate to implement strategies which improve the social-environmental conditions of the city centre due to the higher levels of exposure and lack of resilience, but also need to consider the fate of susceptible groups which are located in its surrounding areas. The development of social services should be prioritized to cope with heat waves in Cologne,

but, as shown in the sensitivity analysis, the environmental factors have a strong influence on the assessment and are integrated in exposure (i.e. the UHI) and in the lack of resilience (i.e. the distribution of the forest cover) in our analysis. This result indicates that the ecological dimension is an important factor and needs to be taken into account in tackling vulnerability to hydro-meteorological hazards.

The qualitative assessment provided additional and complementary information to the quantitative analysis especially regarding the links between ecosystem services from the surrounding areas and the vulnerability of Cologne to heat waves. It stressed how in Cologne it is necessary to acknowledge the cooling functions of urban trees and green areas in urban planning as well as those of grasslands and peri-urban agricultural land, and draw better links between the city core and its surrounding areas, as it is also suggested for Stuttgart (Kazmierczak and Carter, 2010) or Freiburg (Yamamoto, 2006). This consideration supports the more sustainable model of the compact city (Beatley and Manning, 1997). Well designed green corridors could improve microclimate in the inner parts of the city bringing fresh air from the outskirts and bettering the living condition of the high density city centre. Furthermore, even if the urban forest of Cologne covers a relatively high area compared to other German cities, the type of species planted should also be carefully thought through, preferring, a mix of indigenous species, according to our respondents. In this regard, a debate is ongoing as to whether climate change might favour invasive species, thus caution needs to be taken when selecting species for adequate green areas management.

Furthermore, the qualitative data gathered provided insights into additional sources of vulnerability that could originate by the failure of certain ecosystems to provide services to the urban population. It emerged that most of the impacts of extreme heat affect peri-urban ecosystems such as forests and small water bodies while compromising agricultural production and gardening. The benefits derived from recreational activities in these periods can thus be hindered, especially around the city, while water supply seems not to be at risk. This adds to the quantitative assessment and further prompts consideration of the wider urban/rural interface dynamics, moving the focus partly outside the urban core.

Within the set of actions taken from the local government to adapt to climate change plans and strategies are developed in collaboration with the regional or federal authorities and awareness is rising in Cologne with respect to the impact of weather-related hazards such as heat waves[12]. At the Federal level, guiding documents are prepared by the Ministry for Climate Protection, Environment,

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[12] http://www.stadt-koeln.de/3/umwelt/klima/klimawandel/07145/

Agriculture, Nature Conservation and Consumer Protection of the State of NRW (LANUV) which set the strategic framework for cities situated in the region to plan and adapt to climate change, including to the increase of heat related stress[13]. A broad list of measures is here suggested but some seem to be specifically relevant for the Cologne urban area. According to our research, a reduction of the exposure, especially through of green infrastructure and an increase of the connectedness between the city and its green and surrounding areas are urgent measures to be implemented. This overall calls for a broader collaboration between sectors in charge of health, environment as well of urban green and urban planning at the city level. This field based study therefore demonstrates the context specificity of the choice of strategies to cope or adapt to increasing frequency and impacts of heat waves and that resources should be allocated at the local level to conduct such studies to select and prioritize strategies.

Additional guidance and frameworks come from the European level. Several projects, such as the EuroHEAT project, contributed to the implementation of the EC Environment and Health Action Plan. The final report of this project concluded that, although coordination between institutions for timely and appropriate response actions and early warning systems should be priority actions of the health plan to mitigate vulnerability to heat waves, the reduction of exposure through improved urban planning should also be prioritized (WHO, 2007).

In summary, our analysis showed that, while the higher vulnerability of the population of Cologne to heat waves is concentrated in the city centre, policies that aim to tackling it should also take into account the connections and interactions between the city centre, the surrounding districts and its hinterland, reducing the susceptibility of lower status social groups and enhancing ecosystem management.

[13] http://www.umwelt.nrw.de/klima/klimawandel/anpassungspolitik/anpassungsstrategie/index.php

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CHAPTER 3 . The political ecology of peri-urban ecosystem services for city well-being: the case of Barcelona and the Collserola Natural Park

Abstract

Ecosystem services provided to the urban population by urban and peri-urban ecosystems can substantially contribute to the well-being and sustainability of cities. In particular, the preservation of peri-urban parks, such as is the case for the Collserola Natural Park (CNP) adjacent to Barcelona, might result to be vital in maintaining the quality of life of the densely populated urban area. We measured through proxies and indicators the provision of four city specific ecosystem services (namely: flood regulation, air quality regulation, urban cooling and recreation) for Barcelona by including those provided by the CNP, and proved that beyond mere conservation goals, the park constitutes a fundamental component of the livability of Barcelona. Through an analysis of land use changes and an extensive literature review we also showed that Collserola was almost entirely covered by vineyards and dedicated to other agricultural activities until the mid of the 19th century. Consequently, it is now a constructed forest through the actions of political forces and social movements, who claim to preserve it for biodiversity conservation against other opposing socio-economic pressures, such as urban expansion. Our analysis proved that ecosystem services are not simply an independent, objective reality, but that they are the result of environmental as well as socio-economic and political processes.

Keywords

Urban areas, ecosystem services, political ecology, well-being, vulnerability, nature conservation

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3.1 Introduction

In Europe, where three quarters of the population are already concentrated in urban areas and in a growing proportion (EEA, 2006), peri-urban green areas and the ecosystem services (ES) these provide are threatened. Although urban areas are not inherently bad for ecosystems (e.g. in terms of high levels of biodiversity due to high variety of human activities) (Niemelä, 1999), specific urban related processes, like sub-urbanization and high loads of traffic, put large pressures on the environment and on the well-being of populations in cities (MA, 2005). At the city core, air and water pollution and the Urban Heat Island (UHI) effect largely contribute to the degradation of the urban environment, while, in peri-urban areas, it is unplanned urban expansion (or urban sprawl) (EEA, 2006) which directly threatens green areas by fragmenting and isolating natural habitats (Niemelä et al., 2010). At the global scale, we can literally speak of appropriation of ecosystems by cities (Folke et al., 1997).

The state of local and regional ecosystems is particularly important for urban environmental quality, life quality and the sustainability of urban areas, and is increasingly the focus of research programs and urban planning (Chiesura, 2004; Givoni, 1991; Tzoulas et al., 2007; van Kamp et al., 2003). The healthiness and the livability of urban environments are, in fact, the main concern for human well-being in urban areas (MA, 2005). At the scale of the urban core, regulating services, such as water regulation, cooling and air purification, are particularly important and contribute to the enhancement of the health and security of urban populations. Recreational activities and immaterial human needs are also highly important for urban citizens and are often largely supplied by urban and peri-urban parks (Chiesura, 2004). The preservation of neighboring parks seems thus to be vital in maintaining life quality in densely populated urban areas (Bolund and Hunhammar, 1999; Muhamad et al., 2014; Trzyna, 2014).

Within the field of Urban Ecology, it is the branch of Urban Sustainability, as described in Wu (2014), which particularly focuses on ES and human well-being. We aim at contributing to this field through the assessment of ES provided to the inhabitants of Barcelona (Spain) by a peri-urban natural park (the Collserola Natural Park – CNP) and by estimating their contribution to human well-being. However, we adopt a political ecology theoretical framework to show how the supply of ES is ultimately the result of ecologic as well as socio-political processes, which are particularly evident in the urban contexts. It is in fact in urban areas, more than anywhere else, that environments are determined by economic investments, political tensions and decisions, cultural and discursive struggles (Swyngedouw and Heynen, 2003).

In the next sections of the introduction, we review the literature on ES in urban and peri-urban areas and that on urban political ecology studies. In the methods section we introduce the case study area and the methodologies used to: estimate the most relevant ES for the Barcelona urban area (three regulating and one recreational), in order to assess the role of the park in the livability of the city; measure the land use changes which have taken place since the second half of the 19th century, and then specifically in the second half of the 20th century, to assess the change in uses of the area now occupied by the park and document the abandonment of agricultural land and the reforestation process; identify the socio-economic and political drivers behind the changes of land cover in the case study area and detect, when possible, discourses and conflicts which led to the creation of the park, this through a qualitative historical analysis. We then present the results and we conclude with a discussion and conclusions section.

3.1.1 Ecosystem services in urban and peri-urban areas

Despite the apparent human domination of nature in urban areas, cities are constrained and depend on ecosystems. Urban ecosystems provide a large range of services that contribute to human sustenance and well-being such as food, water, fiber and fuel, water and air quality regulation, cooling, soil formation, as well as recreation and spiritual fulfillment (Bolund and Hunhammar, 1999; Gómez-Baggethun and Barton, 2013; Haase et al., 2014; Niemelä et al., 2010). Human ecosystems within urban boundaries are not fully functional and complete ecosystems and are biophysically supported by processes that take place regionally or elsewhere (MA, 2005). Strictly defined urban ecosystems (see for instance Pickett et al., 2001) contribute only marginally to satisfy the demand for environmental services of densely built up areas (Niemelä et al., 2010). For this reason, an increasing number of studies includes or suggests to look at urban hinterlands (i.e. “urban regions with at least 15% of their employed residents working in a certain urban core” (OECD, 2013)) in the study of urban ecosystems (Baró et al., 2014; Depietri et al., 2013; Haase et al., 2012; Larondelle et al., 2014). Urban form is also particularly important and urban shrinkage has been positively related with the supply of ES (Lauf et al., 2014). The intensity and duration of the interactions between the population and the neighboring ecosystems increase the awareness of the population of the wide range of services that ecosystems can provide (Barthel et al., 2010; Muhamad et al., 2014). Urban inhabitants generally acknowledge more the benefits of regulating services whereas the rural population appreciates more the provisioning services (Martín-López et al., 2012). A good connection with well-managed and preserved neighboring ecosystems might be key in guaranteeing and maintaining the well-being of urban populations.

3.1.2 Theory: the political ecology of ecosystem services in urban areas

Political ecologists argue that the environment we generally take for granted is actually socially constructed (Robbins, 2004). The constructed character of natural wilderness has been investigated by numerous authors (Crist, 2004; Demeritt, 2002; Escobar, 1999; Gerber, 1997; Goldman and Schurman, 2000; Robbins, 2004; Stedman, 2003; Woodgate and Redclift, 1998). It refers to the fact that environmental problems are not inevitable or so called "natural" but are the result of social-ecological and political processes that can be traced back and investigated (Robbins, 2004). Also, according to Swyngedouw and Heynen (2003), "natural" processes (or metabolism) "become discursively, politically and economically mobilized and socially appropriated to produce environments that embody and reflect positions of social power". Some authors specifically look at the political ecology of parks and at the socio-economic effects of biodiversity conservation (Adams and Hutton, 2007; Brown, 1998; West et al., 2006). Based on the idea of a pristine nature, this often entails the displacement of native populations in the developing world or the erosion of their livelihoods and welfare. Other authors note that protected areas are often located in the rural periphery of a country, whereas the driving force behind their definition comes usually from the more developed and urbanized core and can also be a source of conflicts (Mather, 1993).

The "production of nature" in the urban context has only recently been investigated by the field of political ecology and by a rather restricted group of studies which focus, for instance, on issues related to water and air (Zimmer, 2010) or on the uneven distribution of resources in urban landscapes (Ajibade and McBean, 2014; Pelling, 1999; Swyngedouw and Heynen, 2003). Byrne and Wolch (2009) look at ethno-racial relations of power within cities stressing how parks in urban areas are not ideologically neutral space nor physically homogeneous and are the result of a differential relation of power and regional racial formations. Other studies looked at the social, differential vulnerability to hazards in the urban context. Ajibade and McBean (2014), for instance, assess the links between poor housing conditions and vulnerability to floods in slum communities in Lagos (Nigeria) with a focus on housing rights. Pelling (1999) deals with the institutional and cultural norms behind the creation of flood risk in urban Guyana from a political ecology perspective finding out that vulnerability was linked to poverty, inadequacy of infrastructures, gender and ethnic issues and a poor civil society. Little work has been done on the political ecology of ES (Barnaud and Antona, 2014; Dempsey and Robertson, 2012) and even less so with applications in urban areas. According to Ernstson and Sörlin (2013), ES in urban areas are influenced by social practice, are part of a historical process and are ultimately, inherently political. Or, as Swyngedouw and Heynen (2003) put it, the social and physical

environment of the city "is the result of a historical geographical process of the urbanization of nature".

The present research is framed in the field of political ecology as it aims at tracing back the "invention" of the CNP and, as a consequence, of the ES it provides to the densely populated metropolitan area of Barcelona. To stress here is that, as we will see, the main reason behind the existence of the CNP is not the range of ES it provides to the city but merely nature conservation purposes. In this sense ES in the peri-urban area of Barcelona are involuntarily socially produced.

Starting by estimating the environmental benefits provided by the ecosystem of the park, we then show how the ES provided to the peri-urban area of Barcelona are fundamental for the well-being of the city population and then that they are the result of socio-economic and political processes. Ecosystem protection and ES, especially in urban areas, more than representing an objective biophysical entity, should in fact be seen as social practices of articulation of value (Ernstson and Sörlin, 2013, 2009), and thus socially and culturally embedded (Barnaud and Antona, 2014). These process are partly supported by urban planning but remain highly politicized especially due to the high interest of capital investment in and around cities (Ernstson, 2013). Ernstson et al. (2008) showed for instance that in the highly contested space of urban Stockholm, the protection of urban green areas depended mainly on the actions of an organized civil society.

More than highlighting injustices made in the name of pseudo, taken for granted truths, our analysis aims at showing how left wing governments and green urbanism first and conservationist and social movements later on, justifying their claims on the preservation of wilderness and biodiversity, have indirectly secured a vital source of ES to the population of Barcelona from an area that has been historically highly transformed by humans and can certainly not be considered a biodiversity hotspot. Despite this, we recognize that the supply of ES is the result of a coevolving process in which social constructivism needs to come into terms with local ecological processes, ultimately characteristic of the particular ecosystem considered.

3.2 Methods

3.2.1 Case study description

The case study area includes the municipality of Barcelona and the whole area of the CNP (Figure 3.1). To assess the role of the park as a source of ES for the inhabitants of the urban core, we first considered the dense urban area of the municipality of Barcelona minus the part of its surface that

falls within the boundaries of the park (scenario 1) (Figure 3.2), and then the park plus the municipality as a single unit (scenario 2) (Figure 3.3).

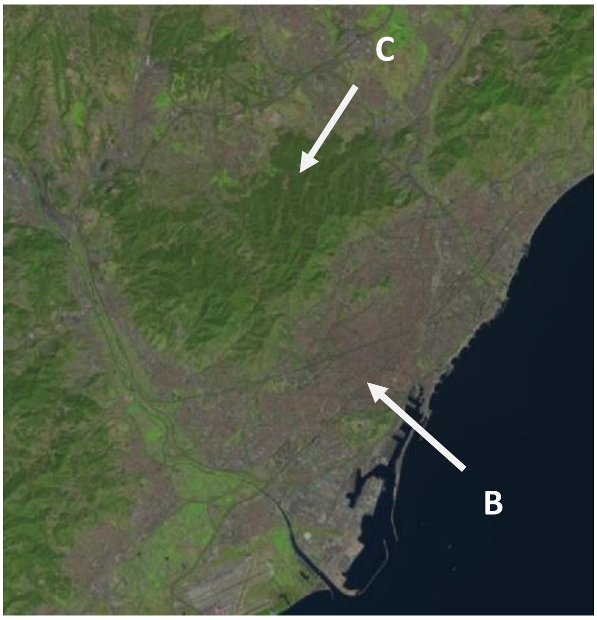


Figure 3.1. Lansat image of Barcelona city (B) and Collserola (C) areas.
(Source: LandsatLook, USGS)

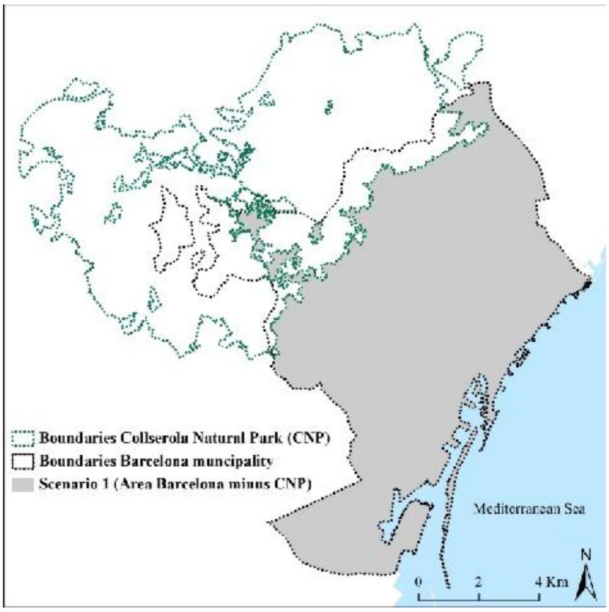


Figure 3.2. Administrative boundaries of the Municipality of Barcelona and the Collserola Park and in grey scenario 1.
(Source: own map)

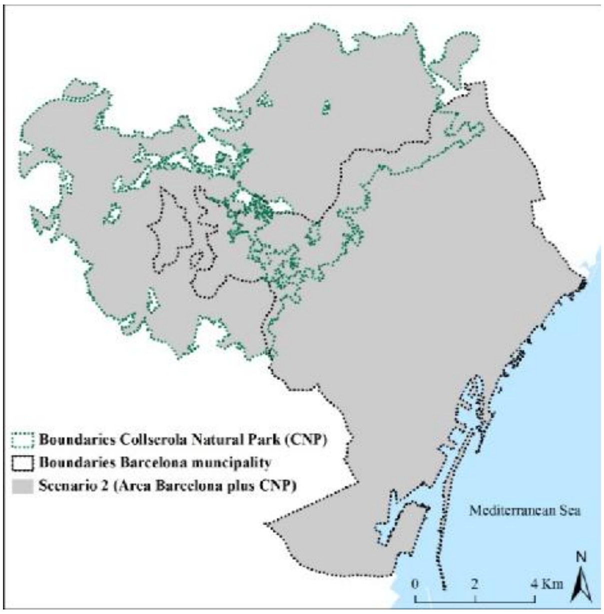
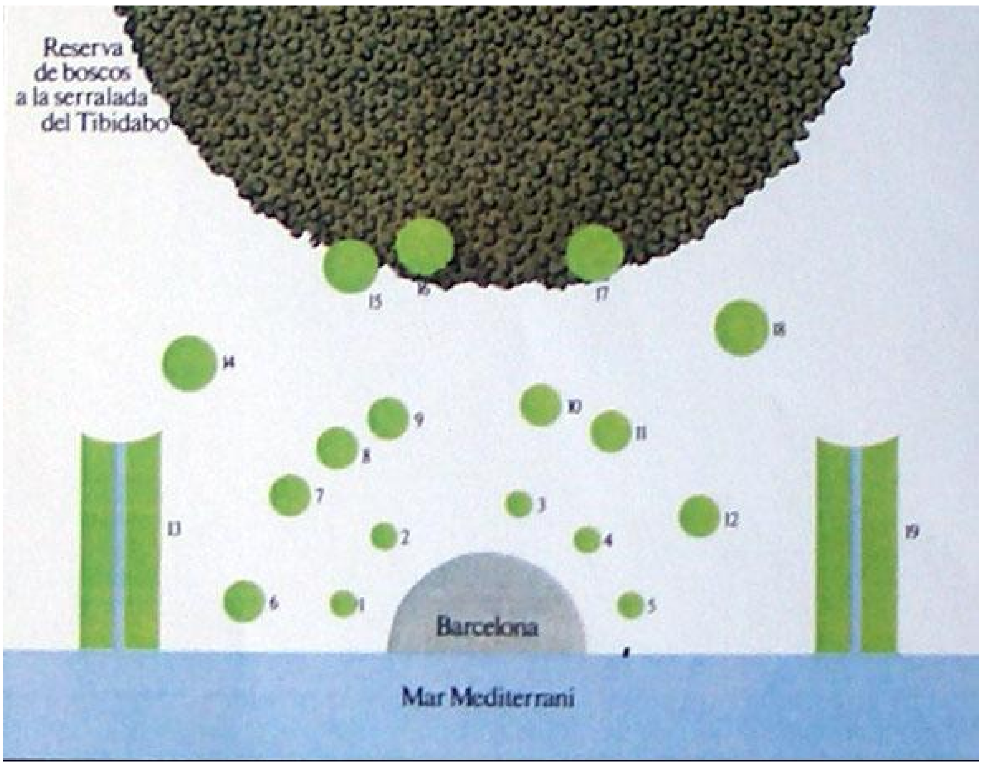


Figure 3.3. Administrative boundaries of the Municipality of Barcelona and the Collserola Park and in grey scenario 2.
(Source: own map)

The municipality of Barcelona (41°23′N, 2°11′E) is the capital city of the autonomous Spanish region of Catalonia. With a population of 1.61 million inhabitants (IDESCAT, 2011a) and a surface of 101.9

km^2, it is part of the largest Mediterranean metropolitan area which comprises about 4.77 million inhabitants (IDESCAT, 2011b). The municipality is bounded by the Llobregat river in the south and the Besòs river in the north, by the mountain range of Collserola on the north-west and the Mediterranean sea to the east. The average yearly temperature recorded between 1987 and 2010 was 18.2°C while most precipitations concentrate in September and October, with their annual average being 565 mm, also recorded between 1987 and 2010 (http://w1.bcn.cat/temps/en/climatologia/clima_barcelona; retrieved on 18th August 2014). The structure of the city, besides its old part which was enclosed within Medieval walls until the year 1854, is now dominated by the Cerdà Plan launched in the 1860s to expand the city (Aibar and Bijker, 1997). Originally, the plan included a grid of 2000 hectares of which 82.35 were mandatory hectares of yards and green spaces between each block, a condition later on partially dropped in the implementation (Aibar and Bijker, 1997; Pallares-Barbera et al., 2011). At present, few dispersed urban parks are located within the boundaries of the city core as a result of the plan of Nicolau Maria Rubió i Tudurí at the beginning of the last century (see Figure 3.4). Nowadays these parks are nearly absent in the immediate sprawling metropolitan periphery (Parés-Franzi et al., 2006).

Note that for scenario 1 we considered solely the 84.6 km^2 of surface occupied by the municipality, excluding the part of it that falls within the borders of the Collserola Park.



1. Parc de Montjuïc, 2. Plaça Letamendi, 3. Sagrada Família, 4. Plaça de les Glòries, 5. Parc de la Ciutadella, 6. Hipòdrom, 7. Turó Parc, 8. Turó Gil, 9. Font del Racó, 10. Vallcarca, 11. Parc Güell,

12. Parc del Guinardó, 13. Parc del Llobregat, 14. Parc de Pedralbes, 15. Parc de Vallvidrera, 16. Parc del Tibidabo, 17. Parc de Sant Medir, 18. Parc d'Horta, 19. Parc del Besòs

Figure 3.4. Rubió i Tudurí's Green Ring proposal. Note that Sierra del Tibidabo was the ancient name of the actual Sierra de Collserola.

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The Collserola Park has a typical Mediterranean climate, with an average annual rainfall of 620 mm (83.1 mm in October only), and an annual mean daily temperature of 15°C (approx. 3 degrees lower than the city) (http://www.parcnaturalcollserola.cat/en/pages/clima-i-meteorologia; retrieved on 18th August 2014). The elevation ranges between 60 m to 512 m (the Tibidabo mount). Most of the Collserola area (38%) is covered by mixed woodlands of Aleppo pine (*Pinus halepensis*) and Holm oak (*Quercus ilex*), but due to past agricultural activity, the vegetation of Collserola is also composed of a diverse mosaic of other land covers ranging from Mediterranean scrub (13%) to savanna and grasslands (2%) (Cahill and Llimona, 2004). Although dominated by Mediterranean species, the deep and steep-sided valleys are populated by animal and plant species typical of central Europe, while some of the sea-facing slopes are covered by African grasses (Douglas and Box, 2000).

The park is situated within one of the most densely populated areas in Europe. The 27 municipalities that surround the park host 52% of the population of the Catalonia region (Barcelona Regional Council, n.d.). Nine of these municipalities (i.e. Barcelona, Montcada i Reixa, Cerdanyola del Vallès, San Cugat del Vallès, El Papiol, Molins del Rei, Sant Feliu del Llobregat, Sant Just Desvern i Esplugues de Llobregat) have part of their territory within the limits of the park. The park is thus enclosed by urban areas and infrastructures. It is crossed by roads and railways built since the beginning of the 19th century (until well into the 19th century the only road crossing the range was in fact built along the Roman road which connected Barcelona with Sant Cugat) (Cañas et al., 1995). As a result, the park is highly ecologically fragmented and isolated (Cahill et al., 2003).

In terms of usage, Collserola, together with the Montserrat and Montseny natural areas, is one of the most visited parks in the Catalan region, with about 2 million visitors per year, seemingly due to its proximity to Barcelona (Creel and Farell, 2008). Also, according to Marià Martí, the director of the Consortium for the management of the natural park, the increased awareness of the population with respect to the benefits that can be derived from frequenting green areas is at the origin of the high number of visits (La Vanguardia, 2013). Cree and Farell (2008) showed that Collserola has the highest annual use value when compared to the other 13 Catalan parks considered.

For scenario 2, which includes the municipality of Barcelona as well as the area of the park, the total area considered comprises a surface of 167.6 km^2.

3.2.2 Assessment of ecosystem services

Numerous indicators and proxies can be used to quantify and assess the range of ecosystem services relevant in urban areas (see Dobbs et al., 2011; Gómez-Baggethun et al., 2013; Haase et al., 2014 for some examples). Few authors based their assessment on ecosystems benefits in urban areas on the concept of “landscape function" (Bastian et al., 2012; Gimona and Horst, 2007; Willemen et al., 2008). We considered four ES in our study, three of which are regulating services (urban cooling, air purification, flood regulation) and one is cultural (recreation). Of these, one service (flood regulation) can be considered an on-site, direct service to the population of Barcelona, while the other three are enjoyed thanks to wind paths and by accessing the nearby natural area. In fact, in a typical summer day, due to winds and the daytime sea-breeze that transport pollutants towards the Collserola range (Huertas et al., 2008; Soriano et al., 2001), Barcelona can benefit from the air quality regulation potential of the vegetation of the park. While, during the afternoon and the nighttime, cooler air flows from the Collserola range into the city (Soriano et al., 2001).

The capacity of urban green areas to uptake pollutants from the air has been assessed for different cities of the world (Cavanagh et al., 2009; Escobedo et al., 2011; Nowak et al., 2006). For Barcelona, air purification delivered by green infrastructure was previously estimated using the i-Tree Eco model (Baró et al., 2014). Starting from the results obtained by Chaparro and Terradas (2009) and especially Baró et al. (2014) who calculated air purification for the entire municipality of Barcelona, air pollution removal potential by vegetation was derived in terms of t $year^{-1}$ km^{-2} of CO, NO_2, PM_{10}, SO_2 and O_3 for both scenario 1 and scenario 2. By doing so, we assumed that the air regulation potential values for each land cover class considered in the study by Baró et al. (2014) were also applicable here for scenario 2 which includes all of Collserola, basing in this way our analysis on the same air pollution levels. Only data from monitoring stations located in the municipality of Barcelona were in fact available.

Urban cooling potential can be calculated through proxies such as: the surface emissivity by land cover type (Larondelle et al., 2014; Stathopoulou and Cartalis, 2007); evapo-transpiration (Larondelle et al., 2014; Larondelle and Haase, 2013; Schwarz et al., 2011); the leaf area index (Knote et al., 2009); or simply through the percentage of tree canopy cover, as a proxy of the cooling potential of tree shadow (Bowler et al., 2010). We calculated both the emissivity (which does not linearly relate to the cooling potential) and the evapo-transpiration capacity (of which the relationship with heat

fluxes is linear), as these provide complementary information (Larondelle et al., 2014). The coefficients of land surface thermal emissions and evapo-transpiration for different land cover types were based on the lookup table derived and applied by Larondelle et al. (2014) (see Table 3.1). The percentage of each land cover type derived by the European Environment Agency (EEA) Urban Atlas[14] (to have the results comparable with those of other European cities) was weighted by the corresponding evapo-transpiration potential (f-value) and surface emissivity coefficient. This was carried out for both the municipality of Barcelona without the area of the municipality included in the CNP (scenario 1) and plus the CNP (scenario 2). Please note that in the results section, the cooling potential calculated through emissivity is expressed as one minus the normalized emissivity through the min-max method were the minimum and maximum emissivity are derived from Table 3.1, and are respectively 131.1 and 145.1. The higher the emissivity, in fact, the lower is the cooling potential.

Table 3.1. Look up table with surface emissivity coefficients and f-values for evapo-transpiration potential by land cover type.
(Source: Larondelle et al., 2014).

The EEA Urban Atlas classes	**Surface emissivity (mean)**	**f-value**
Agricultural + Semi-natural areas + Wetlands	136.7	1.1
Construction sites	134.8	1.0
Continuous Urban Fabric (S.L. > 80%)	143.2	0.8
Discontinuous Dense Urban Fabric (S.L. : 50% - 80%)	143.2	0.8
Discontinuous Low Density Urban Fabric (S.L. : 10% - 30%)	139.4	0.9
Discontinuous Medium Density Urban Fabric (S.L. : 30% - 50%)	139.4	0.9
Discontinuous Very Low Density Urban Fabric (S.L. < 10%)	134.3	1.1
Fast transit roads and associated land	145.1	0.8
Forests	134.7	1.2
Green urban areas	134.3	1.1
Industrial, commercial, public, military and private units	141.5	0.8
Isolated Structures	136.0	1.1
Land without current use	136.0	1.1
Mineral extraction and dump sites	138.0	1.0
Other roads and associated land	145.1	0.8
Port areas	139.9	0.8
Railways and associated land	145.1	0.8
Sports and leisure facilities	138.4	1.0
Water bodies	131.3	1.4

[14] http://www.eea.europa.eu/data-and-maps/data/urban-atlas

The potential supply of recreational opportunities by green space represents the capacity of the ecosystem to provide possibilities for outdoor recreation, nature observation, education, hunting etc. (Niemelä et al., 2010). We calculated this potential on the basis of the land cover types as defined in the Land Cover Map of Catalonia (LCMC)[15] for the year 2009 for both scenario 1 and scenario 2. Several editions of the LCMC are in fact available for the municipality of Barcelona and the CNP. The last, 4th edition corresponds to the year 2009 with a pixel resolution of 0.25 m and 5 hierarchical levels of legend (ranging from 9 to 241 classes). Following other ES assessments in urban areas (see for instance Larondelle and Haase, 2013), all land cover types denoting green spaces (such as urban parks, gardens, forests and agricultural land) were assumed to be capable of providing recreational services. Two simple indicators to quantify recreation potential were calculated: the percent of green covers (from the total area for both scenarios) and the ratio between green cover and population (in m^2 per capita). The selection of green cover classes was based on a query of the LCMC dataset of 2009 and the GIS processing was done with ArcGIS 10 (ESRI). To note is that, according to the classification of recreation areas in urban regions (Niemelä et al., 2010), the Collserola Park would fall into the range of "outdoor recreation areas" for the municipality of Barcelona, as it has a surface larger than 200 hectares, and a distance from the city or housing core ranging between 1 and 15 km.

Water infiltration in soils and groundwater recharge reduce surface runoff and can thus be considered as proxies of the water regulating services provided by ecosystems (Bergkamp and Cross, 2006). Flood regulation can then be calculated as: the percentage contribution of groundwater to base flows (Egoh et al., 2008); in millions of m^3 of groundwater recharge per 1 km^2 of grid cell (Reyers et al., 2009); in terms of the presence and surface of the riparian areas (i.e. "green zones" which lie between stream channels and uplands) (Pert et al., 2010); or in percentage of sealed surface in the urban area (Haase and Nuissl, 2007). Other indicators of flood regulation are the water storage capacity (buffer) in m^3, or performance indicators like "reduction of flood danger and prevented damage to infrastructure" (de Groot et al., 2010). We considered for its simplicity the percentage of sealed surfaces with respect to the total area considered as an indicator of the infiltration capacity of the soil and reduction of runoff. The data analysis was carried out through ArcGIS 10 (ESRI) on the 2009 LCMC. In vegetated areas, in fact, only 5 to 15% of the rainwater flows as surface runoff while the rest evaporates or infiltrates (Bolund and Hunhammar, 1999). In sealed urban areas, on the other hand, 60% of the rain water flows on the surface and is collected by the drainage system (Bolund and Hunhammar, 1999) or overflows. According to Pataki et al. (2011), urban landscapes with 50–90% impervious cover can lose 40–83% of rainfall to surface runoff. In the results section we report the opposite of the percentage of sealed surfaces as the actual capacity of the land to reduce flood impacts.

[15] Provided by CREAF (Centre for Ecological Research and Forestry Applications) http://www.creaf.uab.es/mcsc/

Finally, we carried out a literature review and gathered the results obtained by other studies in other urban areas mainly in western countries but applying similar methods for the assessment of the four ES we considered (see Table 3.3). This is to situate the livability of Barcelona with respect to other cities, with and without the park, and stress its improvement due to the presence of Collserola.

3.2.3 Assessment of land use changes

To show that the forest area of the park is not a native nor a wild forest, the only remnants are in fact present in the area of 113 hectares of indigenous deciduous and evergreen oak forest of Font Groga, close to Sant Cugat (Cañas et al., 1995; CPC, 2011), we used two sources of information. Both assess land use changes in the area now occupied by the CNP in different periods. First, land use information from the mid19th century were derived from the so-called "Amillaramientos" which collected information and data on agricultural activities based on the declarations of the landowners to which taxes would be adjusted. Second, we used the land cover maps of Catalonia available for the years 1956 (Diputació de Barcelona, SITMUN: http://sitmun.diba.cat/sitmun2/inicio.jsp) and 2009 (LCMC dataset), which are both based on aerial photo interpretation.

<u>Amillaramientos</u>

The "Amillaramientos" are documents conserved and made available in national and regional historical archives and report the monetary value as well as the surface occupied by each crop and agricultural activity declared by each landlord in the municipality considered. It was in fact the owner of the land himself who needed to report yearly data on agricultural land in his possession to the municipality. This taxation system substituted, between 1845 and 1900, cadastral information (Vallejo Pousada, 2010). Reliability of land cover information assessed through this mean has some uncertainties as data and information were provided directly by landlords and under little control by the central administration (Vallejo Pousada, 2010). Thus, the declared agricultural land generally provides an underestimated economic and surface value of the agricultural land cover area per municipality and per year. A solution to this is to look at more than one year and compare the total agricultural land to check if there is a difference in the values from a year to another or, more generally, within a decade (Pujantell, 2012).

The municipalities which have part of their territory included in the actual boundaries of the park and for which the "Amillaramientos" were available are: El Papiol, Sant Cugat del Vallès, Cerdanyola del Vallès, Montcada i Reixac and Sant Feliu de Llobregat (see Figure 3.5). For these municipalities,

information was collected from the summaries of the “Amillaramientos” which appear at the end of each report and for the available years (generally two years between 1952 and 1962). The area units in which the data were reported (e.g. “Cuarteras” or “Número de mojadas”) were transformed in hectares. Most of the crops listed fell within these categories: vineyards, different fruit trees, cereals, olive trees, almond and timber. Uncultivated, barren land was excluded from the calculations. The percentages of the reported agricultural land of the two years were calculated with respect to the actual total surface of each municipality. This was possible as, according to the report from the Spanish Ministry for Public Administration (2008) on the changes in the name and composition of Spanish municipalities since 1842, substantial modifications in the boundaries did not occur for the municipalities considered: only a part of the territory of Cerdanyola del Vallès was incorporated in 1994 as part of Badia del Vallès, a new municipality of only 88,7 hectares; and San Feliu de Llobregat now includes a part of Santa Creu d’Olorda, as it was absorbed into the surrounding municipalities in 1920. Finally, the percentages of agricultural land in 2009 for these municipalities were calculated through ArcGIS 10 (ESRI) based on the CREAF land cover data of 2009 mentioned above.

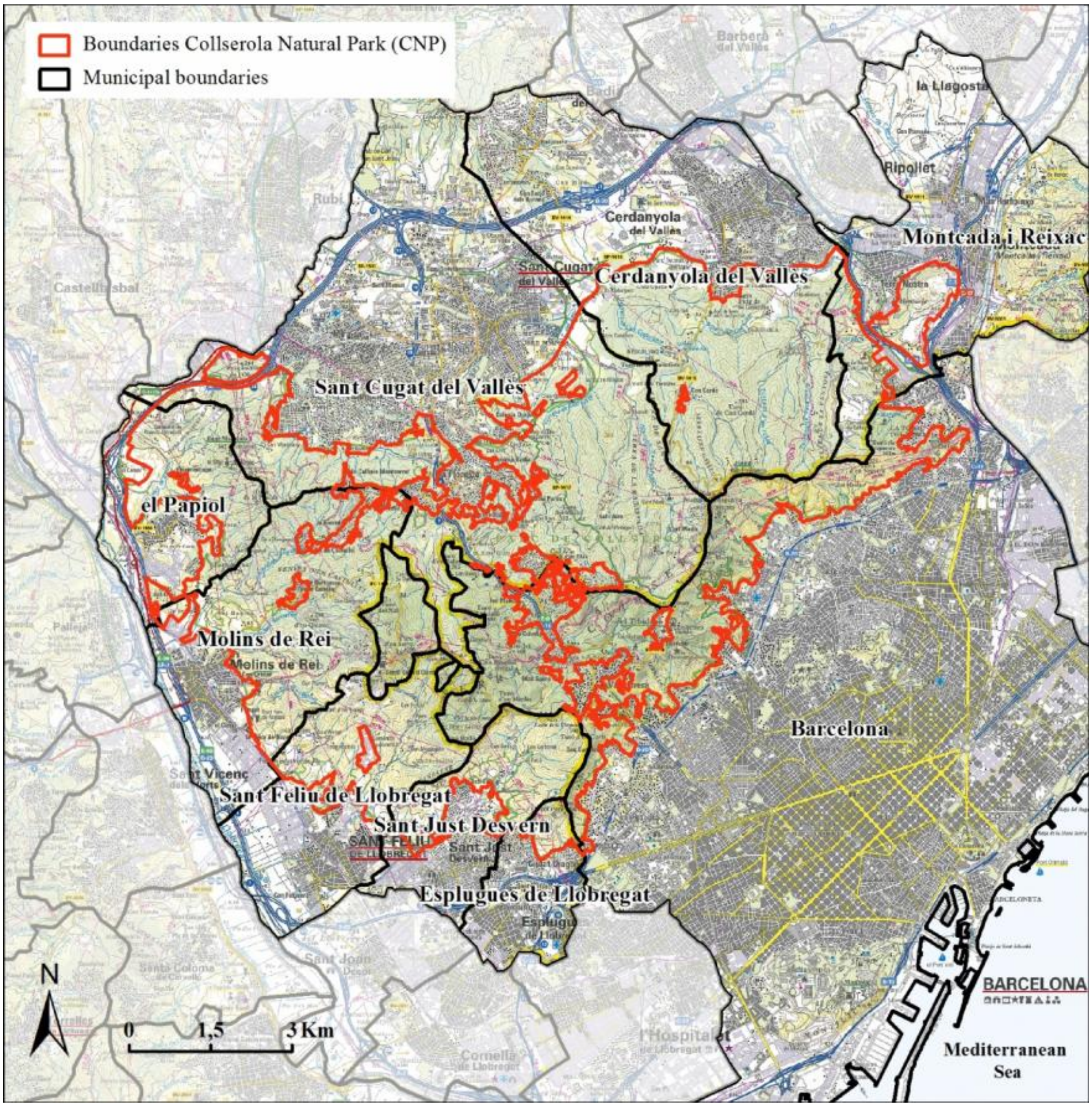

Figure 3.5. Map of the municipalities which have their boundaries partly within the area of the CNP.
(Source: own map)

Land cover change between 1956 and 2009

A land cover map corresponding to the year 1956 was elaborated for the province of Barcelona (NUTS3 region). It was developed from aerial photo interpretation of a set of historical aerial images known as the "American flight". The legend levels match those of the LCMC maps, so land cover change analyses can be easily performed. The land cover change analysis in the CNP was then based on a simple overlay between both datasets (1956 and 2009), using the first level of legend (9 classes) in order to avoid an excessive number of land cover transitions between both years. GIS processing was done with ArcGIS 10 (ESRI).

3.2.4 Historical analysis

The material reviewed for the historical analysis of environmental and socio-economic changes that led to different uses and management of the area now occupied by the Collserola Park spanned between master and doctoral theses, reports, published books, peer-reviewed journal articles, websites and newspaper articles. In this regard, 150 relevant articles were collected from the online newspaper library of the “La Vanguardia” amongst those published between 1909 and May 2014. The keywords used were “Sierra de Collcerola”, “Sierra de Collserola” and from the 1980’s onward “Protección de la Sierra de Collserola”. The texts of the article were analyzed by coding them making use of the software Atlas.ti (Scientific Software Development GmbH) which facilitates the analysis of qualitative data without diminishing the complexity of the information that can be derived. Interviews with principal stakeholders were also carried out.

3.3 Results

3.3.1 Assessment of ecosystem services

In this section we present the results of the ecosystem services assessment we carried out for scenarios 1 and 2 (see Table 3.2) and we compare them then with the results obtained for other cities in which similar methodologies have been applied (see Table 3.3). The aim is to stress the importance of the Collserola Park for the livability of Barcelona. Results indicate that the presence of the area of the park adjacent to the city of Barcelona allows a significant increase in the levels of ES enjoyed by city inhabitants, which enhances the livability of the agglomeration.

Table 3.2. Results of ecosystem services assessment of the municipality of Barcelona plus or minus the CNP obtained with the i-Tree Eco model (for air pollution potential removal) and GIS based applications (for the other ES) as detailed in the Methods section.

Ecosystem service	**Barcelona minus Collserola** (scenario 1 ; 84,65 km^2)	**Barcelona plus Collserola** (scenario 2 ; 167.6 km^2)
Air pollution potential removal (t $year^{-1}$ km^{-2} of CO, NO_2, PM_{10}, SO_2, O_3)	1.85	4.48
Urban cooling (surface emissivity, mean value)	141.75	138.72
Urban cooling (f-value for evapo-transpiration potential of a land use class; mean value)	0.84	1.01

Ecosystem service	**Barcelona minus Collserola** (scenario 1 ; 84,65 km^2)	**Barcelona plus Collserola** (scenario 2 ; 167.6 km^2)
Flood regulation Impervious cover (mean %)	78.60	71.79[16]
Recreation (% green cover from total area)	12.92	53.80
Recreation (Green cover: m^2 per person)	6.79	55.95

(Table 3.2 continued)

[16] Considering only the catchment area affecting the municipality of Barcelona.

Table 3.3. Ecosystem services assessments results for various cities derived from the literature and comparable, in terms of methods used, with the results obtained for Barcelona.

Ecosystem Service	City	Surface (km^2)	Inhabitants (N. circa)	Amount of service provided	Reference
Air pollution removal ($t\ year^{-1}\ km^{-2}$ of NO_2, PM_{10}, SO_2, O_3)	Jacksonville, FL, US	2264.5	828000	4.9	(Nowak et al., 2006)
	Bridgeport, CT, US	50.2	144230	0.43	
	Buffalo, NY,US	139	260000	1.18	
	Salt Lake City, UT, US	284.9	190000	2.70	
Air pollution removal ($t\ year^{-1}\ km^{-2}$)	Chicago, US	606	2715000	9.2	(McPherson et al., 1997)
	Christchurch, New Zealand	452	348500	0.66	(Cavanagh, 2008 cited in Cavanagh et al., 2009)
	Beijing, China	300	4500000	4.20	(Yang et al., 2005)
Cooling capacity (surface emissivity, mean value)	More than 300 European cities	Various	Various	135.99-138.02 (very low)	(Larondelle et al., 2014)
				138.03-139.44 (low)	
				139.45-140.23 (medium low)	
				140.24-140.84 (medium high)	
				140.85-141.50 (high)	
				141.51-142.72 (Very high)	
Cooling capacity (f-value for evapo-transpiration potential of a land use class. Mean value)	More than 300 European cities	Various	Various	0.81-0.85 (very low)	(Larondelle et al., 2014)
				0.87-0.89 (low)	
				0.90-0.92 (medium low)	
				0.93-0.96 (medium high)	
				0.97-1.02 (high)	
				1.03-1.11 (very high)	
Flood regulation (% of sealed soil)	Germany	Various	Various	52.00 (average)	(EEA, 2006)
	Leipzig (Germany)	297.6	531809	58.00–68.00	(Haase and Nuissl, 2007)
	Various European cities	215	851000	22.90 (Min. Stockholm, Sweden)	http://www.eea.europa.eu/data-and-maps/figures/mean-
		414.90	1741246	46.33	

Ecosystem Service	City	Surface (km^2)	Inhabitants (N. circa)	Amount of service provided	Reference
				Vienna (Austria)	soil-sealing-in-european#tab-metadata
		412	3074160	60.43 (Athens, Greece)	
		41.8	726547	69.29 (Tirana, Albania)	
		228	1883500	78.0 (Max. Bucharest, Romania)	
Recreation (% green cover from total area)	Sheffield (UK)	368	513000	45.00	(Fuller et al., 2010 cited in ; Fuller and Gaston, 2009)
	Central built-up area of Guangzhou (China)	235,5	/	31.20	(Jim and Chen, 2006)
	Birmingham (UK)	268	1 M	11.00	(Angold et al., 2006)
	Stockholm (Sweden)	215	851000	26.00	(Bolund and Hunhammar, 1999)
	386 European cities	> 0.25 km^2	> 100000 inhab.	1.90 (Min.: Reggio Calabria, Italy)	(Fuller and Gaston, 2009)
				18.6 (average)	
				46.00 (Max.: Ferrol, Spain)	
Recreation (Green cover: m^2 per person)	386 European cities	> 0.25 km^2	> 100000 inhab.	3.00 to 4.00 (Cádiz, Fuenlabrada and Almería in Spain; and Reggio Calabria in Italy)	(Fuller and Gaston, 2009)
				> 300.00 (Liège in Belgium, Oulu in Finland, and Valenciennes in France)	

(Table 3.3 continued)

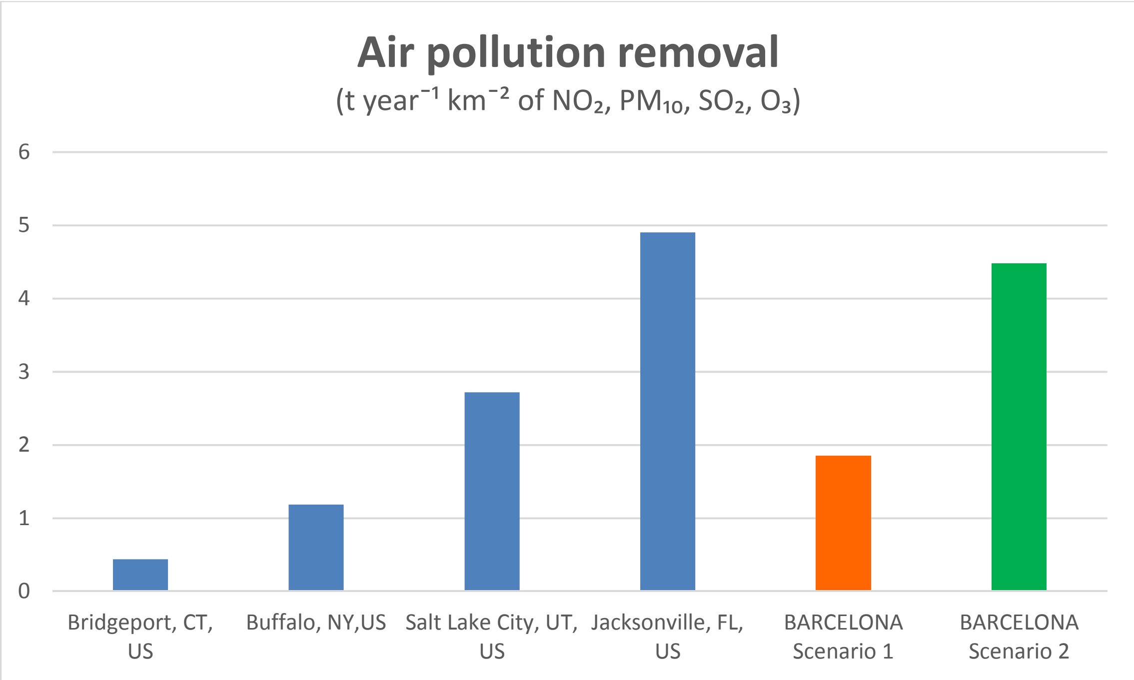

Figure 3.6. Air pollution removal potential, in t year^{-1} km^{-2} of NO_2, PM_{10}, SO_2, O_3 derived for different cities from the literature (see Table 3.3) and from the results obtained for Barcelona scenarios 1 and 2 (Table 3.2).

A study conducted by the Centre de Recerca en Epidemiologia Ambiental (CREAL) showed that 3,500 lives could be saved annually in Barcelona by reducing the current levels of air pollution to WHO standards while significantly reducing morbidity linked to cardio-respiratory diseases (Künzli and Pérez, 2007). Life expectancy would increase by about 14 months and economic benefits would amount to 6.4 million € (Pérez et al., 2009). Air quality in Barcelona is in fact poor. According to WHO guidelines published in 2005, the city presents high levels of PM_{10} concentration when compared to other European cities (WHO, 2005). Pérez et al. (2009) found that, in the 57 municipalities of the Metropolitan area of Barcelona, concentration of PM_{10} for the year 2004 was ranging between 35 and 56 μg/m^3 which are both largely above the WHO recommended level of the of 20 μg/m^3. According to the Generalitat de Catalunya (2012), the air quality of the Barcelona municipality and of the counties of the Baix Llobregat and the Vallès, within which most of the Collserola Park is located, was also poor in 2012. The concentrations of PM_{10} and NO_2 in the air were beyond the quality standards. This fact led, in both areas, to the implementation of a special plan for the improvement of air quality as it was required also for Barcelona. The industrial and urban areas located in the two surrounding river basins emit pollutants that are transported into the Vallés and the Anoia depressions, while the anabatic winds over the southern Collserola slope transport the contaminated air towards the top of the range (Querol, 2001), especially during daytime (Viana et al.,

2005). As mentioned, during daytime the flow of pollutants is in fact transported by the up-slope sea breeze from Barcelona into the Sierra while the opposite happens at nighttime in a typical summer day (Soriano et al., 2001). The service provided by the vegetation of the CNP in removing pollutants from the air thus greatly benefits the municipality of Barcelona and the metropolitan area at large. In fact, from our analysis it resulted that the vegetation of the densely urbanized area (scenario 1) has a low capacity to purifying the air (1.85 t $year^{-1}$ km^{-2} of pollutants) and can be situated in between Buffalo (NY, US) (1.18 t $year^{-1}$ km^{-2}) and Salt Lake City (UT, US) (2.70 t $year^{-1}$ km^{-2}) (see Table 3.2, Table 3.3 and Figure 3.6). These cities, however, host only a fraction of the population of the municipality of Barcelona. The results obtained for scenario 2 (4.48 t $year^{-1}$ km^{-2}) show that the CNP improves the livability of the city providing a source of cleaner air nearby which flows into the city during nighttime. This result is comparable with that obtained through the same method for Jacksonville, FL, US (4.48 t $year^{-1}$ km^{-2}), which is the best performing city in the US according to the available data. However, in 2008 only the municipality produced 744 t of PM_{10} and 10.413 t of NO_x (Ajuntament de Barcelona, 2013) related to which the capacity of the ecosystem to strip pollutants from the air for scenario 1 (188.46 t $year^{-1}$ of CO, NO_2, PM_{10}, SO_2, O_3) and even for scenario 2 (750.96 t $year^{-1}$ of CO, NO_2, PM_{10}, SO_2, O_3) result to play a minor air purification role.

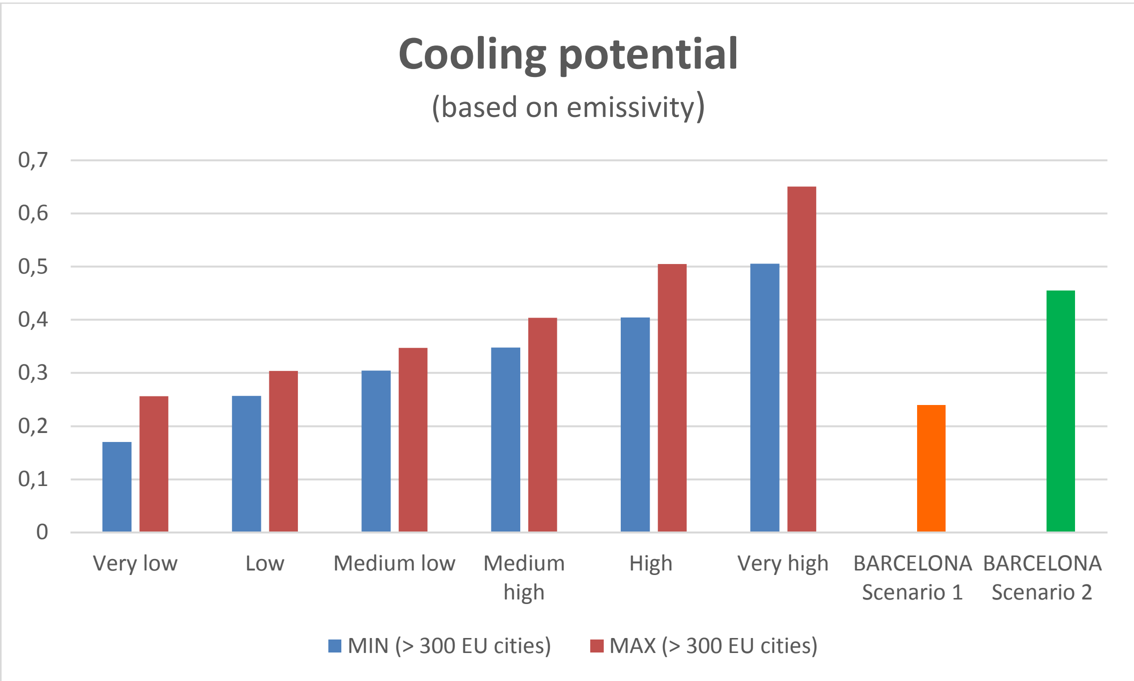

Figure 3.7. Air cooling potential calculated as one minus the normalized emissivity, both derived for different cities from the literature (see Table 3.3) and from our results obtained for Barcelona scenarios 1 and 2 (see Table 3.2).
The scale (from very low to very high) with minimum and maximum ranges for each category is derived from a study by Larondelle et al. (2014) which clustered the results of the mean emissivity of more than 300 European cities.

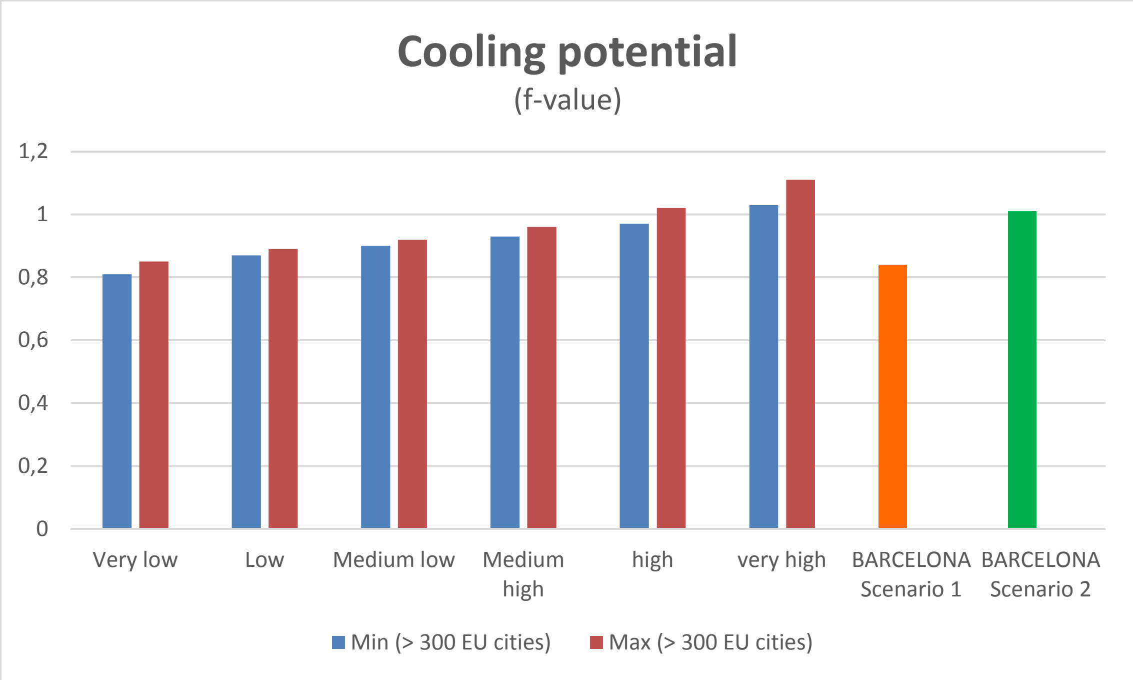

Figure 3.8. Air cooling potential in terms of evapo-transpiration (f-value) derived for different cities from the literature (see Table 3.3) and from our results obtained for Barcelona, scenarios 1 and 2 (see Table 3.2).
The scale (from very low to very high) with min. and max. ranges for each category is derived from the assessment of the cooling capacity due to evapo-transpiration potential in a study which clustered the results obtained for more than 300 European cities by Larondelle et al. (2014).

Regarding urban cooling, the extreme hot temperature threshold beyond which the mortality rate of the population increases sharply was determined by Tobías et al. (2010) at 30.5°C for Barcelona. This threshold was exceeded for 30 days in 2003 and caused 500 excess deaths in the city (Tobías et al., 2010). The CNP represents a place with cooler air which can be easily accessed by the population of Barcelona to alleviate the impacts in case of extreme heat. Colder air also flows at night into the city in the summer months easing high night temperatures (Soriano et al., 2001). While rural inhabitants enjoy some relief at night, urban populations suffer from a persistence of high temperature due to the UHI effect all day long, which is thought to increase impacts on health (Koppe et al., 2004). Including the urban cooling benefits derived by the Collserola Park, Barcelona passes from a very high emissivity (141.75), corresponding to a very low cooling potential, to a low emissivity (138.72), thus benefitting from a high cooling potential with respect to other European cities included in the study by Larondelle et al. (2014) (see Table 3.2, Table 3.3 and Figure 3.7). These results gain in importance if we consider that since 1997 the green surface (in square meters) per inhabitant decreased in the densely populated municipality (Argelich and Recio, 2009). For the calculation of the cooling potential through evapo-transpiration, the f-value passes from being low in scenario 1 (0.84) to high

(1,01) in scenario 2 with respect to more than 300 other cities analyzed by Larondelle et al. (2014) (see Table 3.2, Table 3.3 and Figure 3.8).

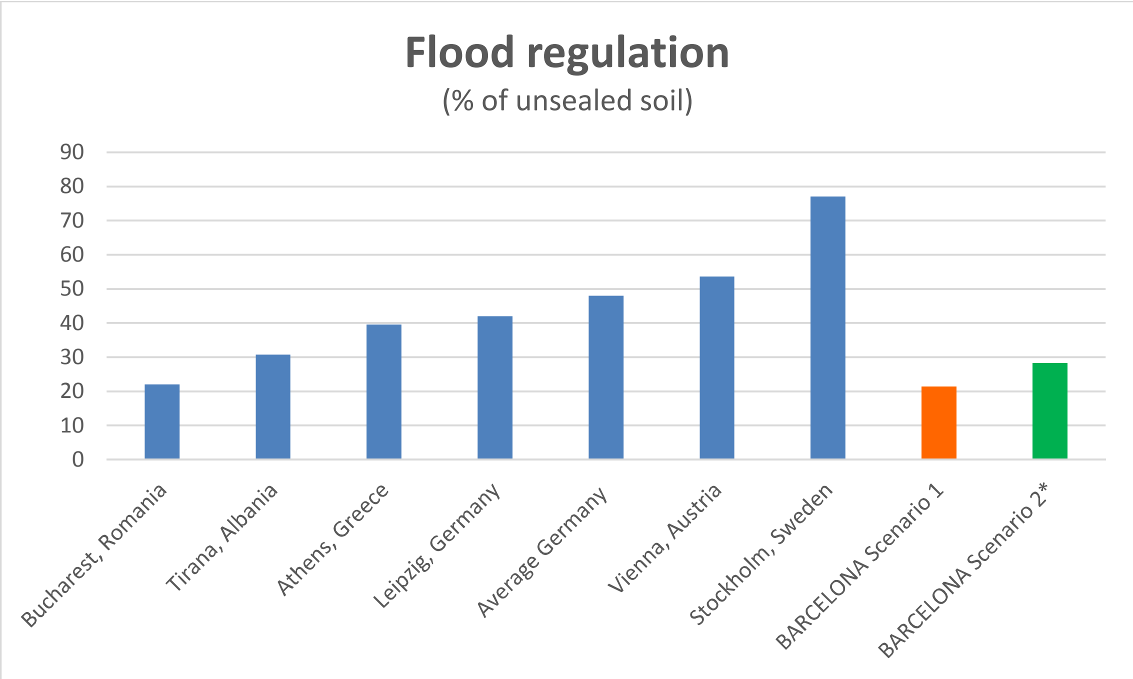

Figure 3.9. Flood regulation potential in terms of 100% minus the % of sealed soil derived for different cities from the literature (Table 3.3) and from our results obtained for Barcelona in scenarios 1 and 2* (Table 3.2).
(N. B. For flood regulation in scenario 2* we took into account only the area of the park adjacent to and that slopes towards Barcelona)

While flash floods downtown had caused significant damages in the past (Barrera et al., 2006), the problem of floods was partly solved when the drainage system was renovated by including underground tanks for the collection and slow release of water in 1992. The fact remains however that the high degree of soil sealing can cause localized floods in case of downpours. From our analysis it resulted that the degree of soil sealing for scenario 1 was 78.60% which is comparable with the highest rate of soil sealing (thus, the lowest flood regulation potential) measured across the main European cities by the European Environment Agency (EEA) (i.e. the city of Bucharest, Romania with 78 % of soil sealing) (see Table 3.3 and Figure 3.9). Flood regulation potential slightly improves when the part of the Collserola range which slopes to the proximity of and towards Barcelona is included. For this case, we obtained a 71.79% of soil sealing with which the flood regulation potential of Barcelona in now comparable to that of Tirana (Albania) with 68.29%, but is still very far from the minimum percentage of the city of Stockholm (22.90%). We then calculated the flood regulation potential including the whole area of Collserola (scenario 2) as the ecosystem in Collserola

contributes in regulating water flows in the two adjacent river basins of the Llobregat and Besòs from which flooding can affect the marginal areas of the city. The result obtained (40.97% of sealed areas) is now comparable to those of other northern European cities like Vienna, London, Amsterdam, Luxembourg and Copenhagen[17] .

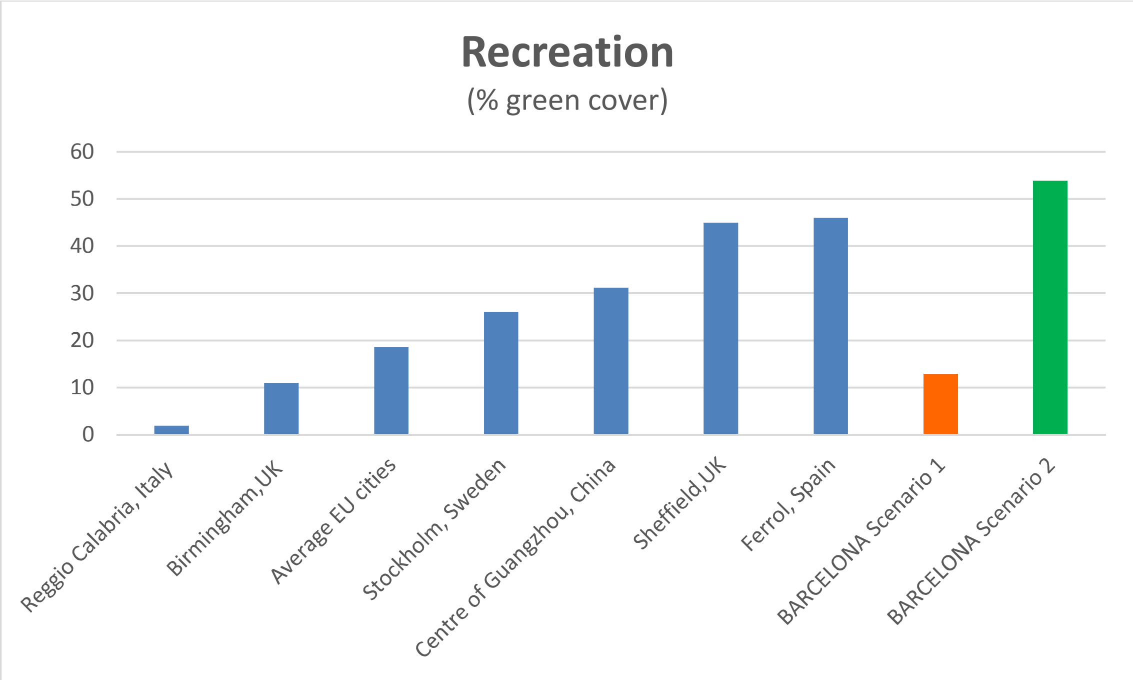

Figure 3.10. Recreation potential in terms of % of green cover derived for different cities from the literature (see Table 3.3) and from our results obtained for Barcelona scenarios 1 and 2 (Table 3.2).

[17] http://www.eea.europa.eu/data-and-maps/figures/mean-soil-sealing-in-european#tab-metadata

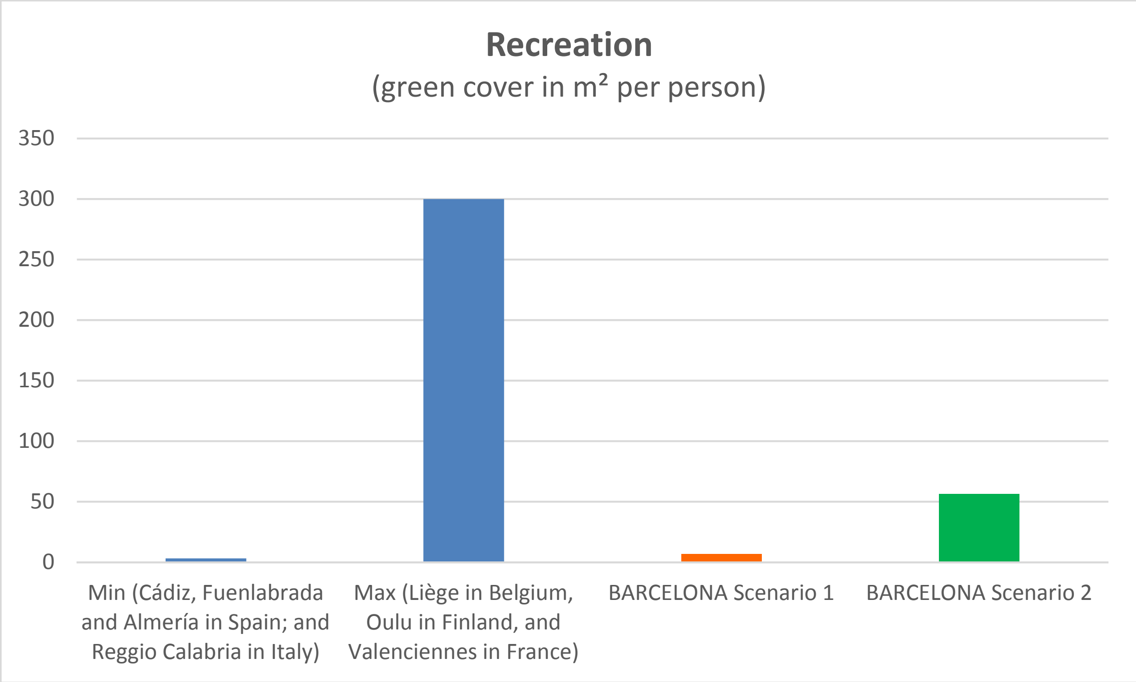

Figure 3.11. Recreation potential in terms of green cover in m^2 per person derived for different cities from the literature, of which only the two extreme cases (min. and max. recreation potential) are indicated here (see Table 3.3), as well as from our results obtained for Barcelona scenarios 1 and 2 (Table 3.2).

Regarding recreation, the high demand for recreational area is ascertained by the elevated number of annual visitors to the Collserola Park, approx. 2 million per year (IERMB, 2009). The percentage of green cover as a proxy of recreation passes from being less than 13% in scenario 1 to approx. 54% in scenario 2. In scenario 1, Barcelona would perform lower than average at the European scale (18.6%), while in scenario 2 it performs better than the maximum with respect to other European cities (see Table 3.3 and Figure 3.10). These results confirm the centrality of the park for the city inhabitants in terms of recreation also considering its easy accessibility due a good and widely connected transport system.

According to various references in the grey literature, WHO and FAO standards recommend between 9 and 15 m^2/capita (Kuchelmeister, 1998; Mercadé Aloy, 2011). The densely populated urban area of Barcelona would be largely inferior to this threshold with approx. 6.8 m^2/capita. Including the Collserola Park, each person benefits instead of approx. 56 m^2 of green area. This nearby source of an environmental service supports a wide range of recreational activities and largely improves the livability of the city well beyond WHO standards. However, to note is that, according to Fuller and

Gaston (2009), some northern European cities can even attain more than 300 m^2/capita of green space (see Figure 3.11 and Table 3.3).

As a note, the CNP is often described as the green lung of Barcelona. However, for what concerns CO_2 sequestration, a study revealed that its carbon sequestration potential would offset the CO_2 production per year of only 2000 inhabitants (Argelich and Recio, 2009). Although this seems rather underestimated if compared with the results obtained by Baró et al. (2014), it confirms one of the conclusions of the study by Pataki et al. (2011) that direct carbon sequestration by urban plants and soils is negligible if compared with the same city's Green House Gas (GHG) emissions.

3.3.2 Land use change

From the "Amirallamientos": second half of the 19th century

From the results obtained through the data collected from the "Amirallamientos", four out of five municipalities had between approx. 50 and 80% of their territory covered by agricultural land in the mid-19th Century (see Table 3.2). The data obtained with this method appear to be reliable as estimates. For all the municipalities, excluding Cerdanyola del Vallès, the percentages of agricultural land were in fact decreasing from a year or date to the other (the case of El Papiol or Sant Cugat del Vallès), which can be expected due to the process of agricultural land abandonment, or resulted to be similar for the two years considered (the case of Montcada i Reixac and Sant Feliu del Llobregat). These minor variations, in fact, tell that the report of the landowners can be considered trustworthy (see also Pujantell, 2012). The percentages obtained for the mid-19th centuries are overall very high if we compare them to the present conditions measured for the year 2009 (see Table 3.4). These results prove that the area now occupied by the park was widely used for agricultural production in the past.

Table 3.4. Percentage of agricultural land in 5 out of 9 municipalities which have now part of their territory located within the CNP and for different years derived both from the "Amirallamentos" and the LCMC land cover map of Catalonia 4th edition of 2009.

Municipality	Surface in hectares	Year	% of agricultural land
El Papiol	896	1852	72.33
		1863	68.71
		2009	12.63
Sant Cugat del Vallès	4,832	1853	58.09
		1862	52.30
		2009	3.37

Municipality	Surface in hectares	Year	% of agricultural land
Cerdanyola del Vallès	3,060	1854	59.17
		2009	8.69
Montcada i Reixac	2,340	1853	74.97
		1862	75.04
		2009	13.87
Sant Feliu del Llobregat	1,181	1861	29.34
		1862	29.54
		2009	15.50

(Table 3.4 continued)

Land use in the second half of the 20th century

The continued agricultural land abandonment is also proved by our GIS-based analysis of land cover changes in the area now occupied by the park between 1956 and 2009. Land covered by crops went from being above 20% of the land of the park in 1956 to about 5% in 2009, while dense forest increased from about 57% to approx. 68.4% between 1956 and 2009 (see Table 3.5 and Figure 3.12).

Table 3.5. Area and percentage of land use type for the years 1956 and 2009 in the Collserola Park.

Land use type	Year 1956		Year 2009	
	Area (ha)	**%**	**Area (ha)**	**%**
Water courses and bodies	/	/	0.68	0.01
Sparse woodland	143.17	1.73	246.65	2.97
Dense woodland	4729.29	57.01	5672.51	68.38
Crops	1714.39	20.67	413.61	4.99
Sealed surfaces	147.94	1.78	411.38	4.96
Bare land	7.65	0.09	21.08	0.25
Scrubland	1405.97	16.95	1354.67	16.33
Grassland	146.88	1.77	174.73	2.11
TOTAL	*8295.30*	*100.00*	*8295.30*	*100.00*

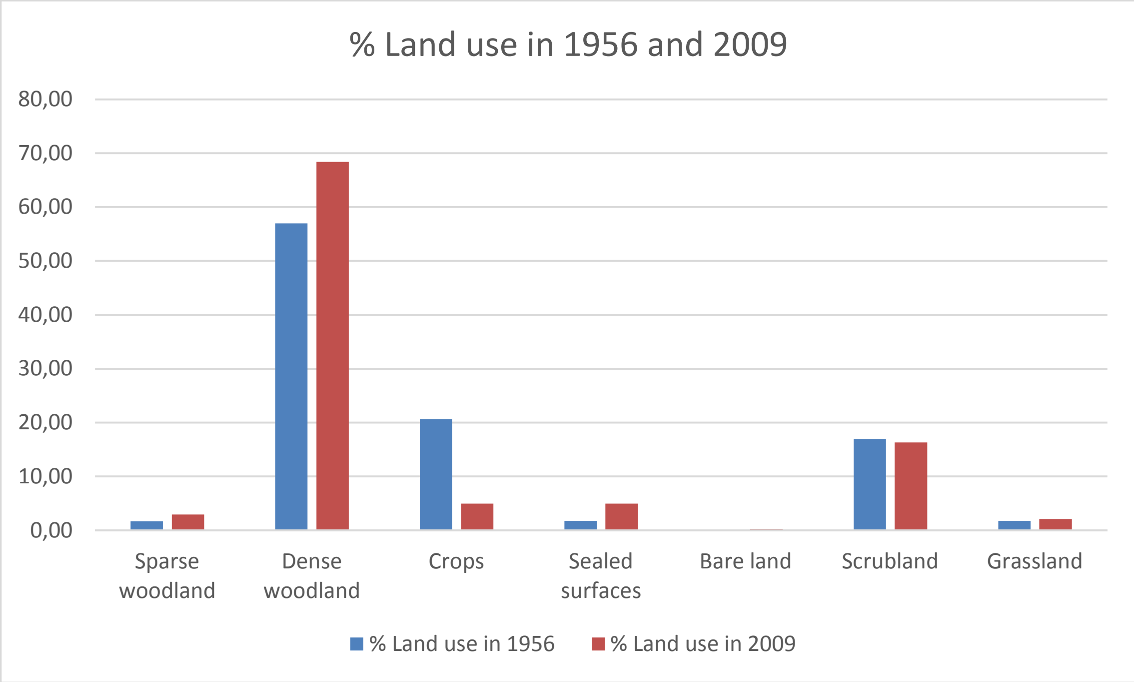

Figure 3.12. Graph of the land cover percentages in the years 1956 and 2009 for the Collserola Park only.

The main changes in land cover occurred through the conversion of scrubland to dense woodland (about 774 hectares). Second and third rank the conversions of crops to dense forest and to scrubland: approx. 16.6 and 14.3 % of the changes respectively (see Table 3.6). Overall, these results depict a thickening of the vegetation from crop to scrubland and to dense woodland. This confirms the process of abandonment of agricultural land followed by reforestation which has been documented also by other studies in the region (Barcelona Regional Council, n.d.; Cañas et al., 1995; Diaz et al., 2008).

Table 3.6. Results of the assessment of the changes from a land cover type to another for the period 1956-2009 in the Collserola Park.

Main changes from one land cover type to another for the period 1956-2009	**Surface (ha)**	**%**
Sparce woodland - Dense woodland	107.32	3.38
Dense woodland - Sparce woodland	65.11	2.05
Dense woodland - Sealed surface	127.46	4.01
Dense woodland - Scrubland	317.89	10.00
Crops - Sparce woodland	96.51	3.04
Crops - Dense woodland	527.67	16.60
Crops - Scrubland	453.45	14.27
Crops - Grassland	107.77	3.39
Scrubland - Dense woodland	773.78	24.34

Main changes from one land cover type to another for the period 1956-2009	Surface (ha)	%
Grassland - Scrubland	80.17	2.52
TOTAL	*2657.14*	*83.60*

(Table 3.6 continued)

3.3.3 Historical analysis of the socio-ecological changes in the area now occupied by the CNP

The aim of this section is to explore how the Collserola range has been the site and is the result of different socio-ecological and political processes which altered the environment, some of which (e.g. agricultural activities) had completely changed its landscape. These changes then led to the present configuration of the Collserola forest, have been intense in the last two centuries and can be structured in three main periods: firstly, the socio-economic change that led to the collapse of agriculture, the abandonment of agricultural land and the reforestation process around the mid-19th century; secondly, the institutional and planning interventions for the protection of the area from the beginning of the 20th century (see Table 3.7); and thirdly the last period of protests from the 1970's against urban encroachment which finally led to the establishment of Collserola as Natural Park.

The conservation of the park justified by its natural value refers in reality to an environment which has been substantially modified through the centuries and has been recently under the pressures of a highly urbanized metropolitan area. More than a pristine environment, it appears, from our analysis, to be a produced forest, the result of highly politicized processes and localized conflicts. These have nonetheless succeeded in guaranteeing an accrued amount and range of ecosystem related benefits that the population of the municipality of Barcelona can enjoy today.

<u>First period: the collapse of agriculture and the reforestation process</u>

The Romans arrived in the Peninsula in 218 BC. In Roman times, the production of amphorae to transport the wine in the area now occupied by the park had taken the semblance of an industrial process (Centellas i Masuet et al., 2008). In the 12th century, vineyards dominated the eastern and the southern part of the range, while croplands spread greatly into the forest after the 16th century (Cañas et al., 1995).

However, according to the Barcelona Regional Council (n.d.), Medieval times had been the most active period in terms of human activity in Collserola. After the Low-Medieval demographic crises, villas had been restructured and expanded on the hills throughout all the 17th and 18th centuries, when

the main economic activity was the cultivation of olive trees, vineyards and wheat (Barcelona Regional Council, n.d.; Cañas et al., 1995). The exploitation of black slate started at the end of the 18th century and intensified all along the 19th and 20th centuries, especially on the side of the Llobregat (Centellas i Masuet et al., 2008). Mines of limestone were also present in the area.

With variations depending on demography, agricultural expansion reached its maximum at the end of the 18th century, when Barcelona got even affected by a shortage of wood supply (Folch i Guillèn, 1977). We have also shown how in the municipalities that have now their boundaries within the park, most of the land was dedicated to agriculture in the 19th century and that this is not the case at present. Agriculture then started to be abandoned in the second half of the 19th century initially due to the Phylloxera plague (Barcelona Regional Council, n.d.; Cañas et al., 1995; Folch i Guillèn, 1977). The removal of the medieval walls in 1854 and the expansion of industrial activities accompanied by urbanization, additionally favored the abandonment of agriculture (Cañas et al., 1995). A study carried out in the nearby Cal Rodó catchment located in the headwaters of the Llobregat watershed assessed land cover changes after abandonment of agro-pastoral activities initiated in the 1860s, when farmers and herders started to migrate to seek jobs linked to the expanding industrial sector in the region (Poyatos et al., 2003). With the development of the textile industry during the 19th century and due to its direct access to the Llobregat valley which connected the city with its hinterland, the importance of the port of Barcelona grew and supplanted Tarragona as principal port of the region (Cardona, 2009). Due to the high rate of immigration and the little space available for agriculture to feed the growing population, most of the food products started to be imported from abroad (Cardona, 2009). These processes also contributed to the abandonment of agricultural land in the area.

Despite the abandonment of agricultural land since mid of 19th century, the area continued to be exploited for timber and charcoal. The pine forest is not in fact the climax vegetation in the region. The spread of white pine (*P. halepensis*) in the abandoned agricultural land was favored by forestry activities as it has a fast growth rate and was used for the production of wood or charcoal (Raspall et al., 2004). This was especially evident during the World Wars which left the slope facing Barcelona quasi bare (Cañas et al., 1995). The advent of gas in the houses in the mid of the 20th century then caused the fall in the prices of these products and their supply stopped being profitable (Raspall et al., 2004).

The abandonment of agriculture in the littoral hills has been the main trend also during the second half of the 20th century as reported by the study of Diaz et al. (2008) on land use changes in Collserola and as confirmed by our results on the changes of land use in the area of the park between 1956 and 2009. In this period too, major reforestation took place involving a significant increase in pine forest

and a decline in grassland, croplands, and scrub vegetated areas (Poyatos et al., 2003). Socio-economic changes and changes in the sources of energy and materials have however been rapid with respect to the temporal scale of changes in the succession of the vegetation and ultimately led to a homogenization of the territory with the consequent increase in forest fires (Argelich and Recio, 2009), as also documented by the numerous newspaper articles collected for this study (e.g. the forest fire that burned 125 ha of park on the 11th of August 1994). Previous studies also found these disturbances to be the most important environmental risk in the area of the park till nowadays, with an average of 61 forest fires per year and a total of 453 ha burned during the period 1993-2004 (Aldeguer et al., 2008). Across the decades, numerous interventions with reforestation were needed due to these hazardous events.

Land abandonment caused overall the loss of some types of ecosystems, especially those requiring higher management practices, which are particularly valuable in terms of biodiversity and the ecosystem services they provide. Examples are the Mediterranean cork oak (*Quercus suber*) savannas, endemic of southwestern Europe and southwestern Africa (Bugalho et al., 2011). Agricultural land also generally increases the social-ecological value of nearby mountainous areas (Pino et al., 2000). An overall enrichment in forest species and, on the other hand, a decrease in total species richness occurred in the area (Pino et al., 2000). In the case of the Collserola Park, silvopastoral and agricultural activities remained relatively intense until the beginning of the 1960s (Raspall et al., 2004). After that, conservation policies specifically aiming at maintaining the diversity of ecosystems were considered and put in place as one of the main objectives of the Consortium in charge of the management of the park. Nonetheless, most of the land got converted from agricultural to forestal. With reforestation, "excursionism" became widely practiced at the end of 19th century and in the first half of the 20th century. Rafael Puig i Valls, a notable Catalan forest engineer who brought to Catalonia conservationists ideas after a visit to the US, was one of the founders of the Catalan Alpine Club and of the Catalan Association for Scientific Excursions (in 1920) (Boada, 1995). The influence of these activities in the park is documented by the numerous newspaper articles that appear on the practice of "excursionism" in Collserola in the first half of the 20th century. Hiking and motorcycling were significant activities practiced in the area of the CNP during the first half of the 20th century by city inhabitants.

<u>Second period: urban expansion and urban planning towards the protection of the area</u>

The very first information we have regarding the protection of the forest in Collserola goes back to 1884 when a commission formed by the Agricultural Institute of Catalonia of Sant Isidre and the

Excursionist Centre of Catalonia, of which Puig i Valls was part, constituted an association against the illegal cut down of forest trees in the Littoral Range (Boada, 1995). The first work of this commission was in fact a plan of 1887 to reforest Collserola (Boada, 1995). The commission also organized a protest at the Spanish Parliament which marks one of the first social events of this kind related to environmental problems in the region (Aldeguer et al., 2008).

It is documented that the links between forest areas and the city of Barcelona have been tight since the last century. At the beginning of the century, the writer and poet Joan Maragall was writing, in an article entitled “La muntanya”, referring to the Tibidabo:

> “ […] how fortunate is a city that is next to a mountain! All its inhabitants will climb it and will return transformed. And in the solitude of the study, in the laid table of the family, in the activity of the industries, in the darkness of the shops, in the noise of the streets and of the large rooms will reign ascended the high view of the mount peak. The straightness of the pine trees, the smell of the bushes, the free harmony of the winds, will live in the soul of the city, so that it will come to feel its mission. That is, to transit on the heights and on the extension of the lands […]”(Maragall, 1909) (own translation)

However, since the beginning of the 20th century, the main threat to the health of the ecosystem of the Collserola range has been urbanization itself (Folch i Guillèn, 1977). At that time, a garden-city with infrastructures for touristic and scientific purposes was planned to be built on the top of the Tibidabo hill (Raspall et al., 2004). Two low density neighborhoods were developed: Vallvidrera (a neighborhood of the Sarrià-Sant Gervasi district of Barcelona) that was starting to be filled with towers, and in La Floresta (a neighborhood of Sant Cugat del Vallès) the first buildings were constructed. Yet, the landscape was still very open and the hills covered by vineyards (Raspall et al., 2004) (see Figure 3.13 and Figure 3.14).

Figure 3.13. View of Vallvidrera and the Tibidabo pick in 1907.
(Source: Raspall et al., 2004)

Figure 3.14. View of Vallvidrera in 1902.
(Source: Raspall et al., 2004)

Figure 3.15. View of Vallvidrera Today
(Source: www.barcelonas.com)

In Barcelona, as elsewhere and as analyzed by Martínez-Alier (1996), two conflicting urban planning models were contending at the end of the 19th and beginning of the 20th century: the rationalist, modern style dominated by cement and glass, with separated zones for working and sleeping and with a sprawling peri-urban area (of which Le Corbusier became the main international representative at a later stage), on one side; and, on the other, a reactionary, more “romantic” approach which aimed at preserving the identity of peri-urban agglomerations and historical environments by maintaining, for instance, green and agricultural belts around the city (a group mainly represented by Patrick

Geddes, his disciple Luis Mumford and the concept of Regional Planning). Seeing the city in its regional context it turns out, Martínez-Alier (1996) argues, that the Regional Planning approach is more ecological and thus scientific as it sees the city located within a region and not as an isolated system, but had failed in its time to become the main model of urban planning in Barcelona. Here, the rationalist approach dominated and was implemented quite early with the Cerdà plan of the Eixample, even if opposed to it was the Geddes' approach represented by Cebrià de Montoliu with the Catalan Garden City Civic Society, who however came later and confined himself to exile in the US beginning of the 20th century (Martínez-Alier, 1996).

Despite this, we argue that the perspective of the Regional approach was still able to influence urban planning in Barcelona as time went on, especially with the preservation of the Collserola range. By highlighting the steps that led to its protection, the park can be seen as the result of a struggle between the two opposed approaches to urban planning mentioned above and the final predominance and application of the more reactionary approach of Patrick Geddes against the rational practice of uncontrolled or sprawling urban expansion. This was possible especially thanks to the work of Nicolau Maria Rubió i Tudurí, a Spanish architect, landscape designer, urban planner and writer, as well as that of his brother Santiago Rubió i Tudurí. These were commissioned with the development of the "Plan de Distribución en Zonas del Territorio Catalán" (Regional Planning) in 1932, which, by following the orientation of the municipality of Barcelona who prepared in 1929 the plan "Gran Barcelona", considered the area of the Collserola range as hardly adaptable to other uses and could therefore be more effectively preserved (Rubió i Tudurí and Rubió i Tudurí, 1932). It suggested a system of protected areas and forest reserves for the city (see Figure 3.4) and the region. These objectives were partly reaffirmed by the work of a Catalan geographer, Pau Vila, who foresaw in 1937 the Collserola range to be a great Metropolitan park (La Vanguardia, 1983a).

Around Barcelona in the 19th century and the first half of the 20th, unplanned urbanization in areas excluded by the Cerdà Plan of 1859, such as on the foothills of Collserola close to Barcelona and the middle hills of Tres Turons, was taking place out of the reach of essential urban infrastructures by low income, immigrant population in search of jobs (Sotoca Garcia and Carracedo García-Villalba, 2011). Besides the plan of Rubió i Tudurí, the expansion of Barcelona continued its colonization of the surrounding hills also during the post-war period (Raspall et al., 2004).

Later on, in 1963, the "Plan General de Ordenación de la Provincia de Barcelona", based on the Regional Planning of Rubió i Tudurí, was prepared and constituted a milestone towards the protection of the Collserola range (see Table 3.7). However, the great acceleration of urbanization processes increased throughout the 1960s and 1970s when the population of the Metropolitan Area of Barcelona

grew from 2 to 2.7 million inhabitants with a growth rate of 3.23% (PMPC, 1990). This quasi-exponential growth highly threatened the Collserola area by urbanizing its periphery and by encircling and isolating it from other surrounding natural areas (Argelich and Recio, 2009). Urbanization within the park developed especially along the axis Vallvidrera-Les Planes-La Floresta-Valldoreix-Sant Cugat, while an extension of the road network in the park further encouraged frequent motor car races.

We have then to wait for 1976 to see the Regional Planning vision translated into legislations. The Plan General Metropolitano (PGM) for Barcelona of 1976, also based on the Plan of Rubió i Tudurí, aimed at halting urbanization in the not-yet urbanized areas by including 130 km^2 of woodland, 28 km^2 of agricultural land, where the Collserola range figures as 50% of the total protected area included in the Plan (PMPC, 1990). Still, the mentality at that time was strongly driven by “developmentalism” and the construction of different infrastructures was included in this Plan.

Later on the area was regulated by the “Pla Especial d'Ordenació i Protecció del Medi Natural del Parc de Collserola” (PEPCo), a plan for the special protection of the area approved in 1987. Besides the considerable dimension of the park and its location at the centre of a Metropolitan area, one of the main reasons advanced for its conservation was the claim of its degree of naturalness (PMPC, 1990). The Plan specifically aimed, in addition to the conservation of biological diversity, at improving the quality of different habitats (i.e. natural, semi-natural and scenic farmlands) acknowledging, in this way, the long and intense human intervention which had occurred in the area (PMPC, 1990). From its implementation onwards, environmentally harmful human activities like motorcycling or hunting, widely practiced in the park in the past, were gradually forbidden or restrained to specific areas (as in the case of hunting), with the aim to facilitate the re-naturalization of the park.

To note is that five out of the nine municipalities which have their boundaries within the park and had decided its creation had left-wing governments at the time of the PEPCo. The Corporación Metropolitana who prepared the plan was, at the beginning of the 1980’s, mostly socialist (La Vanguardia, 1983b), while Pasqual Maragall (socialist party) was mayor of Barcelona at that time. Socialists also opposed to the draft Plan Territorial Metropolitano (PTM) of 1998 which was seen to favor urban sprawl, especially in the Vallès Occidental county.

The PEPCo, however, still encouraged improvements in the transport system and of the connections between the different communities located around the park. Amongst other, three tunnels were still planned to cross the Range and connect the municipality of Barcelona with the region of the Vallès Occidental behind Collserola (PMPC, 1990). It is in fact since the 1950s that the plans for the expansion of urban areas and industries in the Metropolitan region depict the range as an obstacle to

urban growth and development which would be overcome with the drilling and construction of tunnels. This perception of the Sierra continued to be preponderant all along the 1960s-70s. However, among the tunnels planned in the PGM only that of Vallvidrera has been built to date. Nonetheless, the losses in riparian woodlands and crops brought about by its construction have been considerable (Cahill et al., 2003). Concern for the ecological impact of the tunnel of Vallvidrera was in fact raised in the mid-1980s when its construction was started and assigned to a building contractor. Two reports on the environmental impacts of the tunnel were produced and led to conflicting results: the socialist local authorities stressed the serious changes that could occur in Collserola after the construction of the tunnel, while the Barcelona Regional Council stressed more the socio-economic benefits that could derive from it, acknowledging at the same time the potential offered by the park to the surrounding cities' inhabitants in terms of recreational services, water regulation, erosion control and improved air quality. Later on, in 1990, the Ronda del Dalt, a highway which runs through the north of Barcelona, was completed and a year later the Vallvidrera highway, which connects Barcelona with Sant Cugat del Vallès and other cities of the Vallès county, also became operational.

Despite the construction projects, the process of reforestation further intensified after 1987 also thanks to ad-hoc reforestation plans. By contrast, in the Metropolitan area surrounding the park, between 1972-1997, the urbanization process continued to increase, doubling in terms of the surface occupied, going from 23.000 to 48.000 ha and hosting a population which grew from 3.6 to 4.2 million inhabitants respectively (Molina Vacas, 2010). This led to continued pressure on the park.
Illegal urbanization (i.e. in the form of shanty towns not connected to services) had been frequent also in this period due to the expansion of the periphery of the city and the "de facto" tolerance showed by the local administration, as well as the hope to see these constructions legalized in a near future (La Vanguardia, 1985). This was the case for the municipalities of Sant Cugat del Vallès, Molins de Rei and Les Planes. Around a thousand urban plots were planned of which 300 were soon built up (La Vanguardia, 1986a). The illegally built areas were, in a second moment, either expropriated, bought by the Corporación Metropolitana de Barcelona (CMB) or legalized (e.g. the case for Les Planes due to its high proportion of houses already built). Some illegal constructions started to be demolished in the mid-1980s to consolidate the area of the park. For example, 26 illegal constructions were demolished in Turó del Quirze, part of Molins de Rei, in 1986, while an agreement was reached to expropriate 15 constructions in Parc Gavà. Canons i Orioles (Sant Boi de Llobregat) (with 313 lots of which 246 underwent construction) and Can Barat-a (Sant Cugat) (with 104 plots with buildings out of 204) were re-qualified, while Les Planes (Sant Cugat) was legalized as 360 plots out of 400 were already built (La Vanguardia, 1986a, 1986b). Later on, in 1995, the urbanization of Sant Medir was also rendered illegal.

Due to this pressure, the Collserola Park required, differently from the Fontainebleau Park which is its counterpart in Paris and unregulated due to the high social consensus, legal protection and was included in the "Pla d'Espais d'Interès Natural" (PEIN), a plan for natural areas of particular interest approved in December 1992 by the Generalitat de Catalunya. The PEIN further restrains urbanization and construction process in the park.

In September 2006 the park was included in the Natura 2000 European Network of protected areas and, after a series of protests (see next section), the Collserola range was declared natural park on the 19^{th} of October 2010 passing from covering a surface of 7.176 ha to 8.262 ha. Overall, the reasons behind these stricter regulations was the lack of strong social consensus on the preservation of the ecosystem, especially in some boundary areas.

Overall, the preservation of the area along the 20^{th} century can be seen as the product of political decisions taken at the local scale which incorporated the notion of Regional Planning and marked the legislative steps towards the creation of the natural park. In line with the description of Parés-Franzi (2006), in the Metropolitan Region of Barcelona urban parks are in fact more frequent in large, dense municipalities with left/green local governments.

Table 3.7. Main legislative steps towards the creation of the Collserola Natural Park. (Sources: own Table from various sources, including http://www.parcnaturalcollserola.cat/ and http://www.gencat.cat/mediamb/revista/rev27-cast.htm#boada - retrieved on 23rd September 2014)

Date	Body of legislation/steps towards the protection of the park	Actions	Drivers	Authority/author
1929	Plan "Gran Barcelona"	Driver of the transformation of Barcelona towards a metropolitan city. Establishes Montjuic and the urban parks.	International Exposition	Influenced by Jean Claude-Nicolas Forestier and Nicolau Maria Rubió i Tudurí
1932	"Plan de Distribución en Zonas del Territorio Catalán" (Regional Planning)	The area was hardly adaptable to other uses and could be more effectively preserved requesting that the urban planner should take this into account. First attempt to organize and plan the Catalan territory (Font, 2000). Project of the Garden-city.		Nicolau Maria Rubió i Tudurí and Santiago Rubió i Tudurí (commissioned by the Generalidad de Cataluña)
1953	"Plan Comarcal"	Set aside some areas, such as parklands and woodlands, as reserves intended for public use, including Collserola		
1963	"Plan General de Ordenación de la Provincia de Barcelona"	Establishes a list of possible natural parks including Collserola (or the Tibidabo). Based on the "Regional Planning" of Rubió i Tudurí.		Comisión Superior de Ordenación Provincial de Barcelona
1976	Plan General Metropolitano (PGM)	Main reference in terms of territorial planning in the Metropolitan Area of Barcelona. Defines the area needing to be preserved as forestal parks. Article 208, section "Normas urbanísticas" of the "Plano General Metropolitano (PGM), establishes to develop, for each park of the region, a Special plan.	Slow down urban expansion and ensure quality of life in the area.	
2nd April 1985	Art. 22 and 85, Law 7/1985, of the 2nd of April	Establishment of the Patronato Metropolitano del Parque de Collserola		Consejo Metropolitano de la Corporación Metropolitana de Barcolona

Date	Body of legislation/steps towards the protection of the park	Actions	Drivers	Authority/author
1st October 1987	Pla Especial d'Ordenació i Protecció del Medi Natural del Parc de Collserola (PEPCo), (Llei d'Ordenació Territorial de la Generalitat.)	Two objectives: 1) conservation of natural resources and ecological balance; 2) become the main source of leisure activities of the Metropolitan area of Barcelona. It defines a zones for the park: natural areas; semi-natural areas with scenery value; and agricultural zone with scenery value. It also defines the road network within the park.	It plans the preservation of the park for its social and bio-physical value.	Corporación Metropolitana de Barcelona followed by the Entitat Municipal Metropolitana de Barcelona
14th December 1992	Approval of the "Pla d'Espais d'Interès Natural (PEIN)"	Defines 165 natural areas to be preserved in Catalonia including Collserola. For it, it includes 7.654 hectares of which only 6.500 are defined as forestal park and are managed by the Consorci del Parc de Collserola. Definition of the categories of urban soil and land developable for urban purposes. In addition to the PEPCo, it defines which zones can be urbanized and which cannot. It is based on the concept of "Landscape Ecology" and the work of Richard T.T. Forman on the territorial mosaic.	The main reasons behind the conservation of the park are: that it contains still a high diversity of natural landscapes, flora and fauna; that it is an important forestal reserve; and its proximity to the metropolitan area of Barcelona. Its location within a metropolis which threatens the area making it a vulnerable social-ecological system.	Generalitat de Catalunya
5th September 2006	Natura 2000	All the territory defined by the PEIN as part of the park is declared as a Natura 2000.	It was included in the network for the reasons that it presents species and habitats of interest for the European Community.	Generalitat de Catalunya
2010	Plan Territorial Metropolitano de Barcelona	Includes the Sierra of Collserola as an area of special protection in the systems of green areas	For the value of the natural and agricultural land within it.	
19th October 2010	Natural Park	It aims at reinforcing the conservation of the park with respect to a principal urban function. It introduces criminal protection vs. illegal behavior. It was decided by the Generalitat in 2005. The plan at that time was to include in the area of the park in total 1.231 hectares more of which 521 should consist of forest, 11% of the total surface	An additional primary reason cited in the Decree for definition of the area as a Natural Park derives from the pressures which the ecosystem undergoes due to the vicinity to the surrounding urban area (Generalitat de Catalunya, 2010).	Generalitat de Catalunya

(Table 3.7 continued)

Third period: nature conservation, social movements and continuous urban pressure

Since the 1960s urbanization all around the boundaries of the park highly threatened and modified its ecosystem till the present days. From our assessment of land cover changes, it resulted that between 1956 and 2009 sealed surfaces passed from covering 148 ha of the area of the park to approx. 411.4 ha. The effect of urbanization in and around the park caused the isolation and fragmentation of the ecosystem, susceptible to compromise its health. The reduced migration of certain species and the overall reduction in genetic variability which follows can in fact lead to the decline in biodiversity and to the alteration of certain ecosystem processes involved in the supply of services (e.g. nutrient cycling) (Debinski and Holt, 2000). For these reasons, it is documented that natural reserves alone do not suffice to protect biodiversity and ecosystem health as these are often too few, static and isolated, and improved habitat connectivity is needed (Fischer et al., 2006).

The shanty towns which lacked of public facilities and started to be constructed before the definition of the area by the PGM became a strong focus of urban conflicts. Based on this tradition, most of the social movements grew quickly at the end of the 1960s and intensified with the transition to democracy around the mid-1970s (Calavita and Ferrer, 2000). As we have seen, the urban pressure and the permissions from the initial legislation of the second half of the 20th century to build in certain areas of the park often caused the boundaries of it not to be fully defined. Other human activities such as forest burning, the introduction of exotic species (i.e. 62 exotic, invasive species have been found in the Collserola Park (Crespo et al., 2008)), forestry machinery, livestock, hunting uses, industrial activities, disturbances which are specifically common to most urban and peri-urban protected areas (Trzyna, 2014), all contributed to the modification of the environment (Barcelona Regional Council, n.d.). It is against these threats that social movements have mobilized ecological and conservationist arguments in the past decades until today, and have significantly contributed to the preservation of the area. Social movements in Collserola were unsatisfied with the legislation of the 1970s and 1980s, which still allowed urban related construction processes in the area.

As a precursor, the first course in ecology at the University of Barcelona was initiated in 1967 by Ramon Margalef who was notable amongst the first ecologists, highly multidisciplinary in his approach, and well connected to the academic environment overseas (Cardona, 2009). Struggles to reduce sources of pollution for a more sustainable urban transport system or to protect green urban areas have a relatively long tradition in Barcelona (Argelich and Recio, 2009). Green movements in Catalonia appeared simultaneously or after the fall of the Franco regime in 1977. The democratic transition and environmental concerns were often some of the pretexts to protest against the regime (e.g. the case of protests against floods in the Llobregat river). There was in fact an association

between “catalanism”, socialism and environmentalism (Cardona, 2009). The first group of ecologists was created in 1976 by Ramon Folch, from the municipality of Barcelona, and Salvador Filella from the Zoo of Barcelona (Cardona, 2009). The group was called “Lliga per a la Defensa del Patrimoni Natural (DEPANA)”. Most of the environmental movements that developed in Catalonia seem to have mainly a “Not In My Back Yard” (NIMBY)-type of approach, confronting very localized problems (such as the location of infrastructures and changes in the landscape), failing to connect with a broader critique of the processes behind such aggressions (Cardona, 2009). This approach seems to have been adopted also by the social movements for the defense of Collserola but led nonetheless to the implementation of important legislative steps such as the declaration of Collserola as Natural Park.

According to activists, the range was seen by local authorities basically as a reserve for the needs of the urban city activities (e.g. depots, cemeteries, golf courses, more houses, roads, tunnels). Social movements in the defense of Collserola got thus traditionally opposed to different issues such as: the extension of the transport system (e.g. Vial de Cornisa); the construction of tunnels (e.g. of Horta or the central tunnel); the enlargement of the Carretera de las Aigües which, it was argued, would cause more people to easily reach the park; urbanization; the dumping of toxic wastes; hunting in the park; the construction of a Roller coaster in the amusement park of the Tibidabo, which would cause the cut of centennial oaks; or to the fragmentation and isolation of the park, asking instead for a better connection with the two other neighboring parks of Sant Llorenç del Munt and of the Garraf. Actions against these threats are mainly taken by the “Plataforma Cívica per a la Defensa de Collserola” who organized for instance a march which involved approx. 2000 people on the 13th of April 2008 to ask for an increase in the protection of the area against urban expansion. These actions played a significant role in achieving the declaration of the area as a natural park in 2010. In fact, at least since the 1990s, local associations[18] have been pressing the authorities for the enlargement of the park and the establishment of a natural park as the only way to avoid the constructions planned by the PGM of 1976. It was demanded that the area of the park be accrued of 1100 ha to include: the Torre Negra and the Queixalada (Sant Cugat), Turó de Montcada and Can Cuiás (Montcada), Pi de Balç (El Papiol), Torre Vilana, Font del Gos or the Parc del Casteli de l’Oreneta (Barcelona) (La Vanguardia, 1999). The main argument behind these demands was the conservation of a healthy ecosystem and its biodiversity against city-related uses of the land.

A particular case of contested space for preservation vs. urbanization is that of the Torre Negra in the municipality of Sant Cugat for its strategic and sensitive conditions. In the center of the land is the

[18] i.e. Amics de Collserola, Adenc, Depana, Collectiu Agudells-Ecologistes en Acció, Ecopacifistes de Nou Bar, Grup de Natura del Club Muntanyenc Sant Cugat, Grup de Natura Les Planes y Via Verda-Cerdanyola

building which gave its name to the whole area, the Torre Negra, a construction of about a thousand years old. There is also a bicentennial pine called “Pi d’en Xandri” which is considered an emblem for the town of Sant Cugat. According to various newspaper articles appeared on “La Vanguardia”, the tree was attacked in early 1997 and was close to be cut down, an event seemingly related to the ongoing conflict in the area. In the conflict, 43 landlords led by Núñez y Navarro, who owns 60% of the land in question, reclaimed the right to build on this rural area. The municipality of Sant Cugat and some associations (e.g. “Salvem Torre Negra”) got strongly opposed to the project. The local population and social movements have been protesting against the possibility to urbanize the area since the early 1990s. In early 2009, the case reached the Court of Law of Catalonia (the “Tribunal Superior de Justicia de Cataluña”) which defined the land as unsuitable for urbanization. In 2010 the land was included in the area of the Natural Park with no compensation for the property owners. The construction of the road Vial de Cornisa within the park was also cancelled by the Metropolitan authority in 2012 under the pressure of social movements and some Municipalities with their boundaries within the park that had joined the protests.

At present, the ecological equilibrium of the park is threatened by the ever expanding business around recreational activities (such as day and night sportive cycling on well-trodden paths and in the middle of the forest), in addition to isolation as urban pressure is an unsettled threat. Most of the weekend excursions and walks by city people take place in these green spaces (Douglas and Box, 2000). Of the two million annual visitors of Collserola, 80% come from one of the nine municipalities which have boundaries inside the park and 41% come from Barcelona only (IERMB, 2009). Visitors from Barcelona enter the park via Tibidabo (32.5%) or the station at Vallvidrera (17.5%) (IERMB, 2009). The vicinity of the biosphere reserve to the urban area reduces long distance movements: 35.1% of the visitors reach the area on foot and 11% by bicycle while 18.8% of the users from Barcelona get to the park by public transport, taking advantage of the railroad (IERMB, 2009). The question posed is whether the park should remain categorized as natural or be considered more as a metropolitan park. Collserola is in fact considered as one of the largest metropolitan parks in the world (La Vanguardia, 1992).

Today the consortium for the management of the CNP aims at keeping it as the great urban park of the metropolitan area while trying to maintain its natural and cultural values through sustainable use (Barcelona Regional Council, n.d.). The project of the 16 gates of Collserola (“Les 16 portes de Collserola”), for instance, intended to connect and improve the access to the park from the city, thus exemplifying an attitude that aims at opening the park to the city looking at it more as a metropolitan park. The project, with the goal to “re-naturalize” the city, is thought to be a pretext to expand the so

called “green marketing”[19]. It would on the other hand improve the access to the park from the city and diminish the feeling of the lack of green perceived in Barcelona since decades now (La Vanguardia, 1988). This met the opposition from the local social movements which see the ecosystem threatened by a potential further increase in the number of visits. The number of people engaging in group sports has quadrupled between 2005 and 2013, bringing the total number of persons practicing it to 60.000 a year (La Vanguardia, 2013). Social movements complain about the disturbance that this causes to the environment and to nocturnal animals in particular. Since the early 1990s to date, in fact, the increased number in visitors and recreational activities, according to Marià Martí (director of the ‘Consorci del Parc de Collserola’), decreased the number of animals present in the area (La Vanguardia, 2010). Social movements call, on the other hand, for an increase of the area of the Natural Park to 9000 ha.

Overall, local environmental associations protested that the park is seen by local authorities more as a central metropolitan park than as a natural park. Since the mid-1990s the idea to limit the access to the park was taken into consideration also by some politicians to maintain and enhance the ecosystem and its biodiversity (La Vanguardia, 1995), while to date the social value of the park is still not fully defined or recognized and the concept of ES has not been mobilized. It is again the opposition between different ideas of nature (see for instance Gobster, 2001) in the urban contested space that shapes the present understanding and future uses of the park.

3.4 Discussion and conclusions

The assessment of three regulating and one cultural service showed how the livability of Barcelona increases thanks to the presence of the park when compared to other cities. The green area is easily accessible from Barcelona and wind paths contribute to the movement of pollutants from the city to the park and, vice versa, of cooler air from the park to the city, especially in the summer season. Despite the clear relevance of recreational services, the park also contributes to reduce the impact of natural hazards, such as heat waves in the summer and floods in autumn in the city through the regulating services it provides to the population of Barcelona. Our analysis of ES in Barcelona confirms the results obtained by Larondelle et al. (2014) that often city population density is so high that most of the supply of ES happens outside the cities’ administrative borders and, thus, urban areas depend on their hinterlands for a good environmental quality.

[19] http://www.ecologistasenaccion.org/article21684.html (Retrieved on 29th of April 2014)

However, these benefits should not be taken for granted. As demonstrated by the land use and historical analyses, the CNP hosts in fact a constructed forest which is the result of socio-economic and political processes, such as agricultural land abandonment, the prevalence of the Regional urban planning vision and the action of social movements. What appears now as a wild and pristine environment is far from being so. The park is the result of the opposition of different political forces. Our analysis identified three phases through which the area reached the actual appearance, passing from being a source of provisioning services (i.e. food and timber) to one of recreational and regulating services for city inhabitants. The present socio-ecological system of Collserola is in fact first the result of accidents leading to agricultural land abandonment due to the Philloxera plague and to industrialization (first phase), followed by government intervention and the affirmation of the Regional Planning vision of Patrick Geddes in the planning of the Barcelona region (second phase). This then sets the pace for the increased protection of the area as Natural Park reached under the pressure and actions of social movements (third phase).

Ultimately, parks and nature mean different things for different actors and in different epochs. These different meanings led to, and are the product of, conflicts amongst these same different understandings. This is particularly evident in urban areas due to the concentration of human activities (Gobster, 2001). Located at the centre of a Metropolitan area, the CNP is in fact a contested space especially in the last decades.

Finally, although in Collserola the notion of ES has not been mobilized so far for its protection, these services play a significant role in the well-being of the population of Barcelona but their supply cannot be taken for granted or be considered an incontrovertible reality. We argued how ES supplied by the Collserola forest are the result of socio-economic and political processes between conflicting interests and of historical processes which can be traced back. This has implications regarding the commodification of nature. In fact, as the Collserola forest was collectively developed, it would not be appropriate, for example, to enclose it now and claim payments, fees or other pecuniary contributions for its ES.

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CHAPTER 4 . Urban watershed services for improved ecosystem management and risk reduction, assessment methods and policy instruments: state of the art

Abstract

Under scenarios of increasing unplanned urban expansion, environmental degradation and hazard exposure, the vulnerability of urban populations, especially of their poorer segments, needs to be tackled through integrated economic, social and environmental solutions. Basing our analysis on the concept of ecosystem services, we suggest that urban areas would benefit from a shift in perspective towards a more regional approach, which recognizes them as one of many interconnected elements that interact at the watershed level. By integrating an ecosystem approach into the management of water-related services, urban management policies can take a first step towards fostering an improvement of the health of upstream and downstream areas of the watershed, activating environmentally sound practices which aim at guaranteeing the sustainable and cost effective supply of services. These strategies can for instance be supported by using payment schemes for ecosystem services or similar strategies, allowing for the redistribution of resources among communities in the watershed. From our analysis it results that, through the recognition of the primary role played by watershed ecosystems, cities can benefit from an enlarged set of policies, which can help strengthen the supply of essential environmental services, while reducing the vulnerability of its population and contributing to the maintenance of healthy ecosystems.

Keywords

Urban watersheds, ecosystem services, water supply and sanitation, disaster risk reduction, valuation

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4.1 Introduction

Urban areas have established since ancient times strict connections with flowing water and water bodies. Old civilisations have settled along rivers to benefit from drinking water, sanitation and irrigation. Trading relied on waterways for the transportation and exchange of goods. This centrality was partly lost with the industrial revolution (Silva et al., 2006) and the advent of other means of transport. Moreover, as a consequence of industrialization, discharges and abstractions of water changed in quantity and in kind, and rivers have progressively been degraded and polluted. Since the beginning and middle of the 19th century, increasing quantities of nutrients and metals have been released in water bodies through point (industrial activities and urban sewages) and non-point sources (agriculture). Land cover changes in watersheds have occurred and intensified through the conversion of forest areas and wetlands, first into agricultural land and then into sealed surfaces (i.e. buildings and roads). Engineering works, such as the canalisation of river beds and the construction of dikes, have changed the shape and hydrology of rivers around the world.

Taking for granted the unlimited capacity of nature to provide these services, urban areas have been among the main drivers of environmental change in the past century. According to Srinivas (2013) there is currently no single environmental problem whose causes cannot be traced back to urban areas. As expanding cities rely on a wide range of ecosystem services for urbanisation and urban activities, the trend has been to seek for services from ever more distant areas or to substitute them with technological solutions. Both options imply increased costs for short-term measures and, often, additional environmental degradation. However, awareness is growing about the wide range of benefits urban areas actually obtain from surrounding ecosystems, for example from healthy watersheds, in terms of water purification, water regulation, timber, food products, and cultural services (e.g. recreation).

Ecosystems are defined as “a dynamic complex of plant, animal, and microorganism communities and the non-living environment interacting as a functional unit”, of which humans are an integral part (MA, 2005). Cities themselves can be considered as ecosystems (see Pickett et al., 2011 for a definition and description) however, their dependence on the wide range of environmental services that originate at the local as well as at the regional scale, makes them part of larger and more broadly defined ecosystems. There are numerous definitions of what ecosystem services are (see Braat and de Groot, 2012). We adopt the one provided by the MA (2005) which describes them as “the benefits people derive from ecosystems” and distinguishes among provisioning, regulating, supporting and cultural services. The worldwide loss of ecosystem services affects the well-being of human

communities in a variety of ways, including by contributing to an increase of exposure and vulnerability to water scarcity and natural hazards, in particular for poor urban populations.

By bridging environmental and socio-economic perspectives and highlighting the dependence of human communities on well-functioning ecosystems, the ecosystem services concept and framework can promote the integration of environmental issues into policy agendas . This aspect can be of particular interest for urban areas although the approach has often been of conceptual rather than of direct operational value. As Norgaard (2010) highlights, three main limitations must be taken into consideration when making use of the term in environmental science and policy management, namely: 1) the possibility of providing quantitative information on the relation between the characteristics of ecosystems and the services provided is quite limited, as there is little ecological knowledge around the concept of ecosystem services (i.e. ecologists, generally focusing on single species, populations and communities, provide little quantitative insight into the capacities of ecosystems to function and to provide services); 2) existing knowledge on specific ecosystems is difficult to be transferred because of the complexity and the distinctive traits of every single ecosystem (and this seems to be also due to the local, different influences of human history and local distinctiveness of social systems); 3) very little is known about the trade-offs between the provision of different services and no agreement was found in ecological terms concerning particular threshold conditions of a specific ecosystem (Norgaard, 2010). Therefore, despite the numerous attempts to quantify ecosystem services, the author suggests that the concept should be used to inform environmental policies and decisions with caution and as part of a larger solution to fine-tune environmental policies. While significant efforts and resources need to be invested to perform intrinsically complex ecosystem assessments, their role is however essential in a context, as is the one of environmental decision making, in which system uncertainties and decision stakes are high (Funtowicz and Ravetz, 1993).

Among the range of ecosystem services urban areas benefit from, those related to freshwater are of particular importance. According to the MA “four out of every five people live downstream of, and are served by, renewable freshwater services, representing 75% of the total supply” (MA, 2005). On the other hand, “inland water habitats and species are in worse condition than those of forest, grassland, or coastal systems” (MA, 2005). Thus, to draw the tight link between urban areas and more regional ecosystems, we proceed focusing on freshwater services for urban needs, such as water supply (Section 4.2.1), wastewater treatment (Section 4.2.2) and hydro-meteorological hazard mitigation (see Section 4.2.3). These functions are fundamental for human well-being, and will need increased attention as human population continues to concentrate in urban areas. Cities are in fact

already home to more than half of humankind, with their population expected to attain almost 4 billion in 2015[20] (about 55% of the world total population). In addition, environmental change, and in particular climate variability, are threatening the capacity of ecosystems to deliver these services, adding a further reason for urgency to reversing the trend towards their exploitation and degradation.

In the past century, the growth of cities has often been characterized by the partial substitution of ecosystem services with man-made alternatives, through transformation and replacement of the natural infrastructure providing clean water, wastewater remediation or flood protection. This approach frequently implied short-term visions and a limited capacity of adaptation to future changes like those induced by changing climatic conditions. Resorting to hard infrastructures for the management of natural resources and the prevention and mitigation of hazards often results in the degradation of the environment, the loss of local sources of livelihoods and ultimately in a reduction in the resilience and long-term adaptive capacity of the urban social-ecological system (Smith and Barchiesi, s.d.). For instance, most flood protection systems are based on the magnitude of events with lower return periods than those actually occurring in the long run. These structures distort the population's risk perception and have encouraged significant encroachment of floodplains, further exacerbating long-term risk in urban areas. Water diversions from more distant watersheds through engineered solutions have allowed meeting the needs of continuously expanding urban communities, posing however additional ecological and environmental problems. Finally, huge investment in water technology enables rich nations to adapt and cope with water scarcity without however tackling its underlying causes, whereas poorer countries remain vulnerable (Vörösmarty et al., 2010).

The restoration and improved management of ecosystems at the watershed scale through the integration of the notion of ecosystem services in local policy and decision-making aims at reversing this trend and is here proposed as a sustainable, long-term and cost-effective option enabling to better satisfy the multiple needs of urban areas, including their security, while improving the environmental conditions of watersheds. The pressuring need to manage and allocate water resources in a sustainable manner in the face of increasing environmental degradation and social inequalities, can arguably be better satisfied through the adoption of an ecosystem approach rather than by turning to the construction of additional hard infrastructures. Engineering works have demonstrated to have little consideration of the complex socio-ecological and co-evolutionary processes taking place within ecosystems. They often lead to ecological fragmentation, which compromises the ability of ecosystems to support human well-being through a wide range of services. Evidence also suggests that multi-stakeholder, cross-scale, adaptive water management is better suited to the complex and

[20] http://esa.un.org/unup/CD-ROM/Urban-Rural-Population.htm

dynamic nature of healthy ecosystems, but that this long term, flexible adaptation process hardly takes place when technological solution based on hard infrastructures are put in place (Smith and Barchiesi, s.d.). It should nevertheless be considered that the adoption of an ecosystem approach for the satisfaction of urban needs should proceed along a broader redefinition and reduction of the demand of services from urban populations and the implementation of more sustainable urban activities, both at the local as well as at the regional scale.

Stressing the interconnections between urban areas and their surrounding watersheds has the potential to lead to more informed urban management decisions, as trade-offs between urban activities and the provision of ecosystem services at local as well as at the regional level can be more comprehensively highlighted. Conceiving cities as being part of larger ecosystems opens the path to new policy strategies in the face of socio-economic pressures on elements of the watershed that provide services. In order to support this process, valuation techniques are needed to assess the benefits of ecosystems and inform policy and planning decisions at the watershed level.

While Bolund and Hunhammar (1999) and Gómez-Baggethun and Barton (2013) categorise and describe the range of ecosystem services and disservices that originate within the urban and peri-urban area as well as some appropriate valuation methods for this scale, and Brauman et al. (2007) review hydrologic services, we locate cities within the broader watershed unit, emphasizing their connections and dependences on ecosystems through the analysis of water-related services. We also further develop the perspective described in Bahri (2012) on Integrated Urban Water Management, centring our analysis on the ecosystem approach to the management of urban watersheds. The aim is to illustrate how urban social-ecological systems benefit from and can enhance the quality of ecosystems at this broader scale. We argue that boundaries of urban ecosystems are often strictly connected with the watershed level (e.g. through the hydrology of urban areas) and that urban areas as well as regional ecosystems can derive benefits from such a shift of perspective. Amongst the services provided by watersheds (as listed in Table 4.1), we concentrate our analysis on those aspects directly connected to water supply, wastewater treatment and hydro-meteorological hazard regulation (Section 4.2). We then review relevant valuation techniques (Section 4.3) and policy tools (Section 4.4), and provide some conclusions in Section 4.5. This analysis is mainly done on the basis of a literature review.

4.2 Urban watersheds and ecosystem services

Due to their small size, unbalanced composition and extreme fragmentation, natural components of urban ecosystems only play a relatively minor role in providing services and enhancing the resilience of city dwellers. For instance, urban systems, which cover approx. 1% of land area, receive only 0.2% of global precipitation (possibly due to their location mainly in floodplains where precipitations are fewer compared to mountainous areas) and contribute in this same minor proportion to the global runoff (MA, 2005). At the same time, urban water services account for much higher percentages of the total freshwater abstraction, which reach from 20 to 30% of overall abstractions in countries of the European Community (EEA, 2009). Surrounding and more remote ecosystems support the bulk of a city's functions, even though their connections to urban communities are only mediated and indirect. It is at the watershed scale that most of the ecosystem services urban populations rely on originate and it is at this level that the linkages between the natural environment and the well-being of urban communities are most perceptible.

The focus of our analysis at the watershed level is in order to depict the interconnections between urban life, urban management and healthy watershed ecosystems (as "healthiness" is considered as a prerogative for watersheds to be able to provide services). All cities are located in a watershed and derive services from ecosystems that are found within these units. Therefore, while acknowledging the presence of nested hierarchies of urban ecosystems from the local to the regional scale, we concentrate on a shift in the definition of urban ecosystems which goes beyond the local social-ecological system (i.e. defined by ecosystems with a high density of buildings) and treat urban ecosystem at the regional scale, as any other ecosystem (Pickett et al., 2001), and whose boundaries are set by watersheds.

Watersheds, also known as drainage areas, are the land base from which rain or melting snow converge into a single point and drains as surface and/or groundwater in a water body, such as a lake, a wetland, a sea or a groundwater reservoir. It should be noted that the boundaries of surface watersheds and groundwater watersheds do not necessarily coincide. While surface water is conditioned by topographical features, and thus easy to be delimited, the extension of groundwater watersheds is defined by the: "1) hydraulic properties of the aquifer, 2) input to (i.e. recharge) and outflow from (i.e. discharge) the aquifer system, and 3) geological factors such as formations that block the flow of water and tilted formations that create a flow gradient"[21]. For the analysis of ecosystem services and for the management of watersheds, this divergence can pose relevant

[21] http://www.dnr.state.mn.us/watersheds/surface_ground.html

obstacles. To overcome them, it should be considered that a watershed has two components: a surface and a groundwater drainage.

As the boundaries of surface watersheds are based on topographical and physical borders, their size can range from several thousand square kilometres to a few hectares, spanning across administrative and political borders. Watersheds are usually part of larger systems of tributaries and effluents (FAO, 2007) and can include a variety of ecosystems, such as forests, wetlands, grassland, savannas, alongside with urban systems. Within a watershed, all components of the ecosystem affect the delivery of hydrological services to downstream users. Quality and characteristics of soils determine water infiltration and surface runoff, thus defining the retention capacity and the rate at which precipitation waters cross the watershed and their potential for causing inundations. Vegetation increases the rate of evapotranspiration and storage capacity of soils, while improving the water quality by filtration and absorption of nutrients and contaminants. Wetland ecosystems ameliorate water quality through removal of nutrients, principally nitrogen (N) and phosphorous (P) (Fischer et al., 2007), and, in particular floodplains wetlands, increase the retention capacity within the watershed, reducing the risk of flood hazards and increasing dry season flows (Bullock and Acreman, 2003) (see Table 4.1 for a complete list of watershed services).

Table 4.1. Ecosystem services provided by or derived from inland water systems.
(Source: MA, 2005)

Provisioning	
Food	Production of fish, wild game, fruits, grains, etc.
Freshwater	Storage and retention of water for domestic, industrial, and agricultural use
Fiber and fuel	Production of logs, fuel-wood, peat, fodder
Biochemical	Extraction of materials from biota
Genetic materials	Medicine, genes for resistance to plant pathogens, ornamental species, etc.
Biodiversity	Species and gene pool
Regulating	
Climate regulation	Greenhouse gases, temperature, precipitation, and other climatic processes; chemical composition of the atmosphere
Hydrological flows	Groundwater recharge and discharge; storage of water for agriculture or industry
Pollution control and detoxification	Retention, recovery, and removal of excess nutrients and pollutants
Erosion	Retention of soils
Natural hazards	Flood control, storm protection

Cultural	
Spiritual and inspirational	Personal feelings and well-being
Recreational	Opportunities for recreational activities
Aesthetic	Appreciation of natural features
Educational	Opportunities for formal and informal education and training
Supporting	
Soil formation	Sediment retention and accumulation of organic matter
Nutrient cycling	Storage, recycling, processing, and acquisition of nutrients
Pollination	Support for pollinators

(Table 4.1 continued)

While urban areas depend on provisioning and regulating services supplied by surrounding ecosystems, their expansion and activities directly reduce the capacity of watersheds to provide them. At least ever since the industrialization period, urbanization has been one of the main drivers of the degradation of ecosystem features, causing deforestation and soil erosion. As a consequence, runoff and stream flow increase, groundwater table falls, the sedimentation of rivers and river banks increases and hydro-geological hazards become more frequent. Biodiversity, essential for the resilience of ecosystems and for their capacity of providing services, is mostly negatively affected, often severely enough to run the risk of crossing potentially irreversible thresholds (Thompson, 2011), which, in turn, would have extreme consequences on urban areas. A vision that locates urban areas into watershed systems would thus improve the outcome both of urban planning and of watershed management, not least because urban activities drive land use changes at the watershed scale.

4.2.1 Water supply

Cities are located within surface and below-ground watersheds, and it is at these levels that the hydro-geological processes, essential to the creation and regulation of water supply for urban use, take place. As mentioned, at this scale, waters flow and are enriched with salts and minerals essential for life, while vegetation growing on slopes ensures absorption, filtration and release of runoff (FAO, 2007). The state of the ecosystems located in a watershed therefore affects both the quantity and the quality of water that flows within it, and supports in-situ (e.g. hydropower generation, water recreation, transportation and freshwater fish production) as well as extractive (including domestic) use for the local human community (Brauman et al., 2007).

As urban water demand grows with the increase of urban population (Fitzhugh and Richter, 2004) the pressure on the water system also increases. Productive activities such as farming, grazing or industrial manufacture located upstream, while benefiting urban areas, affect water streams, both above and below the surface, often reducing the range and the quality of services provided downstream.

Local governments and policies can be the driver of upstream ecosystem restoration due to the need to guarantee the supply of water of good quality and at low costs to their citizens. For instance, the rapidly growing urban population and the consequent degradation of the broadleaf forest around the Miyun reservoir (China), the primary source of water for Beijing City, have translated over the last decades into increasing stress for the basin's water resources, provoking a long series of urban water crisis (IUCN, 2010). In these circumstances, the need for restoring or enhancing ecosystem functions for improved water supply was urgent and has been favoured through the development of a series of initiatives and compensation schemes instituted by the municipality, as further described in the next sections and Box 4.8.

The cost-effectiveness of ecosystem-based solution for urban water supply in both the short and the long term has further been demonstrated by a number of cases. The city of New York (US), for instance, is now deriving most of its clean water supply from surrounding watersheds, with no need for technological solutions, thanks to the restoration and appropriate management strategies of upstream ecosystems (for a description of the management strategies see Section 4.4, Box 4.9). In Bogotà (Colombia) a high elevation wetland ecosystem (called pàramo) provides the city with clean water with little seasonal variation and minimal need for treatment (Postel and Thompson, 2005), diminishing significantly the costs that the city should otherwise bear for managing water supply. Other case studies of cities relying on healthy ecosystems, in particular on forests, for freshwater supply are Melbourne (Australia), Istanbul (Turkey) and Singapore, and are extensively described in Dudley et al. (2003).

In general, according to a study conducted by the Trust for Public Land and the American Water Works Association on 27 water suppliers, a 10% increase in the forest cover in the source area would reduce approximately the water treatment costs of 20% (Ernst, 2004). Highlighting the value of healthy watersheds for the satisfaction of urban water supply needs at reduced costs can help making the case for the restoration and sustainable management of ecosystems, avoiding the ecological impacts of expanding water supply systems (e.g. trans-basin water diversions and dams).

The importance of healthy watersheds for the supply of freshwater to cities is perhaps the most evident link existing between urban areas and their regional ecosystem. As summarised in Table **4.5**, the valuation of the availability of clean water for urban consumption has been the subject of much of the research on the assessment of urban watershed services.

4.2.2 Wastewater treatment

Cities also produce significant amounts of outputs (i.e. pollution and wastewater). A further advantage of locating human settlements along water courses consists of the possibility to discharge these outputs into the water and have them carried away from the settlements and decomposed. Ecosystems have provided these services throughout human history, and ecosystem-based solutions, especially in association with technological solutions, still are a recurrent management option for the treatment of outputs. Natural and constructed wetlands are in fact capable of removing sediments, nutrients, and other contaminants from water, and have therefore a fundamental role in the treatment of wastewater, and in particular of urban drainage. The use of wetlands for wastewater treatment is also effective in economic terms as these present lower costs of construction, operation (e.g. reduced energy consumption) and maintenance if compared with conventional sewage treatment, while providing a wide range of other services (e.g. recreation). Ecosystem-based solutions for wastewater treatment might therefore be especially important in developing context, where the availability of financial and human resources for technology-intensive options is lower.

It should be emphasized that, when quantities of outputs are increasing, rivers and water bodies are no longer able to provide their ecosystem functions without compromising the health of downstream ecosystems, which, in turn, affects the health of downstream dwellers. Ecosystem-based approaches should be associated with policies aimed at the reduction of pollutant emissions at the source as well as with technological solutions for the depuration of effluents before immission in water courses, wetlands or lakes.

The trade-off of these solutions lays in the increasing request for space compared with technological solutions, which is relevant for urban areas, where the economic value of land is often high. However, there remain currently only few urban wetlands, which offer a high value in terms of recreational opportunities, local livelihoods and flood protection (Boyer and Polasky, 2004). Ecosystem valuation exercises show that the multiple benefits provided by urban wetlands often outweigh the gains linked with infrastructural development initiatives, which, in addition, can have severe impacts on local livelihoods (see cases in Box 4.1 and Box 4.2). Nonetheless, there are only few examples of research

on the role and value of urban wetlands, and most of them are based on hedonic pricing (Boyer and Polasky, 2004) (see next sub-chapter).

Box 4.1. The Nakivubo Swamp, Uganda

The Nakivubo Swamp in Uganda provides wastewater purification, especially through nutrient retention, to the country's capital Kampala's sewage. A Study of the International Union for the Conservation of Nature (IUCN) estimated the wastewater and nutrient retention functions of the wetland through two different methods: the avoided costs of replacing natural wetland functions with manmade alternatives and the foregone expenditures on mitigating or offsetting the effects of wetland loss. The results of the valuation showed an economic value ranging between US$ 1 million and US$ 1.75 million a year, depending on the analysis method used, but which results in both cases in a net benefit (IUCN, 2003). The Wetlands Inspectorate Division and IUCN showed that a sewage treatment plant that would substitute the Wetland's function would cost over US$ 2 million to maintain each year. In addition to requiring the local community to bear a cost for a service the wetland was already providing, the establishment of a treatment plant would also have caused significant loss of livelihoods for the local population (IUCN, 2003).

Box 4.2. The Sanyang wetland, China

For the Sanyang wetland, which is located in the East China coastal zone along the Oujiang river and close to the centre of Wenzhou city, ecosystem services have been indicatively estimated by Tong et al. (2007). The water purification service accounted for 43% of the value of the wetland, circa 3900 US$ ha-1 yr-1, followed by disturbance (hazard) regulation, circa as 1250 US$ ha-1 yr-1 as the Wenzhou city and the Sanyang wetland are occasionally impacted by typhoons, heavy rain, and floods. The wetland has a surface of 1141 hectares which means that the total value of water purification performed by the wetland is US $ 4.45 million a year.

The use of wetlands for urban wastewater treatment has a long history, and has been used extensively by some cities in the early stages of urbanization, for instance in Berlin (Hobrecht, 1884), one of major European cities at the end of the 19th century, or in Australian cities (Brix, 1994). In many

cases these systems have been abandoned completely nowadays, but a scientific review of technologies for the construction of wetlands and the choice of adequate plants has contributed to a revival of these ecosystem functions either as a last step of a biological and/or chemical form of sewerage treatment, or as extensive plants serving particular, for instance seasonal, needs (Brix, 1994).

4.2.3 Hydro-meteorological hazard prevention and mitigation

Over the last decades, disaster[22] occurrences have been steadily on the rise, affecting an increasing number of people and causing an increasing amount of losses (Guha-Sapir et al., 2011) (see Figure 4.1 and Figure 4.2). While there is a certain degree of confidence that the intensity and frequency of hazards have increased since 1950 (IPCC, 2012), it is clear that the raise in natural disaster losses are primarily due to socio-economic drivers (i.e. heightened concentration of vulnerable communities in hazard-prone areas, displacement, discrimination and corruption) and to environmental degradation (Lewis and Kelman, 2012).

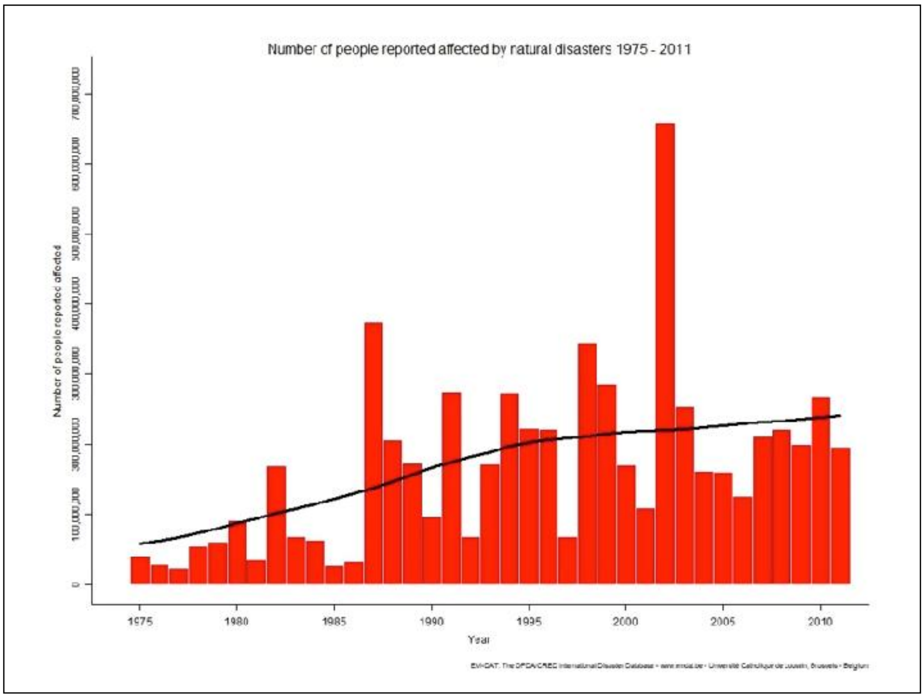

Figure 4.1. Number of people reported affected worldwide by natural disasters between 1975 and 2011.
(Source: "EM-DAT: The OFDA/CRED International Disaster Database www.emdat.be – Université Catholique de Louvain – Brussels, Belgium")

[22]According to CRED: a disaster occurs when at least one of the following four criteria is fulfilled: “10 or more people are reported killed; 100 people are reported affected; a call for international assistance; a declaration of a state of emergency” (http://www.emdat.be/criteria-and-definition)

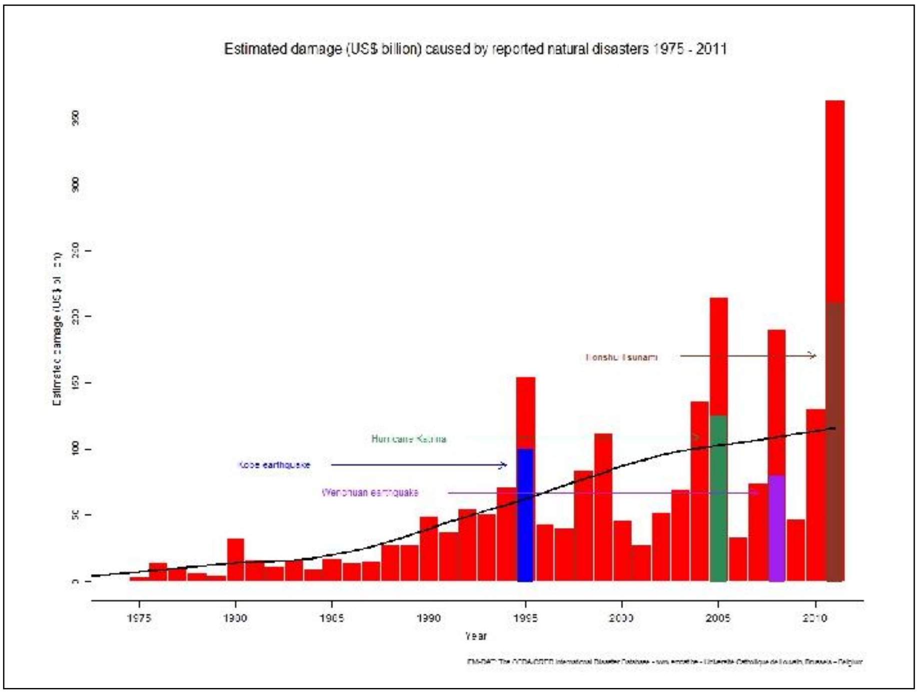

Figure 4.2. Estimated damages in US$ caused by reported natural hazards worldwide between 1975 and 2010.
(Source: "EM-DAT: The OFDA/CRED International Disaster Database www.emdat.be – Université Catholique de Louvain – Brussels, Belgium")

Weather-, climate- or water-related hazards (such as droughts, floods, windstorms, tropical cyclones, storm surges, heat and dry spells, droughts, landslides and wild fires) cause the highest share of damages worldwide. As is shown in Figure 4.3, floods and storms are at the origin of the majority of disasters that occurred between 1980 and 2011. In 2011 only, hydrological[23] disasters were by far the most frequent (52.1%), followed by meteorological[24] ones (25.3%) (Guha-Sapir et al., 2011). This highlights centrality of the processes that take place in watersheds with respect to disaster risk reduction.

[23] “Events caused by deviations in the normal water cycle and/or overflow of bodies of water caused by wind set-up” (e.g. floods, mass movements – wet) (source: http://www.emdat.be/classification)
[24] “Events caused by short-lived/small to meso scale atmospheric processes (in the spectrum from minutes to days)” (e.g. storm) (source: http://www.emdat.be/classification)

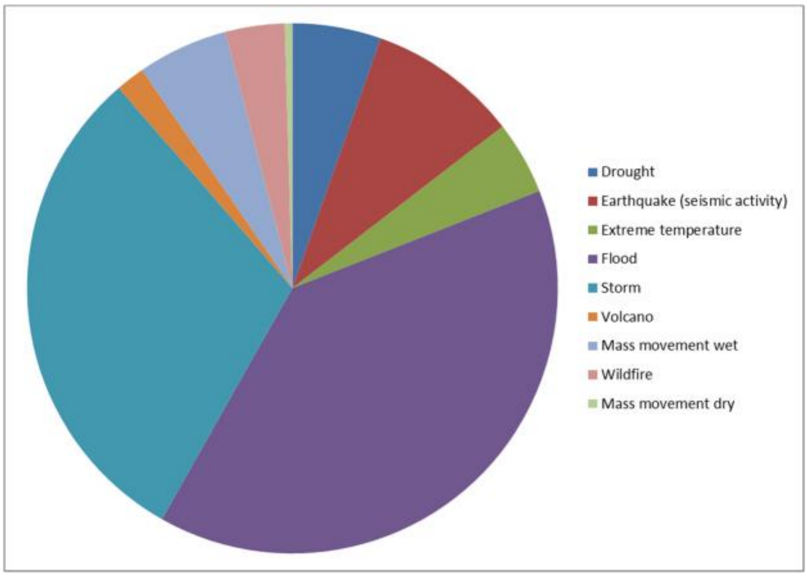

Figure 4.3. Number of disasters per hazard type during the period 1980-2011.
(Source: "EM-DAT: The OFDA/CRED International Disaster Database www.emdat.be – Université Catholique de Louvain – Brussels, Belgium").

Urban areas are no exception to these trends, with a reported growth in the number of disasters and in particular those associated with weather events (such as heavy winds and rains, floods, landslides and fires) (Dodman et al., 2013). The total increases in economic and human damages are in fact mainly associated with growing exposure of vulnerable populations in cities (Lall and Deichmann, 2009). In Colombia and Peru, the urbanization rate of municipalities showed positive correlation to both hazard exposure and vulnerability (Serje, 2010). According to Hupper and Sparks (2006), urbanisation of hazard-prone areas, alongside with environmental degradation are the prominent causes of the higher impacts of hazards (See also Barredo, 2009). These observations are consistent with the interpretation of risk as the product of social processes, which has become prevalent over the past decades. Disasters are the result of the interaction between natural and economic, social and political processes (Cannon, 2008). Urban expansion and development are thus generating new patterns of hazard, exposure and vulnerability (see Box 4.3). In these conditions, the adoption of an ecosystem approach that recognises the functions of in-situ and surrounding ecosystems would significantly reduce exposure and vulnerability of urban populations. It should additionally be noted that, while major hazardous events have the potential of causing widespread destruction in urban areas (e.g. Hurricane Katrina in New Orleans or the Tohoku Earthquake in Sendai), cities are also the scene for many smaller events, mostly concentrated in informal settings were the urban poor reside, which often go completely unrecorded (Dodman et al., 2013).

The regulation of the hydrological cycle at the watershed scale is of fundamental importance for cities. Healthy or well-managed forests and soils can significantly contribute to the regulation of water flows, storing and slowly releasing waters, thus buffering the impacts of extreme events, including in downstream urban areas (see Depietri et al., 2011 for a review of the flood regulating functions of urban watershed ecosystems). In Pakistan, illegal logging and deforestation largely contributed to the devastating effects of the 2010 flood that affected about 20 million people[25] swiped away entire villages[26] and made homeless several million people (Lewis and Kelman, 2012). In Taiwan, the clearing of forests to make space for productive activities and infrastructures has led to reduced slope stability, increased sediment and pollutant delivery downstream, and increased peak flows, a fact that is particularly problematic in a region highly exposed to typhoons and other meteorological hazards (Lu et al., 2001). Though water regulation is a poorly investigated and valued service, there are a number of examples in the literature that demonstrate how watershed restoration can significantly reduce the intensity of weather related events while improving environmental awareness and the reducing long term risk. One example is the Watershed Management Program of Portland (USA), which had the main function to preserve and restore the floodplain to allow flood waters from Johnson Creek to flow freely, while maintaining and restoring biodiversity, improving air and water quality, and providing cultural services. Further examples are presented in Box 4.4 and Box 4.5, in PEDRR (2011), and Table 4.5.

[25] http://www.emdat.be/search-details-disaster-list
[26] 85 villages of Punjab 21 of Baluchistan and 7 villages of Azad Jammu and Kashmir have been affected by the floods (http://www.who.int/hac/crises/pak/sitreps/floods_swat_28july2010.pdf)

Box 4.3. The Marikina River, Philippines

As a result of uncontrolled encroachment and unregulated disposal of waste, the Marikina River, which flows through Marikina City (the Philippines), had become a highly polluted urban waterway, likely to trigger frequent, potentially destructing floods (Yu and Sayor, 2008). Starting in 1993, the “Save the Marikina River” program operated over more than a decade to relocate the population of informal settlements, reduce dumping and establish a recreational park around the river for flood control. Despite being a complex process, the relocation of informal settlements was a success, and 10 years later, the affected communities were satisfied with their safer houses and improved service provision.

Box 4.4. Parque La Agua, Santiago (Chile)

Another case is the Parque La Aguada in Santiago, Chile, currently under construction. It aims at restoring the city’s main ecological corridor to revitalize an abandoned industrial area (World Bank, 2012). Concentrated around Zanjón de la Aguada, a temporary stream, the flood park will cover 60 hectares of river bank, which will provide recreational services during the dry season. The Aguada Flood Park is part of the Santiago Inner Ring Initiative and will cover a 4-kilometer section of the stream, which can no longer accommodate the high-intensity flows of the rainy season. The park also aims to provide economic and social benefits for the adjacent communities (World Bank, 2012).

Box 4.5. Cheonggyecheon Restoration Project (South Corea)

The 2005 Cheonggyecheon Restoration Project created a 6 kilometres public recreation space centred on a seasonal stream in the central business district of Seoul, South Korea. During a period of rapid economic growth, the stream had been transformed into a culvert to make space for transportation infrastructure. In a US$ 900 million effort to improve the environmental quality of Seoul, the metropolitan government removed concrete surfaces and elevated highways to release the historic stream and create a park and floodway, thereby revitalizing the adjacent neighbourhoods (World Bank, 2012).

Droughts and water scarcity also affect urban watersheds. It is in fact estimated that 41% of the world's population lives in river basins where the per capita water supply is so low that disruptive shortages could occur frequently (Fitzhugh and Richter, 2004). Ecosystem-based measures to protect and restore upstream watersheds areas can be implemented to reduce the risk of droughts in cities (see Box 4.6 for an example).

Box 4.6. The Ciudades Y Cuencas Programme (Zampalinamé, Mexico)

In Mexico, the Ciudades Y Cuencas (Cities and Watersheds) Programme promotes the intensification of the relationships between urban citizens and the watershed providing them with freshwater, aiming at raise awareness on the role of watershed ecosystems and to collect resources to contribute to their enhancement. Since 2002, citizens of Zapalinamé (Mexico) pay a voluntary contribution to sustain conservation efforts for the watershed which is providing the city with freshwater. Funding raised by citizens (and integrated by foundations) is employed for environmental management of the natural reserve in the watershed (soil conservation and forest fire control), the constitution of a Water Fund, environmental education and to a small extent for social development projects addressing needs of landowners and communities in the watershed. These interventions are improving water quantity and quality and increasing the city's resilience to droughts. With 15% of the citizens currently paying the voluntary contribution, the initiative is expected to increase awareness about the importance of the natural reserve for the long term protection of the urban water supply (Lechuga Perezanta, 2009).

As mentioned, it is now widely accepted that the impacts of a hazard are the result of the interaction of the hazard itself and of the social and environmental properties of the affected system. Urbanization and urban activities can interact with natural processes in magnifying the impacts of natural hazards, as it has been the case for the cloudburst that killed thousands in Kedarnath and Rambada region of Uttarakhand State (India) on the 15th of June 2013. The disaster was partially attributed to the man-made reduction of the ecosystems' capacity to regulate hydrological extremes, mainly driven by the demand of urban dwellers for hydroelectric power and better infrastructure with little awareness of the potential impacts on upstream ecosystems (Gundimeda, 2013). In addition, when infrastructures are put in place to mitigate the impacts of hazards these often take into account short-term goals and tend to push the most vulnerable fringes of the population to the less desirable, often highly hazard-prone, land, further exacerbating their vulnerability (e.g. the construction of flood levees in New Orleans) (Cutter, 2006). These infrastructures are again often expensive to manage and can further exacerbate environmental degradation. Overall, urban poverty, environmental degradation and disaster risk are closely entangled (Deely et al., 2010).

Highlighting and assessing the interconnections between urban areas and watersheds is essential in designing interventions that preserve or increase the status of ecosystems to reduce the exposure and enhance the resilience of local human communities.

4.3 Valuation methods

Recognition of the value of ecosystems as described in the MA (2005) and in The Economics of Ecosystems and Biodiversity (TEEB) (2012) is growing and attempts for its quantification are increasingly practiced. In this section, we provide an overview on the main valuation techniques in use with some examples of applications to urban watersheds.

In general, ecosystem valuation techniques allow for the quantification and integration of the role ecosystems play in supporting human well-being in a particular location. They offer a series of tools for estimating the amount and distribution of flows of goods and services supplied by the environment and for comparing their evolution under different scenarios. At the watershed scale, valuation techniques need to inform river basin management about which parts of the ecosystem should be prioritized for restoration, improved or protected to guarantee the maintenance of ecosystem functions while supporting agriculture, industry and domestic services (Bergkamp et al., 2000). This is therefore an important step in making the nexus between cities and watershed ecosystems.

Valuation methods for ecosystem services can contribute to slowing down or possibly halting the exploitation and degradation of natural resources and allowing for their better allocation through more informed decisions, at both the individual and the societal level (TEEB, 2012). Though the ecosystem services concept was introduced as an informative notion to raise public interest for biodiversity and ecosystem conservation and restoration, in about three decades ecosystem services have increasingly being valued in monetary terms and, even if to a minor extent, incorporated into markets and payment mechanisms (Gómez-Baggethun et al., 2010). When proceeding to assessments, it needs to be recognised that not all ecosystem values can be expressed in monetary terms, as ethical and societal considerations, albeit of great interest, generally slip out of quantitative approaches. The consideration of non-monetary values alongside with cost-benefit analysis can be achieved recurring, for instance, to multi-criteria, participatory decision-making processes (Bergkamp et al., 2000). Kallis et al. (2013) suggest a framework for ecosystem services valuation which goes beyond the question of the appropriateness of monetary valuation and where environmental improvement and distributive justice are amongst the central criteria considered. As different societies attribute different values to natural goods and services, and as their socio-ecological conditions are in constant evolution, the valuations of ecosystem services are also strongly context-specific exercises (TEEB, 2012).

The value of ecosystems with respect to their services can refer to their biophysical properties or be based on human preferences. With reference to the valuation of services provided by watersheds, a list of the main indicators used for valuation in biophysical terms is reported in IUCN (2006, p. 25) (see Table 4.2), while economic valuation methods of water infrastructures are extensively described in Emerton and Bos (2004). In the next sections we summarise the main valuation methods and link them with some examples and applications to urban watershed services.

Table 4.2. List of main watershed services and related biophysical indicators.
(Source: IUCN, 2006)

Watershed services	Service attributes	State indicator	Sustainable use indicator
Provisioning services			
Water supply	• Precipitation, infiltration, soil water retention, percolation, streamflow, groundwater flow	• Water storage capacity (m^3/m^2)	• Discharge (m^3/year)
	• Biotic and abiotic effects on water quality	• Pollutant concentrations	
Food provision	• Crop, fruit and livestock production	• Agricultural water use (m^3/ha)	• Maximum sustainable water use for irrigation (m^3/year)

Watershed services	Service attributes	State indicator	Sustainable use indicator
	• Edible plants and animals (e.g. fish, algae, invertebrates)	• Fish stock (kg/m^3)	• Net Productivity (kg/ha/year)
Non-food goods	• Production of raw materials (e.g. timber, reeds)	• Amounts available (kg/ha/year)	• Maximum sustainable harvest (kg/ha/year)
	• Production of medicines		
Hydroelectric power	• Flow for energy generation	• Storage capacity of riverbeds and lakes (m^3/km^2)	• Maximum sustainable energy production(kWh/year)
		• Slope (deg), elevation (m)	
Regulating services			
Regulation of water flows	• Retention of rainfall and release (especially by forests and wetlands)	• Infiltration capacity (mm/h)	• Baseflow volume (m^3/year)
	• Water storage by rivers, lakes and wetlands	• Water storage capacity of soils (m^3/m^3)	
	• Groundwater recharge and discharge		
Hazard mitigation	• Reduced flood peaks and storm damage	• Maximum natural water storage capacity (m^3/m^2)	• Size (km^2)and economic value (US$/km^2/year) area protected from flooding
	• Coastal protection		
	• Slope stability		
Control of soil erosion and sedimentation	• Protection of soil by vegetation and soil biota	• Infiltration capacity (mm/h)	• Soil loss (kg/ha/year)
		• Slope length (m)	• Sediment storage (kg/ha/year)
		• Barren land (%)	
Water purification	• Reduced siltation of streams and lakes	• Nitrogen amount (kg/ha)	• Denitrification (kg/ha/year)
	• Nutrient uptake and release by ecosystems	• Total dissolved solids (kg/m^3)	
	• Removal or breakdown of organic matter, salts and pollutants.	• Electric conductivity (µS/cm)	
Supporting services			
Wildlife habitat	• Wildlife and nursery habitats	• Resident and endemic species (number)	• Increase or decline in species population size (number)
		• Surface area per ecosystem type (ha)	
Environmental Flows	• Maintenance of river flow regime	• Area of critical habitats (ha)	• Fish species and population
		• Discharge for each season (m^3/day)	• Total fish catch (t/year)
Cultural and amenity services			
Aesthetic and recreational services	• Landscape quality and features	• Stated appreciation	• Houses on lakeshore (number/km)
		• Recreational value (e.g. entrance fees (US$/visit))	• Visitors (number/year)
Heritage and identity	• Landscape features or species	• Cultural significance and sense of belonging	• Visitors (number/year)
			• Pilgrims (number/year)
Spiritual and artistic inspiration	• Inspirational value of landscape features and species	• Books and paintings using watershed asinspiration	

(Table 4.2 continued)

4.3.1 Monetary valuation methods

For what concerns strictly human preferences, despite a growing interest in non-monetary valuation methods, monetary valuation of ecosystem services remains prevalent in the ecosystem services literature (Haines-Young and Potschin, 2009). This is due to the fact that monetary valuation uses a measure more familiar to people and authorities/policy-makers, and, at present, more directly incorporable in private and public decision-making processes. Especially in urban areas, water is treated as a commodity, and, due to growing population and increased demand, the economic value of ecosystems providing water is substantial (see Box 4.7).

Box 4.7. The Llabcahue watershed, Chile

Núñez et al. (2006) estimated the annual economic value per hectare of native forest in Llancahue watershed (Chile) to be of US$ 162.4 for the summer period and US$ 61.2 for the rest of the year, with respect to their role in contributing to fresh water supplies in Chilean cities. This and other cases indicatively show that the economic value of ecosystems as water infrastructures is relatively high if compared with substitute, engineering solution (Emerton and Bos, 2004)

The Total Economic Value (TEV) is the framework that has been more widely used to estimate ecosystem services in monetary terms. It considers the aggregate amount of use, non-use and option values of the environment, and allows for measuring what individuals and societies gain or lose as ecosystems change. Use values relate to benefits obtained through direct (e.g. production of foods or raw materials) or indirect (e.g. benefits to productive activities through pest control and pollination) interactions with the natural ecosystem (EFTEC, 2005). Use of ecosystems can in turn be consumptive (e.g. use of timber or fuel wood) or non-consumptive (e.g. recreation and education). Non-use values are derived from the simple knowledge of the existence of the ecosystem, or that other people and future generations are or will be able to access the benefits it provides. The TEV framework can also include the ecosystem’s option value (i.e. derived by the possibility of it providing known and unknown benefits in the future) but the opportunity of their inclusion in the TEV measurements is debated (TEEB, 2012).

The valuation methodologies more commonly used are based on estimating the value of an ecosystem service by observing one of the following measures: a) its market value; b) how it influences the

economic choices of people; and c) the people's reactions to simulated changes in its availability. Table 4.3 lists a series of valuation methods, articulated in the mentioned three main categories according to what they aim at observing, what watershed services they can be applied to, and what are their main advantages and limitations. As some valuation approaches are better suited to capture the value of specific ecosystem services, Table 4.4 lists the main water-related services analyzed in the previous chapters with the indication and their most appropriate valuation methods. For instance, for urban wetlands valuation, exercises have been carried out almost exclusively through hedonic pricing.

Monetary valuation methods are then incorporated in policy and decision instruments, as markets or the Payments for Ecosystem Services (PES), described in Section 4.4.

Table 4.3. Overview of monetary valuation methods.
(Based on Pagiola et al., 2004)

Approach	Method	Acronym	Methodology	Application	Example	Advantages and limitations
Direct market valuation - observation of prices, quantity and costs of goods and service on a market	Market price	M	Observe prices of ecosystem good or service in markets	Environmental goods and services traded in markets	Timber and fuelwood production by forests	Data easy to obtain. Inapplicable to non-marketed services and to distorted markets
	Avoided cost	AC	Estimate losses avoided by ecosystem service	Ecosystem services that protect assets and capital	Erosion control by forests	Difficult to capture total damage and to relate it to ecosystem status
	Replacement cost	RpC	Quantify cost of man-made solution to provide the same benefit	Ecosystem services that have a manufactured alternative	Flood control by wetlands (as opposed to engineered structures)	Simple estimation, but depends on human and technological capital of a society, manufactured solution never provides all the benefits of an ecosystem
	Restoration cost	RsC	Quantify cost of restoring lost ecosystem services	Ecosystem services whose loss can be offset or restored	Restoring deforested area	Simple estimation, but full restoration of complex ecosystems is practically unattainable
	Production function	P	Estimate value of a service as an input for the delivery of a service or commodity in a market	Ecosystem services that provide a production input to marketed goods and services	Water purification by wetlands	Implications of ecosystems (and their change) in production are insufficiently understood
Revealed preference - observation of the economic actors' choices associated to a service	Travel cost	TC	Quantify direct and indirect costs to access the ecosystem's site	Areas and sites that provide recreational value	Protected area for recreational or educational purposes	Rely on actual behaviors, but technically difficult, high data requirements and possibly influenced by market failures
	Hedonic pricing	HP	Estimate influence of the ecosystem and of its change on the price of marketed goods and services	Ecosystems that modify the value of marketed good and services	Environmental amenities of buildings and sites for housing purposes	Rely on actual behaviors, but technically difficult, high data requirements and possibly influenced by market failures
Stated preference - observation of the economic actors' choices in a simulated market	Contingent valuation	CV	Estimate directly the people's willingness to pay for a service	Any ecosystem service	Loss of biodiversity	Allow to estimate non-use values, the actors' preferences are hypothetical and non verifiable

Approach	Method	Acronym	Methodology	Application	Example	Advantages and limitations
	Choice modelling	CM	Estimate the people's willingness to pay by ranking different investment options	Any ecosystem service	Loss of biodiversity	Allow to estimate non-use values, the actors' preferences are hypothetical and non verifiable
Others	Benefit Transfer	BT	Use results obtained in one context in a different context	Any service for which suitable comparison studies are available	Estimating the value of one forest using the calculated economic value of a different forest of a similar size and type	Allows estimate the value of ES when access to primary data are non-accessible

(Table 3.2 continued)

Table 4.4. Overview of valuation methods for watershed services.
(Sources Farber et al., 2006; TEEB, 2012)

Type	Service	Valuation methods
Provisioning	Water supply	M, P, RpC, RsC, CV
Regulating	Hazard protection	AC, RpC, CV
	Water regulation	M, P, AC, RpC, CV, HP
	Water purification	P, RsC

Table 4.5. Case study example of application of ES valuation methods in urban watersheds**.** (Source: own table)

Watershed and urban area	**City/Urban area (inhab.)**	**Ecosystem to be recovered or protected**	**Service**	**Valuation methods**	**Value**	**Reference**
Peñablanca Protected Landscape and Seascape	Tuguegarao City, Philippines (136,000 inhab.)	Watershed	Water supply	Contingent valuation	Most of the respondents were willing to pay between UD$ 1 and 1,7 per month	(Amponin et al., 2007)
Layawan Watershed	Oroquieta City, Philippines (68,945 inhab,)	Watershed	Domestic water supply and hazard mitigation	Contingent valuation	More than half of the respondents were willing to pay approx. US$ 1,3 per month per household	(Calderon et al., 2012)
McKenzie Watershed	Eugen-Springfiel Metropolitan area, Oregon USA (200.00 inhab.)	Watershed forest	Water supply	Spatial analysis with Benefit transfer	Between US$ 10 and 48/acre/year	(Schmidt and Batker, 2012)
McKenzie Watershed	Eugen-Springfiel Metropolitan area, Oregon USA (200.00 inhab.)	Watershed forest	Hazard mitigation (flooding and landslides)	Spatial analysis with Benefit transfer	Between US$ 1.40 and 4/acre/year	(Schmidt and Batker, 2012)
McKenzie Watershed	Eugen-Springfiel Metropolitan area, Oregon USA (200.00 inhab.)	Watershed forest	Waste treatment	Spatial analysis with benefit transfer	Between US$ 52 and 182/acre/year	(Schmidt and Batker, 2012)
Chehalis watershed	Hoquiam, Aberdeen, Centralia, and Chehalis (141.00 inhab.)	Wetland	Flood protection	Spatial analysis with benefit transfer	US$ 6,357.71/acre	(Batker et al., 2010)
Sardu Watershed	Dharan, Nepal (118.000 inhab.)	Watershed	Drinking water	Market price	Circa US$ 273.000	(Paudel, 2010)

Watershed and urban area	City/Urban area (inhab.)	Ecosystem to be recovered or protected	Service	Valuation methods	Value	Reference
Johnson Creek, Lents area	Portland, Oregon, USA (576.000 inhab.)	Watershed Wetlands	Water quality services	Contingent valuation and avoided cost	US$ 549 per year per acre of wetland Total: US$ 2,388,982	(Evans, 2004)
Johnson Creek, Lents area	Portland, Oregon, USA (576.000 inhab.)	Watershed Wetlands	Flood protection	Avoided cost or replacement value	US$ 66,700 per 10-yr flood event for all residences Total: $5,437,451 over 100 years	(Evans, 2004)
Cusiles River basin	Matiguás, Nicaragua (9000 inhab.)	Watershed	Water supply	Contingent valuation	Higher willingness to pay under an infrastructure improvement scenario than under a PES approach	(Van Hecken et al., 2012)
Chaina micro-watershed	Villa de Leyva and Chiquiza (Boyacá Department), Colombia (4300 inhab.)	Watershed	Water supply	Contingent valuation	US$1.39/month (with large difference according to the type of users: farmers or recreational house owners)	(Moreno-Sanchez et al., 2012)

(Table 4.5 continued)

Estimation of the value of watershed services to urban areas allow to make explicit the links between the local and regional scales. In-situ contingent valuation seems to be the preferred method to value urban water-related services (see Table 4.5). Benefit transfer is also a valuation technique frequently recurring in the literature but seems to be less explored, possibly for the mentioned context-specificity of ecosystem services valuation.

Due to the context-specific nature of the valuation exercises, their results are extremely difficult to compare and scale up (Farber et al., 2006). In fact, even in the case of direct market observations for goods and services that are traded on global markets, their value will depend on local levels of demand and supply and access to economic assets and natural resources. The supply of one service is often strictly entangled with other ecosystem functions, which increases the overall value of the ecosystem. As an example, the value of a wetland can depend on the fact that it provides some or all of these services: flood control for downstream urban areas, water filtration for near sources of urban drinking water, opportunities for bird and wildlife watching and fishing (Boyer and Polasky, 2004). The value of each will be greatly influenced by the socio-economic context (e.g. distance from one or more cities, their size, their position up/downstream, their economic specificities). Context-specific valuation methods can, on the other hand, inform local policies on micro-level elements such as the amount of money downstream watershed users might be willing to pay to upstream users to maintain healthy ecosystems. The wide variety of estimates listed in Table 4.4 confirms the necessity to carry ad hoc, in-situ studies due to the high variability of the results obtained. With respect to the type of service assessed, water supply is highly considered in urban watershed studies, while there are fewer studies that focus on the hazard regulation and wastewater purification functions of watershed ecosystems for urban areas.

As most methods focus on valuing one or some of the whole range of services provided by an ecosystem, thereby neglecting the environment’s diverse and complex values and benefits to the well-being of human communities, integration of different methodologies and participatory approaches to consider multiple services is often necessary. It should be made clear that monetary valuation need to be accompanied by a broader set of considerations. There is a wide range of situations in which the cost-benefit valuation of ecosystem services is not considered as feasible or an appropriate option (Kallis et al., 2013; TEEB, 2010). It has also been demonstrated that pricing can be counterproductive in terms of biodiversity conservation and equity in the access to resources (Gomez-Baggethun and Ruiz-Perez, 2011). Alternative assessment strategies, based on non-monetary values, are presented in the next section.

4.3.2 Non-monetary valuation methods

Estimating the value of environmental processes for human communities is grounded on biophysical assessments as well as on social and economic analyses. It is therefore inherently multidisciplinary in nature and often best pursued through participatory processes that actively involve stakeholders (Haines-Young and Potschin, 2009). For instance, the social value of a watershed is typically greater than the development value which would benefit a private owner (Boyer and Polasky, 2004). To express these values, participatory valuation exercise can lead to a simple ranking of different benefits provided by a watershed ecosystem to an urban area. As mentioned in Wilson and Howarth (2002), group valuation can be appropriate for ecosystem services that are generally public in nature, for which methods based on the elicitation of individual preferences, such as contingent valuation, might not be adequate. Table 4.6 lists and describes the main participatory valuation methods in use to assess ecosystem services. Few applications of participatory methods to the valuation of urban watershed services are available. Some of them are listed in Table 4.7.

Table 4.6. Participatory and non-monetary valuation methods
(Source: elaborated from TEEB, 2010)

Approach	Method	Acronym	Methodology	Advantages and limitations
Non-monetary valuation methods/participatory appraisal	Individual index based method	IS	Rating and ranking choice models, expert opinion. Or questionnaires to individual stakeholders for semi-structured, narrative or in-depth interviews	Flexible and useful in contexts where there are conflicts between different views and it is necessary to establish the source of the disagreement.
	Individual experts views	IE	Delphi surveys (iterative process including a series of deliberations)	Particularly useful when existing knowledge is limited
	Group-based (e.g. focus groups)	GB	Including voting mechanisms, focus groups, citizens juries, stakeholders analysis	This approach is based on principles of deliberative democracy and the assumption that public decision making should result from open public debate. It is useful to gain insights about institutional linkages and relationships.
	Group stakeholders viewpoints requiring in-depth statistical analysis	Q	Q-methodology (helps determine the nature of individual relationships and perceptions of environmental problems and solutions)	MC assessment is particularly useful when stakeholders identify non-negotiable outcome
	Multi-criteria analysis	MCA	Multi-criteria analysis (helps structure decisions characterized by trade-offs between conflicting objectives, interests, and values; it can be complementary to CBA).	While CBA aims at economic efficiency, MCA includes value expressed in different terms. MCA determines how one services is important with respect to other services (trade-offs)

Table 4.7. Example of non-monetary and participative valuation studies or urban watersheds

Watershed and urban area	City/Urban area (inhab.)	Ecosystem to be recovered or protected	Service(s)	Valuation methods	Value/results	Reference
Lobau floodplain, upper Danube river	Vienna City, Austria	Watershed wetlands	Recreation; groundwater abstraction for drinking water production	Multicriteria decision analysis (MCDA)	The majority of the involved management sectors preferred the higher connectivity options as compared to the Current Status option. Potential conflict between the ecological development and the drinking water production.	(Sanon et al., 2012)
Chicopee Watershed	Boston area, Western Massachusetts, USA (190,600 inhab. in the watershed)	Watershed	Water supply	Non-monetary deliberative method (Deliberative Attribute Prioritization Procedure DAPP: combines a multi-criteria analysis using pair wise comparisons with a process to reach group consensus)	Density of toxic waste sites has the highest priority weight with a value of 0.19. Runoff has then the next highest ranking with a priority weight of 0.15.	(Randhir and Shriver, 2009)

4.4 Policy and economic instruments

Urban water policies based on the results of valuation studies, derived from the application of the methodologies described in the previous sections, are essential to the management of a number of services provided by watersheds: provisioning of clean water in sufficient quantities, release of clean used water to the environment downstream or to the groundwater and protection from droughts and flooding. These services can be provided, and are provided (in particular in most urban areas of the richer countries) through centralised, technology-oriented measures, such as large-scale water retention systems, dams and long-distance pipelines ensuring freshwater provisioning, dikes and drainage systems as flood protection measures, sewerage systems and plants for wastewater reclamation and the reduction of pollution. However, these solutions have often demonstrated to lead to additional environmental degradation and to require continuous and expensive maintenance interventions. As described in the previous sections through numerous case studies, ecosystems provide alternatives or integrations to these technology-based systems. Ecosystems management approaches for resilience in urban areas make use of the existing natural landscape and can significantly decrease the cost and impacts of urban infrastructure projects (Boyer and Polasky, 2004). Even when they are adopted to replace or improve ecosystem services, technological solutions ultimately rely on functioning ecosystems (Brauman et al., 2007).

The separation between the human and ecological dimension that predominated in the past and still exists in development and hazard theories (Khan, 2012; Khan and Crozier, 2009), led scientists to find solutions to environmental problems, including natural hazards, through the modification, substitution and suppression of environmental processes. The application of an ecosystem approach stresses the connection between urban areas and local as well as more distant ecosystems, as are the watersheds in which cities are located and from which they derive important benefits. A better recognition of watershed services, their value and connections to the local urban environment would thus benefit urban areas in terms of social, economic and environmental efficiency, as described in the previous sections.

Ecosystem management strategies can help maximise the resilience-enhancement potential of natural systems for urban dwellers, by making full use of the capacities of wetlands and natural vegetation in water bodies for water reclamation and of green areas as buffer and regulating element against floods, for water retention and for groundwater recharge. The value of urban water-related ecosystems as alternative or complement to technical solutions has been recognized in many cases (see above). Ecosystem-based options may thus represent doubly effective solutions as urban areas not only

benefit from watershed services but are also the drivers of ecosystem use, change and management at this scale.

A particular strategy that found multiple applications at the watershed scale are PES (Dillaha et al., n.d.; Smith et al., 2008). PES are defined as "(a) a voluntary transaction where (b) a well-defined environmental service or a land use likely to secure that service (c) is being 'bought' by a (minimum one) service buyer (d) from a (minimum one) service provider (e) if and only if the service provider secures service provision (conditionality)" (Wunder et al., 2008, p. 835). The literature on this arrangement is vast. Cases of payments for water provisioning services for cities are extensively reviewed in Buric et al. (2011). Most schemes reviewed by these authors were located in South America, in particular in Brazil, and, in the majority of the projects, forestation or reforestation were the main land-use changes implemented. The payments schemes were mainly realised in not extremely degraded watersheds and were driven by some of these aspects:

- avoiding expensive technological solutions for the improvement or conservation of quality of drinking water;
- acting early to protect critical watershed land, i.e. avoid imminent water pollution induced by the change of land-use practice;
- managing the risk of potential water degradation by making a preventive investment into conservation of the current water supply/quality;
- mitigating the effects of watershed degradation in order to improve the quality of water (Buric et al., 2011),

There seem to exist a series of preconditions that facilitate the decision and implementation of payments schemes, in particular the need for a community to prevent or halt initial condition of degradation. However, PES can ultimately facilitate the transfer of resources from urban areas to upstream social-ecological systems, which in turn has the potential to significantly improve the well-being of the downstream urban populations. PES programs have also demonstrated to contribute to curbing urban growth in rapidly urbanizing megacities such as Mexico City (DuBroff, 2009).

As described in Wunder et al. (2008), user-financed PES are generally better targeted and tailored to local conditions if compared to programs initiated by governments (or another third party). Not unlikely the ecosystem valuation exercises and their results, the PES programs demonstrated to be highly context-specific (Wunder and Albán, 2008). Especially in the global South, while economic valuation can be informative on the value attributed to up/downstream services, the actual structure of payments schemes can often be the result of complex social processes involving multiple

stakeholders rather than of a merely technical assessment (Kosoy et al., 2007). For some examples of application of PES schemes see Box 4.7 and Box 4.9.

Box 4.8. The Miyun watershed in China

In response to the watershed degradation, from the mid-1980s the Government instituted strict controls on land and forest use, including a total ban on logging, and invested substantially in a reforestation program. The logging ban caused the forests to be neglected, rather than sustainably managed, with the consequence that they weren't contributing much to soil, water and biodiversity conservation. Also, local communities outside of Beijing were suffering increasing economic hardship, due to the lack of income alternatives to the exploitation of forest products.

However, since 1995, the Beijing Municipality has compensated upstream settlements with the annual payment of US$ 2.5 million for the adoption of soil and water conservation measures and subsidies to farmers who converted paddy fields to dry farmland, forest or grassland. Recognizing the multiple needs and functions associated with a watershed, in 2007 IUCN identified and then introduced through participatory processes a new set of forest management tools that allowed for a shift from a strict protection-oriented approach towards more sustainable resource use by forest-based communities. Local communities are responsible for applying silvicultural treatments that improve forest structure, quality and function. For instance, support has been provided to establish community-based cooperatives for marketing forest goods and services, with the aim of increasing and diversifying local income (IUCN, 2010).

Box 4.9. The New York PES Scheme, USA

Until the XIX century, New York's water supplies depended almost exclusively on a single, inner-city Collect Pond and the city wells were constantly contaminated by the wastewater produced by upstream settlements on the Erie Canal, which resulted in the cholera and yellow fever outbreaks of 1832. In 1842 a first aqueduct was finished, connecting the city to the nearby Croton watershed, which still supplies 10% of the city's freshwater and then to the Catskill-Delaware watershed, which supplies the 90% (Appleton, 2002). By involving the upstream stakeholders into the management of its water resources, the municipality has been able to establish land-use practices and policies that protect the services provided by the watershed ecosystems. In 1997 the city signed a Memorandum of Agreement, committing to invest around US$ 1.5 billion over 10 years to restore and protect the surrounding watersheds, as well as to promote measures to improve the local economies of watershed residents (Postel and Thompson, 2005). A comprehensive study of the National Research Council committee has highlighted a whole range of non-structural measures that have been established for water quality protection, such as land acquisition, buffer zone designations, conservation easements, and zoning ordinances (Pires, 2004). The process has allowed for changes that eliminated the need for industrial water filtration for the downstream megalopolis. It has been noted how the protection of these natural areas through the institution of nature reserves, national parks and wilderness areas allows both for the conservation of local biodiversity and for the enhancement of water resources the city depends on (Postel and Thompson, 2005).

The transfer of resources for environmental management and restoration of upstream areas also prevents land abandonment, which has negative environmental consequences. For instance, in the Miyun Case (see Box 4.8), as in other cases (Harden, 1996; Raj Khanal and Watanabe, 2006), land abandonment and spontaneous forestation as a consequence of restrictive land-use policies in response to overexploitation and degradation of watershed ecosystems has not had positive environmental impacts. Similarly, evidence suggests that cultivation and extensive maintenance of mountain slopes in the Middle Mountains of Nepal guarantees high degrees of stability while rapid de-intensification leads to slope instability (Smadja, 1992). Local food and livelihood security also tend to diminish when agricultural land is abandoned while the occurrence of mountain hazards, such as floods and landslides, increases (Raj Khanal and Watanabe, 2006). Other frequent consequences of land abandonment are "biodiversity loss, increase of fire frequency and intensity, soil erosion and

desertification, loss of cultural and/or aesthetic values, reduction of landscape diversity and reduction of water provision” (Rey Benayas, 2007). The abandonment of agricultural land and subsequent unmanaged reforestation processes have often resulted in the loss of endemic species and the proliferation of invasive, often exotic, ones, causing additional environmental problems. The spread of non-native invasive tree species with high evapotranspiration requirements in the Western Cape watershed (South Africa) has negatively impacted water supply (Postel and Thompson, 2005). Local livelihoods therefore play a major role in healthy watershed management. The abandonment of slopes for floodplains has often “worsened people’s livelihoods, enhanced social conflict and taken critical environments out of community control” (FAO, 2007).

The creation of public parks and the restoration of rivers have been practiced in various urban contexts in which urban planners and water managers needed to prevent and mitigate hydro-meteorological hazards. While one of the earliest recognized successes may be Curitiba (Brazil), there are a handful of cases in Spain (see Box 4.10), Australia (see Box 4.11), the Philippines, Chile, and Korea. Areas dedicated to conservation and watershed services benefiting urban areas have shown to significantly overlap in the 105 cases analysed in a study by (Dudley et al., 2003) where concerns for the integrity of water supply were the main reason for the instauration of protected areas.

Box 4.10. Flash floods in Barcelona, Spain

In Barcelona, the flash-flood-prone Besos River was restored to a meandering low-flow channel within a wider floodway of constructed wetlands (Martín-Vide, 2001). Intense urbanization in the 1960s had led to the encroachment of 300,000 poor residents into the original Besos floodplain. Planning for river restoration began in the mid-1990s in an effort to improve the environmental quality of the city, control floods, and provide a green recreation space for the target municipalities.

Box 4.11. Droughts and flash floods in Melbourne, Australia

In Australia, where frequent droughts and occasional extreme precipitation events have accelerated recognition of the particular importance of water as a natural resource in urban areas, the "Water Sensitive Urban Design principles" (WSUD) are gradually evolving from an experimental stage, where single measures are tested in small parts of the urban areas, into institutionalized practices. The State of Victoria has, for instance, mandated WSUD principles in its State planning provisions (Rijke et al., 2013). Facing recurring water scarcity and threatened by decreasing water availability due to climate change, all major Australian cities have to some extent modified their patterns of water management. Among them, Melbourne with its program "Total Watermark - City as a catchment" (City of Melbourne, 2009) has a forefront role in the implementation of watershed management principles into the urban context. The measures implemented cover both aspects of water quantities and quality, focusing mainly on rain- and stormwater harvesting and increasing water efficiency (considering both households and productive activities). The program is based on targets for the quantitative water balance and for the discharge of pollutants, related to stormwater runoff, aiming at the development of a "water sensitive city" conceived as "a catchment where stormwater and treated wastewater are important water sources" (City of Melbourne, 2009, p. 51). To this aim, a series of measures have been adopted, including non-structural techniques for water efficiency and prevention of stormwater pollution at the source, demand-management strategies, regulation, planning controls and financial incentives (City of Melbourne, 2009).

Evidence thus suggests that, when implementation policies are able to buffer the socio-economic disadvantage generally affecting marginal, upland and lowland communities and when ecosystem service users are willing to pay for improved environmental quality and service delivery, good watershed management and upstream/downstream balance can be achieved. Cities need therefore to be better connected to environmental management strategies and socio-economic practices of upstream and downstream communities.

4.5 Concluding remarks

In this working paper we presented urban areas as parts and defining units of catchments, with their own environmental and water balance, yet inextricably connected to the catchment basin in which they are located. The nexus between the urban system and its surrounding ecosystems are analyzed

in two directions: water quality (nutrients regulation and pollutants removal) and quantity (water supply, drought and flood resilience). This approach aimed at highlighting, through the description of water-related services benefiting urban populations and through numerous related case studies, that urban management has the potential to be the driver of watershed conservation and restoration. Cities and urban areas should be incorporated as central administrative units within the management of watersheds as these derive important services and are most of the time the direct or indirect drivers of environmental degradation upstream and downstream. As suggested in Grimm et al. (2000), the ecology of cities should therefore better include this regional perspective. An important first step is to acknowledge the importance of local as well as of remote ecosystems in decision-making for urban management. This means that, when designing and implementing policies and planning, urban authorities might need to adopt a wider geographical perspective, which we suggest to be the watershed level. With an ever increasingly amount of people settling in urban areas, cities need to become the drivers of this regional ecosystem approach to improve the conditions of local and more distant ecosystems, not least through the transfer of resources. At this regard some summarizing remarks are:

- sustainable watershed management demands the inclusion of different sectors and stakeholders, promoting participative methods and solutions, as it stated in the principles of Integrated Urban Water Management (IUWM) (Bahri, 2012);
- ecosystem-based solutions should be particularly valued by local authorities, as many of the services ecosystems provided are included in their basic mandate. The nature of the competence of local authorities (linked to a territory, rather than to a specific matter), makes the use of integrated solutions generally easier and more effective;
- ecosystem-based solutions provide co-benefits that go well beyond their direct utility here analyzed. Benefits such as recreational, esthetical and spiritual opportunities, incrementing the economic value of properties, fostering the cultural life of urban dwellers and supporting biodiversity and life are some of the desirable side-effects of these interventions;
- integrating ecosystem management and restoration within urban planning and disaster risk reduction measures at the watershed level is a long term, (cost-)effective approach to increasing the resilience of human communities and urban centers, in particular in the face of natural hazards, while enhancing the quality of watersheds ecosystems, as demonstrated by the numerous case studies reported in this paper;
- though cities have been driving environmental degradation in the past century, it is increasingly at the urban level that social, economic and cultural change happens. Even in the lack of

overarching national and international agreements, cities can then play a central role in improving regional ecosystem health through the transfer of resources while diminishing their own expenditures and risk;

- despite the limitations highlighted above, most of the studies available in the literature that attempt to assess urban watershed services rely on pricing. It is recommended that alternative valuation methods be applied to take into consideration a broader range of values and to examine the potential outcomes of different urban management options also according to the expected environmental improvement and consideration of social equity;
- water supply seems to be the most investigated service originating at the watershed level and benefiting urban areas. This can be the driving sector for this regional ecosystem approach. More research with concrete examples needs however to be carried out with respect to the capacity of urban watershed to perform hazard mitigation and wastewater treatment functions.

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CONCLUSIONS

The various chapters of this Thesis have provided some examples of the application of the concept of ES in integrated, policy-oriented studies with respect to some of the most pressings issues that European urban areas face nowadays, such as population well-being and hazards risk. I have also proved how the concept is a promising tool to advance sustainability, especially by leading to policy-relevant conclusions. Much energy has in fact been dedicated so far to the categorization and economic valuation of ES, paying perhaps less attention to the vast range of potentialities that can be derived by applying the integrated ES framework to advance sustainability in a theoretical and policy context. The main contributions are detailed here below.

Opportunities for ES in DRR

The objective of most environmental, integrated studies is to improve the well-being of human populations, which is also the priority of policy interventions. It is for this reason that anthropocentric concepts and approaches, such as the one of ES, are specially suited to complement with ecological information in environmental assessments. Aiming at contributing to the assessment of vulnerability to natural hazards, this Thesis focuses mainly on regulating services for urban areas. The results and evidences reported confirm the importance of preserving well- functioning ecosystems in and around urban areas for DRR and protect human well-being. In Chapter 1 I gathered evidence throughout the results of a large number of studies showing how healthy ecosystems matter for mitigating the impacts of hazards in urban areas, in particular of floods and heat waves. Despite this, I highlight how the vulnerability of the exposed ecosystems needs also to be taken into account to verify that the supply of services in case of extreme events is guaranteed and not affected.

I have then proved in both the cases of Cologne (Chapter 2) and Barcelona (Chapter 3), that urban and peri-urban ecosystems provide the much needed environmental services required to improve human well-being and reduce the vulnerability to hazards of the urban population. In Cologne, urban ecosystem features and the distribution and access to green space were the most important variables in determining the vulnerability of the social-ecological system to heat waves. The integrated assessment showed how, in this case, policies should perhaps target less social factors, such as age or the socio-economic status of the population, and concentrate more on ecosystem distribution, health and connectivity. This might not be the case in other cities and, according to the variables considered, studies might reveal that other types of determinants of vulnerability, such as tight social networks or improved early warnings, might be needed to reduce risk. Nonetheless, the state of ecosystems plays

a role both in determining exposure and lack of resilience of urban populations and should be taken into account amongst other factors.
In Barcelona too I showed how the environmental performance of the city greatly improves thanks to the existence of the adjacent Collserola Natural Park which allows the city to align, in terms of the environmental benefits the population can enjoy, with other greener western cities. This result is even more relevant considering the risk of floods and heat waves that threatens the city. Furthermore, in both case studies I demonstrated how the ecological component of vulnerability is the result of historically traceable processes, mostly linked to the predominance of certain ideas of city in local urban planning. Numerous examples in Chapter 4 also reveal the cost-effectiveness of ecosystem-based strategies due to the lower cost involved in ecosystem restoration, especially in the long run, as well as the advantages of including the provision of complementing, well managed ecosystems functions in the construction of grey infrastructures.
What is to highlight about urban areas is that ecosystems are often degraded by urbanization itself and thus less capable to provide services or even more susceptible to be affected by hazards. Policies aiming at applying ecosystem based approaches in urban areas should therefore carefully tackle ecosystem health, for instance in terms of connectivity at different scales.

Theoretical insights on the ecological dimension of vulnerability of coupled systems to natural hazards

As represented in Figure 0.1.c, ES play a role both in regulating exposure of social-ecological systems to some natural hazards, such as hydro-meteorological ones, and at increasing their resilience. Regarding exposure, ecosystems regulate urban microclimate, diminishing the UHI effect which is responsible for the increasing impacts of heat waves, or improve soil infiltration, reducing pick flood at the small catchment scale. Furthermore, ecosystems increase resilience providing alternative strategies to cope with the hazard. For instance, wetlands might retain water, slowly releasing it whilst reducing water scarcity in periods of droughts. Similarly, access to parks provides relief to heat stress in case of periods of extreme heat, and ecosystems can provide alternative sources of livelihoods when a hazard strikes. Susceptibility of social-ecological systems is, on the other hand, not quite adequately expressed as the quality of the dependence of the human system on the ecosystem as this component describes characteristics of the objects at risk which are, in our case, ultimately human beings.

As mentioned, in Chapter 1 I also highlighted how ecosystems can themselves be affected by hazards, especially if fragmented and degraded. This can in turn affect the urban population through the failure of the supply of services (e.g. water supply or peri-urban agriculture) when a hazard strikes. In the

case of Cologne, I have relied on the knowledge of experts, to show how heat waves affected in the past and could affect in the future, peri-urban agricultural activities and recreational areas, while the city relies on abundant sources of groundwater and is not threatened in this aspect. Generally, policies should therefore adequately preserve and increase the resilience of ecosystems to hazards, tackling ecosystem degradation or unplanned urban expansion. Hazards are in fact part of the biophysical system and, per se, can provide a series of benefits to ecosystems, such as renewal (in the case of forest fires) or nutrients supply (in the case of floods). However, hazards on degraded ecosystems, such as small eutrophic ponds or those affected by fragmentation or high presence of alien species, can lead the ecosystems under stress to cross thresholds, to lose their functions and be temporarily or permanently unable to provide services.

Information about ecosystem vulnerability to hazards is scarce and most of the studies focus on vulnerability to climate change or to land use change, as a consequence to the actual anthropogenic component which becomes embedded and brings changes in the frequency and intensity of weather and climate related hazards as well as changes in exposure. Regime shifts are in fact mainly caused by human action (Folke et al., 2004). Ecosystem vulnerability is an underdeveloped subject even in ecotoxicology due, amongst other, to the difficulty in representing emergent properties of ecosystems (De Lange et al., 2010; Ippolito et al., 2010). Alternatively, community interactions, to account for the health of keystone species (De Lange et al., 2010), could be considered to assess the susceptibility of the exposed ecosystem, together with the number of both introduced and endangered species, per unit of surface (as included in the Environmental Vulnerability Index of UNEP). Furthermore, Folke et al. (2004) identify in the level of biological diversity an indicator of the response capacity of ecosystems, similarly to Naeem and Li (1997), who look at functional diversity or to Walker et al. (1999), who look at functional redundancy with respect to the buffering capacity of the ecosystem against perturbations or environmental variability. This last variable could be even more important than species richness (Díaz and Cabido, 2001; Elmqvist et al., 2003). The EEA integrates in its methodology for the assessment of green infrastructures, indicators of habitat health such as the minimum size suitable to host mammals, as this requires large and well-connected natural areas for their survival and movement, to account for the actual capacity to provide services (EEA, 2014), which can also be used as a measure of the resilience of the system.

This type of information would be the base to include the degree of ecosystem vulnerability as a measure of its capacity to overcome external shocks and continue in supplying services, also in the short-medium term. For this purpose, I suggest to consider the revised MOVE framework depicted in Figure 0.1.c, in which the introduction of alien species, of endangered ones and the health of

keystone species would be a measure of susceptibility of the exposed and degraded ecosystem, while response diversity or habitat health would constitute the measure of the system's resilience. Note that these ecological variables complement information on the coupling between the social and ecological system, expressed in terms of ES, as the dependences of the social system on the ecosystem serve as a basis to account for the actual capacity of the degraded system to continue to supply ES when affected by a hazard.

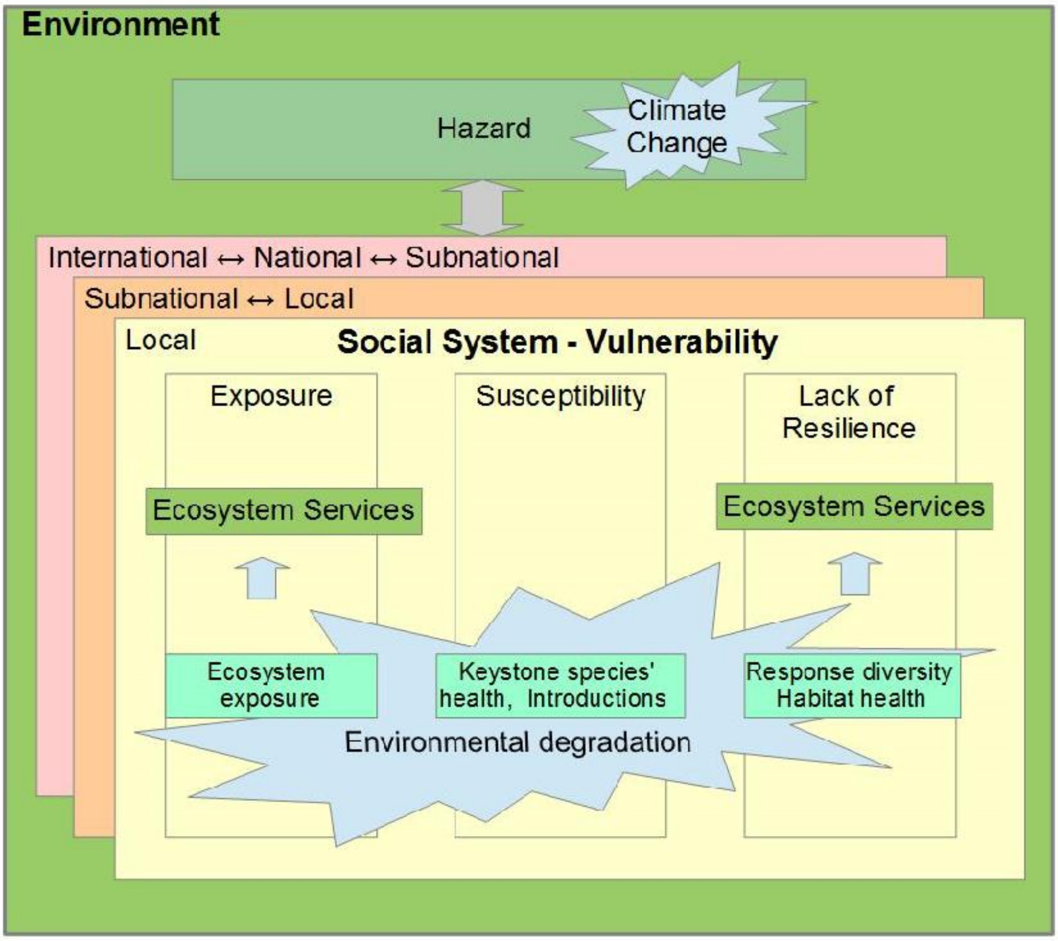

Figure 0.1.c. Revised MOVE Vulnerability Framework with a focus on the social-ecological component.

Thus, regardless of the increasing spread of holistic and integrated approaches, which get perhaps progressively detached from the original disciplines, these might still need to be solidly grounded in the findings and value systems of disciplinary approaches and cannot be reduced to a single valuation system or paradigm. That means for instance that to assess the vulnerability of the population of an urban area, ecological knowledge is still needed to estimate ecosystem health. This is ever truer in and around urban areas where ecosystems are highly altered.

ES in urban areas as social constructions

Chapter 3 aimed at contributing to the field of political ecology presenting a case in the urban context looking at how ES are ultimately socially constructed through a series of historically traceable processes and decisions. In our analysis we have shown how the social construction of nature in urban and peri-urban areas can happen passively as a consequence of socio-economic processes, or actively based on different ideas of nature brought by planners or by social movements. The conservation of the Collserola Natural Park adjacent to Barcelona necessitated, in fact, the continuous intervention and update of the legislation, as well as the action of social movements against an ever expanding urbanization. In this way, ideas of nature were brought into existence through power struggles. These processes are ever more intense and detectable in the urban context due to the highly contested space and the high value of land. However, disputes about nature conservation in the urban context have been paid little attention in political ecology.

On the other hand, the vision of nature as detached from human influence, advanced by some authorities or social movements, often reinforces the definition of a landscape as separated by its historical and management context. We have seen how ideas of a well preserved or pristine nature for conservation in the case of Barcelona are not backed up by historical and ecological evidence. The success in the implementation of these ideas has led nonetheless to a certain configuration of the ecosystem, ultimately determining its capacity to supply services to the city. Some of the examples presented in Chapter 4 about Payments for ES also illustrate how the management of water and land at the watershed scale is a matter of political decisions. The implementation of these schemes often entails a change in the type and beneficiaries of the services provided by the watershed, shifting from a rather rural use to the goal of satisfying more multiple or urban needs.

ES, therefore, are not simply external to human control but are instead often the result of the confrontation between power relations and conflicts. Recognizing that human well-being depends on certain ideas of nature could lead to more transparent processes for sustainability. I in fact stress here the idea that the recognition of the ecosystem as a social practice of articulation of value could significantly contribute to foster sustainability: besides improving our knowledge on how to manage ecosystems and their services, we can and should in fact ask what kind of ES we need or want, remaining of course within the local environmental limits. The recognition of ES as the product of historical, socio-political and biophysical processes for a given area could serve in fact as a base for discussion for a deliberative process of planning and management in which politics are also accounted for.

Scales and ES: urban ecosystems redefined

In this Thesis, I adopted a multi-scalar approach to the assessment of ES for urban areas to identify, amongst the ES analysed, the appropriate scale of service provision, giving, when possible, also an estimation of its effectiveness. The findings are summarized in Table 0.1.c and can be considered, with case by case variations, as representative for medium to large European cities.
Urban cooling in densely built urban ecosystems supplied by well distributed urban parks and street trees within the urban fabric has come out as the most effective benefit at the very local scale. Other services provided at this scale, though relevant, have demonstrated to be less successful in providing benefits. For instance, cities might more effectively combat air pollution through its reduction at the source or by establishing pedestrian zones in the city core. Planting trees might in fact not suffice to counter the threat, as we have seen for the case of Barcelona. Similarly, at the urban scale a mix of soft (e.g. green roofs) and hard infrastructures (i.e. a drainage system), or of Sustainable Urban Drainage Systems (SUDSs), might be necessary for flash flood mitigation due to the extremely high degree of sealed surfaces in urban areas.

On the other hand, peri-urban ecosystems, especially if well connected to the urban core of densely populated cities and depending on the configuration of the biophysical environment, appear to provide the bulk of the services analysed (i.e. urban cooling, flash flood mitigation, wastewater treatment, food supply and recreation). Moreover, the regional, watershed scale is relevant for the supply of services linked to water regulation and purification. Again, to note is that these ecosystems would not always suffice in their purification functions to process the wastes of industrial cities or treat wastewater, and that policies targeting the reduction of the emissions at the source are in fact necessary. Concerning other threats to urban well-being such as water scarcity, the UHI, flash floods, these could on the other hand be significantly and satisfactorily tackled through adequate planning and ecosystem restoration.

What this analysis reveals is that the different scales of urban ecosystems appear to be complementary in terms of the ES they provide, a trait that should perhaps be emphasized in urban planning. While Alberti et al. (2003) see city resilience as dependent on the capacity to maintain simultaneously ecosystems and human functions, I stress the connection between the city core and healthy surrounding ecosystems to be central. Locating the cities within their surroundings and regional systems seems to be an unavoidable perspective to improve the quality of life in urban areas.

Table 0.1.c. Scales, relevance and effectiveness in ES supply to urban populations as analysed in this Thesis.
(/ not relevant, + relevant, ++ very relevant, n. c. not considered).

Ecosystem service	Urban ecosystem	Peri-urban ecosystem	Regional ecosystem
Urban cooling	++	++	+
Air purification	+	+	/
Flash flood mitigation	+	++	+
Riverine flood mitigation	/	+	++
Water supply	+	+	++
Wastewater treatment	/	++	+
Food supply	+	++	n.c.
Recreation	+	++	n.c.

The examples of Barcelona and Cologne dealt with in this Thesis show, for instance, the importance of the presence of green belts and forests in and especially around the urban core. This has implications as the high price of land in urban areas often discourages its use for planting vegetation. Therefore, planning interventions need to be well targeted. Furthermore, these findings support the compact city model if surrounded by a well-managed forest. Urban sprawl not only often increases the risk to hazards of urban populations, but also damages the surrounding ecosystems likely to provide important services required to enhance and ensure human well-being. I have proved in Chapter 3 that the Regional Planning approach of Geddes and Mumford implemented in Barcelona by Rubió I Tudurí appears to be justified not so much because of the wilderness of the peri-urban park but because of the ES it provides to the city, not least in terms of recreational opportunities. These considerations are even more relevant bearing in mind that at present one of the main threats facing urban areas is disaster risk which demands sustainable solutions at broader scales than the mere urban core.

Finally, humans constitute an integral part of ecosystems, but social and economic systems can be described as subsystems of the ecological systems according to the field of Ecological Economics. Similarly, urban systems can be seen as part of larger systems (the peri-urban area, as seen in Chapter 3, or the watershed, as seen in Chapter 4). I have shown that cities, by adopting a broader spatial perspective, could benefit from an enlarged set of policies based on the concept of ES which would place them as having a primary role in the environmental restoration of surrounding and more distant ecosystems.

Concluding remarks and future research

In this Thesis I explored through different integrated frameworks, some aspects of the well-being and vulnerability to hazards of urban populations, mainly in Europe. However, as most of the arguments treated here are emerging or growing fields of study, the results reached should be intended as a base for future research. This applies, for instance, to the theoretical work on the social-ecological dimension of vulnerability to hazards or on the political ecology of ES, but also to the multi-scalar approach to urban ES assessment. These approaches have nonetheless demonstrated to have high potentialities to draw policy-relevant conclusions and to support the effective and sustainable ecosystem management to ensure urban well-being.

Some suggestions and remarks for future research are offered as follows:

- Increase in policy-relevant studies through the application of the ES concept beyond monetary valuation or classification exercises;
- Consider ecosystem health and its services in the integrated assessment of vulnerability as these can play a primary role in shaping the spatial vulnerability of populations. Frameworks such as the one I suggest here could help in taking into account the different factors involved;
- Diminish the understatement of the historical and political dimensions of ES as these dimensions can be a base for discussion in deliberative processes for sustainability;
- Enlarge the multi-scale approach to urban ecosystems as delineated in this Thesis across the chapters in terms of, for instance, other types of services or case studies that can increase the efficiency of policies and decisions.

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Ecosystem Services In Practice: Well-Being And Vulnerability Of Two European Urban Areas

by Yaella Depietri

This Thesis explores how ecosystems and their services improve the livability of cities and reduce the vulnerability of urban populations to hydro-meteorological hazards. It focuses on two European medium-sized urban areas: Cologne in Germany and Barcelona in Spain. Through integrated approaches it explores how and why cities can benefit, in terms of well-being and reduction of vulnerability, from a broader range of policies that adequately account for ecological aspects at different geographical scales.

The research concludes that the regulating functions of ecosystems often provide efficient, cost-effective alternatives or complementary solutions to hard infrastructures for the wellbeing of populations and disaster risk reduction in urban areas. Empirical research in both case studies shows how urban and peri-urban green areas provide services which reduce the exposure of the population to hazards and improve the quality of life of densely inhabited cities. Beyond the local scale this research also explores the peri-urban and more regional, watershed scale as important levels for the supply of services to urban areas.

From a theoretical perspective, the Thesis sheds light on the social-ecological coupling which shapes vulnerability to natural hazard by analyzing in depth the role played by the poorly explored ecological dimension. It also adds to the literature on the political ecology of ecosystem services in urban areas showing how these benefits are ultimately socially constructed and thus need to be part of a broader decision making process.

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